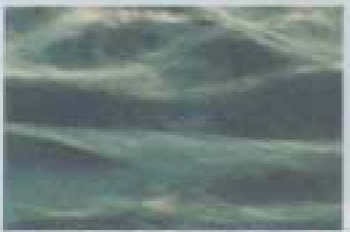
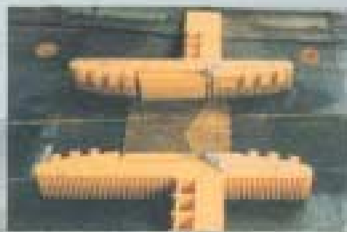
ABP Southampton

REVIEW OF COASTAL HABITAT CREATION, RESTORATION AND RECHARGE SCHEMES

1998

Report No. R.909





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ABP Research & Consultancy Ltd
Pathfinder House
Maritime Way
SOUTHAMPTON
Hampshire
SO14 3AE

Tel: +44(0)23 8033 8100
Fax: +44(0)23 8033 8040
Email: abpr&c@research.abports.co.uk

SUMMARY

An extensive literature review has been undertaken to identify the essential components that contribute to the creation of successful habitats and to evaluate the successes and failures of previous habitat creation/restoration schemes. In addition, data has been gathered and sites visited to compile a set of UK case studies (Section 13). This information has been used to determine the requirements necessary to successfully create a wetland habitat.

The review begins by providing the background to wetland creation and restoration based on experience from around the world, in particular in the USA (Section 2). The various aims and objectives of creation and restoration schemes are investigated and summarised and some of the techniques used to meet these aims and objectives are introduced. The review concludes that wetland habitats afford the opportunity to satisfy multiple objectives, including:

- Coastal defence and flood alleviation;
- Water quality improvement;
- Fishery and shell fishery production;
- Ground water recharge;
- Tourism;
- Habitat development for nature conservation;
- Beneficial use of dredged material;
- Provision of educational and research opportunities;
- Archaeological conservation;
- Public access;
- Recreation;
- Shore stabilisation; and
- Enhancement of urban landscapes.

The majority of schemes undertaken recently, both in the USA and UK, have sought to meet several of these objectives. However, the primary objective of many coastal habitat creation and restoration schemes undertaken in the world to date has been to provide habitat for nature conservation.

The remainder of the review is divided into various sections. Firstly, the physical considerations for the creation of coastal habitats are addressed under the broad headings of hydraulics, morphology and hydrology (Section 3) and sedimentology (Section 4). Following this, some of the fundamental ecological processes that drive habitat function are discussed (Section 5). Subsequent sections specifically address the ecological processes with respect to vegetation, micro-organisms and plankton, benthos, birds and fish (Sections 6-10).

The use of buffer habitats and environmental corridors (and other protective measures) to reduce pressures on newly created sites are discussed (Section 11) followed by the monitoring and possible maintenance requirements of created and managed habitats (Section 12). Finally a description of a number of case studies of created and managed habitats in the UK is provided in Section 13.

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1. INTRODUCTION

1.1 PURPOSE OF THIS REVIEW

The aim of this review is to document the achievements in the field of coastal habitat creation/restoration in the UK and abroad. The review aims to identify the essential components that contribute to the creation of successful habitat and to evaluate the successes and failures of previous habitat creation/restoration schemes. This review is one of the outputs from a research project commissioned by ABP Southampton, undertaken by a consortium comprising ABP Research & Consultancy Ltd (ABP Research), Ecological Planning and Research (EPR), GeoSea, Canada and the Cambridge Coastal Research Unit (CCRU) to undertake a review of literature on coastal habitat creation/restoration and intertidal recharge schemes.

1.2 STRUCTURE OF REVIEW

The literature review identified many sources of information relating to the creation, restoration and recharge of estuarine habitats. **Section 2** of this review provides a brief background to the development of the concept of habitat creation, outlining the diverse range of schemes that have been undertaken to date. **Section 3** considers the physical aspects of the creation of coastal habitats under the broad headings of hydraulics, morphology and hydrology. Sedimentology is covered in detail in **Section 4**. **Section 5,** reviews some of the fundamental ecological processes that drive habitat function including natural succession and community development, food chains and energy flow. Subsequent sections specifically address the above ecological processes with respect to vegetation (**Section 6**), micro-organisms and plankton (**Section 7**), benthos (**Section 8**), birds (**Section 9**) and fish (**Section 10**).

The use of buffer habitats and environmental corridors (and other protective measures) to reduce pressures on newly created sites are discussed in **Section 11**, and the monitoring and possible maintenance requirements of created and managed habitats is discussed in **Section 12**. **Section 13** contains a detailed description of 12 case studies of created and managed habitats in the UK which have been visited in order to make an assessment of the components which have contributed to the outcome (success or failure) of the restoration scheme. A summary of the main conclusions that can be drawn from the literature review and case studies is presented in **Section 14**. This section also identifies information gaps and unanswered questions that should be examined in Stage 2 of this study to increase our understanding of the requirements for the creation of successful coastal habitats of wildlife value. The full list of references used in the literature review is contained in **Section 15**.

A database bibliography of habitat creation references has been prepared to accompany this report and future work on the creation, restoration and recharge of intertidal habitats. The bibliography contains details of the author(s), title, reference source, keywords, and abstract for all relevant pieces of literature used in this review.

2. BACKGROUND

To avoid ambiguity and confusion it is first necessary to:

- Define and discuss the terminology that will be used throughout this literature review (see Appendix I for definitions);
- Provide a brief background to the historical development of habitat creation schemes (Section 2.1); and
- Identify the types and objectives of projects undertaken and the extent of their documentation in literature (Section 2.2-2.4).

The six key terms considered to require definition are creation, restoration, recharge, mitigation, habitat value and success and these are presented in Appendix I.

2.1 WETLAND CREATION AND RESTORATION

2.1.1 Why Create and Restore Coastal Wetlands?

Over recent history there has been significant destruction of coastal habitats worldwide, primarily as a result of drainage and reclamation activities for agricultural, urban or industrial development. Indeed reclamation of intertidal habitats has been a major feature of coastal management practice for the past 500 years.

The majority of reclamation has occurred in Europe, particularly the coastal lowlands of the southern North Sea. In some UK estuaries, such as in the Tees, intertidal mudflats have been completely degraded (Davidson and Evans, 1987).

Over the past century up to 90% of California's coastal wetlands have been lost or severely degraded to the point where they provide limited habitat value (Tsihrintzis *et al*, 1996) and 40% of the French (Brittany's) coastal wetlands have been destroyed since 1960 (Williams, 1994).

The removal of healthy sedimentary foreshore has three effects:

- It removes trophic energy from the ecosystem;
- It reduces the efficiency with which the shoreline can adjust to tidal and wave forces; and
- It reduces the ability of the system to recycle nutrients and other organic / inorganic compounds essential for a healthy ecosystem.

Coastal erosion, in some cases caused or exacerbated by anthropogenic activity, has also contributed to the high rate of loss of coastal habitat over the last few decades (Brampton, 1992; Burd, 1988). Growing awareness and concern over the ecological and economic damage caused by this large scale removal of wetland habitats, together with concern regarding sea level rise, has led to the recognition of the need for a means to alleviate some of this damage. In response, the concept of habitat creation and restoration was conceived in the USA around 20 years ago. Wetland restoration is becoming more popular in a number of countries, including the UK, The Netherlands, Germany and Japan (Williams, 1994), particularly in response to climate change.

2.1.2 Wetland Restoration in the USA

In the United States, significant areas of wetland habitat have been lost to development and the remaining areas are now considered to be highly valuable. As a result, any new proposals for development are required to mitigate the loss of habitat by restoring or creating new wetlands nearby. To date thousands of habitat restoration schemes have been undertaken in the USA; for example between 1987 and 1989 the US Fish and Wildlife Service restored almost 55000 acres of wetland habitat (Kuslar and Kentula, 1990).

Most creation and restoration schemes have been primarily used as compensation and mitigation for loss of habitat elsewhere. However, as the benefits of other wetlands functions have become appreciated, new imperatives for habitat creation have developed with a wide range of objectives, including:

- Coastal defence;
- Flood alleviation;
- Water quality improvement;
- Fishery and shell fishery production;
- Ground water recharge; and
- Tourism.

Creation of marsh habitat is also used as a shore stabilisation technique, particularly for stabilising newly created islands of dredged material and as a means of improving local water quality (Weckman and Sales, 1993).

Restoration and creation schemes have been promoted by way of various legislative requirements, particularly the Clean Water Act. In 1992, the National Research Council made recommendations that a national aquatic ecosystem restoration strategy be developed for the USA. Although no national strategy has been established to date, the level of interest and funding for conservation and restoration of wetland habitats has been raised considerably (Zedler, 1996).

The increasing development and adoption of creation and restoration schemes has not been without criticism, particularly over the lack of success of certain restoration schemes implemented in the US during the 1980’s. There has been some controversy over mitigation schemes with claims made of poor planning, inadequate monitoring, the low ecological value of restored habitats and disinterest among those carrying out schemes in achieving a positive outcome (Race, 1985; Grenell, 1994). These criticisms have served to identify the need for a more thorough and regulated approach to habitat restoration which recognises the need for careful planning, clear objectives, long-term monitoring and a sound understanding of the physical and ecological function of the habitats to be restored or created (Section 2.4).

2.1.3 Wetland Restoration in Europe

From its roots in the west coast USA, a movement has begun to restore and create coastal wetlands throughout the world. This movement is now growing in many countries, including the UK, The Netherlands, Germany and Japan (Williams, 1994). The implementation of habitat creation and restoration schemes in Europe is becoming increasingly common and has been given added impetus by the predicted increase in the rate of sea level rise due to global warming. The predictions of the Intergovernmental Panel for Climatic Change of an average 6mm/year increase in sea level has meant that many nations are considering restoring intertidal habitat along their shorelines to provide a natural barrier to sea level rise. Furthermore, the adoption of a retreat option to flood defence rather than costly and, it is argued, unsustainable, hard defence structures is now more often promoted. This approach to saltmarsh restoration is known as "managed retreat" in the UK and "ont-poldering or de-poldering" in the Netherlands.

The response of individual nations to the problems caused by inappropriate reclamation policies in the past and the threat of sea level rise has been very different. Only recently has there been an international attempt to pool information concerning methods and objectives for these responses.

In the UK, managed retreat has been promoted by the Ministry of Agriculture, Fisheries and Food (MAFF) through the Habitat Scheme that was launched in 1994. The scheme provides a means by which areas of saltmarsh can be created and extended on suitable agricultural land immediately adjacent to the coast, normally by the realignment of coastal defences inland (MAFF, 1996).

The managed retreat programme, which began as a habitat creation exercise, has more recently developed a more fundamental objective; that of the restoration of coastal function. The efficiency with which estuaries in particular can respond to tidal and wave forces is seriously compromised by the removal of intertidal areas and the restoration of these areas lost to reclamation is seen as a major goal, providing benefits to all users. In this context, Government research in the UK has been channelled into investigating estuarine functional morphology, particularly through a MAFF research programme that is developing predictive models of sustainable estuarine morphology. The results of this programme may allow managed retreat to be seen as merely one approach to the restoration of fully functional estuaries in which navigation, water quality, flood defence and conservation uses are maximised. At the present time the objective of a fully co-ordinated estuarine restoration programme is still remote (Pethick and Burd, 1996).

2.1.4 "No Net Loss" Policy

During the past decade, Canada and the USA have moved towards a goal of "no net loss" of wetlands (Lynch Stewart, 1992). The concept of no net loss is simply defined in the US Wetlands Action Plan as a process whereby "wetland losses must be offset by wetland gains”; this recognises that unavoidable wetland losses must be compensated with wetland restoration (US Fish and Wildlife Service, 1989).

The adoption of a no net loss policy has progressed further through the development of mitigation land banking. This more proactive approach to wetland creation or restoration compensates for proposed wetland losses from future developments as part of a credit programme. A developer can purchase a credit from “a bank” that has restored or created large areas of wetland which can be used to mitigate for a number of developments off-site and in advance of habitat loss. This process facilitates the strategic planning of habitat creation on a regional basis and provides a number of ecological benefits. However, successful mitigation banking requires implementation by a responsible agency, up-front financing and need to be regulated (Hugget, 1996).

The EC has proposed four main principles for implementing a no net loss policy in Europe, namely:

- No further loss of wetlands;
- No further wetland degradation;
- Wise use of wetlands; and
- Wetland improvement and restoration.

Habitat restoration and creation plays an essential role in this strategy which states that further losses of wetlands will only be allowed for imperative reasons of overriding public interest. Losses must be compensated by the restoration of former wetlands or the creation of new wetlands of at least the same surface area and at least performing the same functions and providing the same ecological value.

In the UK, the nature conservation authorities (English Nature, Countryside Council for Wales and Scottish Natural Heritage) have adopted a no net loss policy. The policy is considered to be an important step forward for wetland conservation purposes in Europe, but it is recognised that its development and implementation for coastal wetlands will not be easy (Hugget, 1996). Although full implementation of a no net loss strategy throughout Europe may be some way off, the requirement for developers to create and restore wetland to mitigate for habitat loss now exists and such mitigative schemes are becoming increasingly common in the UK.

2.2 SCOPE OF LITERATURE REVIEW AND RESEARCH

In response to the increased interest in habitat creation and restoration over the past decade, there has been growing emphasis on research, information exchange and guidance. As a result a wide range of government agencies, scientific institutions and private sector organisations have initiated research programmes, published reports and articles, and held various meetings and symposia to discuss habitat creation and restoration and related issues. The implementation of habitat creation schemes varies greatly from country to country and region to region, as does the reporting of experiences in scientific literature. For example, there is a much smaller literature base covering coastal and estuarine habitat creation in the UK where fewer schemes have been undertaken in comparison to the United States. Similarly, within the UK, literature tends to focus on the North Sea coast, where sea defence problems have arisen as a result of extensive saltmarsh erosion and subsequently more schemes have been implemented.

The emphasis of work throughout the world has been on the creation and restoration of wetland habitat in general; this includes saltmarsh, freshwater marsh, swamps, bogs and similar areas (Kusler and Kentula, 1990; Marble, 1992, Williams, 1994). Very little literature specifically documents the implementation of intertidal flat creation schemes (Ray *et al*, 1994; Dearnaley *et al*, 1995). However, the creation of intertidal flats and tidal creeks have been more widely covered in literature describing saltmarsh restoration (Cammen *et al*, 1974; Broome, 1990; Frenkel and Morlan, 1991; IECS 1991-1996; Rumrill and Cornu, 1995; Havens *et al*, 1995, Pethick and Burd, 1995; Dixon and Weight, 1995; Zedler, 1996; Tsihrintzis *et al*, 1996; Port of Everett, 1996; Carpenter and Brampton, 1996;) and island creation schemes (Landin, 1991; Gallene, 1991; Maynord *et al*, 1992). The majority of literature available focuses on the creation of *Spartina* marshes.

Due to the relatively recent interest in habitat creation, few evaluations have been made of the long-term outcome of schemes (over 10 years), particularly with regard to the ecological value of the newly created habitat. Those which have been documented have been concerned with the long-term evaluation of *Spartina* saltmarsh restoration schemes, although the results of post-construction monitoring of other habitat types has been undertaken including intertidal flats created with dredged material (PIANC, 1992), mixed saltmarsh (Joenje, 1979) and bird islands (Landin, 1991). Several references were found which documented the design of habitat creation schemes which have yet to be constructed (Rumrill and Cornu, 1995; Tsihrintzis *et al*, 1996) and others which have recently been created but as yet no monitoring results have been collated (I. Black, *pers. comm*. 1996).

Despite the wealth of available information documenting the implementation of individual habitat creation or restoration schemes, information providing comprehensive coverage of all aspects of project design, construction and evaluation, including hydrology, sedimentology, morphology, ecological considerations (habitat requirements, recolonisation rates and community development), and monitoring and management is lacking. A major report has been compiled from work undertaken during the Wetlands Research Plan that was initiated in 1986 by the US Environmental Protection Agency (Kusler and Kentula, 1990). This status report contains regional summaries of coastal wetland restoration and creation experiences, identifies the lessons learned and covers a wide range of topics of general application to creation schemes, including hydrology, habitat value, monitoring and management.

Kusler and Kentula's report incorporates and builds on a number of well known compilations of wetland creation information in the USA which have also been consulted in the present review. They include the work of the US Army Corps of Engineers Dredged Material Programme (Newling and Landin, 1985, Ray *et al*, 1994), and the "Creation and restoration of coastal plant communities" (Lewis, 1990).

A recent publication by Zedler (1996) is one of the only comprehensive reviews focusing on the restoration of tidal wetland systems in estuaries, covering the scientific perspective of encouraging intertidal habitat development, problems encountered and lessons learned, illustrated with numerous case studies from Southern California.

Design guidelines for constructed and restored coastal wetlands have been produced in the USA (Zedler, 1984; Marble 1992, Tsihrintzis *et al*, 1996), although most published guidelines and case studies pertain to wetlands in general or freshwater systems only. Much work in the US has covered the types of information that are necessary to ensure a successful scheme, including Harvey *et al,* (1983), Zedler (1984), Coats *et al*, (1989) and Weckman and Sales (1993). No comparable reviews or guidelines for coastal creation/restoration schemes have been attempted in the UK, although there are valuable lessons to be learned from projects and research undertaken here, the majority of which involves the creation and restoration of saltmarsh for the objective of coastal defence and nature conservation.

A study has been undertaken by the Institute of Estuarine and Coastal Studies to investigate the lessons that may be learnt from the numerous sites where sea defences have failed in the past and the land behind has reverted to saltmarsh. Although such saltmarsh re-creation had taken place naturally the results were considered to be analogous in some degree to the results that may be expected in the managed retreat schemes. As a result of this study a number of practical recommendations were drawn together and published by English Nature in their Campaign for a Living Coast series (Burd, 1995). Methods of saltmarsh restoration and recharge are presented and assessed in the Environment Agency document “A guide to the understanding and management of saltmarshes” (NRA, 1995) and a review of saltmarsh restoration and recharge schemes has been undertaken for the Environment Agency entitled “Maintenance and enhancement of saltmarshes” (Carpenter and Brampton, 1996).

2.3 SCHEME OBJECTIVES AND TECHNIQUES

The number of coastal creation and restoration schemes is continuously increasing in the UK and abroad. Created and restored wetland habitats afford the opportunity to deliver multiple objectives including:

- Coastal defence and flood alleviation;
- Water quality improvement (e.g. Weckman and Sales, 1993);
- Fishery and shell fishery production;
- Ground water recharge;
- Tourism;
- Habitat development for nature conservation;
- Beneficial use of dredged material;
- Provision of educational and research opportunities (e.g. Tsihrintzis *et al*, 1996; Zedler, 1996)
- Archaeological conservation;
- Public access;

- Recreation;
- Shore stabilisation technique (e.g. Weckman and Sales, 1993); and
- Enhancement of urban landscapes.

The majority of schemes undertaken recently, both in the USA and UK, have sought to meet several of these objectives. However, the primary objective of many coastal habitat creation and restoration schemes undertaken in the world to date has been to provide habitat for nature conservation.

2.3.1 Provision of Habitat for Estuarine Nature Conservation

The provision of habitat often occurs as part of conservation and mitigation schemes associated with developments. Particularly widely recognised is the primary importance of the provision of intertidal flats and saltmarsh as feeding, roosting and breeding habitats for birds. Intertidal habitats have been created which support typical estuarine benthic communities, wading bird and waterfowl populations, including migratory and breeding birds, and a mix of estuarine and migratory fish species.

A large number of mitigation schemes have been implemented in ports, harbours and estuaries in the USA with the primary objective of replacing intertidal flats and saltmarsh habitat lost due to development (Dial *et al*, 1986; Broome, 1990). By the late 1980's, however, there had been no large-scale attempts to create or restore intertidal mudflats to provide animal and bird habitat in the UK. Exceptions are a *Spartina*-clearing project in Lindisfarne and feasibility studies for the creation of intertidal habitat in compensation for port development in Seal Sands, Teeside, although the latter was never completed (Davidson and Evans, 1987).

A small-scale mitigation scheme has recently been undertaken in Parkstone Bay, Poole Harbour, to replace ecologically important mudflats lost during the creation of Parkstone Marina (Dearnaley *et al*, 1995). Although few intertidal habitat creation schemes are known to have been undertaken in the UK for conservation purposes, the practice of mitigating for port and harbour developments is more common. For example, a lagoon habitat for waterfowl has recently been created in Felixstowe Docks and artificial cliffs have been successfully created in Lowestoft Docks as a nesting area for kittewakes. However, this practice is likely to become more widespread in the future. This is exemplified by Cardiff Bay, where the completion of a barrage in 1997 will result in the loss of intertidal mudflats of importance as habitat for a variety of wading bird species. To mitigate the effects of the scheme, an alternative compensation package has been proposed to attract some of the same species that will be directly impacted by the scheme, but also to attract other different bird species. Three different types of habitat are to be created; reed bed, wet and flooded grassland, and saline lagoons (Broadcast Reporting Service, 1996).

Techniques

Krone (1993) describes a general approach to wetland restoration work as "Restoration requires, as a minimum, the provision of hydraulic and sediment regimes that promote the growth of plants, that establishes the wetland on the evolutionary path of natural wetlands, and that will continue to provide a healthy habitat with minimum management and maintenance." Various methods and techniques have been adopted worldwide to achieve the restoration of intertidal habitats for flora and fauna including the following:

- The creation of saltmarsh, intertidal flats and tidal creek systems by excavating and grading coastal uplands to lower the level to that of intertidal elevations, providing conditions for saltmarsh development either natural or through revegetation planting programmes (Dial and Deis, 1986).

- The creation and restoration of saltmarsh habitat by the re-introduction of tidal regimes to previously enclosed or reclaimed land by managed retreat or the use of sluices (Bell, 1996; Pethick, 1994; IECS, 1991-1996; Dixon and Weight, 1995; I. Black, *pers. comm.* 1996);

- The creation, restoration, and recharge of intertidal flats and/or saltmarsh habitats by the placement of sediment on intertidal and/or subtidal habitat to raise the level to intertidal elevations, providing conditions for natural saltmarsh development and/or vegetation planting (Ray *et al*, 1994; Dearnaley *et al*, 1995; Carpenter and Brampton, 1996; M. Dixon, *pers. comm.* 1996);

- The creation of subtidal and intertidal habitat for fish, shellfish and birds, for example by planting eelgrass beds (Fonseca and Kenworthy, 1985) or through the placement of sediment material on estuary and sea bed to form berms or beds, such as the creation of oyster beds in the Blackwater Estuary (M. Dixon, *pers. comm.* 1996);

- The placement of sediments (dredged material) to create coastal and estuarial islands to provide new feeding, roosting and breeding habitat for birds (Landin, 1991);

- The creation and excavation of lagoon systems or permanent pools for birds, fish and other marine communities (E. Wiseman, *pers. comm.* 1996).

The most common technique used to create and restore coastal habitats is the revegetation of intertidal areas by sowing seeds or transplanting small plugs of saltmarsh plant species, usually *Spartina*. A wide variety of wetland habitats have been constructed using revegetation programmes with a range of saltmarsh species in addition to *Spartina*, including mixed saltmarsh, eelgrass, reedbed and mangrove species.

Projects which have allowed the natural recolonisation of intertidal areas with saltmarsh species are more rare in the US (Frenkel and Morlan, 1991), although they are increasing in number as the benefits of adopting a natural approach are becoming more widely recognised (Section 6). Various fertiliser and soil treatment methods have been researched and applied in the field to enhance the rates of revegetation (Webb and Newling, 1985). Most projects rely on benthic fauna to move or colonise a new intertidal area over a period of time, and the gradual use of the habitat by insects, birds and mammals from the surrounding area as communities develop and the food web becomes established. At present there have been no successful transplants of invertebrate fauna to newly created habitats, although attempts have been made (Josselyn *et al*, 1990).

Although many of the practical methods of saltmarsh restoration used in the USA are directly applicable to the UK, many of the more specific recommendations in the US work are not applicable because of differences in the environment and species involved. One of the primary differences, for example, is that in the USA *Spartina alterniflora* represents the main climax vegetation type, together with *Salicornia* spp. In contrast, in British saltmarshes *Spartina* and *Salicornia* are both representative of the pioneer and low marsh zones, with much more species-rich communities found towards the upper parts of the marsh. The picture for habitat creation in the UK is therefore substantially more complex than that of the United States (Pethick and Burd, 1996).

2.3.2 Coastal and Flood Defence

The second most common objective of habitat creation and restoration schemes in the UK is to enhance flood defence effectiveness. Saltmarshes are widely recognised for their ability to dissipate wave energy and stabilise sediments, thereby reducing erosion (NRA, 1995), and offering protection to seawalls or embankments from wave attack (Brampton, 1992; IECS, 1993a). In the UK, saltmarsh restoration is increasingly applied as a means of providing a long-term, sustainable approach to flood and sea defence with the added advantage of enhancing the conservation value of a natural habitat (NRA, 1995).

Over recent decades there have been considerable and widespread erosional losses of saltmarsh along the UK coastline, particularly in Essex, Hampshire and the Severn Estuary (Brampton, 1992). In many estuarine areas, the landward transgression of saltmarshes in response to rising sea level is interrupted due to the presence of flood defences. Therefore, as the outer marshes erode, the inner marshes are unable to transgress, leading to progressive loss of saltmarsh area. This process is known as ‘coastal squeeze’ (IECS, 1992). The loss of saltmarsh in Essex and Kent during the period 1973-1988 was shown to be as high as 40% (Burd, 1988).

In Europe, the high rate of loss of coastal habitat due to erosion and development and concern regarding sea level rise has prompted interest in experimental work to find innovative methods to reduce erosion and promote saltmarsh restoration (IECS, 1993a). These include managed retreat, de-polderisation and sediment recharge techniques.

Managed retreat

Saltmarsh restoration is increasingly applied as a means of providing a long-term, sustainable approach to flood and sea defence with the added advantage of enhancing the conservation value of a natural habitat (NRA, 1995).

There are two main rationales for use of the managed retreat technique in Britain; enhancement of flood defence effectiveness and habitat re-creation as a replacement for habitat losses. The restoration of saltmarshes to provide a flood defence benefit is a relatively new technique in Britain, although there are already several active schemes underway. The majority of these are located in Essex where sea defence problems have arisen as a result of extensive saltmarsh erosion, and therefore there have been more opportunities for improving sea defences by actively re-creating saltmarsh.

De-polderisation

The objectives of managed retreat, or de-poldering in the Netherlands, are quite distinct from those in the USA or the UK. Reduction of sedimentation in the estuarine subtidal channel using an extension of the flooding area or de-polderisation was first proposed by Pieters *et al*, 1991, 1992 and Pieters, 1993. They suggested that as well as the obvious benefits of habitat creation several other benefits may accrue. For example:

- Diversion of sediment from major channels to de-polder sites can reduce the necessity for maintenance dredging for navigation purposes;
- A reduction in flood risk upstream due to increased frictional drag and the water storage provided by the de-polder areas;
- An improvement in water quality due to deposition of sediment and associated contaminants within the de-polder site.

During the past few years more quantitative research has been carried out on the use of de-poldering as a management tool and the acceptance of the idea by the management authorities has grown, both in the Netherlands and in Belgium. For example, de-poldering is now part of the *Nature Restoration Plan Westerschelde,* a project in which national, regional and local authorities are involved, while in Belgium, de-polderisation is now seen as a method for the reduction of flood risks. This innovative use of tidal wetland restoration in order to improve the physical characteristics of the estuary for navigation and to reduce flood risk is, at the same time, associated with a general improvement in the ecological status of the Westerschelde. This is a complementary relationship in which the economic gains are able to support nature conservation interests.

Sediment or foreshore recharge

Sediment recharge or foreshore nourishment is an innovative technique to combat erosion of intertidal flats and saltmarsh by providing sediment suitable for raising foreshore levels in the vicinity of recharge sites (IECS, 1993a). The first trial schemes were implemented at three sites in Essex in 1990 and the technique has been applied on a largely experimental basis in over 15 locations in Essex and Suffolk (Carpenter and Brampton, 1996). "The objective behind tidal flat regeneration is to actively manipulate the regime such that low, concave, eroded shores adopt a high, convex, accretionary shape and remain stable" (Kirby, 1995). The ideal shape will vary depending on the objective of the restoration.

2.3.3 Beneficial Use of Dredged Material

Dredged materials have been extensively used in the restoration and creation of coastal and river habitats, providing an alternative disposal method and a beneficial use for the material. The concept of beneficial use is not new and in the USA, for example, over 16000ha of riparian and coastal wetlands have been restored or created from dredged material (PIANC, 1992). During the past two decades in particular, a number of innovative beneficial use schemes have been undertaken in coastal regions of the USA. These include the construction of saltmarshes, intertidal flats, bird islands, berms, shellfish and seagrass beds, and the restoration of eroding, subsiding and impacted wetlands and other natural habitats. A considerable amount of information is available on the establishment of wetland habitats using dredged spoil, with the best known research being undertaken by the US Army Corps of Engineers Waterways Experiment Station which has over 30 years experience in this field of research (Webb and Newling, 1985; Simmers *et al*, 1987; Olin *et al*, 1994; Ray *et al*, 1994; Brandon *et al*, 1996). The US Army Corps have developed a strategic framework for the selection of suitable habitat development schemes in North America, including mudflat, marsh, island and subtidal habitat options.

In the USA the use of contaminated dredged material for habitat creation has been studied for the past 10 years and is considered to be a major innovation in beneficial use (Brandon *et al*, 1992).

Similar projects in the UK are still rare in comparison. Several small scale intertidal recharge and creation schemes have been undertaken along the Suffolk and Essex coasts providing a beneficial use of material dredged from Harwich Harbour and the Blackwater Estuary for coastal defence and nature conservation purposes (Legget and Dixon, 1994; Kirby, 1995; Carpenter and Brompton, 1996). Dredged material has been used in the creation of mudflats in Poole Harbour and for the restoration and stabilisation of saltmarsh in Maldon, Blackwater Estuary (Dearnaley *et al*, 1995). A case study undertaken to investigate the potential for the use of clean dredged material for the creation of mudflat and saltmarsh habitat in a south coast bay in the UK concluded that there is significant scope for such projects in such locations (Paipai, 1995).

2.3.4 Water Quality

Coastal habitats have been restored and created throughout the USA and Europe for the purpose of enhancing water quality by nutrient removal, contaminant retention and sediment trapping (Josselyn *et al*, 1990). Interest in this technique in the UK was inspired following a visit by British Water Authorities to Germany in 1985, and since then reedbed treatment schemes have increased in number and are widely recognised for their benefit as a means of tertiary treatment for waste water (Collinson *et al*, 1994). The removal of contaminants and pathogens by tidal wetlands is rare, although the use of coastal freshwater wetlands for waste water improvement is more widely adopted, as indicated by the schemes in Humboldt Bay, Northern California (Zedler, 1996) and Ballona wetland, Southern California (Tsihrintzis *et al*, 1996).

2.3.5 Education and Research

Habitat creation, restoration and recharge schemes can also provide educational opportunities. For example, educational programmes can increase public awareness of the value of coastal wetland resources. The various schemes also contribute to this expanding field of research through pilot research programmes and the data collected as part of the monitoring programmes (Tsihrintzis *et al*, 1996; Zedler, 1996).

2.3.6 Other Objectives

Created and restored habitats also provide a number of other functions including archaeological conservation, public access, recreation and the enhancement of urban landscapes.

2.4 HABITAT CREATION AND RESTORATION: A SUCCESS?

2.4.1 Evaluation of Success

An important question that needs to be considered when reviewing a wide range of habitat creation schemes is the definition of success and the criteria by which it is measured. The criteria used to evaluate levels of success are scheme specific, reflecting whether objectives have been met or not. Most intertidal habitat restoration studies under review expressed success in terms of the percentage of vegetation cover and saltmarsh biomass over given periods of time. Although these criteria relate to certain characteristics of natural saltmarshes, such measures of success overlook the complexity of ecosystems, failing to indicate whether the scheme is functioning properly or whether it will persist over time. In certain cases there has been a failure to meet specific goals over predetermined periods of time, despite the total or partial revegetation of the site. The literature review has revealed numerous and varied reasons for scheme failure, which includes improper site conditions, failure to protect the site from on-site and off-site impacts and inadequate expertise and information during design and construction. Kuslar and Kentula (1990) recommend that success should ideally be measured in terms of the degree to which the functional replacement of natural systems has been achieved, a process that is difficult to assess and quantify.

2.4.2 Natural Versus Created Coastal Habitats

It is generally agreed that no habitat, coastal or otherwise, can be recreated exactly. This is because:

- There is often little knowledge of the original character of a site;
- A large degree of variation exists in both natural and created/restored habitats;
- Both intact and disturbed wetland ecosystems are constantly changing in response to natural and anthropogenic factors; and
- There is a complex interaction of hydrology, soils, nutrients, vegetation and animals in coastal systems which may have developed over many years (Kusler and Kentula, 1990; Frenkel and Morlan 1991; Roberts, 1991).

Nevertheless, there is much evidence to suggest that with careful design and consideration of existing and desired physical and ecological conditions, it is possible to create or restore coastal wetlands which provide similar functions to the natural habitat (Conservation Foundation, 1988). This does not mean that a created site resembling a natural site will necessarily represent equal habitat value (Kusler and Kentula, 1990). The ability of man-made coastal wetlands to replace the habitat value of areas destroyed by unavoidable development can be increased by carefully incorporating species habitat requirements into project design and locating wetlands where their habitat value can be realised (Roberts, 1991).

The review of available literature found few studies that have made long-term evaluations of the value of habitat creation schemes, through monitoring of man-made sites and comparison with adjacent natural habitats (Roberts, 1991). Consequently, the question of how long it takes for a constructed coastal habitat to achieve the same functional level as a similar natural habitat remains largely unanswered (Havens *et al*, 1995). Given the uncertainty surrounding habitat creation, some have argued that caution should be exercised against the widespread adoption of marsh creation as a mitigation strategy (Huggett, 1996).

3. HYDRAULICS, MORPHOLOGY AND HYDROLOGY

3.1 INTRODUCTION

Hydraulic characteristics such as hydroperiod (frequency and duration of tidal inundation) are the critical basic elements in the functional ability of estuarine habitats (Marble, 1992). The success of intertidal creation and restoration schemes is largely determined by the hydraulic design of the site. It is the hydraulic design that ultimately influences aspects such as intertidal morphology, sedimentology and ecology. Many saltmarsh restoration schemes have failed due to the inadequate consideration of the hydraulic, geomorphic and hydrological functioning of the designed system (Williams, 1994).

Within an estuary system there are a number of linkages between hydrodynamic and morphological systems at local and estuary wide scales. The morphological evolution at a particular site will be partly dependent on the temporal and spatial changes to the local hydrodynamics. However, the hydrodynamics and morphology of a particular site will also be sensitive to the short and long term changes occurring within the system as a whole. Furthermore, the impact of any recharge or habitat creation scheme may reach beyond the local site of interest and may impact on larger scale mechanisms present in an estuary. Therefore any investigation into the short and long term morphological evolution at a particular site will require an integrated approach which determines changes that are driven by local considerations and also those which will result from temporal changes to the system as a whole.

Consequently, this section initially documents the methods that have been developed to assist in the prediction of large-scale estuarine morphology in response to the hydrodynamic climate (Section 3.2). Such tools may subsequently be employed to investigate the impact of proposed schemes on the predicted long-term morphological evolution of the estuary as a whole. In turn such tools will help to determine the sensitivity of any proposed scheme in relation to the long-term morphological evolution of the estuary.

The review goes on to discuss methods of predicting local short and longterm morphological evolution in response to potential recharge or habitat creation schemes. As mentioned, natural mudflat and saltmarsh areas evolve largely in response to the local tidal and wave hydrodynamics (Goodwin, 1994). Hence understanding the local intertidal hydrodynamics and the morphology aids in the determination of the likely temporal response within recharge areas, which may dictate the potential success for any proposed scheme. Such methods involve both conceptual understanding and numerical predictions. Key hydrodynamic and morphological considerations include:

- The previous morphodynamic form of the site including past changes and configurations of the intertidal regions such as elevations, habitat boundaries and creek networks.
- The sediment properties.
- The cross sectional form of intertidal profiles.

- Waves, tides and sediment transport.
- The role of creeks in both intertidal flats and saltmarshes affect sediment, nutrient and pollutant exchange and hence it is important to consider the design requirements of such features (Section 3.3.2).
- The freshwater input and the degree of tidal mixing that also influence habitat type and stability (Section 3.3.3).

Successful habitat creation also benefits from the appraisal of existing schemes to determine practical considerations (Section 3.4). Such studies have revealed a number of important design considerations:

- Site elevation within the tidal frame, the incorrect choice of which has been the most frequent cause for site failure (Section 3.4.1).
- Topography and gradient which are particularly important for species diversity (Section 3.4.2).
- Saltmarsh creek design (Section 3.4.3).
- Wave and tidal climate along with sediment properties, which together govern the stability of the recharge sites following recharge (Section 3.4.4).
- Breach design and tidal prisms that can also influence vegetation diversity and the direction of sediment flux into or out of the area (Section 3.4.6).
- The actual size of the restored habitat is also a factor in governing the stability of the designed habitat in relation to prevailing hydrodynamic conditions (Section 3.4.7).

3.2 LARGE SCALE ESTUARINE MORPHOLOGY IN RESPONSE TO THE HYDRODYNAMIC CLIMATE

In recent years the interaction between nearshore hydrodynamics and coastal morphology has witnessed considerable research. In particular, the mechanisms present within estuaries and tidal basins have been heavily researched. This has been in response to a number of driving factors:

- The need to establish the long-term environmental impacts of anthropogenic activities such as land reclamation and maintenance dredging.
- Increased pressure to regenerate saltmarsh and mudflat habitats from previously reclaimed land.
- The need to understand the implications that may result from the managed retreat of coastal defences.

To investigate past and future long-term changes to the morphology of the estuary, the concept of minimum entropy production in natural systems (Leopold and Langbein, 1962) has been utilised. This concept has been developed, as a morphological stability analysis technique, for the more general case of a bi-directional variable discharge along a channel reach such as an estuary (ABP Research, 1996b). This technique requires the use of a hydrodynamic numerical model of the estuary to simulate the tidal flows for a number of historical bathymetries. The resulting hydraulic predictions are then used to enable the evaluation of the long-term morphological evolution.

In the evolution of a stationary geomorphological state, the entropy production per unit volume within the system will tend to evolve to a minimum compatible with the conditions imposed on the system (Prigogine, 1955). This suggests that, in the long-term, a natural system will tend to evolve in an attempt to achieve the most probable distribution of tidal energy in which entropy is maximised. This most probable distribution is referred to as the most probable state for the estuary and is described below. The most probable state represents a morphologic end point, to which the estuary is evolving.

The time taken to evolve to this most probable state will be dependant on constraints imposed upon the system. Such constraints include elements such as the underlying geology, sediment supply, and sediment redistribution by dredging. Furthermore, these constraints may be significant enough to prevent the evolution to the most probable state, or may induce a switch to some other steady state. Another complication is that the energy available to the system may vary during the evolutionary timescale; for example sea level rise due to global warming will change the tidal conditions and hence one component of the energy input to the system.

Leopold and Langbein (1962) developed the concept of minimum entropy production per unit discharge with regard to a river system. This concept was developed by ABP Research for the more general case of a bi-directional variable discharge along a channel reach (ABP Research, 1996b).

The development of this theory suggested that for minimum production of entropy per unit discharge, the energy distribution along an estuary may be represented as;

$$\ln \int HQ\,dt = C^1 x + D^1 \quad (1)$$

or re-writing:

$$\int HQdt = \exp(C^1 x + D^1) \quad (2)$$

where C^1 and D^1 are constants, and

$$\int HQ\,dt$$

is the sum of the energy passing through a section at a distance ‘x’ from the mouth of the estuary for a complete tide.

Equation (2) represents the most probable distribution of tidal energy within an estuary. This distribution takes the form of an exponential decay of tidal energy upstream. The theory implies that the estuary morphology will evolve in an attempt to achieve this minimal production of entropy. This may be achieved via mutual co-adjustment of channel morphology and tidal characteristics. However, this distribution may not be achievable due to natural or anthropogenic constraints.

The application of this theory to the Humber Estuary produced favourable results (ABP Research, 1996a) with a close to exponential decay of tidal energy transfer in the up-estuary direction. Further research has suggested that the up-estuary decay of tidal energy may be accompanied by a similar distribution in the cross-estuary direction. Although the investigation into the distribution of tidal energy in the cross channel direction is at a fairly early stage this hypothesis helps to explain the modelled cross channel decay of bed shear stresses in areas such as the Severn Estuary. The hypothesis also explains the presence of dissipative channels within natural mudflat and saltmarsh areas.

Furthermore, tidal creeks that form within saltmarshes (NRA, 1995) tend to exhibit similar morphological characteristics which are similar to those of estuaries, although they are of a smaller scale (French, 1996). These characteristics include near exponential decay in width, or cross sectional area, in the up inter-tidal direction. The formation of geomorphological features may thus be explained by the cross channel tidal energy distribution. Such features tend to induce the evolution of a more probable distribution of tidal or wave energy.

From the above discussion, it is apparent that morphological relationships can be used to determine stable morphologies with respect to the long term evolution of estuaries as a whole. Furthermore the future development of the entropic model developed at ABP Research (1996b) may aid in the determination of cross channel energy distributions. This may assist in the determination of long term stability of intertidal morphologies. Such tools also aid in the determination of the most probable morphology within such areas and consequently aid in determining the response of intertidal areas which have been created or reintroduced.

3.3 INTERTIDAL MORPHOLOGY IN RESPONSE TO THE HYDRODYNAMIC CLIMATE

A great deal of work has been undertaken in an attempt to understand the stability of tidal wetlands in response to tidal and wave climates (Goodwin, 1994). A large proportion of this research has been undertaken in the US where there has been increased pressure in the past twenty years for wetland regeneration for mitigation purposes. This section describes the various hydrodynamic and morphological criteria that need to be investigated for the successful design of a restoration or reclamation scheme, including:

- Hydrodynamic and morphological interactions.
- The role of creek systems.
- Hydrological influences on habitat type and stability.

3.3.1 Hydrodynamic and Morphological Interaction in Intertidal Wetlands

A report prepared for the NRA (1995) documents the primary activities that should be undertaken in considering a saltmarsh recharge or regeneration scheme in the UK.

The report states that such schemes are largely experimental in the UK and thus any scheme should be accompanied by an extensive monitoring programme to determine its effectiveness.

The initial design of a site should involve the following investigations:

- Historic maps and aerial photographs.
- Bathymetric/topographic charts.
- Analysis of grading curves and sediment properties.
- Modelling of tides, waves, sediment transport and habitat types.

The following sections discuss these issues in more detail. Alternative aspects that are considered to be essential within the NRA (1995) report include the analysis of pollutants and invertebrate communities, these aspects are discussed in Sections 4, 5 and 8.

Historic maps and aerial photographs

It is suggested that in the initial stage of any investigation (NRA, 1995), the interrogation of historic maps and aerial photographs should be undertaken. This should help to establish:

- Changes to high and low water marks.
- Previous spatial coverage of mudflats and saltmarshes prior to any land reclamation.
- Previous mudflat/saltmarsh elevations in the previous natural state.
- The location and orientation of previous saltmarsh creeks.

Information on previous creek morphology can be invaluable in determining the stable design of any man made creeks. However, an important caveat is that the information gained from aerial photographs of arable farmland may be particularly sensitive to the period of capture with respect to the farming year.

Bathymetric/topographic charts

Analysis of an accurate bathymetric/topographic chart relative to ordinance datum enables the current degree of tidal inundation to be determined. Such information also assists in the approximate estimation of the volumes of fine or coarse sediments required to achieve intertidal elevations suitable for habitat creation. These elevations may possibly be inferred from the historical analysis or from data from neighbouring or equivalent areas or from hydrodynamic/ morphological analysis.

Detailed topographic surveys will show the cross sectional form of the intertidal profiles at a site. Studies in Cardiff Bay (Kirby, 1995) suggests that sections through existing mudflats can indicate if it is existing in an erosional of accretionary state. Kirby (1995) suggests that this may be inferred from whether the cross-sections are concave or convex upward in shape. Kirby suggests that concave upwards profile forms are generally present in the unstable erosional mudflats. Kirby (1995) also suggests that such information will help to determine the stable cross-sectional profile that should be achieved by intertidal recharge.

Kirby (1995) discusses means by which such profiles may be determined. It is suggested that the initial stage is to determine if there are any stable convex upward mudflats locally. If such profiles exist these will give a good indicator of the profile required. Further work may include the application of a simple mathematical model, which utilises local input data to calculate the revised shape:

$$h' = \exp[4K(1 - y)]\, y^2 \quad (3)$$

where:

h’	=	h/h_o,
y	=	y/y_o,
K	=	Ki, y_{ou} is the non-dimensional wave parameter, which scales K_i by the length profile,
h	=	water depth, h_o is the depth at the seaward limit of profile, h’ is the non-dimensional water depth, expressed as a ratio or percentage,
K_I	=	profile-averaged wave attenuation coefficient,
K	=	non-dimensional wave attenuation parameter,
y	=	horizontal axis normal to shoreline located at the mean water level/offshore distance,
y_o	=	offshore distance at the offshore limit/profile length,
y	=	non-dimensional value of the y co-ordinate.

Comparison of the model results with natural profiles from many sites around the world show good agreement (Lee, 1994). However, Kirby (1995) does not suggest methods to determine mudflat profiles for sites that are sheltered and experience low wave energy input. Such areas often occur within inner estuarine areas within the UK. It is likely that tidal currents within such areas provide a significant contribution to the energy dissipation responsible for stable mudflat cross-sections.

Analysis of grading curves and sediment properties

The analysis of grading curves and sediment properties of recharge sediments can yield valuable data with regard to the likely stability of the material and the *in-situ* sediment transport processes. Further testing should be undertaken to determine pollutant or contaminant levels that may hinder or prevent the colonisation of the site.

Modelling of tides, waves and sediment transport

The numerical modelling of tides, waves and sediment transport is of fundamental importance to the success of the project as it will attempt to determine the hydrodynamic/ morphological environment that will eventually become stable. Such modelling may be employed to investigate in more detail the mechanisms present at the site. This may be subsequent to the initial design utilising the hypsometric methods suggested above (Kirby, 1995), or alternative regime theory, to initially determine approximate profiles or morphologies. Consequently there follows a brief discussion of the modelling tools that are presently available including the recognised limitations in using such models.

Although long term morphological modelling is not highlighted in the NRA (1995) guidelines, such predictions are often inherent in design guidelines (for example groyne spacings, wavebreaks, polders and sediment transport fields). Despite these guidelines being available there are inevitably some limitations in applying these general rules to sites of varying characteristics. For this reason the alternative tools available in order to establish likely future morphological changes are also discussed.

One and two-dimensional numerical models have been successfully applied to investigate hydrodynamic, sediment transport and water quality characteristics within tidal wetlands and lagoons (Goodwin, 1994; Fischer, 1969). It has been suggested that in tidal wetlands, particularly lagoons, the two-dimensional approach is more valid (Goodwin, 1994; Falconer, 1986). However, this rule is not fixed. For instance, work in the UK (Woolnough *et al.*, 1995) has resulted in development of simple 1D hydrodynamic and sediment transport models. These models have allowed valuable insights into the mechanisms of fine sediment deposition and thus morphological evolution of tidal salt marshes and creeks.

Goodwin (1994) notes that there are particular numerical refinements required for the representation of flows across intertidal areas, as opposed to more general estuarine studies. These refinements include wetting and drying properties, and the resolution of narrow channels and variations to friction parameters across flood plains. In recent years, research has resulted in the improvements to the numerical representation of wetting and drying of intertidal areas. These improvements have been applied and tested with respect to estuaries and natural tidal basins within the UK (Falconer and Owens, 1987; Falconer and Chen, 1991).

Despite the numerical constraints of utilising hydrodynamic models within wetlands, these tools have been applied, within the US and more recently within the UK, to determine exposure condition and sediment deposition rates for intertidal sites (Falconer, 1982 and 1984; ABP Research, 1996b). Thus with the increasing sophistication of numerical models, the understanding of the hydrodynamics and sedimentary processes present within the intertidal is becoming more complete.

Research in the USA has resulted in the correlation between hydrodynamic exposure conditions, such as tidal range and periods of inundation (hydroperiods), and stable habitat types (Cowardin *et al*, 1979). Thus, knowing the habitat type that is desired, a numerical model may be utilised to determine the stability of the wetlands with respect to the predicted hydrodynamics. Having established the stable intertidal morphology, the water level predictions may be utilised to produce hydroperiods. Then, applying the conclusions from the monitoring of wetlands, the habitat that is stable in such an environment may be determined (Haltiner and Williams, 1987; Goodwin, 1994).

Unfortunately this approach has an inherent problem, with respect to the confidence to which the spatial extent of future habitat may be determined. This problem arises from the predictions of the numerical model siltation/deposition rates that tend to have broad confidence limits. These confidence limits are governed by the present level of understanding of sediment transport mechanisms. Since tidal wetlands are generally very flat areas, Goodwin (1994) suggests these are generally less than 1% slope, then small errors in predicted bed levels may induce significant errors between the predicted and actual horizontal extent of habitat boundaries.

In the UK, this approach is further limited because the research, which correlates exposure conditions and hydroperiod to the type and stability of various wetland habitats, is at a relatively early stage. However, recent work has helped to address this problem (Gray, 1992). For example, Gray (1992) found good correlation between the upper and lower limit levels for *Spartina* and the tidal range, effective fetch and estuary plan area for 107 sections throughout 27 estuaries in the UK. Gray (1992) subsequently published equations relating these characteristics:

$$L.L. = -0.805 + 0.366SR + 0.053F + 0.135 \operatorname{Log}_e A \qquad (4)$$

$$U.L. = 4.74 + 0.483SR + 0.068F - 0.199L \qquad (5)$$

where:

L.L., U.L.	=	upper and lower limit of *Spartina* (in m AOD),
SR	=	spring tidal range (m),
F	=	fetch in the direction of the transect (km),
A	=	is the area of estuary in km^2,
L	=	is the latitude (in degrees N expressed as a decimal).

These equations yielded regression coefficients of 0.93 and 0.90 respectively for the dataset investigated.

In the UK, large areas of intertidal mudflat are exposed to relatively long fetches. Recent work with regard to these areas suggests that mudflat morphologies can be more sensitive to wave action than thought previously (Pethick, 1992a; 1992b; 1994). Studies in the Blackwater estuary (Pethick, 1992a; 1992b) suggest that mudflats may respond in a similar manner to beaches. Thus, extreme events may result in the movements of material down the intertidal, whilst during periods of more moderate wave climate activity there is a build up of material in the middle and upper intertidal.

The implication is that the intertidal morphology continually re-adjusts in response to the varying wave energy. This suggests that mudflat areas respond to changes in the short-term wave climate fairly rapidly, thereby reducing the impact of extreme conditions. It is further suggested that such relatively short term changes to the mudflat morphology may induce fairly rapid movements of mudflat/ saltmarsh boundaries (Pethick, 1994). Thus it is suggested that for schemes proposed in the UK, an assessment of the likely wave climate and its influence on the tidal flat morphology will be essential for the potential success of a project.

Within more sheltered areas, such as those often found within inner estuarine areas throughout the UK, the wave climate is generally of lower energy and the temporal variations are less. Saltmarsh morphologies within these areas will show less variability and the location of the saltmarsh/mudflat boundary may be more constant (Pethick, 1994). Such wave effects have been less well researched in the US since the majority of reported wetland regeneration scheme are protected by nearshore bars and lie within tidal basins with narrow entrances. Within such basins the wave climate may be of only secondary significance relative to the tidal flows present.

Numerical models are available which simulate the interaction of tidal and wave effects (DHI, 1996). These models have been applied successfully to determine fine cohesive siltation rates within intertidal areas for a number of studies throughout the UK (ABP Research, 1993; 1996a; 1996b). These models have been calibrated and verified using current data, suspended sediment concentrations and measured deposition rates within the intertidal. Thus, with the advances in the understanding of wave/ current interaction and how sediment transport varies in such environments, these numerical tools provide a means of determining the stability and short term morphological evolution of any proposed recharge scheme.

Models that have been developed to investigate hydrodynamics and siltation in intertidal areas often utilise considerable site-specific data to produce reliable results. Such site-specific information includes:

- current and wave measurements;
- suspended sediment sampling; and
- the estimation of deposition rates via traps or trial pits.

Further data may be required from:

- consolidation testing, in order to determine likely temporal changes to in-situ cohesive sediment strength; and
- *in-situ* shear strength testing, to determine re-erosion coefficients.

For areas that are to be reinstated or created, data may be unavailable at the proposed site. In such cases, data may be gathered from neighbouring areas that are of a similar nature to the proposed final climate in the scheme. If sites of this nature are not present, then it may be necessary to determine information from alternative sites that are comparable in nature. Care should be taken to ensure that sites chosen exhibit similar hydrodynamic properties to the desired final state, and that the sediment deposits are of a similar nature to those present or those to be introduced at the design site.

3.3.2 Tidal and Saltmarsh Creeks

French (1996) considered that the main functions of creek channels are to:

- Supply and disperse fine sediments;
- Aid removal of water for efficient drainage and de-watering of sediments; and
- Reduce and dissipate tidal energy inputs, which may decrease saltmarsh erosion elsewhere.

Major creek channels within the upper intertidal marshlands, may be designed from the empirical relationships between cross sectional area of flows and tidal prism. Such relationships offer an alternative to, or may supplement, the modelling approach discussed above and are similar to those discussed in Section 3.2 for the estuarine system as a whole. For a database of the Californian wetlands, the primary relationships are summarised below;

$$D = 0.16\, P_d^{0.24} \quad B = 0.10\, P_d^{0.49} \quad A = 0.025\, P_d^{0.61} \quad (6)$$

where:

D	=	is the maximum depth of channel,
B	=	is the bank top width of channel,
A	=	is the cross sectional area of the channel, and
P_d	=	is the diurnal tidal prism.

These equations may be used to determine the size and shape of a marsh channel that would be expected to develop as a result of a recharge programme. Such equations have been employed with varying degrees of success in the US. It should be noted that the design of constructed marsh channels may also require numerical modelling to supplement these simple equations. The need for such analysis was illustrated by the failure of a project in Mizzi Marsh, California (Goodwin and Williams, 1992). In this scheme, insufficient scour led to reduced rates of erosion and the inaccurate prediction of marsh channel geometries. This subsequently inhibited the growth of salt marsh vegetation. The dredging/excavation of artificial channels subsequently overcame this problem.

The theory described in Section 3.2, with regard to the most probable distribution of energy within estuaries, may also be applied to saltmarsh and tidal creeks. Through this approach it is possible to determine the long term geometry and the relative stability of artificial creeks.

Groen (1967) suggests that the primary factor in the net direction of sediment transport in tidal creeks is tidal asymmetry. Ebb dominated channels tend to occur in tidal inlets with extensive mudflats and intertidal areas. Such systems result in the net export of material from the creek channel that may induce erosion. This erosion may be more limited for cohesive sediments than for sandier non-cohesive ones. Such limited erosion may result in the restriction of the degree of tidal inundation at the site (Haltiner and Williams, 1987). Flood dominated channels tend to be deeper and wider with more vertical bank slopes. In such channels there is a net import of sediment into the creek (Goodwin, 1994; Boon and Byrne, 1981). Such flood dominance may induce rapid siltation of the designed creek (Haltiner and Williams, 1987).

From this discussion, it is apparent that an equilibrium system will be some way between these two environments (Haltiner and Williams, 1987). The tidal domination for particular creek designs may be determined with the aid of numerical or analytical solutions. By varying the design of such creeks it is possible to determine probable future quasi-equilibrium morphology with no tidal dominance. Theoretically it would also be possible to design a creek network with some flood dominant channels and some ebb-dominant ones. This would then mean that for the whole network, there was a balance between the material introduced through flood dominated channels and that expelled through neighbouring ebb-dominated channels.

Secondary creek channels, which result from bifurcations, may also be designed via these means. The branching of creeks results in;

- The efficient dissipation of tidal energy throughout the upper intertidal;
- Reductions to wave energy propagation;
- Efficient intertidal drainage; and
- The supply of water and sediments throughout the areas of saltmarsh.

French (1996) outlines details of determining appropriate junction angles according to the ratio of tidal discharges within any two-network branches.

3.3.3 Hydrological Influences on Habitat Type and Stability

In creating suitable habitat for particular flora and fauna it is necessary to determine the impact of freshwater inputs to a particular site, since species may be sensitive to salinity levels. Important considerations include:

- The estuary wide salinity relations;
- The local freshwater input; and
- The local input of contaminants.

Within estuaries as a whole the salinity characteristics depend largely on the quantity of freshwater input and the volume of energy available to achieve mixing of the fresh and saline waters. Pritchard (1955) classifies estuaries into three general groups with respect to their vertical salinity characteristics, these being:

- Well mixed;
- Partially mixed; and
- Salt wedges.

Generally, macrotidal estuaries tend to have large amount of energy available and thus tend to be fairly well mixed. Conversely, low energy tidal regimes tend to result in greater degrees of stratification such as those of the partially mixed and salt wedge types. The vertical salinity distributions for a particular estuary may also vary with tidal range, with a system exhibiting partially mixed characteristics during times of neap tides and low freshwater inputs to a salt wedge system during neap tides and high freshwater inputs (HR Wallingford, 1992). Furthermore, lateral density effects have been shown to occur in some systems (Dyer and Taylor, 1973), in turn these density effects have resulted in inducing cross estuarine currents, which may in turn modify the channel morphology.

On a more localised level, freshwater inputs into a particular site may arise from surface runoff or ground water sources. Various degrees of fresh water input may result in modifications to the mechanisms that determine the intertidal morphology. A common example of this is the deltaic formation of braided channels within the intertidal close to river mouths. Such braided channels have the property of dissipating the stream energy, from a particular freshwater input over a relatively large area, as opposed to a narrow fluvial channel. This reduces the energy dissipation per unit area and hence the impact of high fluvial discharges on the intertidal morphology.

Freshwater inputs may also introduce organic or inorganic sediments to a particular site as well as pollutants. Pollutants may be in the form of dissolved or undissolved materials along with litter. Species may be very sensitive to salinity or pollutant concentrations and thus any study into habitat creation scheme will need to quantify the magnitude and quality of freshwater inputs to a particular site and the likely salinity distribution during particular conditions. Numerical models can and have been employed to investigate salinity and pollutant distributions in both the horizontal and vertical plane (Falconer, 1986). In some cases it may be necessary to divert a particular source of freshwater in order to achieve an environment suitable for a particular type of habitat. This may be for reasons of water quality, salinity or morphological evolution.

Certain habitats such as lagoons may require periodic breaching or pumping in order to increase salinity levels and maintain brackish waters. This may be required due to the temporal build up of freshwater inputs to the system. The simulation of waves and currents induced at such breaches has been modelled numerically (Wright *et al*, 1976; Roelvink and Broker, 1993). The morphologies within lagoons have often developed as a result of a previously energetic hydrodynamic climate, such as the case for the linear sluiced, former tidal creeks, which are common in Essex. Consequently the morphological characteristics tend to vary considerably from site to site. A more detailed discussion of the morphological, hydrological and salinity characteristics for 26 lagoonal habitats within East Anglia is given in Barnes (1987).

3.4 PRACTICAL CONSIDERATIONS AND PROBLEMS IN THE CREATION INTERTIDAL HABITATS

Hydraulic problems are the most commonly documented problems during the implementation of restoration or creation of coastal wetlands (Williams, 1994; Zedler, 1996). This is partly because these problems are readily observed and the subsequent effects on the whole system may be dramatic. Unsuitable hydrology may result from improper planning or implementation, particularly with regard to the over or under excavation of intertidal sites. Such mistakes can create major problems for:

- Species which are sensitive to micro topographic variations (Gray, 1992; Gray and Scott, 1977; Brereton, 1971);
- Sedimentation rates; and
- Tidal access.

This section therefore discusses the appropriate design of:

- Site elevation;
- Topography and gradient;
- Creek networks;
- Wave and tidal climate;
- Sediment properties;
- Breach design and tidal prisms; and
- Spatial coverage.

3.4.1 Wetland Elevations

Inappropriate wetland elevations have been a significant factor in the failure of a number of restoration and creation projects in the US (Frenkel and Morlan 1991; Dial and Deis, 1986; see Appendix II). In a comparative study of 38 man-made coastal marshes in Florida, Roberts (1991) concluded that the most common cause for site failure was incorrect elevation of the site and subsequent erosion due to wave action. For example:

- The Bay Point Marsh creation scheme in Panama City became regularly flooded at low tide and failed as a result of incorrect elevation and a poorly designed tidal connection;

- Mariner's Square Marsh in Cocoa Beach, Florida was almost entirely lost due to erosion by severe wave action and, in little over one year after construction, it was reduced to only a narrow coastal strip or even altogether absent in some places; and

- The Salmon River saltmarsh restoration scheme, Oregon, USA (Appendix II) illustrates a common problem encountered in the restoration of dyked wetlands which have been prevented from tidally flushing for a period of years. In this case substrate subsidence lowered the elevation of the site and slowed the restoration effect (Zedler 1996).

Generally, sites that are too high (usually those that have been filled with dredged material) are found not to develop adequate drainage systems and lacked habitat diversity. Conversely, those sites which were lower than the recommended level developed numerous branching channels and a variety of marsh habitats consistent with a natural system (Weckman and Sales, 1993).

In the UK, MAFF suggest that land most suitable for the restoration of saltmarsh lies between elevations of high water neap tides and high water spring tides which equates to approximately 450 and 500 tidal inundation each year (MAFF, 1996). Appropriate elevations and tidal inundations for saltmarsh establishment are discussed fully in Section 6. A study of historic flood defence failures revealed that site elevation was the most important factor in determining the outcome of subsequent uncontrolled marsh restorations (IECS, 1994). The present elevations of the most successful marshes were higher than 2.34mODN, which correspond to approximately 1.77mODN at the time of breach. The study concluded that for saltmarsh restoration schemes elevation should be higher than 2.1mODN (i.e. covered by between 400 and 500 tides per year) and that lower elevations will result in the creation of mudflat.

3.4.2 Topography and Gradient

The choice of appropriate topographic gradient is critical to establishing the balance between estuarine hydraulics and morphology (Cylinder *et al*, 1995). Upper intertidal gradients are also a major control on colonising plant species that help stabilise the slopes and thus control erosion. In the restoration of Sweetwater/Paradise marsh, grading plans were not followed in the construction of the islands and channels (Appendix II). This resulted in a reversal of planned water flows in some areas, increasing erosion along many of the creek banks, and significantly altering the graded wetlands (Josselyn *et al*, 1990).

Topography is therefore an important factor in the creation of intertidal habitat. Generally, sites with a gradual, “natural” slope across the marsh surface provide a range of tidal inundations that promote a richer and more diverse saltmarsh (IECS, 1994; MAFF, 1996). The surface of the site should be graded to be relatively (but not completely) flat, and sloped towards the tidal channels. If the surface is too flat, or not graded to the creeks, the water was found to become impounded and prevent vegetation growth, exemplified in the managed retreat site at Northey Island (Section 13.3). In intertidal recharge schemes using fine sediments the design and creation of recharge profiles is not straight forward and Toft and Maddrell (1995) recommend that the system be allowed to naturally develop a new equilibrium in response to the hydrological conditions of the site. Various workers have recommended different wetland slopes that they considered stable:

- For the creation of a salt marsh site with dredged material in Bolivar Peninsula, Texas, a 0.7% slope was considered suitable for saltmarsh establishment (Webb and Newling, 1985);
- Woodhouse (1979) recommended a slope of between 1-3%;
- Zedler (1984) recommended a slope of between 0-2%;
- Knutson *et al* (1990) recommended a steeper design slope of 6-7%; and
- Reimold and Cobler (1986) recommend slopes ranging from 1:5 to 1:15 for increasing wetland vegetation diversity and decreasing erosion potential.

3.4.3 Saltmarsh Creeks

Creek networks are important in influencing the flooding and drainage of the marsh in order to produce the required hydraulic, sediment and water quality regimes. Tidal waters are carried from the estuary to the marsh through primary creek channels, while smaller secondary creeks distribute the water within the marsh plain (Haltiner and Williams, 1987). Although it is possible to allow the tidal flows to establish a new drainage pattern by natural erosion and deposition, this is a long-term process and in the meantime can result in poor circulation and can delay vegetation establishment (Haltiner and Williams, 1987).

An artificially constructed marsh may therefore require an artificial drainage network to produce adequate circulation, sediment supply and velocity of tidal flows. Indeed, in restoration schemes higher success rates have been achieved by the extension of a functional system by incorporating tidal creeks into the created system (Shisler, 1990). Creeks have been recommended for a number of reasons. Garbisch (1986) recommended creeks to control litter and for their value in the exchange of nutrients and increased habitat diversity. The U.S. Army Corps of Engineers (1986) recommended creeks to increase tidal circulation and to increase marsh productivity. It is generally accepted that meanders within creeks significantly reduce the up-creek propagation of wave energy and thus help to create a more stable environment (Allen and Pye, 1992).

The creation of an artificial drainage network is a complex process and requires studies of hydraulic design (e.g. Bertolotti and Crumpley, 1991; Haltiner and Williams, 1987). However, some general principles have been outlined. For example Krone (1993) recommended that:

- Channel cross sections should be oversized rather than undersized since sedimentation will allow channel to reach equilibrium; and
- Channel dimensions should be sufficient to produce nearly a full tidal range at the most remote portions of the marsh.

Additionally Haltiner and Williams (1987) recommended that:

- The junctions between the main creeks should be at 120° angles;
- The junctions of creeks with smaller channels should be at right angles;
- Side channels bed gradients should be graded into the main creeks to reduce ponding;
- The creek network should contain meanders to maximise the inundation areas; and
- No part of the marsh plain is more than 30m from a creek.

In relation to the last point, it is interesting to note that other workers have recommended optimum inundation distances between creeks much greater than the 30m suggested by Haltiner and Williams (1987). One such approach to the design of marsh creeks in retreat sites in the UK, was based upon suspended sediment concentration and vegetation density (Pethick and Burd, 1995). The greater distances recommended in this work possibly arose due to the higher sediment concentrations and different vegetation types.

One method of establishing a creek network, used in the US, is by placing bales of hay along the desired creek lines. The bales prevent sedimentation along these lines as the rest of the marsh accretes. Eventually the bales break down to leave unvegetated depressions and organic matter on the marsh surface. However, once the creek system is in place and opened to tidal flow the conformation may alter dramatically if the system was not designed at the correct equilibrium conformation. At Warm Springs, California, for example, increased flow through the breaches and drainage channels expanded the designed channel by 3-4 times the original (Morrison, 1988). The importance of accommodating natural sedimentation and erosion of tidal channels was also recognised by Tsihrintzis *et al* (1996) in the design of the Ballona scheme in Southern California.

3.4.4 Wave and Tidal Climate (Site Stability and Protection)

Sites are particularly sensitive to hydraulic factors during and immediately following construction. This may be due to a number of reasons, including poor initial strength of placed material or lack of any saltmarsh flora. Wave climate is an important factor in determining site suitability (Woodhouse, 1979). Wave climate depends on:

- Wind speed and direction;
- Bathymetric and topographic configuration;
- Water level; and
- Fetch.

In a relatively high wave energy environment some method of reducing the energy may therefore be required. This may be necessary during the period of stabilisation/consolidation and plant colonisation. Many innovative schemes have been employed to provide protection against high wave energies and thus erosion, whilst not severely inhibiting tidal flushing. Potential methods include:

- The planting of vegetation;
- The use of grounded barges as wave breaks; and
- The construction of levees or dykes.

One example of the latter was undertaken along the Bolivar Peninsula, Texas, where a sandbag dyke was constructed around the site (Webb and Newling, 1985).

In the US planting of *Spartina*, *Zostera* or *Halodule* is often used to protect islands of dredged material, or to slow foreshore erosion (Fonseca *et al*, 1982; Fonseca *et al*, 1984; Fonseca and Kenworthy, 1985). However, high vegetation mortalities may result from even moderate and high wave energy situations unless some form of additional wavebreak is also used. Many different methods of wavebreak construction have been proposed including those of Allen and Webb (1983), Allen and Shirley (1988) and Knutson *et al* (1990).

In order to reduce the impact of internally generated wind waves a series of low-elevation levees are often constructed within saltmarsh restoration sites (Abbe *et al*, 1991; Barker *et al*, 1993; Weckman and Sales, 1993). These provide protection to the perimeter walls, promote sediment accretion and produce a further range of habitats for vegetation colonisation. At the Tollesbury managed retreat site, hedgerows and trees have been left standing following the breach of the flood defence embankment due to public pressure. These hedgerows now act as natural wavebreaks to the relatively large fetch that is present across the site (I. Black, *pers. comm.* 1996).

Further examples of methods of protecting new sites against erosion are discussed in Section 4.5.5.

3.4.5 Sediment Properties

Although peak bed shear stresses will tend to be the ultimate control on the stability of tidal mudflats, the sediment properties are also important. The erodability of material will depend on the degree of consolidation and thus shear strength. Studies undertaken in France have generally indicated that peak tidal currents should be in the range of $0.5\text{-}0.7ms^{-1}$ in order to achieve successful mudflat creation. Currents in this range are reported to be sufficient to sustain mudflat habitat, whilst preventing sediment accretion which would promote colonisation by pioneer plants and the development of saltmarsh.

Kirby (1995) suggests various methods that are available to increase the strength of dredged material prior to placement and thereby potentially reduce erosion after placement. These methods include:

- The use of mechanical/chemical de-watering process plants;
- The use of temporary holding areas ashore; and
- The use of varying degrees of coarser material in order to assist in situ draining.

However, the successful site colonisation by particular flora/fauna may require sediments with particular characteristics. Consequently, this may limit the effectiveness of any options that change the particle size distribution of sediment.

3.4.6 Breach Design and Tidal Prisms

In managed retreat schemes, the size of the breach or culvert opening has been found to affect both the circulation and the tidal range within the restored/created marsh (Harvey *et al*, 1983; Haltiner and Williams, 1987). In a reduced tidal cycle the marsh plants may occupy different elevation ranges.

Comparison of the potential tidal prism within the site (calculated from the site geometry and the local tidal regime), with the actual tidal prism (measured from tidal fluctuations within the site) provides an indication of how far the site is from reaching a steady state (Barker *et al*, 1993). Changes to channel geometries occur until this steady state is reached, although in some cases this state may never be achieved due to disturbance or sea level rise for example. There is a strong correlation between tidal prism and both equilibrium channel depth and equilibrium cross-section area (Gale and Williams, 1988). Channel sections were considered to be in equilibrium if no significant changes had occurred over a period of approximately 25 years.

The development of an equilibrium geometry in a natural creek system evolves over time, and is dictated by the following parameters (Haltiner and Williams, 1987):

- Tidal characteristics in the estuary (mainly tidal amplitude);
- Potential tidal prism inland of the creek location;
- Sediment characteristics (mainly size);
- Sediment concentrations, supply and siltation rates; and
- Biological characteristics (e.g. vegetation or burrowing benthic organisms).

3.4.7 Spatial Coverage

The ability of the marsh to withstand wave stress depends on factors such as:

- Its growth stage;
- The density and vigour of vegetation; and
- Overall width.

In order to increase the likelihood of success of created wildlife habitats, the construction of small, narrow, fringe-marshes should be avoided. The minimum practical width being between 6m (Knutson and Woodhouse, 1983) and 10m (Knutson *et al*, 1990). Roberts (1991) recommended that block-shaped salt marshes of at least 0.5ha in size were necessary for the successful provision of fish and bird habitat. Many saltmarsh creation schemes in Florida were considered to be too small to provide the habitat area required by common marsh dwelling birds or fish (Roberts, 1991).

4. SEDIMENTOLOGY

4.1 INTRODUCTION

Sedimentological considerations are critical to the outcome of intertidal creations and restorations (Davidson and Evans, 1987). The failure to achieve preferred sediment characteristics is a common and widely documented cause of habitat restoration scheme failure (Zedler 1996, Broome, 1990). The continued presence of intertidal mudflats in the highly dynamic estuarine environment depends upon the balance between sedimentation and erosion. It is therefore necessary to understand the sedimentary processes in estuarine environments to select the appropriate habitat creation and recharge techniques (Toft and Maddrell, 1995). The relationships between hydrology, morphology and sedimentology have been investigated in Section 3.

The present section discusses:

- Sedimentary processes in estuarine and intertidal habitats, including nutrient cycling, consolidation, and the erosion and deposition of sediments;
- Sediment processes in habitat creation and recharge schemes;
- Rates of sediment maturation in created habitats, including consolidation, accretion and organic contents; and
- Practical considerations and problems, including sediment characteristics, sediment chemistry, organic content, contaminants, sedimentation and sediment placement techniques.

4.2 SEDIMENTARY PROCESSES IN ESTUARINE AND INTERTIDAL HABITATS

When a column of suspended sediment settles in still water there is a sequence of settling, deposition and consolidation (Dyer, 1986). Once the sediments are deposited they become colonised by various flora and fauna which leads to the build up of organic matter. This section considers each of these aspects in turn.

4.2.1 Erosion and Deposition

Deposition occurs when bed-shear stress falls below a critical value for deposition (Krone, 1963). Erosion occurs when the strength of the sediment bed is exceeded by the applied hydrodynamic shear stress. The deposition of cohesive particles, necessary for intertidal mudflat formation, is dependent on the process of flocculation. Flocculation occurs when electro-static charges between individual grains cause them to coalesce to form clusters of particles or flocs, which settle more rapidly than individual particles. In contrast, the deposition of non-cohesive sediments such as sand depends solely on the size of the individual particles (Dyer, 1986, Pye and French, 1993).

Deposition rates are likely to vary both spatially and temporally in both natural and recreated marsh habitats. Accretion rates vary seasonally in response to:

- Tidal range;
- Wave energy;
- Fresh water inputs;
- Suspended sediment concentrations;
- Water temperature;
- Plant productivity; and
- Benthic faunal populations.

Accretion rates vary spatially in response primarily to:

- Current speeds;
- Wave energy;
- Fresh water inputs;
- Suspended sediment concentrations; and
- Plant types.

On the basis of laboratory investigations work, (Pethick *et al.* 1990) proposed that the sediment flux from a marsh creek across a marsh surface is finite and that deposition rates are linearly related to suspended sediment concentrations. This leads to decreasing amounts of sediment in the water column further away from creeks. This arises because the highest deposition rates occur near the creeks, which results in the largest reduction in the suspended sediment concentrations in the overlying water column, which decrease the amount of sediment available for deposition rates further away from the creek. The rate of decrease of the suspended sediment concentration, and therefore the deposition rate, is related to the initial sediment concentration and the density of the vegetation through which the flow is occurring.

4.2.2 Consolidation

When cohesive particles (flocs and aggregates) are deposited on the intertidal they form a loosely packed layer with a continuous open network structure. Consolidation begins when the weight of the accumulation particles above causes the collapse of the network, due to particle sliding and un-rigid bonds. During this collapse pore water is expelled from the network (Been and Sills, 1981; Toorman and Huysentruyt, 1994). Initially the density of the newly deposited fines or the fluid mud is uniform until the coarser, denser particles and flocs settle to the bottom and begin to form a sediment. A region of intermediate density forms on top of the denser bottom material and beneath the suspension. Eventually only one region of sediment exists above the dense base layer, and this zone then begins to slowly consolidate through self-weight and expulsion of the pore waters.

Once deposited, the cohesive nature of mudflat sediments means that the stability of such sediment depends upon their bulk properties. The sediment size and the chemical and physical cohesion of grains determine the strength of the deposited bed sediment. The cohesive properties of the sediments of intertidal mudflat develop slowly as consolidation occurs. In the natural environment consolidation is influenced by a number of factors including:

- Tidal inundation;
- Drainage;
- Biological processes; and
- Air temperature.

4.2.3 Nutrient Cycles and Organic Content

High primary productivity in wetlands causes an accumulation of organic matter in the sediments. Such accumulation results from the addition of *in situ* plant biomass as well as organic material settling out of the water column. Wetland sediments serve as reservoirs of organic matter for estuarine systems (Boorman *et al*, 1994). Wetlands also represent important sinks in the global carbon cycle and Schlesinger (1977) estimated that they contain approximately 10% of the earth's land soil organic carbon. The organic contents of sediments influences many aspects of saltmarsh ecosystems, including (Zedler, 1996):

- Altering sediment porosity and water holding capacity;
- Influencing nutrient dynamics;
- Controlling the growth rates of plants and algae; and
- By influencing the abundance, composition and productivity of benthic invertebrates.

Nutrient cycles in saltmarshes are complex being dependent on numerous physical, chemical and biological processes. Tidal exchange influences the influx and efflux of nutrients from the soil and estuarine waters and is dependant on site elevation and vicinity of creeks (Broome, 1990). The alternating aerobic and anaerobic conditions that result from regular tidal flooding affect the biogeochemical cycles of nutrients and consequently marsh productivity.

Interactions between nutrient dynamics, organic matter and redox conditions control plant growth rates. Low organic matter in soils produces low rates of nitrogen fixation, and therefore low supplies of nitrogen. Conversely, in soils with significant organic matter negative redox potentials arise and may restrict the growth of some marsh plants such as *Salicornia* (PERL, 1990).

4.3 SEDIMENTARY PROCESSES IN HABITAT CREATION AND RECHARGE SCHEMES

The choice of sediments for intertidal recharge may be governed by local availability. However there are advantages and disadvantages of using either coarse or fine recharge material. A general problem related to recharge, whether it be with coarse or fine material, is the smothering of existing saltmarsh plants and benthic fauna at the recharge site. This is especially likely to be an issue if the initial placement occurs at too high a rate.

The recharge of muddy intertidal habitats with coarser materials, such as sand and gravel, changes the physical, ecological and aesthetic characteristics of the placement site. In terms of the physical processes, the placement of sands and gravels disturbs the existing sediment processes by decreasing the resuspension of fine-grained materials and hence the mobility of the foreshore profile. Such a change offers benefits in terms of flood defence and a number of schemes using coarse material have been successfully undertaken to protect saltmarshes from wave attack and erosion (Carpenter and Brampton, 1996; M. Dixon, *pers. comm.* 1996).

However, the use of coarse sediments for the recharge of muddy intertidal habitats is generally less acceptable on conservation grounds. Such changes in the intertidal habitat at the recharge site are likely to be considered unacceptable in areas of high nature conservation and landscape importance. In terms of ecological characteristics, the high biomass communities typical of muddy habitats, are replaced with lower biomass, higher diversity communities, associated with coarser sediments. This alteration of the structure and composition of benthic communities may result in a reduction in the food supply and available feeding area for birds and fish. This may in turn cause a change in the composition of species using the area. However, mounds of coarse materials may also provide alternative habitats for breeding and roosting birds.

A more natural approach to intertidal recharge is the use of fine materials with similar grain sizes to the existing intertidal mudflats (mixtures of sands, silts and clays). A small number of experimental recharge schemes using fine materials have been undertaken in the UK, although levels of success have been difficult to assess and highly varied (Kirby, 1995; Carpenter and Brampton, 1996; I. Black, *pers. comm.* 1996; M. Dixon, *pers. comm.* 1996). The advantages of using fine cohesive sediments can be summarised as follows:

- Fine materials can be retained within the intertidal zone and recycled into the saltmarsh/mudflat system, replacing lost intertidal area;
- Clean (uncontaminated) fine dredged materials are able to support high biomass benthic communities, similar to natural intertidal flats, and can be recolonised by fauna at the recharge site and from adjacent areas;
- The recharged intertidal habitat will closely resemble natural intertidal flats, both in appearance and function; and
- Saltmarsh regeneration is encouraged through the regrowth of existing saltmarsh plants and the recolonisation of recharged areas.

Despite the many advantages of using fine dredged materials, there are a number of potential problems associated with the approach and problems have been encountered in past schemes. These disadvantages or constraints can be summarised as follows:

- After placement dredged cohesive sediments require a consolidation period during which they are less stable/ more erodible than coarser material. This increases the risk of sediment loss from the disposal site.

- This loss of sediment increases the potential for sediment to be drawn into the subtidal zone and returned to the dredged channel thereby increasing sedimentation rates. This may have a number of effects including:
  - reduction in navigable depth,
  - the smothering of adjacent benthic communities, particularly sensitive shell fish beds,
  - increased turbidity which can reduce photosynthetic activity in phytoplankton and algae, and disturb sensitive marine communities (e.g. filter feeding organisms).

The various techniques that have been adopted in response to this problem are discussed in Section 4.5.5, and in more detail with regard to the individual schemes in Section 13.

4.4 RATES OF SEDIMENT MATURATION IN CREATED HABITATS

Natural habitats such as saltmarshes generally form over periods from 10's to 1000's of years. In comparison habitat creation or restoration schemes often involve stepwise changes in habitat type over a matter of months to years. As such sediment properties will differ to the natural habitats which the artificial scheme is aiming to create. This section considers the sediment maturation of artificial habitats with respect to three processes:

- Deposition;
- Consolidation; and
- Organic content.

4.4.1 Deposition

The experience gained from the Northey Island managed retreat scheme demonstrated high initial accretion rates in the summer following the breach of sea defences and the reintroduction of tidal influence (Pethick, 1994). Additionally, data gathered at the site lent some support for the model for decreasing SSC's and deposition rates away from creeks, described in Section 4.2.2. The surface topography data demonstrated that accretion rates in the site were highest at the entrance to the restored site but decreased thereafter to a zero accretion rate at 140m from the entrance. Laboratory results using natural marsh vegetation in a flume suggested that zero accretion would be achieved at 75m from the 'entrance' in a *Halimione* marsh, given a 50ppm initial suspended sediment concentration. The Northey Island data showed that an initial suspended sediment concentration of 100ppm resulted in a 140m extinction distance for accretion. However the vegetation type at the field site was dominated by the much smaller *Salicornia* rather than *Halimione.* As such, *Salicornia* would be expected to result in much lower rates of deposition.

Although these results are far from conclusive they do indicate that a predictive model can be developed for marsh creek density based on the principles outlined here, which would maximise the rate and area over which deposition takes place for restored marsh sites.

4.4.2 Consolidation

In an artificially recharged site, the rates of sediment consolidation will depend greatly on the method used in laying down the sediment (see Section 4.4.3). It is important to account for substrate consolidation at the design stage because this affects the final elevation of the site. Considering this slow consolidation of sediments, it may be some time before the outcome of intertidal recharge schemes and habitat creation projects utilising dredged material can be evaluated (Toft and Maddrell, 1995).

4.4.3 Organic Content

The build up of organic material in natural saltmarsh sediments occurs over time scales in the range of centuries and millennia, rather than the timescales of years and decades involved in restoration and creation schemes. Whether constructed marshes can become an organic carbon sink with a reservoir capacity similar to natural marshes is an important question from a management perspective (Havens *et al*, 1995).

The length of time it takes for constructed saltmarshes to resemble natural marshes has been estimated based upon the levels of organic carbon in the sediments (Cammen *et al*, 1974; Seneca *et al*, 1976; Webb and Newling, 1985). The organic contents of both natural and man-made marshes can show great variation (Roberts, 1991). The length of time required for the development or maturation of sediments varies greatly from scheme to scheme. However, many studies show that constructed wetlands lack organic matter, especially in the initial stages of development (Havens *et al*, 1995):

- The work of Cammen *et al* (1974) confirmed the wide variations in the timescale required for the accumulation of organic matter in the sediments of constructed marshes, with estimates of 3.7-4.5 and 22-26 years for two different sites;

- Seneca *et al* (1976) estimated that it can take between 4 and 25 years for carbon content in constructed marshes to reach similar levels to natural sites;

- In Florida, the average organic matter content in the 38 created marshes studied ranged from between 0.2 to 14.4 % and showed no direct increase with age. These results were thought to reflect the inadequate time received for saltmarsh development in many of the schemes (some of which had been in existence for little over 1 year) (Roberts, 1991);

- In a created saltmarsh at Bolivar Point, Texas, sediment organic levels, 3 years after construction, were lower than those found in the neighbouring natural marshes. However, the rapid rate of accumulation of organic matter was such that natural levels were predicted to be reached 5 to 7 years after construction (Webb and Newling, 1985); and

- In a tidal creek and marsh system, 5 years after its excavation from coastal upland, the differences in organic content of the surface sediments were negligible when compared to two adjacent natural marshes. However, at depths between 4 and 16cm, organic contents were greatly reduced at the created site. The lack of an organic rich peat substrate in this "root zone" was considered to account for the poor development of *Spartina* marsh (Havens *et al*, 1995). However despite this, the benthic communities in the non-deficient upper sediments of the unvegetated marsh and tidal channels were found to resemble those of natural marshes (Havens *et al*, 1995).

4.5 PRACTICAL CONSIDERATIONS AND PROBLEMS IN THE CREATION OF INTERTIDAL HABITATS

This section considers the practical considerations and problems related to habitat creation for restoration schemes. It draws on the experiences gained from a number schemes around the world and considers the following aspects:

- Soil and sediment characteristics;
- Sediment chemistry;
- Sediment organic content;
- Sediment quality;
- Sedimentation; and
- Sediment placement.

Each of these aspects are discussed in greater detail below.

4.5.1 Soil and Sediment Characteristics

Sediment and soil characteristics can affect:

- Substrate stability;
- Ease of planting vegetation (Knutson *et al*, 1990);
- Nutrient availability; and
- The placement technique.

It is widely recommended to use similar sediments to those that are being replaced when creating or restoring intertidal habitats for conservation and mitigation purposes. For example, in Florida sands and loams are the most commonly used sediments for the creation of coastal marshes. Such sediments also form the natural salt marsh soil types in the area (Roberts, 1991). Generally, loose loam to clay soils are the most suitable for marsh plant growth (Harvey *et al*, 1983). The clay and organic matter in these soils provides ion exchange sites that buffer the soil against salinity fluctuations (Long and Mason, 1983) and allow nutrients to be passed to the plants (Harvey *et al*, 1983).

The use of coarse sediments can cause the poor development of floral and faunal communities in created habitats. Such sediments differ from the natural fine textured marsh soils in a number of ways. Comparatively, coarse sediments have:

- Impaired water absorption, impoundment and retention qualities;
- Altered salinity levels (Zedler, 1996); and
- Reduced nutrient levels.

According to PERL (1990), the most important variables in predicting the ability of a site to support a functional saltmarsh are:

- Soil salinity;
- pH;
- Nitrogen dynamics;
- Organic matter concentration; and
- Redox potential.

Soil texture and salinity can be important in determining the species established on the restored or created area. If these characteristics are known, it is easier to predict the species composition and ultimate direction of restoration (Frenkel and Morlan 1991). The tolerable range of pH is thought to be fairly broad for halophytes, with growth being possible from pH 4 to 9. However, most nutrients are more readily taken up by plants when the pH is between 6 and 8 (Harvey *et al*, 1983). The redox potential is also important in controlling the cycling and mobility of nitrogen, sulphur, and heavy metals (PERL, 1990). Nutrient supply is not thought to be a problem, especially if dredged material is used. However appropriate fertilisers can be used if necessary in extremely nutrient poor sediments (Knutson *et al*, 1990) (see Section 4.5.3).

Cohesive soils are more difficult to plant than loose sandy soils, but offer advantages in that their stability allows time for plants to establish without substrate movement. If dredgings are used as fill material, soft estuarine muds are most suitable since overly firm or stiff muds can prove difficult for plant colonisation (Haltiner and Williams, 1987).

The type of sediment or substrate used for habitat creation schemes influences the manner of placement. In the USA, experience illustrates that at sites constructed of soft dredged materials, such as silts and clays, their lack of bearing capacity can make mechanical operations such as grading and contouring difficult. Conversely, on eroding shorelines, silt and clay sediments may become compacted which can hinder planting schemes and make the construction of sites more difficult (Broome, 1990). Although soils with higher sand contents provide a more stable environment on which to operate equipment, they have a number of disadvantages in terms of subsequent plant colonisation (see above).

4.5.2 Sediment Chemistry

The chemistry of substrates may undergo changes during and immediately following the construction process and their characteristics may differ from natural wetlands in terms of:

- Acidity;
- Hydricity; and
- Salinity.

The substrates of newly excavated sites may become acidic when exposed to air, as sulphides accumulated in the soil are oxidised into sulphuric acids. This problem was encountered during the Pacific Coast Terminals Saltmarsh compensation scheme (Appendix II).

When a new wetland is created with dredged materials, hydric soil conditions and the appropriate hydrological conditions must be introduced to the site. The period of time required for soil to assume hydric characteristics after becoming saturated is unknown. However, current data suggests that the minimum time is fifteen years (PIANC, 1992).

Although natural marsh soils are usually highly saline, field and laboratory studies have shown that rates of vegetation establishment are improved in less saline substrates (Zedler, 1984). Even US smooth cordgrass, which is able to grow in marine salinities of up to 35psu, establishes and grows more rapidly in lower salinities (Knutson *et al*, 1990).

Salinity problems were encountered during the restoration of tidal marsh at the Ballona wetland in Southern California. At this site, the lack of tidal flushing led to the development of hypersaline soil conditions (up to 50 psu) in the sediments (Tsihrintzis *et al*, 1996). To improve the success of transplants or colonisation, three methods are currently under consideration for the treatment of hyper saline areas:

- Artificial flushing with freshwater;
- Treatment with freshwater/hydroax injections; and
- Extended period of natural tidal flushing.

In terms of freshwater flushing, Zedler (1996) recommended watering the site with soaker hoses or flooding the area with fresh water to reduce salinities before exposing the site to tidal inundation.

4.5.3 Sediment Organic Content

Lack of organic content in sediments affects floral and faunal activities directly, through nutrient availability, as well as indirectly, through altering the sediment density and ability of burrowing fauna to colonise the sediment (Section 8). As discussed in Section 4.4.3, the time taken for sediment organic content in constructed marshes to reach that of natural marshes, is greater in the root zone than in the surface sediments (Havens *et al*, 1995). Havens *et al* (1995) suggested that a solution to this problem would be substrate enhancement. Havens *et al* considered a suitable option for this would be to supply wetlands with highly organic substrate preferably similar sediments derived from wetland, in the constructive phase. This would provide:

- Peat soils with high organic carbon contents;
- An inherent seed bank; and
- A developed micro-organism population.

In the restoration scheme described in Section 4.4.3 (Havens *et al*, 1995), the fact that *Spartina* marsh was poorly developed, whilst benthic communities were found to be fully developed, suggested that the perceived success of a restoration scheme depends upon the group of flora or fauna being targeted. This then influences whether substrate enhancement is considered as part of a habitat creation or restoration.

In the United States there has been some debate as to whether sites of wetland creation need “top soiling” with fertile mineral loam soils to enhance habitat development. A number of workers (e.g. Webb and Newling, 1985; Garbisch, 1993), have considered it unnecessary and unsuitable for a number of reasons:

- Wetlands have historically developed on infertile alluvial soils;
- Organic matter rapidly accumulates in newly developing wetlands (e.g. Webb and Newling, 1985); and
- The high levels of nutrients that may result can lead to the development of unnatural floral and faunal growth patterns.

For these reasons, substrate enhancement has therefore been considered only to be necessary for nutrient-deficient soils on high energy shores (Knutson and Woodhouse, 1983).

4.5.4 Sediment Quality

Analysis of the quality of the sediment is an important consideration in the creation of wildlife habitat, particularly in beneficial use projects. This applies equally to schemes that add material to form areas of intertidal, or those which remove it to form creeks and lagoons. In urban and industrialised areas, the excavation of coastal habitats may inadvertently cause the resuspension of contaminants stored in the sediments, possibly during previous reclamation activities or if the previous habitat had been used as a landfill (Zedler, 1996). The use of dredged material is particularly common in habitat creation schemes. However, dredged material derived from industrial and urban areas may in certain cases, be heavily contaminated with a variety of substances which are potentially toxic to plant and animal life, including:

- Heavy metals;
- Nutrients;
- Oils;
- Pesticides; and
- Persistent organic substances e.g. PCBs (polychlorinated biphenyls) and TBT (tributyl tin).

These toxic substances may be taken up from the sediments and accumulate in the tissues of plants and animals (bioaccumulation), and may become amplified up the food chain (biomagnification). It is, therefore, necessary to characterise the dredged material to evaluate its suitability for intertidal recharge and to determine similarities with the potential receiving area. Dredged materials are generally considered suitable if pollutant concentrations are less than the recipient sediments. Contaminants have a higher affinity for finer sediments, such as silts and clays, where they are generally found in higher concentrations. Contamination is not normally a problem if the recharge material consists of sands and gravels.

4.5.5 Sedimentation

The rate of natural sedimentation at a site depends on a number of factors (Barker *et al*, 1993):

- Elevation;
- Local topography;
- Proximity to the breach;
- Concentration of sediment load in the flooding water;
- Tidal prism; and
- Wave energy.

In set back sites where breaches have been made in an existing sea wall, the majority of the sediment is deposited in the breach area. There is a gradation of particle size away from the breach; coarser sediment is deposited nearer to the breach and finer sediment is deposited further away.

Excessive sedimentation in tidal channels excavated from non-tidal areas can cause the creek system to be filled in and sand bars to form across the mouths of channel. Such sedimentation can lead to a reduction in tidal flows (Zedler, 1996). This is a particular problem where sites are dredged or excavated deeper than the channel that feeds them. High rates of sedimentation may also result from development activities in the catchment area or from periods of flooding.

Although areas of accretion and channels which are "silting-up" can be maintained through dredging activities, such activity is undesirable since it is likely to disturb flora and fauna (Zedler, 1996). It has been suggested that where sedimentation events can be predicted, it may be possible to compensate by over excavating channels at the construction site (Zedler, 1996).

4.5.6 Sediment Placement

A problem associated with the use of fine sediments in recharge and creation schemes is how to keep them in place while the processes of consolidation and stabilisation take place. The demand for an ecologically sensitive means of dredged material placement has lead to the trial of innovative disposal techniques. Such techniques aim to:

- Reduce the impact of habitat construction on sensitive intertidal habitats; and
- Aid recovery of the reconstructed habitat.

Details of various restoration and creation schemes are discussed in Section 13. There are a number of methods for retaining the fine sediments at the recharge site (Carpenter and Brampton, 1996):

- Bunds;
- Silt curtains;
- Hazel fencing;
- Straw or coconut matting;
- Mounds or walls of coarser dredged material;
- Thin layer placement or rainbowing; and
- Reduce wave energy (see Section 3.4.4).

Thin-layer disposal of dredged material has been under trial as a mechanism of saltmarsh and intertidal flat recharge. This method is now being applied on an experimental basis in the Hamford Water, Blackwater Estuary and the Stour and Orwell Estuaries (Carpenter and Brampton, 1996; M, Dixon, *pers. comm*. 1996). This method of disposal has been termed "rainbowing" due to the appearance of the sediment and water mix being pumped at high pressures into the air and onto the intertidal area. Even saltmarshes which have reverted to open water can be restored if sufficient quantities of dredged material are available (Olin *et al*, 1994).

Initial findings of an experimental scheme in the Medway indicate that intertidal recharge at a slow rate can be achieved using fine materials without the need for bunds (Pethick, *pers. comm*. 1996).

Preliminary trials to investigate the technological and ecological feasibility of the use of hovercraft for dredged material transport and distribution in wetlands have been made by the US Army Corps of Engineers to reduce the impacts of conventional transport methods (Olin *et al*, 1994).

The timing of the placement of material at a site should avoid periods of high physical disturbance, such as the winter storms events, as well as ecologically sensitive periods (Section 5). The optimum time to deposit dredged material is late summer (August) so that the sediment has time to consolidate before the main release period for the seeds of saltmarsh plant species (Section 6.4).

5. ECOLOGICAL PROCESSES

5.1 INTRODUCTION

In order to successfully create or recharge intertidal habitat it is of primary importance to have an understanding of the ecological functioning of the estuarine ecosystem. Of particular relevance to this review are the role of estuarine floral and faunal communities in the food chain, the process of natural succession and the development of intertidal communities. Central to the basic design of intertidal habitat is an understanding of the ecological mechanisms by which pioneer plant and animal species colonise a new area, and the spatial and temporal relationship of such species to more established communities.

This section discusses habitat creation schemes in the context of the following:

- Natural succession and community development;
- Food chains and energy flows;
- Habitat preference and requirements;
- Recolonisation rates; and
- Factors affecting recolonisation rates and community structure.

5.2 NATURAL SUCCESSION AND COMMUNITY DEVELOPMENT

Change in an intertidal environment is often linked to the concept of succession, whereby individuals of different species interact to produce changes in the species composition. An example of ecological succession is the progressive development of saltmarsh from intertidal mud to the climax vegetation of the grazed upper marsh.

In the early stages of saltmarsh development, intertidal mudflats are initially invaded by colonising species (or pioneer species), for example *Salicornia europaea*, that are able to tolerate frequent inundation with saline water. The presence of such species serves to stabilise the substratum and promote accretion of sediment, gradually raising the surface level of the intertidal zone. Species that are better adapted to the slightly elevated intertidal therefore begin to colonise the area. These species are often better adapted to the less frequent or shorter duration of tidal inundation that results from the creation of the raised intertidal area than the pioneer species. As a result, the pioneer species are lost from the saltmarsh community as they are less well adapted to survive in the altered physical environment and they are replaced due to competitive interaction with other, more tolerant, species. Typically, this process continues until the marsh surface is raised above the height of the highest tides, eventually allowing non-halophytic species to invade the raised surface.

The result of the above process is a gradation of community types within a saltmarsh habitat complex, with a transition from more halophytic (salt tolerant) species dominating the lower saltmarsh zones with plant species that are less tolerant of immersion in saline water becoming more dominant at higher levels in the saltmarsh.

In addition to the zonation of vegetation within the saltmarsh habitat, benthic fauna also display transitional distributions across saltmarsh. These distributions are due to a range of physical and chemical factors, including:

- Salinity;
- Sedimentation;
- The degree of shelter;
- Chemical composition of the sediment;
- Temperature; and
- Oxygen concentration.

The distribution of some key animal species or groups of species across a typical marsh transition is shown in Figure 5.1.

The diversity of benthic organisms within saltmarsh communities also varies with distance from the mouth of the estuary. Generally, diversity decreases upstream from the mouth of the estuary as a result of changes in salinity and substrate conditions (Davidson *et al*, 1991).

Although evidence of the ecological succession process described above can be recognised on many saltmarshes, consistent evidence of a full linear succession as described above is lacking for both fauna and flora (Long and Mason 1983). It is generally agreed that such a theoretical process is an oversimplification of successional changes that occur in ecological communities, although the extent of oversimplification suggested varies from study to study (Chapman 1960, Long and Mason 1983).

The fact that community change over time does not necessarily occur in a predictable or orderly way has implications for the design of intertidal habitat creation. In particular, difficulties have been experienced in predicting the patterns of vegetation colonisation for given creation schemes (Niering, 1990). Consideration of this process is therefore important in any habitat creation scheme (US Fish and Wildlife Service, 1989).

Despite the uncertainties inherent in creating intertidal habitat, an understanding of the theory of natural succession allows managers to identify the features of the new habitat that are most likely to change in a predictable way.

5.2.1 Regional Perspective

There is considerable geographical variation between the floral and faunal communities of estuarine ecosystems. For example, the saltmarshes of the south and east coasts of England provide a link with those of the warmer south including the Mediterranean, while those in the north which are more restricted, show affinities with those of the Arctic (Toft and Maddrell, 1995). In the south of England, species such as sea-lavender, sea purslane and cordgrass are particularly prevalent. However, these species cease to be a major component of the vegetation north of a line drawn between the Solway in the west and the Firth of Forth in the east. Given this variability, it is necessary that a regional approach should be adopted in the planning of mitigation projects which considers the necessity to sustain biodiversity on a national scale.

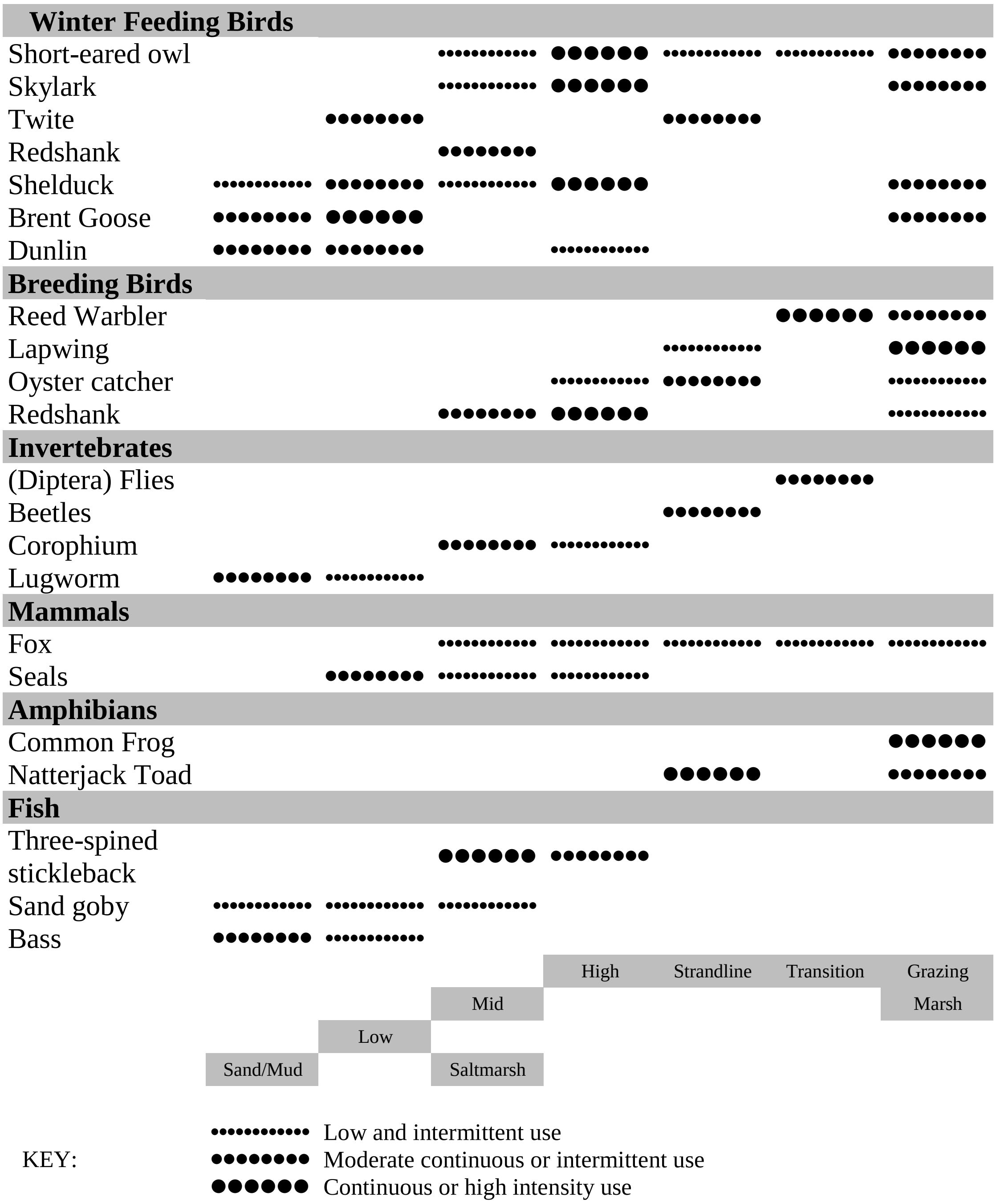

Figure 5.1 **Distribution of some key animal species or groups of species across a typical saltmarsh. (Source: Toft and Maddrell, 1995)**

5.3 FOOD CHAINS AND ENERGY FLOWS

The estuarine environment is temporally variable in terms of physical and chemical characteristics due to tidal activity and therefore relatively few species are successful in adapting to this environment. However, those species that are adapted to this environment are often present in high abundance.

In the intertidal estuarine environment, benthic algae, plants and microscopic algae are responsible for primary production. In the uppermost regions of the intertidal zone, saltmarsh plants often account for most of the primary productivity, whereas further down the intertidal, algae growing on the surface of the mud, such as *Enteromorpha* spp. are more dominant. Eelgrass (*Zostera* spp.) are true flowering plants and these species can also make a contribution to primary production in estuarine areas. Single celled epibenthic algae (microphytobenthos) live attached to sediment particles, and some species can vertically migrate within the upper few centimetres of the mud, moving down when the intertidal is immersed to avoid being washed away and moving up when the mud is uncovered to photosynthesise at the surface.

The primary producers described above, along with the phytoplankton in the water column, constitute the basis of the food chain in the estuarine ecosystem. Much of the primary productivity enters the food chain as detritus. Detritus, which is largely comprised of plant material, can be defined as “all types of biogenic material in various stages of microbial decomposition which represents a potential energy source for consumer species” (McLusky, 1989). Bacteria and other microbial organisms decompose this material and there are strong links between phytoplankton, benthic microalgae, plant fragments and their decomposers. In fact, the relationship is so intertwined that the food for primary consumer animals is generally called particulate organic matter, without regard to its exact origin (McLusky, 1989).

Primary consumer animals are mostly found on the bed of the estuary where they form benthic communities. Such communities comprise, for example, Polychaeta, Oligochaeta and Mollusca. These animals constitute the food resource for higher predators such as estuarine fish and birds.

The interaction of the various components of the estuarine ecosystem in terms of food chains and energy flows, or the functioning of the system, is therefore as important as the structure of the community (for example, the species composition and diversity). “Ecological restoration means more than simply replacing the dominant plant cover or habitat structure. Systems must be measured to assure that not only are the appropriate structures in place, but that these structures are functioning normally over an appropriate ecological time scale” (Pratt, 1994). This requires that any changes occurring in the constructed wetland are a result of environmental variations rather than shortcomings in the original construction of the habitat.

As ecosystems mature, diversity increases at many organisational levels, in terms of species richness, landscape heterogeneity, community richness, life cycles, biogeochemical cycling, and food web complexity. It takes many years for a restoration site to develop into a fully functioning, self-sustaining ecosystem. According to Zedler (1996) no system can be said to have done so, as all those monitored continue to change up to 15 years after the initial restoration work. It is a requirement of the monitoring that the difference between developmental and natural changes is distinguished.

5.4 HABITAT PREFERENCE AND REQUIREMENTS

The biological preference and requirements of plant and animal communities and particular target species need to be considered in the creation of coastal habitat systems (US Fish and Wildlife Service, 1989). The requirements of target species include:

- nutritional;
- chronological (reproduction and migration);
- behavioural or social aspects (foraging techniques and breeding behaviour); and
- the significance of location and status (endangered or rare, recreational or commercial value).

Certain food chain functions can be restored or created with relative ease, whilst other, more subtle, ecological functions are more difficult to recreate due to a lack of basic understanding (Kusler and Kentula, 1990).

Various schemes have gauged success by comparing the created site with “control” or reference areas in the immediate vicinity (Webb and Newling, 1985; Roberts, 1991). However, it is not always possible to locate local control areas and in these cases the best representative ecosystems in the region can be used (Pratt, 1994).

5.5 RECOLONISATION RATES

An important question to address when considering the establishment of communities on created habitats is the rate at which communities similar to those in comparable habitats nearby develop. Basic theoretical ecological concepts provide several predictions that apply to the design of created habitats (Pratt, 1994). Colonisation of newly created habitats generally follows the dynamics suggested by MacArthur and Wilson’s theory of island biogeography (1967). Their equilibrium model states that species colonisation and accumulation at a new habitat patch is a saturation process which initially occurs at a high rate. Subsequently, as the number of resident species rises, the rate of immigration of new, unrepresented species diminishes. The immigration rate reaches zero when all species from surrounding areas are present (Begon *et al*, 1990) (Figure 5.2). Community structure changes once equilibrium is reached as a result of interactions between species.

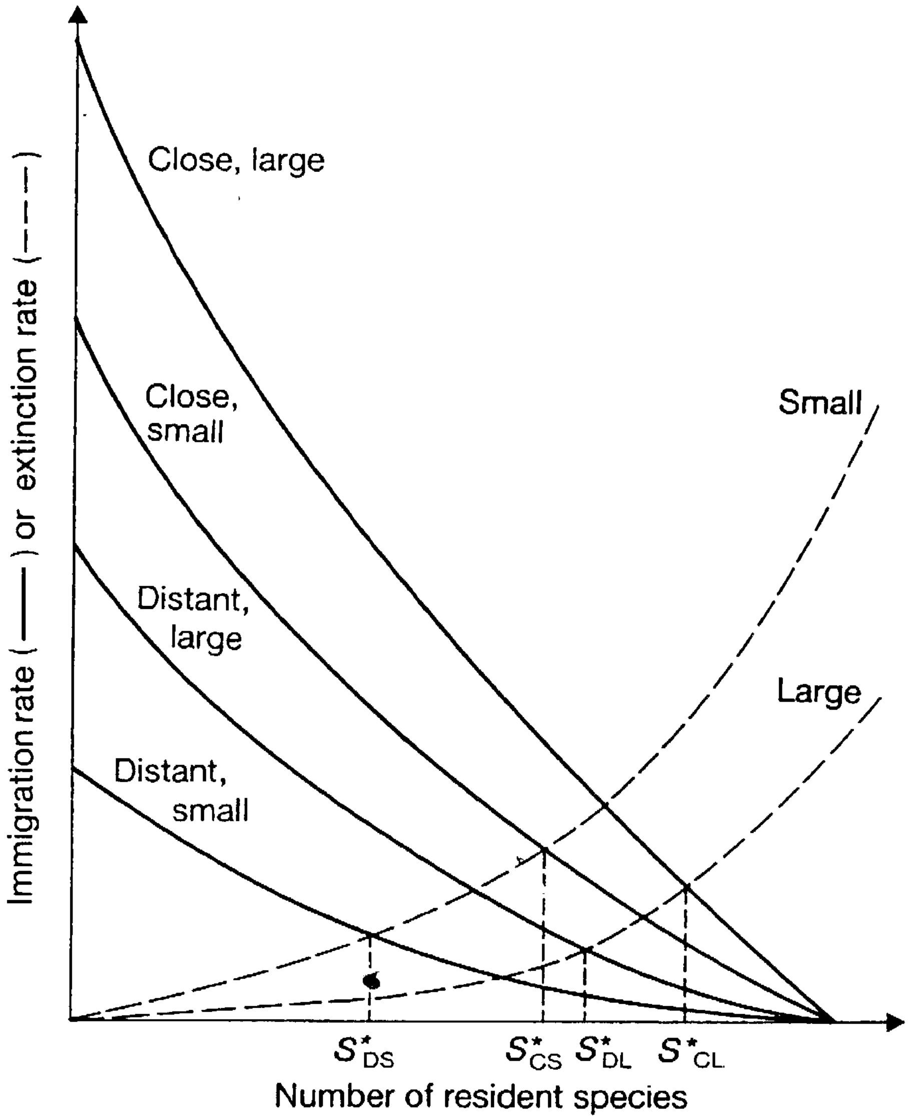

Figure 5.2 The equilibrium model based on the theory of island biogeography (Source: MacArthur and Wilson, 1967). The balance between immigration and extinction on small and large and on close and distant islands. In each case, s* = the equilibrium species richness

The recolonisation of a newly created area of intertidal mud in an estuarine system is likely to follow a fairly predictable sequence of events. Following placement of the mud, the species richness is likely to be low, although this will obviously depend on the source of the material. Subsequently, the mud will be colonised by those species present in neighbouring habitats and will largely occur by recruitment of individuals from the planktonic phase in the water column. The initial colonisers are often termed opportunistic species capable of high rates of reproduction and fast growth rates. Therefore, these species are capable of building up abundant populations relatively quickly (within a period of weeks). Following this initial phase, other, competitively superior, species are likely to colonise the sediment resulting in an increase in the species richness of the community. These species are likely to be slower growing but longer lived than the opportunistic species. Eventually, a point is reached where, in theory, the rate of colonisation by new species is zero and the community structure is determined by inter-specific competition.

There is little information regarding the time taken for mitigation sites to develop wetland habitat values comparable to natural wetlands, although the time is likely to be highly variable depending on the type of habitat created and the environmental conditions at each specific site. For example, Josselyn *et al* (1990) report that high marsh sites in the west of the United States are likely to be rapidly colonised by vegetation within a few years whereas lower portions are more slowly revegetated, unless artificial propagation is undertaken.

5.6 FACTORS AFFECTING RECOLONISATION RATE AND COMMUNITY STRUCTURE

The literature review has identified a number of important considerations in the design and construction of schemes that influence rate of recolonisation of newly created intertidal habitats and the ecology of the wider estuarine system, namely:

- site size;
- site location;
- habitat type and tidal influence;
- replacement of flora and fauna
- management and enhancement of natural processes; and
- timing.

5.6.1 Site Size

According to the island biogeographic model of colonisation, habitat heterogeneity increases with habitat size. Consequently larger habitats hold a greater diversity of species and diverse habitats have a greater chance of persistence (Willard and Hiller, 1990; Pratt, 1994). This is a theory that can be applied to habitat creation schemes.

In order to increase the likelihood of success of habitats created for conservation purposes, the construction of small, narrow, fringe-marshes should be avoided. Roberts (1991) recommended that block-shaped salt marshes of at least 0.5ha in size were necessary for the successful provision of fish and bird habitat. Many saltmarsh creation schemes in Florida were considered to be too small to provide the habitat area required by common marsh dwelling birds or fish (Roberts, 1991).

5.6.2 Site Location

The number of colonising species is dependent on the size of the habitat and its distance from the source of colonising flora and fauna. Therefore, the location of the proposed creation or restoration site is an important element in determining the rate of recolonisation from surrounding areas and ultimately a project's potential for success (US Fish and Wildlife Service, 1989; Roberts, 1991).

Close proximity to high quality natural habitats both aids and encourages recolonisation. For example, the ecological success of a scheme to rebuild Carrolls Channel on the Columbia River, USA, was attributed to the ecologically rich surrounding area (US Fish and Wildlife Service, 1989). The channel was created by removing infilled material, restoring previous elevations and allowing natural deposition of silt over the area from the Columbia River. The silts were colonised by a number of desirable emergent species from adjacent habitat and three years post-construction the diversity and value of the Carroll Channel's plant and animal communities were far greater than expected.

By locating intertidal replacement sites in areas where they can be hydrologically connected to existing wetland, the created habitat can be used to increase the diversity of the larger area by possibly providing missing habitat requirements (Marble, 1992).

5.6.3 Habitat Type and Tidal Influence

The ecological characteristics of a created estuarine habitat will be greatly determined by the hydrology of that habitat and the nature of the estuarine inlet in terms of tidal flushing which may vary from a fully tidal or brackish system, to a closed system with a high freshwater influence. Certain coastal habitats may be closed to tidal influence at certain periods of the tidal cycle or time of year, to varying extents and durations. Habitats with reduced tidal flushing, such as brackish lagoons, can be susceptible to freshwater inundation during the winter and evaporation and hypersalinity in the summer months.

Boland and Zedler (1996) investigated the influence of varying tidal regimes on the flora and fauna of four created habitat sites in Southern California. These sites were divided into three hydrological types; fully tidal inlets (Tijuana Estuary), frequently closed inlets and lagoons (San Dieguito and Los Penasquitos lagoons), and inlets with a berm, weir or tide gate which allowed water to flow out, but excluded or reduced tidal inflow (Ballona Wetland). Their studies strongly indicated a relationship between hydrology and biodiversity, with fully tidal sites supporting a higher diversity of species of saltmarsh plants, benthic invertebrates and fish, than frequently closed lagoons. Fully tidal habitats with open inlets are less likely to experience the environmental extremes that may develop in areas with reduced tidal flushing.

5.6.4 Replacement of Flora and Fauna

Habitat creation involves an unavoidable trade-off between new and existing habitats. For example, an upland habitat can be lowered and graded in order to provide the correct elevation for saltmarsh development. However, this will cause the displacement of coastal grassland and terrestrial plants, some of which may be of significant nature conservation importance. Davidson and Evans (1987) strongly recommended that restoration or creation of a wetland should not damage any features of existing conservation importance on that site. Suitable sites should not displace highly productive or rare communities. Transplanting schemes may be devised to re-establish communities of conservation importance within the new habitat creation schemes or elsewhere. Transplantation has been successfully achieved for a managed retreat scheme at Tollesbury, Essex, where the saltmarsh present on the sea wall to be breached was successfully transplanted to a nearby location (I. Black, *pers. comm*. 1996).

Davidson and Evans (1987) also consider the identity of the waders that will use newly created mudflats for feeding in relation to those using the lost habitat. They concluded that the wintering waders of a particular species using the new site may not be those that were using the site prior to the scheme, particularly if the artificial habitat is created before the loss of the old. Hence, whilst the development of a compensatory site may ensure the maintenance of the overall population sizes of waders, they may not always aid the continued survival of those individuals that formerly fed on the destroyed area.

5.6.5 Management and Enhancement of Natural Processes

The ecological recovery of a disturbed habitat, including newly created and restored sites, depends upon a number of biological factors, such as the sources and transport of propagules which in certain cases may require management to facilitate the natural process (Pratt, 1994). Target species with long life spans and poor dispersal mechanisms might need particular management through introduction. Pratt (1994) recommends that newly restored wetlands should be inoculated with material from similar aquatic ecosystems to ensure that microbes and invertebrates will effectively colonise the site. However, this is not considered necessary in situations where the created site is adjacent to established sites and where opportunity therefore exists for exchange and transport of natural propagules between the existing and created site (I. Black, *pers. comm*. 1996).

5.6.6 Timing

Certain times of the year are more conducive to wetland creation or restoration. It is widely suggested that the timing of schemes should be organised to avoid periods when estuarine flora and fauna are particularly sensitive to disturbance. Schemes should be scheduled to enhance the natural response and recovery rate of the estuarine flora and fauna (Josselyn *et al*, 1990). Examples of timing considerations include:

- the avoidance of periods of heavy use of intertidal habitat by breeding and overwintering birds;
- the avoidance of spawning and breeding periods of estuarine invertebrate and fish species; and
- implementation sufficiently in advance of the main germination period of saltmarsh species in spring so that sediments may undergo some degree of consolidation and stabilisation.

Intertidal habitats must be created well in advance of the date of loss of the area for which they are intended to compensate. The lead time will usually exceed a minimum of about five years prior to the loss of the site. This length of time is required to:

- identify suitable sites;
- acquire, design and construct the site; and
- allow settlement and stabilisation of a sufficient depth of suitable sediments.

As mentioned above, the time taken for mitigation sites to develop wetland habitat values comparable to natural wetlands is unknown and is likely to be highly variable. Differences in construction techniques, variation in plant establishment, and differences in the colonisation of new sites by animals makes it difficult to predict the rate of succession in restoration sites. Plant establishment in tidal marshes usually takes two to four years, though it can be accelerated by planting. If natural seed sources are not readily available, plant establishment may take five to seven years (Josselyn *et al,* 1990).

6. VEGETATION

6.1 BACKGROUND

6.1.1 Introduction

Within a saltmarsh habitat complex, halophytic plant species and communities display a transition, from a marine to a terrestrial habitat. This transition is important to the stabilisation of newly deposited sediments and provides the fundamental basis for food chains that operate across the intertidal ecosystem (see Section 5.2).

For any scheme where the objective is to create an intertidal habitat that includes the full transition from bare mudflats to the upper saltmarsh, a full appreciation of the physical conditions that determine plant zonation is required. A number of studies have specifically focused on this area of saltmarsh development and include both academic research and actual habitat creation and restoration schemes.

6.1.2 Research Background

United States
Since the early 1970's, a large number of intertidal habitat creation schemes have been undertaken in coastal areas of the USA. These schemes, which include the Ballona wetland scheme in Southern California (Tsihrintzis *et al,* 1996) and Archie Creek Marsh, Tampa, Florida (Lewis, 1981) have tended to focus, although not exclusively, on the restoration of degraded saltmarshes.

Procedures for creating and restoring saltmarshes have mostly been developed with the primary objective of stabilising both dredged material and eroding shorelines. This has largely been achieved by the planting of *Spartina alterniflora* (a cordgrass native to the eastern USA) although stands of *Juncus roemerianus* and mangrove species have also been used (Roberts, 1991). As a result, little information is available on natural colonisation of pioneer plant species and associated communities within a restored or created intertidal habitat, or the factors that may influence it.

More recently, schemes have been undertaken in the USA that mitigate the loss of wetlands and wetland resources caused by coastal development projects and other anthropogenic factors. As mitigation with specific nature conservation objectives, evaluation of the success or failure of this type of scheme has begun to consider the relative habitat value of created or restored saltmarshes in relation to natural habitat (Webb and Newling, 1985; Roberts, 1991). However, such schemes remain very much based on the planting of *Spartina alterniflora*, and are of limited value in drawing parallels to habitat creation schemes proposed for the UK.

Europe and the United Kingdom
A small number of intertidal habitat creation schemes have been undertaken in Europe. In the Netherlands, previously reclaimed land in the Zuiderzee has been allowed to revert to tidal flats and channels (Joenje, 1979). Monitoring of the

colonisation rates of intertidal plant communities has occurred and are of relevance to the UK due to the close geographical proximity. Within the UK itself, a number of managed retreat schemes have taken place, and include the Tollesbury Fleet scheme, Northey Island and St. Leonards (Abbots Hall). However, the relatively recent nature of these schemes has meant that little published information is currently available.

Considerable research work has been undertaken in the UK on the ecology of intertidal and saltmarsh areas, particularly on the north-western coast of the British Isles (Gray and Bunce 1972).

The plant that has been most frequently used in saltmarsh restoration in the UK is *Spartina anglica* (Environmental Advisory Unit, 1989). This species is a hybrid between the native small cordgrass *Spartina maritima* and the alien *Spartina alterniflora,* accidentally introduced from America prior to 1870. *Spartina anglica* was planted between the 1930's and 1950's to protect coastal areas from erosion and to provide new saltmarsh for eventual reclamation. Since then, *S. anglica* has spread around the coast, often at the expense of mixed saltmarsh communities and intertidal mudflats. The loss of intertidal flats has been of particular concern for sites traditionally used as a feeding resource by wintering wildfowl. As a result, there have been several attempts to control *S. anglica* by spraying with herbicide (Goss-Custard and Moser, 1988; Davidson *et al*, 1991).

The low conservation value of *Spartina* marsh and the highly invasive nature of this species mean that any further planting should be viewed with caution (Environmental Advisory Unit, 1989). There are very few accounts of saltmarsh restoration with species other than *Spartina* in the UK. The only example found is reported by the Environmental Advisory Unit of Liverpool University (1989) who cite the use of *Puccinellia maritima* turfs to encourage the spread of saltmarsh in the Ribble Estuary, although no further details are given.

In the few instances of managed retreat in the UK, schemes have relied on natural colonisation of pioneer plant species for development of the desired saltmarsh communities (M. Dixon, *pers. comm.* 1996). Such schemes include the Tollesbury Fleet managed retreat scheme in the Blackwater Estuary, Essex which relies on a good local source of colonising plant species. However, for this particular scheme, plugs of saltmarsh vegetation from neighbouring saltmarsh were also established, but were generally unsuccessful with the high initial rates of sedimentation occurring at the site, resulting in their burial. The vegetation that now grows at Tollesbury, eighteen months after the breach of the sea wall, is generally attributable to natural colonisation (I. Black, *pers. comm.* 1996)

6.1.3 Scope of Discussion

The following section reviews both the factors that enable plant colonisation, and those that influence the process of ecological succession within a saltmarsh (Section 5.2). These factors are illustrated with intertidal habitat creation schemes and research studies that have monitored plant establishment in new or restored intertidal areas.

Given the low conservation value of the *Spartina* marsh within the UK, this review places an emphasis on the colonisation of native saltmarsh species and considers those schemes and research projects that have characterised their habitat requirements and recolonisation rates. Reference to the habitat requirements and colonisation rates of *Spartina anglica* is also made, but should be considered in isolation from other pioneer plant species of much higher nature conservation value. The section concludes with a practical consideration of both positive and negative factors of saltmarsh vegetation establishment, and is based on recommendations from previous intertidal habitat creation schemes.

6.2 HABITAT PREFERENCE AND REQUIREMENTS OF VEGETATION

6.2.1 Environmental Factors Affecting the Transition Vegetation of the Saltmarsh

There is general agreement that the main factors affecting the zonation of halophytic plant species within a saltmarsh habitat relate to frequency of tidal inundation and associated effects of salinity, and tidal scouring (Brereton, 1971; Gray and Bunce, 1972; Frenkel and Morlan, 1991; Pieters *et al*, 1991). A general study by Brereton (1971) in Foryd Bay, Caervarnonshire, exemplifies these points and demonstrates that patterns of zonation are determined by the overall level of waterlogging as controlled by the height of the marsh surface relative to the sea level. Within each of the successional phases, he states that species distributions are controlled by variations in salinity and waterlogging and these, in turn, are associated with local differences in soil texture. This was also reflected in the development of four zoned plant communities at Salmon River, Oregon after managed retreat (Frenkel and Morlan, 1991).

The above findings are in broad agreement with a study by Gray and Bunce (1972) that investigated the effect of localised soil variation on saltmarsh development. The study concluded that the main environmental trend of the saltmarsh vegetation in Morecambe Bay was an expression of the pedogenic sequence that develops in association with the primary successions and the increased silting they cause. However, the study also made reference to the importance of factors related to the organic and nutritional status of the soils and it is suggested that the factors related to submergence (Brereton, 1971) are ‘superimposed’ on, and interact with, edaphic factors. Areas with similar patterns of submergence may therefore show considerable variation in vegetation associated with local discontinuities in soil type (Brereton, 1971; Gray, 1992).

The significance of the effect of tidal inundation in influencing the plant zonation is clearly demonstrated by Chapman (1960) who used the frequency of submergence to characterise vegetation type. Chapman states that the line of demarcation between upper and lower marshes lies at about mean high water. Typically, lower marshes will undergo more than 360 submergences per annum, and their maximum period of continuous exposure will never exceed nine days. Upper marshes, however, will undergo less than 360 submergences on an annual basis, with their minimum period of continuous exposure exceeding 10 days.

Other physical factors which have been demonstrated to affect the zonation of saltmarsh vegetation include the fetch associated with tidal action in relation to *Spartina anglica* (Gray, 1992), and sediment supply in relation to the ability of newly establishing plants to photosynthesise (NRA, 1995)

Whilst the studies described above consider some of the factors that may generally influence zonation of vegetation in a saltmarsh habitat, many more studies focus on a relatively small number of species and the factors that influence their establishment. Where documentation exists for plant succession in intertidal habitat creation schemes, the recent nature of such projects dictates that information is almost entirely confined to pioneer plant species. The following section characterises the habitat requirements of some of the more common saltmarsh species found within the UK for both establishment and colonisation.

6.2.2 Requirements for the Establishment of Eelgrass (*Zostera*) Species

The role of eelgrass *Zostera* spp. in the development of the saltmarsh is not clear. Prater (1981) and Brooks (1979) both record these species as one of the first plants associated with early stabilisation of mud and sandflats, growing around mid-tide level on muddy shores, and sometimes contributing to colonisation by *Salicornia* spp. (Brooks, 1979; Prater, 1981). However, a review by the NRA (1995) states that *Zostera* spp. are either permanently submerged or exposed only at low water on spring tides and are therefore considered by most researchers to be functionally and biologically independent of saltmarshes.

6.2.3 Requirements for the Colonisation of *Salicornia* Species

In 1935, Wiehe carried out an investigation in the Dovey salt marshes that examined the relationship between the density and vigour of populations of *Salicornia europaea*, a common pioneer halophyte, and the frequency of submergence by tides. Wiehe demonstrated that there was a high correlation between the density of individuals of *Salicornia europaea* and the number of days that they are free from submergence by the tide. This allowed him to conclude that mechanical removal of seedlings before they have developed an anchoring root system is a more important factor limiting the seaward growth of pioneer species than the duration of tidal inundation (NRA, 1995).

Pye and French (1993) confirm the above conclusion with their observation that the initiation of a new marsh by colonisation of pioneer marsh vegetation can only occur if the surface elevation is high enough to allow a sufficiently long period of exposure. Furthermore, the sediment surface should be sufficiently stable to allow successful germination and seedling establishment. They further state that germination requires a period of two or three days when the upper tidal flat remains uncovered by neap tides. For this reason, pioneer zone development in most areas tends to occur between the levels of Mean High Water Neap Tides and Mean High Water (Chapman 1960; Pye and French 1993). However, it is not sufficient to simply establish the minimum period of continuous exposure required for the successful establishment of a species, such a period must also coincide with the time of germination. The minimum required exposure period for seedlings of some species is therefore likely to fluctuate somewhat from year to year, depending upon the chances of a neap tide coinciding with germination (Chapman, 1960). Once *Salicornia europaea* has become established, frequent submergence by the tide does not inhibit its growth (Wiehe, 1935).

6.2.4 Requirements for the Establishment of Other Pioneer Plant Species

A study by Brereton (1971) in Foryd Bay, Caervarnonshire concluded that surface plasticity, or the movement of tidal water over sediments, is probably the dominant feature affecting the establishment and survival of both *Salicornia* spp. and *Puccinellia maritima*. Comparable information for species other than *Salicornia* spp. and *P. maritima* is not readily available from saltmarsh habitats. However, germination experiments by Chapman (1960) for *Aster tripolium,* a species of the lower to middle marsh, have shown that normally five days must elapse prior to tidal inundation before a seedling is adequately rooted.

Recent research (NRA, 1995) has shown that 70% to 96% of the total variation in the upper and lower limits of many dominant saltmarsh species can be explained by a simple linear regression equation. As an example, the upper and lower limits of *Puccinellia maritima* are given as:

- Upper limit (m) = 0.12 + 1.80 MHNW
- Lower limit (m) = 0.23 + 1.39 MHNW

The review notes that the residual variation that cannot be explained by the level of the tide is likely to be explained from partial correlations such as fetch and the spring tidal range (NRA, 1995).

6.2.5 Requirements for the Establishment of *Spartina anglica*

Ranwell (1963) considered the establishment of *Spartina anglica* in different zones of an intertidal area at Bridgewater Bay, Somerset. He concluded that seedlings germinating from seed were very rare at the extreme seaward edge, although became more frequent further away from the influence of the tide. More important for *S. anglica* establishment in the lower reaches of the marsh were the *Spartina* plant fragments, which can rapidly take root, that were often found in the area after storms.

Successful seaward establishment by *Spartina anglica* at Poole Harbour, Dorset occurred at approximately Ordnance Datum, with the landward limit restricted to between 0.5m and 0.9m above OD. This lower limit has been related to a limit in the duration of tidal submergence of 6 hours that *S. anglica* can tolerate, although it has been shown to withstand a period of 9 hours submergence at equinoctial spring tides (Hubbard, 1965). The establishment of *S. anglica* may also be limited by wave action in exposed sites (Hubbard, 1965). In addition, on more established sediments, seed or plant fragments may become entangled in *Zostera* spp. and algal growth (Hubbard, 1965).

Studies have also been undertaken that have attempted to establish equations to explain the variation in elevational limits in *S. anglica* and a variety of physical parameters. This showed that whilst variation in tidal range was able to explain 86% to 89% of the variation in limits of this species, the addition of other variables did help to improve the prediction. In particular, the spring tidal range, the fetch in the direction of the transects taken, and estuary area were all significant to a lesser degree in predicting the variation of elevational limits at which *S. anglica* occurred (Gray, 1992).

6.3 COLONISATION RATES AND SALTMARSH DEVELOPMENT

Although the majority of work undertaken for colonisation of new marshes by plant species concentrates on *Salicornia* spp., a study by Brereton, (1971) at Foryd Bay, Caervarnonshire also considers *Puccinellia maritima*. He showed that in the initial stages of succession the distribution of both *Salicornia europaea* and *P. maritima* was random, although both species formed a more aggregated distribution with time. Brereton demonstrated that after establishment of both species, patterns of colonisation reflected the greater relative tolerance of *P. maritima* to water-logged conditions than *S. europaea*, and the greater tolerance of *S. europaea* to more saline conditions.

6.3.1 Colonisation of *Salicornia* spp. and Other Pioneer Plant Species

The importance of *Salicornia* spp. as a pioneer species of the tidal flats is shown by an intertidal
habitat creation scheme in the Netherlands (Joenje, 1979). The construction of recent embankments within the Lauwerszeepoldeer has created tidal flats, channels and sands of little or no agricultural value. The character and rates of plant colonisation occurring on these areas has been closely monitored. Colonisation of the tidal flats began with a sparse halophytic vegetation mainly consisting of species of Chenopodiaceae, notably *Salicornia dolichostachya*, *S. europaea*, *Suaeda maritima* and *Atriplex hastata*. In the first year, plant densities ranged from 0 to 300 per ha. This variability in density was considered to reflect two factors; the silt content of the sediments and the distance of the site from nearby saltmarshes. *Salicornia* spp. were found to reach highest densities on silty sand whilst *Suaeda maritima*, *A. hastata* and *Spartina* became established in sandy areas.

In the initial year of study, *Salicornia europaea* colonisation was about twice that of *Salicornia dolichostachya*. In the following years, plants of *S. europaea* were the more abundant and persisted in the more silty areas. In more sandy habitats, although *S. dolichostachya* was initially more successful, it was replaced by *S. europaea* within a few years (Joenje, 1979). Joenje concluded that salt rising with the capillary water in the silty habitats was and still is the dominating environmental factor that markedly affects the establishment of new species.

6.3.2 Colonisation Rates of *Spartina anglica*

Although the spread of *Spartina* has attracted much interest, there are few estuaries for which there are reliable data on its rate of spread (Goss-Custard and Moser, 1988). A study by Hubbard (1965) concerning the patterns of invasion of *Spartina anglica* in Poole Harbour concluded that although the initial establishment probably occurred as a result of plant fragments, the rate of spread of this species suggested that colonisation occurred as a result of seedling production. *S. anglica* establishment and colonisation in Poole Harbour was most successful on the mudflats of sheltered inlets and bays, and was less successful on sandy shores (Hubbard, 1965).

6.3.3 Other Factors Likely to Affect Colonisation of Saltmarsh Species

In a comparison of man made *Spartina* marshes with naturally occurring *Spartina* marshes in the USA, Roberts (1991) found an unexpected difference in the height of the three youngest age classes of *Spartina*. He attributed this to higher levels of nutrients in the created marsh and noted that particularly robust stands of *Spartina* were located at the base of sloping lawns and were probably receiving runoff containing fertiliser. He also suggested that the slow release fertiliser applied when the created marsh was planted may have further contributed.

Anecdotal evidence from the Tollesbury managed retreat scheme, Essex, indicates that the saltmarsh vegetation that is becoming established is influenced by the enhanced nutrient levels of what was formerly, before breach of the seawall, productive agricultural land. This appears to be evident in the high levels of vigour in the newly established vegetation (I. Black, *pers. Comm.* 1996).

Zedler (1996) reflects on the significance of organic matter and cites it as one of the three key factors (with elevation and drainage) that influence saltmarsh productivity. Zedler also mentions that nitrogen is a limiting factor in saltmarshes and that low nitrogen levels tend to be characteristic of constructed wetlands along the coast.

6.3.4 The Development of Saltmarsh Communities in Created and Restored Intertidal Habitats

Documentation of the development of saltmarsh communities and the subsequent outcome (success or failure) of intertidal habitat creation schemes with respect to vegetation colonisation relates almost entirely to the creation of *Spartina* marshes in the USA. Consideration of the success of vegetation colonisation alone is somewhat artificial, in that success of the scheme is more usually judged in terms of the habitat value of the site. However, where colonisation success has been judged, the parameters used for assessment are most typically stem density and plant height.

Roberts (1991) compares a number of created *Spartina* marshes with naturally occurring *Spartina* marshes. He concludes that where fully covered with *Spartina*, it is not possible to visually distinguish naturally occurring marshes from created marshes. However, differences in stem density and plant height were found, with both parameters being greater on created marshes than in natural marshes, although both parameters were highly variable.

This variability is further reflected in other studies. Havens *et al*, 1995 found no difference between the stem densities of created and natural *Spartina* marshes whilst Webb and Newling (1985) found a significantly lower stem density for created marshes. Whilst analysis of vegetation characteristics can quantify structural differences between sites, the success of a habitat creation scheme relies on the relation of such physical differences to habitat value. Roberts (1991) suggests that vegetation density and structure is probably one of the main factors influencing site use by many marsh dwellers and concludes that if properly planned, constructed and maintained, man-made *Spartina* marshes have a habitat value for animal species similar to that of naturally occurring *Spartina* marshes.

Whilst success of vegetation in habitat creation schemes usually concentrates on the success of planted species, successful establishment and colonisation of plant species can also be achieved through reliance on the natural influx of local propagules. It has been stated that usually too much emphasis is placed on the revegetation aspects of the detailed plan. If the hydrology and other physical aspects are conducive to colonisation of a particular area, target plant species are likely to establish naturally given a nearby source of propagules. The majority of intertidal recharge and managed retreat schemes undertaken in certain UK estuaries, for example in Essex and Suffolk, rely on the natural colonisation and development of saltmarsh species, with apparent success (M. Dixon, *pers. comm.* 1996).

The above phenomenon is clearly illustrated by the Salmon River Estuary Project, Oregon where no transplanting was undertaken for a scheme that restored reclaimed agricultural land to saltmarsh. Here, naturally occurring and dispersed local propagules were readily available, and a full array of native saltmarsh species became established (Frenkel and Morlan, 1991). The time scale for this occurrence is not known. The success of this project was attributed to readily available natural propagules and hydrological conditions adequate for natural saltmarsh development. Planting is not always necessary. Frenkel and Morlan (1991) concluded from their study that with an adequate local seed source and the correct hydrological conditions a full array of native saltmarsh species can be established. Natural colonisation of vegetation has also been shown in certain cases to be more productive than planted sites (Shisler and Charette, 1984; cited in Kusler and Kentula, 1990).

6.4 PRACTICAL CONSIDERATIONS FOR ESTABLISHING SALTMARSH VEGETATION

Prediction of the probable species likely to colonise a newly created intertidal area, and the level at which they are likely to establish, can only be attempted when a number of physical characters or parameters are known for the site (Frenkel and Morlan, 1991). Knowledge of these parameters and their relation to the establishment of specific species can be used to the advantage of habitat creation schemes in two main ways:

- Where it is hoped to encourage particular plant species or communities, a scheme can be designed so as to provide the most appropriate physical conditions for successful establishment of the desired species; and
- Where there is an active requirement that a particular species will not colonise newly created areas, a scheme can be designed so that the only substrate provided is hostile to successful establishment of the species in question.

The parameters identified by this literature review as of importance to pioneer plant establishment in a saltmarsh habitat are:

- Elevation;
- Exposure of intertidal area.

6.4.1 Elevation

Elevation is critical in determining saltmarsh diversity and composition (Webb and Newling, 1985) and is possibly the single most important overall factor in wetland creation (Garbisch, 1986). Saltmarsh vegetation can establish itself when mudflats accrete to a level around Mean High Water Neaps and has an upper limit of around Mean High Water Springs (Legget and Dixon, 1994). This is because the height of the created intertidal area relative to sea level determines a number of physical conditions important to plant establishment, and includes:

- Interval between tidal inundations - pioneer plant species have a critical length of time for germination of seeds and rooting of seedlings. Tidal inundation before successful rooting has occurred results in the loss of seedlings to the sea (Brereton, 1971).

- Duration of submergence - halophytic plants vary in their tolerance to submergence by the tide. Submergence results in a decreased gaseous exchange and reduction in photosynthetic capacity (Brereton, 1971).

- The frequency of submergence by the sea affects the salinity of the habitat, to which halophytic plant species have different tolerances. In areas of frequent inundation, where salt concentration is high, species vary in their ability to establish according to the extent of their physiological adaptation to saline conditions (Brereton, 1971; Joene, 1979).

The portion of the intertidal zone that is suitable for plant establishment is dependent on the plant species selected, local tidal conditions and regional trends (Knutson *et al*, 1990). It is therefore necessary for elevation requirements of marsh vegetation to be established locally. For example, in the USA *Spartina* occupies a much greater range within the intertidal area than in the UK (A. Gray, *pers. comm*. 1996), and there are therefore significantly more niches for *Spartina* in the USA than in the UK. PERL (1990) found that changes in the topography of 10cm substantially altered the vegetation composition, and Zedler (1984) recommended that the marsh surface should be contoured to provide a range of elevations which allow for diversity of plant establishment.

6.4.2 Exposure of Intertidal Area

The establishment and colonisation of saltmarsh vegetation will vary according to the degree of exposure of newly created habitat to both wave action and effects of the weather. An exposed area will suffer greater erosive processes and this will directly affect pioneer vegetation in the following ways:

- Establishment of a given species is likely to occur at a higher point above sea level for an exposed intertidal habitat than an equivalent habitat more sheltered from weather effects;

- A more exposed area will suffer greater levels of erosive action from weather influenced wave action at lower levels on its intertidal profile than a comparable area in a more sheltered location;

- Given that species require a critical length of time free of erosive processes for successful establishment (Chapman, 1960), an exposed shore must necessarily provide these conditions at a higher level than a more sheltered shore;

- For intertidal areas of similar aspect and shelter from the weather, exposure of the substratum to erosive processes may vary according to differences in the steepness of the intertidal profile. Tidal scour, an erosive process affecting intertidal areas, is as a function of water velocity and varies with the steepness of the intertidal profile;

- Steeper intertidal slopes result in a more rapid movement of tidal water over the surface sediments, causing increased levels of tidal scour. The erosive power of tidal scour is therefore directly related to the steepness of the intertidal profile;

- The exposure of a site may be measured in terms of fetch, whereby areas exposed to larger stretches of open water will experience larger wind-induced waves which is reported to be one of the most important factors in determining sites which are suitable for saltmarsh creation. It has been suggested that in the UK the critical length of fetch for colonisation of saltmarsh is 2km;

- As an erosive process, tidal scour will affect the height above OD at which establishment of plant species occurs in a similar way to exposure to weather. Pioneer plant species able to establish at a given level on a gradually sloping intertidal area will be forced to locate further up the profile of another, steeper intertidal area in order to find equivalent conditions suitable for establishment; and

- Once established, exposed plant communities are more likely to suffer setback from storms and other factors contributing to the force of coastal erosive process, than those established on more sheltered shores.

6.4.3 Substratum Type

The particle size of sediments used to create an area of intertidal habitat has been shown to significantly effect both the species composition of colonising vegetation and the time taken for establishment of the first pioneer species (Joenje, 1979). This can be attributed to two reasons:

- Sediments vary in the length of time they require for stabilisation and consolidation. Prior to consolidation, surface stability is likely to be insufficient for successful establishment of pioneer plant species; and

- Salt rising in the capillary water of more silty habitats is likely to affect species colonisation according to their physiological tolerance of salt (Joenje, 1979).

6.4.4 Local Variations within Substrate of Created Habitat

Variations in the distribution of a colonising species along the horizontal axis of the created intertidal may reflect local influences in the substratum. Typically, this may reflect:

- Waterlogging - some pioneer plant species are more tolerant to waterlogging than others and their distribution would reflect locally saturated areas (Brereton, 1971);

- Salinity - as explained above, the tolerance of halophytic plant species to salt varies (Brereton, 1971). Whilst this is generally reflected in zonation of plant species relative to the frequency of inundation, it may also be important in schemes where a freshwater influence is intended; and

- Nutrient levels - where variation exists in the nutrient levels of soils, they are unlikely to affect rates of colonisation in the first instance, however, once established variation is likely to be seen in the size and density of plant species.

7. MICRO-ORGANISMS AND PLANKTON

7.1 INTRODUCTION

Despite the extensive search of literature for this review, no reference was found to habitat creation schemes with specific design considerations for the requirements of estuarine microflora and fauna and no monitoring schemes recorded the long-term development or recovery of these communities post-construction. In the absence of such information, the following sections will provide a brief discussion of:

- The ecological importance of plankton and micro-organisms; and
- The environmental factors that influence the distribution and density of micro-organisms in intertidal sediments and estuarine waters.

This information is taken from literature documenting ecological studies in estuaries in the UK. These reports provide some insight into the requirements of estuarine micro-organisms, inhabiting intertidal areas. The following sections have been presented in such a way as to highlight the ecological importance of plankton and micro-organisms for benthic intertidal communities and hence their role in the success of habitat creation and restoration schemes.

7.1.1 Plankton

Plankton are organisms that live suspended in seawater and are generally incapable of moving against water currents and can be defined as either phytoplankton or zooplankton. Phytoplankton are the photosynthetic plants within the plankton whereas zooplankton constitute the animal component. Two groups can be identified within the zooplankton; those species that are permanent members of the zooplankton community, such as the calanoid copepods *Acartia* and *Eurytemora* (known as the holoplankton) and the temporary members such as the larvae of benthic species that are present in the zooplankton during their dispersal phase (or meroplankton) (McLusky, 1989).

7.1.2 Ecological Importance of Plankton

Although phytoplankton is an important component of the estuarine ecosystem, its role is not as dominant as in the marine ecosystem. The main reason for this is that although nutrients are essential for the development of phytoplankton populations present in estuaries, there are other factors limiting phytoplankton growth. Of most importance in the estuarine ecosystem is the relatively high levels of turbidity that are characteristic of temperate estuaries. In addition, the growth rate of phytoplankton may be less than the flushing time of the estuary (McLusky, 1989). However, the importance of phytoplankton is discussed below in relation to its contribution to the productivity of benthic communities and hence the development of habitat creation and enhancement schemes.

Zooplankton populations are similarly less important components of the food chain in estuaries when compared with the marine environment. This is partly due to the fact that phytoplankton populations (one of their main food sources) are relatively low in estuaries in comparison with the marine environment and partly because strong tidal currents and water exchange continuously carries zooplankton out of estuaries. However, some species of zooplankton are believed to have evolved mechanisms to enable them to retain their position in estuaries through vertical migration in the water column in relation to the direction of tidal flow at different stages of the tidal cycle. The relationship between phytoplankton and zooplankton in contributing to benthic productivity is discussed in this section.

The main contribution of plankton to benthic community productivity is through the production and sedimentation of detritus. Phytoplankton are either consumed by zooplankton and egested as faecal pellets or they die; in both cases they sink to the bed of the estuary as detritus. The contribution of detritus to the productivity of the benthic ecosystem is discussed under Section 7.1.4 below.

7.1.3 Micro-organisms

A wide diversity of micro-organisms are present within the estuarine ecosystem, both in the water column and on and within the sediment. Micro-organisms fulfil important functions within estuaries; of particular relevance to this section is their role in the productivity of intertidal areas and the overlying water column. It is in this respect that they contribute towards the development and functioning of intertidal mudflat and saltmarsh communities. Although habitat creation and restoration schemes are not specifically designed to create habitat for micro-organisms, the role of micro-organisms is crucial to the development of productive intertidal communities. For this reason, this section discusses the range of micro-organisms present within the estuarine system and concentrates on those that play an important part in the functioning of intertidal areas.

The term ‘micro-organism’ is a generic one that encompasses a wide diversity of bacteria, plants and animals. The following micro-organisms are of relevance in contributing towards the productivity of habitat creation or restoration schemes:

- Bacteria;
- Ciliates; and
- Benthic microalgae.

For the purposes of this review, it is not necessary to discuss the role of each of the above groups specifically; it is more informative to consider the importance of micro-organisms as a whole. However, where necessary, reference will be made to specific groups of organisms where they perform a unique function in the system.

7.1.4 Ecological Importance of Micro-organisms

Within the intertidal environment, micro-organisms perform an important role in that they constitute the basis of the food chain for deposit feeding organisms. Deposit feeders ingest sediment and detritus from which they derive their nutrition. However, the ability of deposit feeders to assimilate this material is low and their main source of nutrition is from the bacteria, which they can readily digest, that are adsorbed onto the surface of the detritus. This method of feeding is known as the microbial stripping hypothesis (Levinton, 1995). Therefore, detritus is of little nutritional use to deposit feeders unless it is decomposed by microbes and converted into the more readily digestible microbial tissue. Micro-organisms therefore form a major component of the basis of the food chain for deposit feeding organisms that are often an important component of the benthic invertebrate community.

The ecological importance of bacteria in terms of providing a food resource for higher predators varies depending on the nature of the environment and the relative amounts of various types of vegetation that are present. On sandy intertidal areas, the amount of vegetation is low and micro-organisms constitute the majority of the organic matter required by deposit feeders for nutrition. In contrast, mudflats that are in the vicinity of saltmarshes will contain a mixture of forms of particulate organic matter such as seaweeds, fragments of decaying saltmarsh vegetation combined with bacteria, diatoms and other micro-organisms. Here, the relative importance of bacteria in terms of a source of nutrition for deposit feeders is likely to be lower than for sandflats because, although the ability to assimilate detritus directly is low, its high abundance means that significant quantities may be ingested. Furthermore, some components of the detritus, for example seaweeds, are more digestible than others, such as saltmarsh vegetation.

In shallow water and on intertidal areas, diatoms are an important food source for surface feeders. Diatoms are microscopic single celled plants encased in a siliceous structure. Diatoms are able to migrate within the upper layers of sediment, tending to move down into the sediment when intertidal areas are covered with water to prevent them from being washed away and moving up to the sediment surface when the mud is uncovered in order to photosynthesise. Under certain conditions, a golden brown film of diatoms can often be seen on the surface of intertidal mudflats. Whether they are present within the sediment, adsorbed onto the surface of sediment particles or present on the surface of the mud, diatoms can constitute an important feeding resource for benthic invertebrate communities.

In addition to providing a feeding resource, mats of diatoms on the surface of the sediment act to stabilise the sediment. In certain situations, where diatom density reaches sufficiently high levels, this can reduce the rate of erosion of intertidal muds, although the extent to which this occurs will depend on the hydrodynamic conditions.

Diatoms and other benthic microalgae also play an important role in developing and maintaining an oxygenated zone on the surface of intertidal estuarine sediments. Along with tidal action which supplies a source of oxygen from the water column to intertidal sediments, benthic microalgae may be the main source of oxygen for the sediment surface through the process of photosynthesis (McLusky, 1989).

7.2 HABITAT PREFERENCE AND REQUIREMENTS

In an investigation of the distribution of micro-organisms in intertidal mudflats of the Stour Estuary, Jackson (1985) indicated that there was a relationship between the viable population of microbes and elevation, as numbers of viable microbes decreased with distance downshore. Nutrient rich sediments at the landward edge of the saltmarsh were found to hold larger more viable populations of microbes than non-enriched areas (Jackson, 1985).

7.2.1 Exposure and Sediments

Barnes *et al* (1976) reported that variations in the small amounts of fine particulate matter and exposure to wave action exerts a profound influence on the micro ecology of medium-coarse sands in three different stages of saltmarsh development. For example, ciliate protozoans were most abundant and diverse in the most sheltered and organically rich sediments of upper marsh creek systems whereas nematodes were most diverse in drainage channel sediments adjacent to pioneer marsh (*Salicornia* spp.).

7.2.2 Light

The amount of light reaching the surface of intertidal mudflats is likely to be an important factor controlling the development of benthic microalgal populations. In the estuarine environment, the natural high turbidity of the water column, in particular over intertidal mudflats when they are covered, is likely to have a limiting effect on the growth rate of benthic microalgae by reducing the penetration of light through the water column. However, many microalgal species move down into the sediment during periods when the mudflats are covered in order to avoid being washed away and therefore will not photosynthesise during these periods. When the intertidal areas are uncovered, they move onto the surface of the mudflat to photosynthesise and during these period turbidity of the water column will obviously have no limiting effect.

Water column turbidity can also limit the growth of phytoplankton populations for the same reasons as for benthic microalgae. However, the relatively high flushing of phytoplankton from estuaries is likely to be of more importance in limiting the development of significant populations in the estuarine environment.

8. BENTHOS

8.1 INTRODUCTION

The benthos plays an essential role in the food webs of estuarine and coastal ecosystems. Benthic communities of saltmarsh are important in the decomposition of dead vegetation and recycling carbon and nutrients, providing food to invertebrates in the intertidal mudflats and creeks. Invertebrate animals of the intertidal flats, such as the polychaete worms and molluscs, also have a significant role in sedimentary processes, by stabilising deposited sediments (Legget and Dixon, 1994). The invertebrates and algae of the intertidal flats provide a major food source for a number of predators, including birds, crabs and fish.

Relatively few studies have investigated the natural development of created coastal ecosystems in great detail. A number of recent investigations have focused on the rates of colonisation and accumulation of benthic invertebrate communities in the sediments of created tidal creeks and saltmarsh channels (Havens *et al*, 1995; Zedler, 1996). Various schemes have gauged the long-term success of created intertidal habitat by comparing the development of the benthic communities at created sites with "control" or reference areas in the immediate vicinity (Webb and Newling, 1985; Roberts, 1991). However, it is not always possible to locate local control areas and in these cases the best representative ecosystems in the region can be used (Pratt, 1994).

In the absence of direct evidence of natural rates of colonisation of invertebrate communities in created or restored intertidal habitats, it is informative to investigate the factors determining the distribution, abundance and diversity of estuarine invertebrates in natural coastal habitats and in situations similar to the disturbed conditions of newly created habitat. Such disturbed conditions are found at dredged material disposal sites or in dredged channels.

8.2 HABITAT PREFERENCE AND REQUIREMENTS

8.2.1 Zonation

The distribution, or zonation, of invertebrates in the intertidal zone is influenced by a wide variety of ecological factors, such as feeding requirements, physiological stress, predation and disturbance which are influenced by physical factors, such as tidal inundation, salinity, sediment composition and structure, exposure, wave action and elevation. A number of these factors are discussed in the following sub-sections.

8.2.2 Tidal Inundation

The distribution of invertebrate fauna that inhabit intertidal flats is largely controlled by the tolerance of the various species to the physiological stresses caused by exposure to air (Peterson, 1991). Benthic organisms that inhabit higher levels in the intertidal zone experience longer periods of exposure to air and tend to burrow down into the sediments to avoid stresses such as desiccation, temperature fluctuations, osmotic shock and UV radiation, and subsequently are subject to reduced predation pressures. Alternatively, tube-dwelling (amphipods and worms) and shell-bearing (mollusc and gastropods) invertebrates are more tolerant to stress which enables them to inhabit sediment at higher elevations. Species with lower tolerance to the relatively high variability in environmental parameters that are characteristic of higher levels on the intertidal zone, such as sea anemones and free living polychaete worms, tend to be found at lower levels on the shore where conditions are less variable.

8.2.3 Salinity

The daily variations in salinity of estuarine water affects the abundance and diversity of the species within intertidal habitats and tidal channels. Most estuarine invertebrates are species which thrive under marine conditions and their abundance generally decreases with extended periods of salinity of less than one-third that of seawater (approximately 10psu) (Zedler, 1996).

However, there are many species that are specialised for the conditions that prevail in the estuarine environment and are therefore characteristic of intertidal areas in estuaries. Examples of these species include the ragworm *Nereis (Hediste) diversicolor* and the burrowing mud shrimp *Corophium volutator*. These species can thrive under variable and low salinity conditions accounts state that *N. diversicolor* is absent from the seaward end of estuaries (Anderson, 1990).

Given that invertebrate species differ in their tolerance to salinity, consideration should be made to the salinity content of the new substrate at the creation site.

8.2.4 Influence of Sediments on Invertebrate Communities

Investigations of the intertidal invertebrates in the Wash (Goss-Custard and Yates, 1992) and Morecambe Bay (Anderson, 1990) demonstrated that typical invertebrate species vary according to two key environmental factors; elevation of the shore level and sediment particle size.

Results from a wide range of investigations of the benthic communities inhabiting mudflats have shown that the fine sediments are typically dominated by mud snails *Hydrobia ulvae* and the burrrowing amphipod *Corophium volutator* which are often present in very high densities. Polychaete worms are also abundant in intertidal flats, particularly the ragworm *Nereis (Hediste) diversicolor* and in sandier substrata the lugworm *Arenicola marina* often dominates.

Wave action on soft-sediment shores often creates a gradient in sediment size from fine sediments on the lower shore to coarser sediments further upshore.

The preferences of intertidal animals for particular sediment types is reflected in their feeding and behavioural requirements. Two broad groups of invertebrates can be identified on the basis of the method of feeding. Firstly, suspension feeding organisms, for example the mussel *Mytilus edulis*, feed by filtering fine particulate matter from the water column. As their feeding apparatus would become blocked by high concentrations of suspended material, they tend to occur in areas characterised by coarser sediments where resuspension of sediment is likely to be less than in areas of finer sediments. In contrast, deposit feeders, for example the lugworm *Arenicola marina*, ingest large quantities of mud and digest any organic material, expelling the mud as a cast. Deposit feeders tend to inhabit finer sediment where there is a relatively large amount of organic matter in comparison to areas of coarser sediment.

In a survey of the benthic communities of the intertidal sandflats of Morecambe Bay, Anderson (1990) demonstrated that sediments with a silt content of 30% or more provided the most favourable conditions for the development of abundant estuarine fauna.

8.2.5 Vegetation

A number of ecological studies have revealed that the distribution and diversity of benthic fauna in intertidal areas is directly related to the plant communities present (e.g. Jackson, 1985). Saltmarsh plants may influence macroinvertebrates in several ways, by providing them with substrate, food, protection from predation and by influencing the temperature, humidity and light intensity at the sediment surface.

Benthic surveys of the intertidal flats of the Wadden Sea illustrate the change in invertebrate community structure and function within different levels of saltmarsh (Toft and Maddrell, 1995). Diversity of species increases with development of the saltmarsh, with there being more plant eaters and detritus feeding invertebrates in the mid-upper marshes which are associated with the digestion and decomposition of vegetation. As discussed in Section 6, colonisation of intertidal areas by vegetation results in changes in the sediment characteristics, including sediment structure and stability, topography, climate, organic content and food availability and subsequently a distinct change in benthic communities can be observed across the intertidal flat/pioneer saltmarsh interface.

8.2.6 Invertebrate Communities of Algal Mats

The distribution of green algae can have a pronounced affect on benthic communities inhabiting mud and sandflats. Nicholls *et al*, (1981) observed that the presence of dense algal mats on the intertidal flats of Langstone Harbour, Hampshire reduced biomass and diversity of mud-dwelling invertebrates, including ragworm and lugworm, but increased the number and biomass of epibenthic animals and worm species adapted to anaerobic conditions. The spread of these algal mats over the Harbour's intertidal mudflats in response to elevated nutrient levels has reduced the feeding area available for certain estuarine waders (Tubbs and Tubbs, 1980; Nicholls *et al*, 1981).

8.2.7 Predation

Certain macroinvertebrate species require the cover afforded by dense saltmarsh vegetation to escape predation from birds and fish, such as the shore crab *Carcinus maenas* and the rough periwinkle *Littorina saxatilis* (Toft and Maddrell, 1995).

8.3 RECOLONISATION RATES AND COMMUNITY DEVELOPMENT

8.3.1 Invertebrate Recolonisation of Created Habitats

Little is known of the rates of recolonisation of newly created mudflats and the succession of colonising species is rarely monitored in detail. Experiments reported in Davidson and Evans, 1987 at Seal Sands mudflats in the Tees Estuary indicated that rates of recolonisation for some invertebrates may be slow even when there are adjacent communities which can repopulate the area. It was estimated that it takes between 2 and 3 years for invertebrates in northern UK estuaries to colonise intertidal areas and grow to sufficient densities to feed wading birds (Davidson and Evans, 1987).

Monitoring programmes in a constructed tidal marsh and two neighbouring natural marshes in Sarah's Creek, Oregon, USA revealed that colonisation of excavated tidal creeks (1m in depth) and adjacent *Spartina* marsh by benthic communities was relatively fast (Havens *et al*, 1995). Evidence suggested that within 5 years post creation there was little differences in the infaunal community structures of the constructed intertidal habitat and the natural sites.

Ray *et al* (1994) observed that polychaete worms, particularly *Capitella* spp., *Pygospio elegans* and *Polydora quadrilobata* were the first to colonise created mudflats constructed of dredged silty materials in Sheep Island, USA. A year later the polychaete *Exogene hebes* and the amphipod *Corophium volutator* dominated benthic communities. Softshell clams and sandworms recolonised in these constructed mudflats in substantial numbers and were present in greater abundance than reference natural mudflats four years post-construction, which reflects their preference for finer sediments.

Ray *et al* (1994) monitored the development of benthic communities in intertidal mudflat in Sheep Island near Jonesport, Maine. Data from monitoring programmes suggested that within 2 years post construction Sheep Island supported a diverse infaunal community, however the number of species and individuals present was approximately half those of a nearby reference natural marsh. Over the following 2 years differences in species richness and abundance between the constructed and natural mudflats decreased and until by four-years post-construction the difference in abundance was negligible.

Not all created habitats are successful in providing suitable conditions for invertebrate colonisation and there is some evidence to suggest that many constructed tidal wetlands support lower abundances of invertebrates or vastly different species of invertebrates than natural wetlands. In a study of benthic communities of six constructed saltmarshes in North Carolina (ranging between 1 and 17 years after construction), Sacco (1989) found that the communities at constructed sites had similar species composition and structure (deposit feeders, suspension feeders and carnivores), but animal densities were consistently lower. Similar observations have been made in created *Spartina* marshes in San Diego Bay after four years (Zedler, 1996).

The lower densities of animals recorded at the three year old Bolivar Point created marsh were attributed to differences in sediment characteristics from the natural marsh soils which are finer organic soils and the high utilisation of the new site by predators. These characteristics are typical of young marshes and faunal communities at the created marsh were expected to become increasingly similar to neighbouring natural marshes as the marsh matures and sediments increase in organic matter and fine materials (Webb and Newling, 1985). Subtle differences between faunal communities inhabiting a constructed mudflat at Beals Island in the mid-1960s and nearby natural intertidal flats were attributed to differences in sediment and bait digging activities (Ray *et al*, 1994).

8.3.2 The Recolonisation and Recovery of Disturbed Habitats

Studies of the recovery of invertebrate communities after dredging operations provide a number of parallels with the recovery of communities following a newly created or restored habitat. A report on the effects of the extraction of marine sediments on fisheries by the International Council for the Exploration of the Sea (ICES, 1992) states that the recovery of these disturbed habitats ultimately depends upon the characteristics discussed in the following subsections.

The nature of the new sediment at the extraction site
The characteristics of the new sediment surfaces that are exposed or subsequently accumulate at the excavation site will strongly influence the structure and composition of the colonising benthic communities. Exposure or accumulation of a substrate similar to that which originally existed at the site promotes recolonisation by a similar invertebrate communities to those originally present.

The complete recovery of animal communities following a dredging event in soft sediment environments may take any period of time from 1 month to more than 15 years, depending upon the nature of the sediment. For example, Van der Veer *et al* (1985) recorded that the recovery of benthic invertebrates in a channel in the estuarine Dutch Wadden Sea occurred within 1 year of the removal of sediments from this highly mobile sand environment. This contrasts greatly with recolonisation rates in tidal flat habitats where rates of sediment transport were reduced and dredged areas were not completely infilled and recolonised after eleven years (ICES, 1992). Studies undertaken by Stickney and Perlmutter (1975) in the Georgia Estuary system, USA, suggest that dredging has only short term effects on the benthos of the silt and clay sediments and that although almost complete removal of organisms occurs during the excavation of sediments, recovery begins within 1 month. Within 2 months the diversity and species composition were reported to be similar to pre-dredge conditions and there was little change in the sediment composition.

The sources and types of colonising species

Patterns of invertebrate recolonisation after dredging operations have been likened to those which occur after the cessation of organic pollution or after severe storm events (ICES, 1992). With the exception of certain invertebrate species, such as deep burrowing animals and mobile epifaunal species, which may survive a dredging event through avoidance, dredging may initially result in the complete defaunation of the excavation site. Mobile epibenthos are the first adult species available for colonisation of the newly exposed sediments inhabiting surrounding undisturbed sediments. Whereas recruitment of adult invertebrates through the sediments is highly constrained by the proximity of undisturbed communities to the excavation site, recolonisation by invertebrate larvae are not as limited.

More rapid recovery has been observed in areas that are exposed to periodic disturbances, such as maintenance dredging activities, where the sediments become dominated by opportunistic colonisers, typically polychaete worm species. A study of benthic recolonisation in regularly dredged channels indicates that invertebrate communities are adept at readjustment to disturbance (ICES, 1992).

The nature and magnitude of the disturbance

The recruitment of animals to the newly exposed sediments depends upon the magnitude of the effect of sediment redeposition upon the wider benthic ecosystem of the surrounding area. In turn, this depends upon the nature of the fauna, the deposition rate of the sediment, and the increase in the turbidity of the water column relative to the natural turbidity of the estuary. The relevance of these findings to the recolonisation of newly exposed sediments of a created or recharged intertidal mudflat is clear. Ultimately the rate and extent of recovery of benthic ecosystems will depend upon the scale of the disturbance, whatever the nature of its source, dredging operation, placement of sediment on intertidal saltmarsh and intertidal flats or the creation of intertidal habitat by the excavation of terrestrial habitat.

8.4 PRACTICAL CONSIDERATIONS AND PROBLEMS

8.4.1 Hydrology

Benthic communities have been reported to be adversely affected by drastic fluctuations in tidal levels of greater than 3 feet (approximately 1m) in range (Marble, 1992). Large water level fluctuations can expose spawning areas and generally reduce invertebrate density.

8.4.2 Substrate

As mentioned above, the nature of the substrate greatly influences the structure of the benthic community that is likely to subsequently develop. For example, different species have preferences for different habitat types. Muddy sediment is likely to support a different composition of species in comparison to sandy mud or sand, for example. Furthermore, the species abundance and diversity of the habitat creation scheme will be affected by the nature of the substrate, with muddy sediment, for example, having the potential to support a more diverse invertebrate population than a sandy sediment.

An important consideration in the design of habitat creation schemes is the value of the habitat that will be created, in terms of its ability to support benthic invertebrates and subsequently higher predators, in comparison with the habitat that will be covered by the scheme. For example, an intertidal area that comprises consolidated mud or clay is unlikely to be able to support a diverse benthic community and its value as a feeding area for birds, for example, will be limited. If the habitat creation scheme proposes to create an area of soft mud then this area, over time, is likely to support a more diverse community than was originally present and the area therefore has enhanced potential for feeding birds.

8.4.3 Sediment Placement

Davidson and Evans (1987) stress the importance that intertidal habitats must be constructed in such a way as to ensure the settlement and stabilisation of the finer sediments most suitable for supporting high densities of intertidal invertebrates.

The placement of dredged material on intertidal habitats causes considerable disturbance and potential mortality to the faunal communities that inhabit them. The extent of disturbance to the invertebrates *in situ* depends on the rate of feed of material. According to Toft and Maddrell (1995), communities may survive increases in surface elevation of 1 or 2cm, but may not be able to adjust to changes that result from the single placement of larger amounts.

8.4.4 Water Quality

Low levels of suspended solids (>80mg/l) have a high correlation with aquatic diversity and abundance (Marble, 1992).

9. BIRDS

9.1 BACKGROUND

9.1.1 Introduction

British intertidal areas are of national and international importance for migrant and wintering waterfowl who use the intertidal mud and sand flats to feed, and surrounding terrestrial habitats to roost (Davidson *et al*, 1991). Breeding birds are also an important feature of the conservation interest of intertidal zones, where the mosaic of terrestrial habitats surrounding such areas provides nesting sites for a variety of bird species. The intertidal mudflats are of importance for the rich feeding grounds they provide for both adults and juveniles alike (Davidson *et al*, 1991).

The importance of intertidal mud flats as a resource for feeding birds is an indirect consequence of the high levels of organic matter associated with the sediments from which they are formed. This organic material provides the basis of a substrate that is capable of supporting a large biomass of macrobenthos (see Section 8) which, on exposure of the flats at low tide, becomes vulnerable to predation by wading bird species.

9.1.2 Scope of Discussion

For any habitat creation proposals whose objectives include the provision of bird habitat, close consideration of the habitat preferences of all bird species targeted by the scheme is required. Given the importance of intertidal areas as a food resource for both wintering and breeding bird species alike, the design of intertidal habitat is likely to place a large emphasis on the provision of intertidal mud or sand flats for feeding purposes. Habitat preference of bird species will therefore relate primarily to diet and other aspects of feeding ecology, although nesting requirements for breeding birds in the upper reaches of the saltmarsh should also be considered.

The majority of habitat creation or restoration schemes of intertidal habitat have focused on restoring the upper reaches of the saltmarsh, providing vegetated saltmarsh habitat that has most often been established to reduce or prevent erosive processes. Provision of bird habitat is not therefore one of the original objectives of many schemes and documentation of the bird use of habitats is mostly written up as a secondary occurrence. This is typified by the managed retreat scheme in Tollesbury, Essex, which although did not include specific objectives to create intertidal bird habitat, is extensively used by a variety of wading species (I. Black, *pers. comm.* 1996).

However, where there has been specific loss of bird habitat through either natural or anthropogenic factors, schemes have been instigated to restore or mitigate the loss.

9.2 HABITAT PREFERENCES AND REQUIREMENTS OF FEEDING BIRDS

As part of the intertidal food chain, waders occupy one of the higher trophic levels, preying on benthic invertebrates that feed on sediment associated organic detritus. Wader distribution and densities can therefore be expected to bear a direct relation to the density of their major prey species and this has been demonstrated in a number of studies (Bryant, 1979; Goss-Custard *et al*, 1977a).

Therefore, of great significance to the design and planning of intertidal creation schemes are a number of studies that have characterised the diet of wading birds and considered aspects of their feeding ecology and behaviour around the British coast (Goss-Custard and Durrel, 1990). These studies have typically centred on up to ten species of waders with a large proportion of the total study effort given to four species; redshank, curlew, dunlin and oystercatcher. Of the remaining species, bar-tailed godwit, turnstone and grey plover are also considered. The following discussion reviews the food preferences of these seven waders species typical of intertidal habitats within the UK. The food preferences reviewed below are sourced from a number of studies carried out in a variety of coastal locations.

- **Oystercatcher *Haematopus ostralegus***
  Oystercatchers feed almost entirely on bivalve molluscs (Goss-Custard *et al*, 1977a, 1977b; Bryant, 1979), although diet does vary according to substrate type and location on the intertidal profile. Typically, oystercatcher diet includes mussels *Mytilus edulis*, cockles *Cerastoderma edule* and the Baltic tellin *Macoma balthica*. Diet differences also occur according to geographical location. In the Wash, the cockle is the most important prey for oystercatchers, providing them with most of their food on mud and sandflats (Goss-Custard *et al*, 1977b). However, in Essex and Suffolk, the density of oystercatchers is only correlated with the density of mussels (Goss-Custard *et al*, 1977a).

- **Dunlin *Calidris alpina***
  A study undertaken for the Wash showed that dunlin feed mainly on the middle and upper levels of the intertidal area (Goss-Custard *et al*, 1977a) and closely follow the tidal perimeter (Bryant, 1979). Goss-Custard *et al,* (1977a, b) record the main food items of dunlin as laver spire shell *Hydrobia ulvae,* ragworm *Nereis (Hediste) diversicolor*, *Nephthys* spp. (occasionally) and *Macoma balthica* (Prater, 1981).

- **Grey Plover *Pluvalis squatarola***
  Goss-Custard, Jones and Newbery (1977b) found it difficult to identify the main feeding areas of this species in the Wash because the birds occurred at such low densities. However, they were able to report that grey plovers appeared to favour beds of the polychaete *Lanice conchigela* for feeding.

- **Redshank *Tringa totanus***
  A range of invertebrates are recorded as prey species for redshank which characteristically take food items close to the mud surface (Bryant, 1979). Prater (1981) reports that in general, two crustacean species seem particularly important to redshank; the amphipod *Corophium volulator* and the brown shrimp *Crangon crangon.* However, a study by Goss-Custard, Kay, and Blindell (1977a), which whilst agreeing as to the relative importance of *C. volulator,* also suggests that the polychaete worm *Nereis (Hediste) diversicolor* is of importance as a prey species in south-east England. The diet of redshank is also reported to include crabs *Carcinus maenas,* and small fish taken from creeks and pools at the edge of the sea, especially in the autumn (Goss-Custard *et al,* 1977b; Prater, 1981). Redshank mostly use the mid and upper shore for feeding. Redshank is the only common wader to feed in saltmarsh creek, where sight lines are more restricted (Prater, 1981).

- **Curlew *Numenius arquata***
  Several studies conclude that curlew feed mostly on large polychaetes such as *Nereis (Hediste) diversicolor* and *Arenicola marina* (Goss-Custard *et al,* 1977a, b; Bryant 1979). Curlew are also recorded as taking some large bivalves such as *Scrobicularia plana* and also the polychaete *Cirriformia tentaculata* (Goss-Custard *et al*, 1977a). A study by Goss-Custard, Jones and Newbery (1977b) showed that in the Wash, large numbers of *Lanice conchilega* were taken when exposed at the bottom of the beach. Typically, curlew feed at the tide edge, where prey is most abundant and where they are most likely to encounter a variety of prey types at high densities (Goss-Custard *et al*, 1977b).

- **Turnstone *Arenaria interpres***
  There is little literature documenting the food preferences of turnstone although Goss-Custard, Jones and Newbery (1977b) note that in the Wash, many fed amongst the mussel beds and took a variety of prey, including *Cerastoderma* spp.

- **Brent Goose *Branta bernicla bernicla***
  Prater (1981) records the eelgrass *Zostera* spp. as being of major importance as a food source for the brent goose. They also feed on algae such as *Enteromorpha* and *Ulva* spp. (Tubbs, 1980). At the Tollesbury managed retreat scheme, brent geese have been noted as feeding on the recently established *Enteromorpha* at the breach in the sea wall (I. Black, *pers. com.* 1996).

- **Bar-tailed Godwit *Limosa lapponica***
  A study undertaken in the Wash by Goss-Custard, Jones and Newbery (1977b) found that bar-tailed godwit feed extensively on *Lanice conchilega*. However, when feeding upshore, the majority of their feeding effort was focused on bivalves, mostly *Macoma balthica*. Bar-tailed godwit were also occasionally seen to take large numbers of the polychaete *Scoloplos* spp.

9.2.1 Summary of Wader Feeding Preferences

The literature review suggests that certain waders such as oystercatcher, dunlin and the bar-tailed godwit have a more restricted diet than other species of wader such as redshank, curlew and grey plover. However, from the review it is possible to conclude that certain macrobenthic invertebrates, summarised in the following table, are of widespread importance to wading bird species as a whole.

Table 9.1 Macrobenthic invertebrates of widespread importance to wading bird species

Invertebrate Prey	Wading Bird Predators
Mytilus edulis	Oystercatcher
Macoma balthica	Oystercatcher, Dunlin, Godwit
Cerastoderma edule	Oystercatcher, Turnstone
Scrobicularia plana	Curlew
Hydrobia ulvae	Dunlin
Neiris diversicolor	Dunlin, Redshank, Curlew
Lanice conchigela	Grey Plover, Curlew, Godwit
Arenicola marina	Curlew
Cirriformia tentaculata	Curlew
Corophium volulator	Redshank
Crangon crangon	Redshank

When considering the food preferences of wading bird species in the design of effective proposals for intertidal habitat, it is necessary to consider the validity of extrapolating information collected of diet in a particular estuary to other estuaries in the UK. Bryant (1979) deliberates this question in some depth and concludes that the close correspondence between individual papers concerning diet analyses for wader species across the country suggests that the diets of birds wintering in Britain are uniform within species. He goes on to conclude that although diets are undoubtedly locally affected by availability and abundance of prey, there is sufficient agreement between diet studies for published data to be used as a guide to the potential prey of waders in the Forth, provided the invertebrates concerned are *frequent in the area of study*. For purposes of intertidal habitat creation, this suggests that macrobenthic invertebrate communities colonising newly created mudflats are likely to be used according to documented preferences of wading bird species, although within the constraints of local variation in species diversity.

9.2.2 Implications of Sediment Type for Wader Prey Availability

The relationship between bird numbers and sediment composition is important because prey densities are largely determined by the sediment type (Goss-Custard and Yates, 1992). This has important implications for any habitat creation scheme that aims to provide intertidal feeding habitat for wading bird species. The relationship between invertebrate fauna and sediment characteristics is discussed in detail in

Section 8, however the following discussion provides an account of how this relationship influences the distribution and density of feeding waders.

Goss-Custard and Yates (1992) were able to demonstrate that the main invertebrate prey species of waders using the Wash varied according to two environmental gradients; shore level and sediment particle size. In the Wash, new mudflats are developing down shore of recent reclamations. Whilst shore level and particle size characteristics remain similar between the old and the developing mudflats, shorebird utilisation of the new intertidal areas also remains broadly similar. This was further emphasised by comparisons of wader surveys carried out in the periods 1972-74 and 1985-87 which showed that the main changes of shorebird distribution between the two survey dates could be attributed to change in the distribution of sediments.

In contrast to the relatively sheltered Wash, the exposure of the intertidal flats in the Severn estuary to prevailing gales, along with the large tidal range causes removal of large areas of mud on a fairly frequent basis, along with many of the associated benthic invertebrates. With the loss of prey species, shorebirds are then forced to move to other areas, which although more sheltered, represent habitat of lower value to feeding waders. Goss-Custard and Yates (1992) conclude that the stability of bird distribution is primarily linked to the variations in prey distribution. These in turn, depend upon the stability of the intertidal habitat sediments.

9.2.3 Other Factors Affecting Wader Utilisation of Intertidal Areas

Local variations in use of intertidal habitat by many wading bird species have been linked to the distribution of preferred food items. Goss-Custard *et al*, (1977a) found that differences in the density of redshank, curlew, dunlin and oystercatchers between estuaries located in south-east England, could be linked to differences in density of prey. However, this relationship was not found for knot, suggesting that other factors are also important in explaining patterns of wader utilisation of intertidal areas. Documentation of other factors affecting wader feeding patterns is not widespread, but possible factors include the following:

- **Topography**
  Bryant (1979) notes that topographical features at estuarine sites may have an effect on wader usage and concludes that the intertidal area at MLWS, the ratio of area: length of shore and coverage sequence were all determining factors in the feeding distribution of knot.

- **Disturbance**
  The close proximity of lights, noise and motion may affect wader use of an area and Zedler (1996) notes how difficult it is to buffer such influences from created intertidal habitats.

- **Wader behaviour**
  Feeding densities may also reflect behavioural mechanisms as well as prey density. Hence the lowest feeding densities in relation to *Nereis (Hediste)*

diversicolor are for the largest most demanding species such as curlew, whilst the highest densities are observed for the smaller, less space demanding species such as dunlin (Bryant, 1979).

- **Sediments**
  In a study that develops a strategy for predicting wader densities in the Severn Estuary, it was found that characteristics of the degree of consolidation of sediments influenced the densities of redshank and the particle size of sediments affected the density of bar-tailed godwit (Goss-Custard *et al*, 1991).

9.3 HABITAT PREFERENCES AND REQUIREMENTS OF BREEDING BIRDS

Breeding bird species have very specific habitat requirements for food resources and nest sites and, for purposes of intertidal habitat creation, habitat diversity is likely to be one of the key factors in ensuring that the range of these needs is met (Weller, 1991). Design of intertidal breeding habitat should incorporate the following technical considerations:

- The provision of gradually sloping shorelines to allow access to the water;
- The provision of suitable substrate for nests and young;
- The provision of areas not less than 1 to 3m above the highest water levels to allow nesting without risk of nest wash-out; and
- Breeding habitat should be relatively isolated from human disturbance during the nesting season (PIANC, 1992).

The use of a habitat complex rather than a habitat type to provide the range of factors required by a breeding bird species is illustrated by the restoration of black duck intertidal habitat at Bodkin Island, USA (Landin, 1991). The primary objective of this scheme was to compensate for breeding habitat lost through naturally occurring erosive processes (Landin, 1991). The project involved deposition of clean, sandy dredged material to form a six acre island of varied elevation with distinct nesting and feeding habitats for black duck. The success of this project is not yet known.

The successful design of habitat creation schemes for breeding bird species requires identification of specific habitat preferences. Of the species whose feeding preferences are reviewed earlier in this section, redshank and oystercatchers most frequently use saltmarshes for breeding, although Davidson *et al* (1991) also record infrequent use of intertidal areas by breeding curlew.

Redshank breed at very high densities on the upper reaches of some saltmarshes and in some cases are the most abundant breeding wader (Long and Mason, 1983; Davidson *et al*, 1991). Long and Mason (1983) further observe that redshank have a preference for the slightly higher land that often borders tidal creeks. At Morecambe Bay, redshank prefer higher saltmarsh vegetation approximately 40cm in height for nesting (Greenhalgh, 1969).

Orplands Marsh, Blackwater is one of the few documented managed retreat schemes in the UK which incorporates a specific objective to create a new important high level marsh of value to both overwintering and summer breeding birds. The success of the created habitat for bird use is currently being monitored by English Nature (M. Dixon, *pers. comm.* 1996).

9.4 BIRD USE OF CREATED AND RESTORED HABITAT

In 1982, 16.5 hectares of tidal wetlands in Upper Newport Bay, southern California were restored from silted up salt evaporation ponds and salt-flats of previous importance as bird feeding habitat. The site was excavated to mid-tidal level and now has unrestricted tidal flow over silt and sandy mud sediments. Three years after restoration, the site is extensively used by wintering waders, and especially those that feed on smaller prey (Davidson and Evans, 1987). However, differences exist in the interspecific use of the site by birds and these are attributed to a failure to establish high invertebrate populations.

Restoration has also been achieved on a limited scale at Lindisfarne, Northumberland where loss of intertidal feeding habitat occurred as a result of colonisation of mudflats by *Spartina anglica*. The intertidal flats were cleared with the use of a herbicide. Several wader species readily used the newly-exposed flats after clearance, and fed on these areas in preference to the adjacent open mudflats that had never been colonised by *S. anglica.* This has been attributed to a probable temporary elevation of prey densities following clearance (Davidson and Evans, 1987).

A study in the USA considered the avian utilisation of natural and man made saltmarshes. It concluded that the numbers of some bird species (for example, red-winged blackbirds) in the impounded, artificial marshes were up to five times as great as that in the naturally occurring marshes, although certain species were *primarily* found in the latter (for example, clapper rail and sharp-tailed sparrow) (Burger, 1982). The study considered the saltmarshes in isolation of any habitat creation schemes so that no evaluation of success or failure was made in relation to scheme objectives. However, the study did suggest that conservationists should encourage the maintenance of natural saltmarshes because they are necessary habitats for some species, despite the fact that diversity and avian biomass may be less on natural saltmarsh areas as compared to managed areas.

Because few of the early intertidal habitat creation schemes included objectives for creation of bird habitat, there are now few accounts and evaluations of the relative success of provision of bird habitat in an artificial situation.

9.5 PRACTICAL CONSIDERATIONS FOR ESTABLISHMENT OF INTERTIDAL BIRD HABITAT

Prediction of the bird species likely to use a newly created intertidal area, as either feeding or breeding habitat, will indirectly depend on a number of the physical characteristics of the site. This relates to the position of bird species as consumers within the intertidal food chain; invertebrate species must colonise before birds are able to use mud flats as a food resource, and plant species must colonise before bird species are able to use the habitat to roost or nest. Bird species are therefore best considered as secondary colonisers.

The literature review has identified a number of physical characteristics that are likely to affect eventual bird use of a created habitat. These characteristics are discussed in the following sub-sections.

9.5.1 Elevation

The elevation of intertidal sediments directly influences the establishment and colonisation rates of invertebrate and plant species upon which bird species depend (Goss-Custard and Yates, 1992). The importance of elevation in determining eventual bird use of the created habitat can be summarised as follows:

- Establishment of pioneer plant species occurs at an elevation of Mean High Water Neaps (Chapman, 1960; Pye and French, 1993). Elevation of the created habitat will therefore distinguish those areas of importance as a potential food resource for wading bird species, because they remain unvegetated mudflats, from those areas that will progressively lose their feeding importance as they become increasingly vegetated;

- Elevation of the created habitat will determine the extent of upper saltmarsh, of importance to breeding bird species such as redshank; and

- Elevation within the intertidal profile will affect the invertebrate species able to colonise the area (Anderson, 1972) and therefore the patterns of bird feeding, according to their prey preference.

9.5.2 Sediment Type

Goss-Custard and Yates (1992) have shown that bird feeding patterns are directly related to the distribution and density of their main invertebrate prey species, and that in turn, the distribution of these invertebrate species is directly related to sediment types and distribution.

Sediment type determines the nature of the benthic community that will develop in a particular area. This will affect the bird species using the created habitat, and will favour those species whose main prey item is best able to colonise the sediment type used for the scheme.

9.5.3 Exposure

Exposure of the site to the weather and wave action serves to regulate the rates of colonisation, and elevation at which establishment occurs, of both intertidal invertebrate and plant species. This will affect use of the site by bird species in the following ways:

- An intertidal habitat exposed to the weather is likely to suffer periodic loss of benthic invertebrate populations, reducing its value as a resource for feeding bird species;

- An intertidal habitat more likely to suffer extremes of weather will be of reduced value to breeding bird species. The upper marsh vegetation, of importance as breeding habitat, is likely to suffer periodic damage, reducing its value as cover to nesting birds and nests themselves as the risk of inundation is higher than average; and

- An intertidal habitat exposed to the effects of wave action will suffer a reduced extent of upper saltmarsh and therefore breeding habitat as the elevation at which specific plant species are able to establish is pushed upwards.

9.5.4 Other Considerations

Other more general considerations for compensation of bird habitat lost to development with new, created habitat are listed below (Davidson and Evans, 1987):

- A created or restored intertidal area should aim to provide similar substrate types and invertebrate fauna to the site to be destroyed, otherwise it is unlikely to support the species of waders that will be displaced;

- Full compensation should aim to provide habitat for all birds displaced from the destroyed site. In practice, this means that the new wetland should be at least as large as the destroyed site, unless greater densities of invertebrate foods can be established, and which may allow waders to feed at higher than previous densities without intraspecific interference;

- Compensation should focus on schemes, which provide the same habitat to that which is lost where possible. Provision of alternative habitats will provide for species other than those to be displaced by the proposed development; and

- Consideration should be given to the creation of intertidal habitats in advance of the loss of the are for which they area intended to compensate. This would enable settlement and stabilisation of a sufficient depth of suitable sediments and allow colonisation and growth of sufficient densities of invertebrate animals large enough to provide food for waders.

10. FISH

10.1 INTRODUCTION

The shallow waters provided by estuarine saltmarshes and intertidal flats are important for all coastal fish species at some stage in their life cycle (Marble, 1992). Saltmarsh habitat provides estuarine fish with nutrients and detritus. These substances, along with the invertebrates of intertidal flats and estuarine phytoplankton and algae, are critical to the fishery food chain. Saltmarshes also provide shelter and cover for young fish, such as bass, which spawn in southern estuaries of the UK. There are a number of specialised saltmarsh fish species including the three-spined stickleback, which is associated with the brackish water of the upper marsh (Toft and Maddrell, 1995). Little is known about the relationship between the productivity of saltmarshes and their wider role in estuarine productivity and the provision of food and nursery grounds for fish and shellfish.

10.2 HABITAT PREFERENCE AND REQUIREMENTS

Fish species diversity increases from the freshwater influence at the head of the estuary towards the marine conditions found at the mouth of the estuary. In a review of the status of estuarine fish in the UK, Potts and Swaby (1993) identified 41 species that they considered estuarine in nature, being dependent upon estuaries at some time in their life cycle and showing physiological adaptations towards their estuarine environment. Of the 41 species listed, 21 are migratory, eight anadromous (migrating from the sea into the freshwater parts of rivers to breed) and two, the eel (*Anguilla anguilla*) and flounder (*Platichthys flesus*) are catadromous (migrating out of estuaries to spawn at sea). Maitland (1974) previously recognised only five species of truly estuarine fish in the British Isles that were dependent on estuarine conditions in order to complete their life cycle, and a further 13 species that were thought to occur regularly in estuarine waters. A description of the estuarine species is given in Table 10.1 (derived from Campbell, 1976 and Davidson *et al*, 1991)

The review carried out by Potts and Swaby (1993) covered only 22 estuaries in England. It was found that a number of these estuaries had not been subject to detailed examination, therefore where low species diversity was found it was not clear whether this resulted from a low intensity of survey work or from environmental conditions.

The number of species found in an estuary depends upon many factors including:

- Habitat diversity;
- Estuary size and shape;
- Structural complexity;
- Tidal amplitude; and
- Freshwater runoff.

Table 10.1 Habitat preference of estuarine fish species identified by Maitland 1974 (derived from Campbell, 1976 and Davidson *et al*, 1991)

Species	Description
Sea bass *Dicentrarchus labrax*	This species occurs over sand, shingle and rocks from shallow water down to about 100m. It is widespread and common in estuaries, particularly in southern and western Britain. The sea bass uses estuaries as spawning and nursery grounds, with the late postlarvae congregating around the saltwater/freshwater boundary.
Common goby *Pomatoschistus microps*	This species is found in intertidal sandy-bottomed pools on muddy saltings, breeding prolifically throughout the summer, hollowing out a small 'nest' under a pebble or shell. It is abundant in the UK and has a significant impact, both as predator and prey, on the ecosystems of British estuaries.
Thick-lipped grey mullet *Chelon labrosus*	This species of mullet is found among rocks and seaweeds in shallow water and estuaries, often near sewage outfalls. It is widespread and common in the UK.
Thin-lipped grey mullet *Liza ramada*	*Liza ramada* penetrates as far as the freshwater reaches and is restricted to coasts between south Wales and Essex. It is not very common.
Golden mullet *Liza aurata*	This fish is also uncommon and restricted to coasts between south Wales and Essex.

In the (unlikely) absence of human factors it has been suggested that between 30 and 40 estuarine fish species may be present (Potts and Swaby, 1993). The tidal channel is important to the passage of fish, such as salmonids, during their breeding migrations. The strength of the tidal flow affects the movement of fish and the freshwater draining into the upper end of the estuary will influence the suitability of the ecosystem as a habitat for fishes. Climatic conditions also alter habitat conditions, for example, strong winter rains or snow melt waters in northern latitudes can flush out the maritime influence.

10.2.1 Feeding Preferences

One of the major factors affecting habitat selection by juvenile fish is foraging profitability (Holbrook and Schmitt, 1984; Schmitt and Holbrook, 1985). Estuarine fishes are dependent upon the feeding grounds provided by the invertebrate rich intertidal flats and saltmarsh habitats at high tides. Feeding preferences and the distribution of food species are therefore major factors in the value of a habitat for fish.

As part of the intertidal food chain, fish occupy many trophic levels, including higher levels, feeding on phytoplankton, zooplankton, algae, invertebrates and other fish. Feeding habits include predators (sharks, pikes, anglerfish, archerfish), grazers, food strainers (plankton filtration) and suckers, and parasites such as the sea lamprey (Lagler *et al*, 1962). Many fish feed on surface vegetable matter or graze algal films from benthic mudflats. Some species tend to aggregate in the mouths of estuaries to feed, for example young labrids are often found occupying the mouths of estuaries before moving to the coastal inshore waters.

The intertidal substrate is very important in determining benthic communities and therefore demersal fish species. An experiment in France found that when sand and gravel overlying a rocky substrate were removed from an area, a hard-ground fauna developed which was of less food value to demersal fish species than the previous soft-bottom fauna (ICES, 1992).

10.2.2 Access

Foraging fish regularly move from tidal creeks onto vegetated saltmarsh during rising tides, preferentially using permanent creek channels with shallow sloping profiles or areas with abundant submerged aquatic vegetation, such as seagrass (Rozas *et al*, 1988). There is evidence that dense vegetation inhibits the foraging efficiency of some piscivores and that topological features of the marsh, such as rivulets, facilitate the use of vegetated wetland surface for foraging by juvenile fish (Havens *et al*, 1995).

10.2.3 Cover

Another major factor that is considered important in habitat selection by juvenile fish is the risk of predation (Holbrook and Schmitt, 1984; Schmitt and Holbrook, 1985). Partially submerged vegetation is reported to provide optimal environments for most aquatic invertebrate and juvenile fish communities. The provision of saltmarsh cover for fish increases habitat complexity that can lead to a richer species diversity and provide an escape from predators. Cover is also reported to provide increased substrate for food items and for egg attachment under certain circumstances (US Fish and Wildlife Service, 1989).

Tidal marshes and creeks have been shown to be important habitats for the juvenile and larval stages of marine and estuarine fish (Clark and Hannon, 1969; Weinstein, 1979). The risk of predation is influenced by the morphology and microtopography of the marsh. Rivulets, for example, provide refuge for juvenile fish during low tides (Havens *et al*, 1995). Fish utilisation of a creek system may be affected by the sinuosity of the creek, channel depth, bank stability and the adjacent vegetated marsh surface (McIvor and Odum, 1988).

10.2.4 Nursery Areas

The estuarine resource provides nursery areas for juvenile fishes, including sole, plaice, sprat, cod, dab, flounder, bass, herring and other species. Estuaries are particularly important to sea bass as spawning and nursery grounds. The late postlarvae bass appear to congregate around the saltwater/freshwater boundary at the top end of many estuaries (Davidson *et al*, 1991). Juvenile Dover sole have been observed to use the mudflats of the Tamar Estuary as a nursery area. During their time in the estuary individual fish appear to restrict themselves to a single mudflat with only limited movement of fish between adjacent mudflat areas. These populations are thus very susceptible to localised disturbance or the destruction of mudflat areas.

10.2.5 Water Quality

Estuarine fish show some tolerance of high turbidity, temperature extremes, and a wide range of salinities and dissolved oxygen levels; the distribution of species in an estuary is largely determined by these limits. Chemical reactivity, tidal movement and freshwater discharge interact to produce complex conditions that vary between one estuary and another, and also within any particular estuary on an hourly, monthly or annual basis due to tidal cycles, climatic conditions and so on. Water movement, critical for the distribution and dispersal of suspended matter and pollutants, is affected by the tidal amplitude and geomorphology of an estuary. The number of variables illustrates the problem in defining water quality and relating it to the distribution of fish species within an estuary. The most serious impact of water quality in estuarine fish habitats results from the deoxygenation of the water, which kills fish communities and the invertebrates upon which they feed.

10.3 FISH USE OF CREATED HABITATS AND PRACTICAL CONSIDERATIONS FOR THE CREATION OF FISH HABITAT

The creation and enhancement of subtidal and intertidal habitat for fish takes many forms, and schemes have been undertaken for a range of purposes such as the replacement of lost habitat and the increase of fish catches (Pratt, 1994). The placement of mounds of dredged material on the seabed create artificial reef systems with sheltered areas of water on their lee side which may provide refuge and resting habitat for fish. These mounds or berms also alter flow patterns of water currents, creating eddies which may concentrate food organisms (PIANC, 1992). In addition, the sediments that comprise the mound, depending on their characteristics, may serve as foraging habitat for fish.

Although much work has been undertaken into the creation of subtidal habitat, such as offshore berms, the following discussion will focus on the creation of intertidal habitats and the factors that are likely to affect their use by estuarine fish.

A review by Roberts (1991) of man-made coastal marshes in Florida provides some insight into factors that affect the value of created habitat for fish. A wide range of factors are thought to influence the use of a site by fish, including both on site conditions, such as vegetation cover and size of the marsh, and nearby habitat conditions. The least successful created fish habitats were found to share a number of common design characteristics; they were all small (0.02 to 0.04ha) and narrow, they were all fringe marshes, and were located in areas with few remaining wetlands. These poorly designed fringe marshes lacked the complexity of natural and successful man-made marshes thus limiting opportunities for fish foraging and cover. The narrow saltmarsh fringes also lacked an elevational gradient that resulted in their rapid inundation on flood tides and drying as the tide ebbs, providing fish with inadequate opportunity to seek alternative cover. In addition, the likelihood of fish predation was greater in narrow marshes, which concentrated them in a relatively uncovered small area.

10.3.1 Habitat Diversity and Size

Roberts (1991) concluded that for a created habitat to be of value for fish and wildlife it should be at least 0.5ha in size. Fish populations benefit from a mixture of coastal habitats, typically containing tidal inlets, permanent pools (lagoons), *Spartina* marsh, a variety of lower and higher saltmarsh communities and seagrass beds. This is particularly relevant as certain fish species are associated with specific areas of marsh habitats, for example tidal creeks or permanent pools, while others are transients. Marble (1992) suggests that the key to designing for aquatic diversity is to provide diversity in the habitat conditions that will in turn create a diversity of organisms.

Marble (1992) also reported that a higher diversity of fish and their invertebrate food species are present in saltmarsh which support some areas of permanent water or are hydrologically connected to permanent water. It is recommended that in restored saltmarshes, the proportion of vegetated areas to open water areas should range from approximately 70:30 to 40:60. The provision of permanent deep water in the form of unvegetated creeks, channels and/or pools throughout a saltmarsh habitat is important to allow fish movement for longer periods of time and refuge even during the lowest tidal conditions. The presence of continuous areas of dense vegetation can be detrimental to fish. Marble (1992) also suggests that in estuarine saltmarsh areas where surface water is present at specified tides, a mosaic of small patches of vegetation interspersed with open water should be provided by varying the elevation of the sediment surface.

10.3.2 Substrate and Invertebrate Prey

Feeding preferences and the distribution of food species are major considerations in the creation and restoration of habitats for fish. Fine organic sediments of estuarine mudflats generally support high densities of invertebrate food species for fish and other higher predators. The highly productive nature of these organic materials make them the most desirable materials proposed for restoration schemes. (Section 5.2) (Marble, 1992). Coarser sand substrates generally support lower densities of macroinvertebrate food species, mainly because of their instability.

To create a suitable fish habitat it is important to determine the relative importance of vascular plants and algae as the base of the estuarine food web that supports fish species. In order to do this the principal foods of estuarine fishes must be known. A study at the Tijuana Estuary (Zedler, 1996) found that macroalgae, cyanobacteria, and vascular plants (*Spartina*) all contributed to the food web base. If all habitats in the estuarine ecosystem appear to support producers that contribute to the food web, it follows that all intertidal habitats (tidal creeks, saltmarshes, pools) should be protected, restored, and managed when the objective is to develop or enhance fish habitat.

10.3.3 Cover

The created site should be given a slight gradient to allow gradual flooding and drying which will enable foraging fish to find alternative cover. The benefits of creating intertidal and subtidal eelgrass habitat for fish have been documented. Eelgrass *Zostera* spp. beds provide various estuarine fish species with shelter, cover, and refuge which is particularly important to fish subject to predation. The rich epifaunal and diverse benthic communities associated with eelgrass habitats provide an abundant food source for grazing and foraging fish species. Shallow and intertidal fish habitats created with fine grained materials have also been stabilised and protected from erosion by planting eelgrasses or by capping with coarser materials, such as shells, which also serve to enhance fish habitat by providing greater habitat diversity (PIANC, 1992).

10.3.4 Water Quality

Turbidity can be controlled to some extent by keeping the gradient of the intertidal flat and saltmarsh gradual (Marble, 1992). A gradual slope across an intertidal area will tend to dissipate wave energy more effectively than a steeply-sloping intertidal area. The result will be that a gently sloping intertidal will be subject to lower turbulence thus limiting the resuspension of fine material from the surface of the intertidal area.

11. BUFFER ZONES

11.1 INTRODUCTION

11.1.1 Background

The external factors affecting the evolution of dynamic coastal systems, either natural or created, or the ability of species to adapt to these changes, is not always possible to predict and control. This is particularly true of intertidal habitats located in urban areas where the risks of disturbance are great and space for the natural adjustment of the ecosystem to external stresses is often restricted. Such external forces may include catastrophic events, such as hurricanes, drought, fire, infestation and major pollution events as well as the more common gradual and cumulative impacts, including sea level rise, sedimentation, trampling and urban and agricultural run off water causing water pollution. Willard and Hiller (1990) recommend that habitat creation project designers and managers should plan for extreme circumstances by including mechanisms for wetland adjustment and persistence and by maintaining multiple sites as refuge to spread the risk of catastrophe. This can be achieved through the following means:

- The creation of buffer zones and corridors;
- Maintaining adjacent natural marshlands as reserve sites;
- Recreating temporal and spatial habitat variability; and
- Planning for worst case scenarios (cumulative impacts).

"Managed habitats may fail by being too constant, changing the wrong way or changing too fast" (Willard and Hiller, 1990). In the past poor management of habitat creation schemes has resulted in failure to reach the potential wildlife value by limiting its ability to adapt. Dykes, banks and other structures have been constructed in the flood plain which restrict the expansion or migration of saltmarsh inshore during high water periods (100 or 50 year high water level) therefore limiting adaptability (Willard and Hiller, 1990). Such steep confinements do not allow saltmarsh vegetation to adjust to changing hydroregimes and therefore force the loss of the flora and fauna of both saltmarsh and intertidal flats. This process is commonly known as "coastal squeeze" and is widely reported at many degraded saltmarsh sites along the east coast of England where coastal defence structures, such as seawalls, prevent the landward migration of intertidal habitats in response to coastal retreat and erosion.

11.1.2 Ecological Buffer Capacity Concept

Jorgensen (1990) developed the ecological buffer capacity concept to predict ecosystem reactions to perturbation while acknowledging that a system will always be changing. This concept can be used to develop an understanding of the ecological recovery of a newly created or restored intertidal habitat. The concept can be defined as follows.

Buffer capacity = forcing function/state variable
Forcing functions are the external variables which drive change in a system (for example, sea level, turbidity, waste water discharge) and state variables are the internal factors of a system which respond to them (for example, concentration of zooplankton, nutrient levels, density of invertebrates). The model can be used to consider all combinations of variables and forcing functions in a multi-dimensional manner and suggests that for a given change there are many buffer capacities which are not constant (Figure 11.1). It is important to understand the relationship between specific forcing functions and state variables for environmental management in general, including habitat creation, to determine under what conditions buffer capacities are low or large (Jorgensen, 1990).

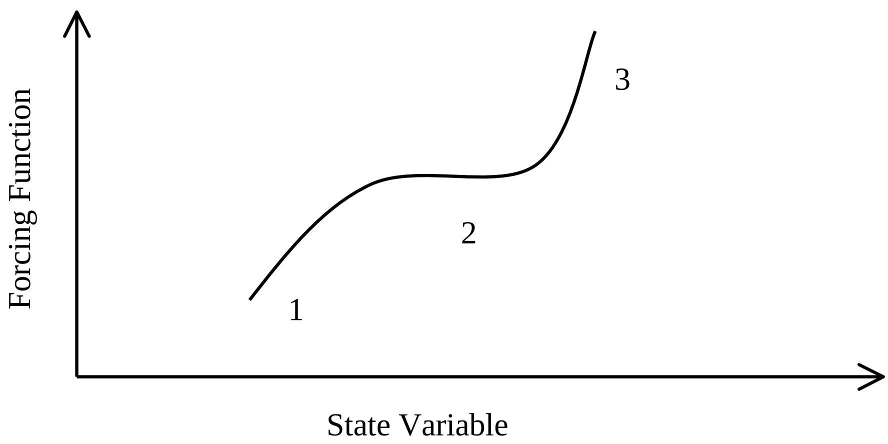

Figure 11.1 Ecological buffer capacity (Source: Jorgensen, 1990). At Points 1 and 3 the buffer capacity is high, at Point 2 it is low

11.2 BUFFER ZONES AND ENVIRONMENTAL CORRIDORS

11.2.1 The Benefits of Providing Buffer Zones and Environmental Corridors

Jorgensen (1990) defines buffer zones as “an area of land left unimpaired between the natural habitat type and a neighbouring developed area”. Buffers act as a zone of protection between the fragile newly created habitat and urban areas, providing a refuge for plants and animals between their preferred habitat and human activities. Protection can be afforded from the pressures exerted by adjacent urban areas, such as trampling, disturbance from traffic, sediment, excessive nutrients, contaminants and pesticides. Buffer zones around newly created and restored habitats are required to:

- Provide the space to allow for the expansion and adjustment of intertidal communities in response to changing hydroperiod and other external forces, avoiding the loss of habitat area, flora and fauna; and

- Function as high tide refuges and as habitats for upland species that play important roles in the adjacent wetland (Zedler, 1996).

The benefits of including environmental linkages or corridors between created habitats and the surrounding natural habitats has been stressed by several authors (Willard and Hiller, 1990; Roberts, 1991; Pratt, 1994). Ecosystems are linked together, often in an interdependent manner, and ignoring these linkages could cause the creation of a poorly functioning habitat with a low probability of persistence (Pratt, 1994). Environmental corridors reduce the isolation of created habitats in addition to acting as buffers. These corridors may provide water connections between different portions within the created marsh habitat and with nearby natural marshes. These act as reserves or refuges for plants and animals which disperse by water and facilitates recolonisation of the new site. Buffers and environmental corridors increase the effective size of a site and, as stated previously, larger habitats generally provide greater habitat diversity which improves the ability to spread risk and consequently increases the likelihood of survival for a variety of species (Willard and Hiller, 1990). Therefore the use of buffer zones increases the probability of habitat success and allows the development of self-sustaining and persistent features. In addition to providing a zone in which the habitat and its biota may adjust in response to external changes, it provides resident animals an area of refuge during construction and alteration of the existing habitat.

11.2.2 Buffer Zones: The Lessons Learned

Few references give detailed information of specific buffer zone requirements or design considerations as to the type and size of buffer zones for their creation (Shisler 1990). However, the need for such protective barriers are clearly reported by various authors (Kusler and Kentula, 1990; Willard and Hiller, 1990). The addition of only a modest buffer around a newly created wetland is suggested to provide a large improvement in the persistence of the site (Willard and Hiller, 1990).

The importance of the inclusion of buffer zones in the design of habitat creation schemes is illustrated by the Hayward Regional Shoreline Marsh Scheme, South California (see Appendix II) which demonstrated that in urban areas a large buffer zone is required to facilitate even limited use by birds (Zedler, 1996). The close proximity of public access trails to the saltmarsh site were not screened by buffer vegetation and as a consequence few birds remained in the marsh when visitors were present in the park. It is important to restrict access to the wetlands under restoration or creation until vegetated buffers have fully filled in. Inadequate public awareness and the degraded appearance of wetlands during the revegetation period often seems to invite off road vehicle activity that can permanently disrupt existing wetland areas as well as newly restored ones.

Zedler (1996) reports that in the absence of native tress to act as natural buffers between coastal wetlands and adjacent urban developments in Southern California, there is no easy way to screen wetlands from noise, lights and motion because fencing is costly and areas adjacent to wetlands are too saline for woody plants.

In order to protect newly created saltmarsh systems in Ballona wetland, Southern California from uncontrolled and untreated storm water, an adjacent freshwater wetland system was constructed to provide a buffer (Tsihrintzis *et al*, 1996). In addition, various measures were adopted to protect the site against human impacts, including the construction of a series of berms or 1m walls along roads, urban developments and communities. This saltmarsh restoration scheme was designed to incorporate buffer areas sufficient enough to respond to the 50 year storm event (Tsihrintzis *et al*, 1996).

Further research is needed to help set the widths of buffers and determine allowable activities within buffer zones. Until such information is available, it makes sense to use the broadest buffer zones that can be justified (Zedler, 1996).

The role of a created habitat in the local and regional landscape should be considered in the planning process, and is dependent upon its proximity to other habitats and its persistence (Willard and Hiller, 1990).

12. MONITORING, MAINTENANCE AND MANAGEMENT

12.1 THE IMPORTANCE OF INFORMATION AND MONITORING

Information is a key concern for wetland creation and restoration programmes; "gathering it, organising it, making it available to others as a body of experience from which to draw and keeping it current" (Grenell, 1994).

Field monitoring is vital to evaluate project performance and to guide management decisions (PIANC, 1992; Zedler, 1996). For example, a scheme must be monitored in order to determine whether goals are being achieved so that mid-course corrections can be made if necessary to ensure a scheme's success.

A review of restoration schemes carried out as part of the California State Coastal Conservancies Wetland Programme concludes that a lack of transference of experience gained from past schemes to new schemes was the least effective part of the programme (Josselyn *et al*, 1990; Grenell, 1994). This was attributed, at least in part, to a lack of monitoring, uniform data collection and reporting in literature. In addition, limited staff effort was directed towards information dissemination and budget restrictions were reported to further restrict the ability to pass on information. The review recommends that uniform data collection and database creation are essential in order to take advantage of information available from wetland restoration and enhancement activities.

The importance of pre-construction monitoring is illustrated by the Humbolt Bay saltmarsh mitigation scheme where inadequate knowledge of site conditions resulted in the development of a poorly functioning site which required costly remedial action (Grenell, 1994).

12.2 MONITORING PROGRAMMES

Monitoring programmes need to be geared to specific, realistic and flexible goals. Specific site requirements and the type of habitat creation scheme must be considered when deciding the appropriate and efficient monitoring strategy.

The monitoring criteria need not comprise a long list of parameters, but it should include measures that reflect system structure and function. Structural parameters, such as species diversity and community composition, can be used to indicate habitat function. One of the most important functions associated with well developed communities is the biomass of harvestable species (Pratt, 1994).

In the USA, mitigation monitoring is usually required for 5 to 10 years post-construction, although recently federal and state agencies have been requiring longer monitoring programmes. Monitoring is recommended by Frenkel and Morlan (1991) for at least 10 years. In the case of the Salmon River restoration project they observed changes in species cover, biomass and surface elevation 10 years after dyke breaching. Rates of change diminished thereafter and subsequently so did the monitoring effort.

The Environment Agency generally monitors intertidal recharge sites for one year before construction and five years post construction. Post construction monitoring is undertaken twice yearly, including sediment profiles and benthic surveys (M. Dixon, *pers. comm.* 1996).

Monitoring plans for coastal habitat creation/restoration schemes are very varied, tailored to meet specific project objectives, however most monitoring schemes usually take two general forms; physical and ecological.

12.2.1 Physical Monitoring

Physical monitoring determines whether the engineering integrity of the site is maintained, whether the structure of the site is eroding, migrating, accreting or otherwise changing in configuration and elevation (PIANC, 1992).

To measure change at the site, replicated samples must be collected at the same site over a period of time. A tried and tested method for achieving physical sampling has been to establish a grid in and around the project site, upon which predetermined sampling sites are located and given xy co-ordinates. The size of the grid needs to be large enough to cover the project site and the potential area for subsequent material spread. The complexity of the grid depends upon the sensitivity of the site and in past schemes survey lines have been spaced from between 25m and 300m apart (PIANC, 1992).

Physical monitoring tools include bathymetric surveys, sediment sampling, tide, current and wave gauges, settling plates, seabed drifters and aerial photography.

12.2.2 Bathymetric Survey

Bathymetric surveys form the basis for any physical monitoring programmes. Repetitive pre- and post-construction bathymetric surveys provide an evaluation of changes in volumes and elevation over time and the overall integrity of the site.

12.2.3 Sediment Sampling

Sediment sampling may take the form of grab samples, which can be analysed for grain size and changes in sediment type due to mixing, sorting and erosion. Alternatively, core samples can be taken which give information on densities, shear strength and vertical stratification from which the degree of consolidation and erosion potential can be calculated.

12.2.4 Tide, Current and Wave Gauges

Current and wave gauges measure the forces that move sediment. "The presence of these gauges can sometimes mean the difference between success and failure of a monitoring plan" (PIANC, 1992). Near tidal inlets and deep waters a current gauge is most useful, whereas in shallow water sites a wave gauge is more useful depending on the most dominant hydrological force.

12.2.5 Aerial Photography

Photographic evidence, particularly aerial photography, is a valuable tool for evaluating the progress of a scheme as it matures and evolves. It can be used to provide a visual comparison of large-scale changes in and around a site over time. Depending upon the resolution of the techniques adopted, aerial photography may also be used to monitor changes in vegetation cover and recolonisation rates. Low altitude aerial photographs were used to study the physical characteristics of a created tidal marsh in Sarah's Creek, Virginia (Havens *et al*, 1995).

12.2.6 Ecological Monitoring

A basic knowledge and understanding is required of the ecological characteristics of the site before habitat construction. Such characteristics include the distribution, abundance and diversity of marsh vegetation, benthic communities, fish, birds and other coastal wildlife at the site in different seasons. Comparisons of pre and post construction monitoring data will determine any substantial changes in communities at the site.

Monitoring for vegetation and wildlife is usually undertaken by measuring along fixed transect lines established across sites prior to construction. Measurements are taken at regular intervals for a given period of years. The nature of the monitoring that is implemented will depend on the type of restoration project being undertaken. However, it is likely that ecological monitoring will largely be focussed on vegetation, benthic invertebrates and birds. Monitoring of vegetation and benthic invertebrates would include standard community parameters such as species diversity, abundance and biomass. Bird monitoring would comprise low water and high water counts to determine the use of the area by feeding and roosting birds. It may also be necessary to monitor the site for its use by breeding birds.

Depending on the nature and objectives of the habitat creation/restoration scheme, it may also be appropriate to monitor fish populations. However, the presence of fish species does not necessarily indicate functional support of those species. Documentation would require showing that the habitat provided food, refuges from predators (e.g. presence of submerged vegetation), and places for egg laying (Zedler 1996).

12.2.7 Sediment/Soil Data

Additional monitoring of sediment and water characteristics could be undertaken to provide information on the functioning of the habitat creation scheme and to determine whether or not management measures may need to be introduced in order to alter the environmental conditions to encourage or enhance the development of benthic communities. Sediment parameters that could be monitored include soil texture, soil chemistry (pH, salinity), soil elevation, soil migration, organic content, sediment contaminants (for example heavy metals, TBT, PCB, PAH etc). Water quality parameters include salinity, organic content (BOD), dissolved oxygen, turbidity (suspended solids), nutrients and contaminants. It should be noted that it is unlikely that it will be necessary to monitor all of the above parameters, with only those parameters that are likely to be an issue in the particular area or will affect the development of a particular characteristic of the scheme requiring monitoring. Therefore, the monitoring that is proposed should be focussed and relevant to the scheme in question.

12.3 MAINTENANCE REQUIREMENTS

Although many habitat creation and restoration schemes described in the literature are self sustaining, others require human intervention in order to enable a satisfactory outcome (Grenell, 1994). Maintenance may be required if monitoring reveals that the created wetland is not functioning as intended and that the project is not achieving its goals. This is particularly true in wetlands created or restored for the purpose of waste water treatment.

In the schemes undertaken as part of the California State Coastal Conservancy Wetland Programme, hydrological functioning of created and restored wetlands was identified as the most common problem encountered which required human intervention (Josselyn *et al*, 1990, Grenell, 1994). In tidal marsh restorations, tide gates, sluices and weirs may require frequent adjustments to maintain the correct hydrological and tidal regime. This is exemplified by the Abbot’s Hall saltmarsh restoration scheme in the Blackwater Estuary where modification of sluice pipes was required in order to retain water within the site over several tidal cycles and to raise water levels to cover the whole site (I. Black, *pers. comm.* 1996; Section 13.6).

Sustained management may be required for the removal of sediment originating from the surrounding watershed which can cause the gradual siltation of the newly created site with time or to maintain tidal circulation by periodic breaching of sediment bars which may develop (Grenell, 1994).

Intervention has also been necessary where revegetation schemes have failed due to invasion by exotic species which require control (Grenell, 1994).

12.4 MANAGEMENT AND PLANNING CONSIDERATIONS

12.4.1 Construction Impacts

The NRA guidelines on saltmarsh restoration and management warn that physical damage to the site during construction may reduce the success of restoration (NRA, 1995). They recommend that the following factors should always be considered in the creation, restoration and recharge of wetlands with most techniques adopted:

- sources of material;
- trampling and compaction of marsh and mudflat surfaces;
- disturbance of wildlife;
- alteration of the natural environment characteristics;
- timing and construction of maintenance works;
- visual intrusion;
- fuel spillage;
- removal of unnecessary construction materials;
- minimising presence of toxic substances; and
- minimising long term losses of material.

13. UK CASE STUDIES

In this section, a number of habitat creation and saltmarsh restoration schemes around the coast of the UK have been examined in detail, in order to investigate the different techniques adopted, and to assess the components which have contributed to the success and/or failure of the various schemes. A synopsis has been made of each scheme, and various aspects of the schemes have been examined under the following sub-headings:

- Site description and background;
- Responsible authorities, contact, contractor and funding body;
- Objectives of the scheme;
- Methodology and technical aspects of site design;
- Time needed to create the habitat;
- Problems encountered during construction;
- Monitoring programme;
- Results of the scheme;
- Maintenance needs and possible improvements; and
- Figures showing the location and design of the site and photographic documentation.

The various schemes included in this section are summarised in Table 13.1. Figures are enclosed throughout this section. A bibliography of further case studies in the UK and abroad is contained in Appendix II.

Table 13.1 A selection of UK case studies

Location	Type of Scheme	Funding/Responsible Authority
Parkstone Bay, Poole Harbour	Mudflat creation	DoE and marina owner
Maldon Marshes, Blackwater Estuary	Saltmarsh restoration	EA and boatyard owner
Northey Island, Blackwater Estuary	Managed retreat	EA, EN and NT
Tollesbury, Blackwater Estuary	Managed retreat	MAFF and EN
Orplands Marsh, Blackwater Estuary	Managed retreat	MAFF, EA and landowner
Abbots Hall (St. Leonards), Blackwater Estuary	Saltmarsh restoration	MAFF, EA and landowner
Blaxton Meadow, Plym Estuary	Saltmarsh restoration	MAFF and NT
Horsey Island, Hamford Water	Saltmarsh restoration/ Intertidal recharge	EA and EN
Pewet Island, Blackwater Estuary	Intertidal recharge	EA
Trimley Marshes, Orwell Estuary	Intertidal recharge	EA
Parkstone Quay, Stour Estuary	Intertidal recharge	EA
Medway Estuary	Intertidal recharge	MAFF, Medway Ports, EA and EN

13.1 PARKSTONE YACHT CLUB MUDFLAT CREATION SCHEME, POOLE HARBOUR

13.1.1 Site Description and Background

Parkstone Yacht Club obtained planning permission for the construction of a Yacht Haven on the northern shore of Poole Harbour in Parkstone Bay in 1990 (Figure 13.1). The Yacht Haven (Figure 13.2) was constructed in the winter of 1994/95, and as part of the planning consent there was the requirement to provide an area of intertidal mudflat to replace that lost to the development.

The area of the created mudflat is approximately $6500m^2$ and experiences a tidal range of around 4m. Photographs of the scheme are shown in Figure 13.3. The Yacht Haven was designed by Hartmans and hydraulic studies and breakwater design was undertaken by HR Wallingford.

13.1.2 Scheme Objective

The overall objective of the scheme was to incorporate the construction of a breakwater around the Yacht Haven and the generation of an intertidal mudflat area to replace the area lost in the development of the Yacht Haven. No specific objectives or goals were set for the scheme.

13.1.3 Methodology

The mudflat was built on the inside of a rubble mound breakwater which protects the Haven from wave action from the south and west and which is held in position by steel sheet piling (15m long) inshore of the breakwater. The mudflat is approximately 325m long by 20m wide. The sheet pile wall is at a level of +1.2m CD, which is the level of mean low water on a neap tide, and at the breakwater edge it is +2.0m CD, slightly below mean high water on a spring tide (Figure 13.2).

The construction of a temporary roadway was necessary to facilitate the driving of the inner piled wall. The piles were then tied together involving the drilling of holes through the top of alternate piles and bolting them together. The mudflat was coarsely levelled with a grader. Mud from the original intertidal area had been temporarily stored above the high water level and was placed on the area by dumper trucks.

Dredged material from the construction site was used to construct the intertidal mudflat. Material was dredged using a trailing suction hopper dredger. A total of $7000m^3$ of dredged material was used inshore of the sheet piled wall for the initial fill and $3000m^3$ from the existing intertidal area was used for the top layer. The material is of variable consistency, being very dense clay in some areas and very soft in others. The lowest bulk density measure was $1400kg/m^3$.

13.1.4 Construction Problems

Construction of the sheet piling took longer than anticipated. This was for a number of reasons:

- At one stage the pile driver toppled into the water;
- The subsoil was found to be denser than expected;
- The planning consent stipulated work could only be carried out within office hours; and
- Water levels were often higher than predicted.

During the construction phase, additional dredged material was placed on to the eastern edge of the breakwater which raised the level of the tidal flats to an elevation which was too high (Figure 13.3), as a result this section of mudflat has experienced less tidal inundation, surface sediments are coarser and the flats are ecologically less developed than those on the western half of the breakwater.

13.1.5 Monitoring

Various monitoring studies were undertaken before the construction of the Haven including a preliminary environmental impact assessment, a study of the hydraulic impact of the Haven, and an assessment of the breakwater/mudflat design.

Ongoing post-construction monitoring includes:

- Hydraulic monitoring;
- *In-situ* measurements of bulk density of the placed material;
- Surface sediment analysis (heavy metal concentration, organic content, particle size analysis and characterisation);
- Biological sampling to investigate vegetation and invertebrate colonisation of the mudflat; and
- A photographic record of the development of the mudflat.

There is no formal programme for the monitoring, which takes place on an opportune basis. However, throughout the first year of construction monitoring was undertaken every three months.

13.1.6 Results

Biological surveys, undertaken by A level students at Ferndown School, indicate that the new mudflats on the western end of the breakwater support more polychaete worms than the original foreshore. Vegetation surveys undertaken by a local botanist recorded 13 species of flowering plants growing on the western breakwater in autumn 1996 where the previous year there had been none. The species are all typically found around the beaches and saltmarshes of Poole Harbour and a large number belong to the family Chenopodiacea. Plants recorded include sea purslane, glasswort and other

Salicornia species, seablite, sea mayweed, sea aster, and the saltmarsh grasses *Agropyron* and *Puccinellia*.

Although no bird counts have been made, the site is used as a roost by a mixture of seabirds, including gulls, swans, and typical waders, such as oystercatchers and ringed plover. Nests have been observed in the far western corner of the breakwater. The main roosting area is located to the west of the bend in the breakwater.

13.1.7 Reference Sources

David Newman, Secretary of Parkstone Yacht Club; *pers. comm.* 1996.

Dearnaley and Burt, 1996; Dearnaley *et al*, 1995; Tarraway, 1996.

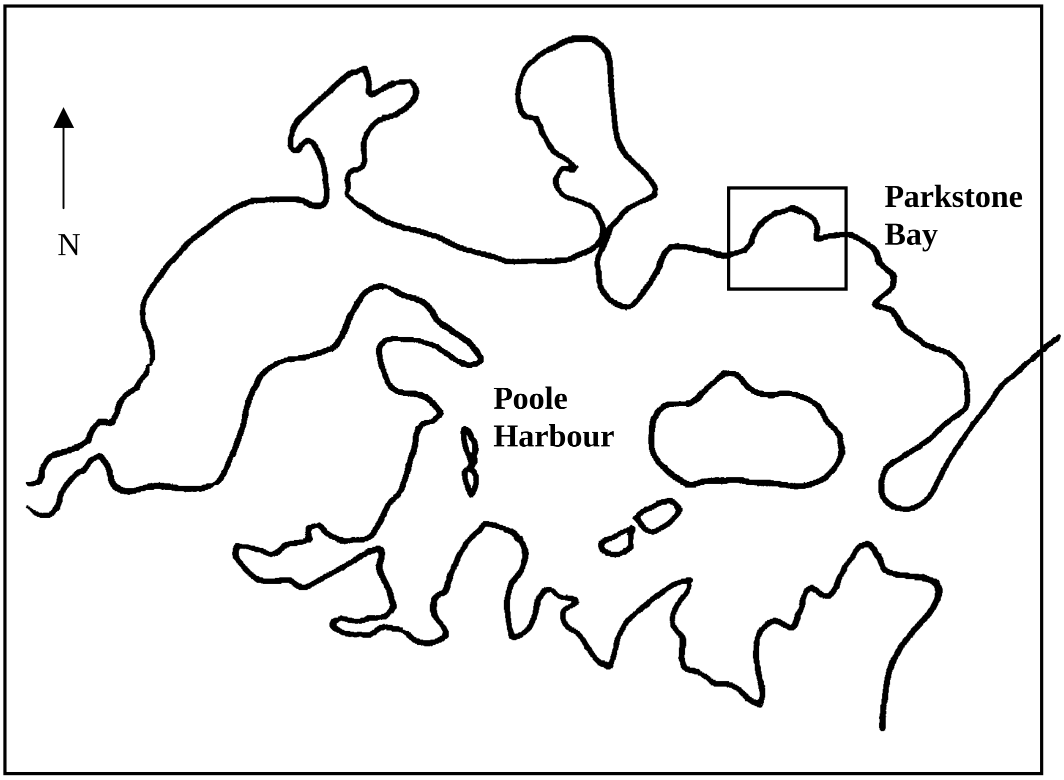

Figure 13.1 Location of Parkstone Yacht Club mudflat creation scheme, Poole Harbour.

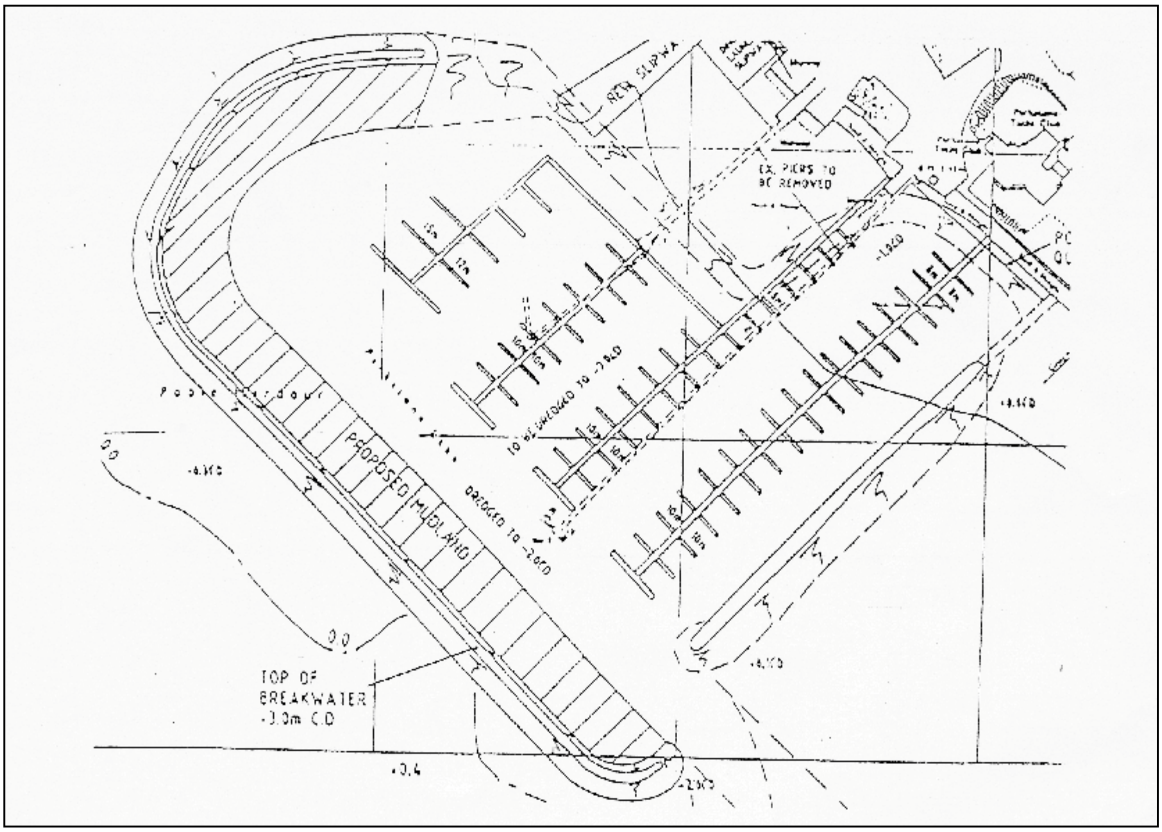

Figure 13.2 Plan of Parkstone Yacht Haven and proposed artificial mudflat

Figure 13.3 Parkstone Yacht Club mudflat creation scheme, Poole Harbour (December 1996)

13.2 MALDON SALTMARSH RESTORATION SCHEME, BLACKWATER ESTUARY

13.2.1 Site Description and Background

This saltmarsh restoration scheme is located at Hedgecocks Boatyard, Hythe Quay in the Blackwater Estuary (Figure 13.4). The area has been experiencing a general decrease in area of active saltmarsh and breaches in the salting separating the main river channel from Heybridge Creek have also been increasing in magnitude. It was feared that this pattern of erosion would result in a loss of depth at Maldon Quayside and accelerated loss of the remaining saltmarsh. Cliffing of the saltmarsh edge, caused by wave and current energy, had also rendered the saltmarsh banks susceptible to undercutting mass failure. The scheme has been funded and undertaken by the Environment Agency and the boatyard owner.

13.2.2 Scheme Objectives

The objectives of the scheme were as follows:

- To provide a beneficial use of maintenance dredgings from the boatyard to maintain and reconstruct the adjacent saltmarsh;
- To reduce erosion through the closure of the saltmarsh breaches situated in the Maldon salting separating the main river channel from Heybridge Creek;
- Cliff stabilisation; and
- Extension of the saltmarsh boundaries opposite the boatyard and at several locations downstream to produce a new area of intertidal mud.

13.2.3 Methodology

In Spring 1993, the four gaps in the salting between the river channel and Heybridge Creek were plugged with wooden planking bolted to timber piles at 2.4m centres, and infilled either side with dredged material. The material was composed of cohesive silts and clays removed from maintenance dredgings from the berths of Hedgecock's Boatyard on the southern bank of the river.

The sediment was removed using a grab and therefore the silt/clay was firm enough to use in marsh reconstruction purposes. Dredged spoil was deposited in front of the eroding cliffed edge in order to create a sloping, rather than vertical, bank profile (Figure 13.5).

The edge of the saltmarsh has been extended in a piecemeal fashion through the deposition of dredged spoil along the south side of the main channel.

13.2.4 Monitoring

A monitoring programme was designed to investigate plant colonisation and the potential ecotoxicology of the dredged material used to infill the breaches in the saltmarsh. The results will be compared to those from a control site in an area of natural saltmarsh. This monitoring programme comprises:

- Charting plant colonisation on the deposit sites;
- Analysis of contaminant concentration and comparison with Netherlands standards;
- Estimation of potential ecotoxicology; and
- Investigation of bioaccumulation.

The monitoring is undertaken at approximately six monthly intervals in the months of March and August. Assessment of the engineering success of the scheme is on the basis of opportune observations and discussion with boatyard operators.

13.2.5 Results

The results of the monitoring programme for the various aspects of the scheme are discussed below.

Closure of the saltmarsh breaches
The breaches were successfully infilled and tidal flow through them was therefore eliminated. There is evidence of recolonisation of the infill areas with saltmarsh vegetation. The infilled areas are not as species rich as the adjacent natural saltmarsh; however this would be expected given their relative ages.

Cliff stabilisation
Cliff stabilisation appears to be successful in engineering terms although detailed profile monitoring would be required to determine whether lateral erosion has been halted.

Saltmarsh extension
At some sites where extension of the saltmarsh has been attempted little revegetation has occurred. It is believed that the elevation of the placed mudflat surface is too low and hence the frequency and duration of tidal inundation is not suitable for the development of saltmarsh vegetation. Algal mats are the only form of plant life over the majority of the mud mounds.

13.2.6 Possible Improvements

To encourage colonisation it is thought that the height of the mudflat should be within the range of elevations between MHWS and MHWN (which at Maldon is approximately 1.7m ODN to 2.7m ODN).

13.2.7 Reference Sources

Dearnaley and Burt, 1996.

Dearnaley *et al*, 1995.

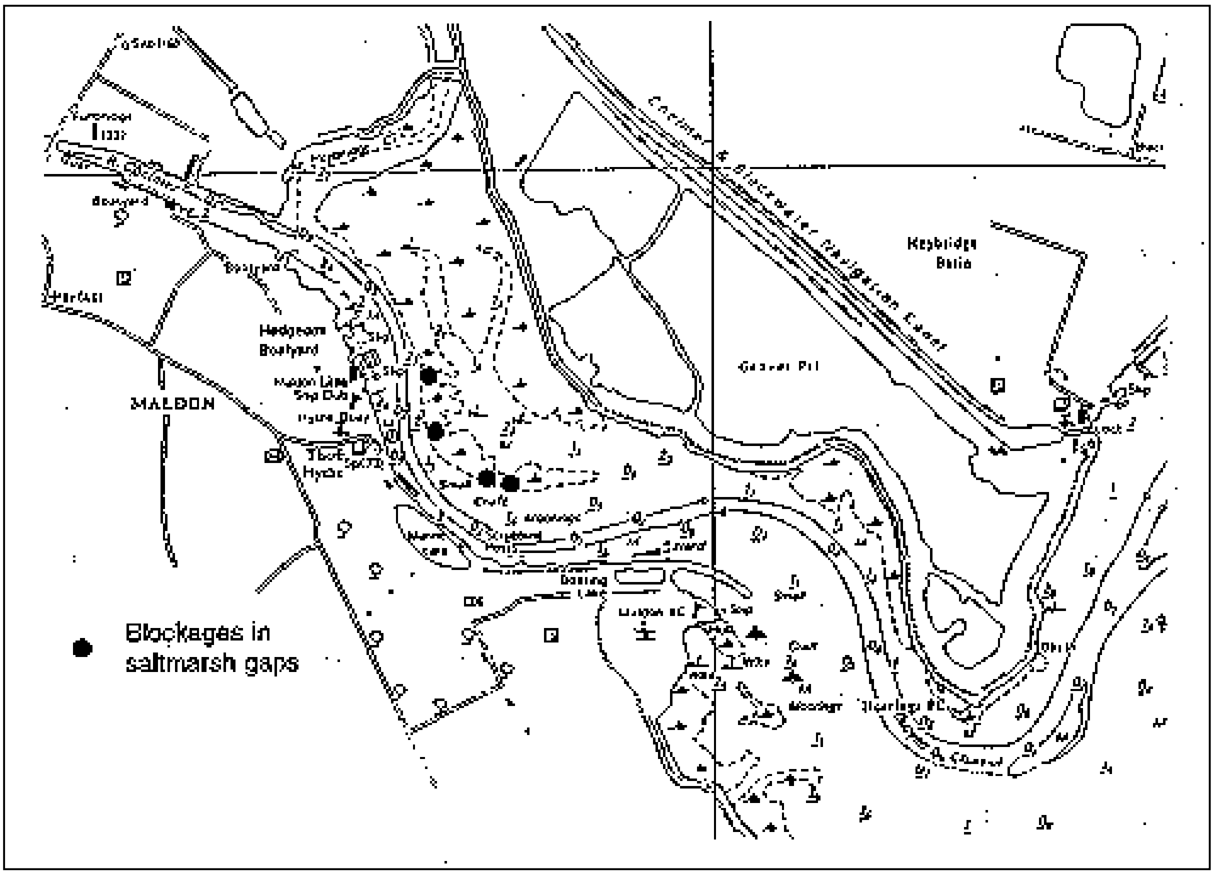

Figure 13.4 Location of Maldon saltmarsh restoration managed retreat schemes, Blackwater Estuary

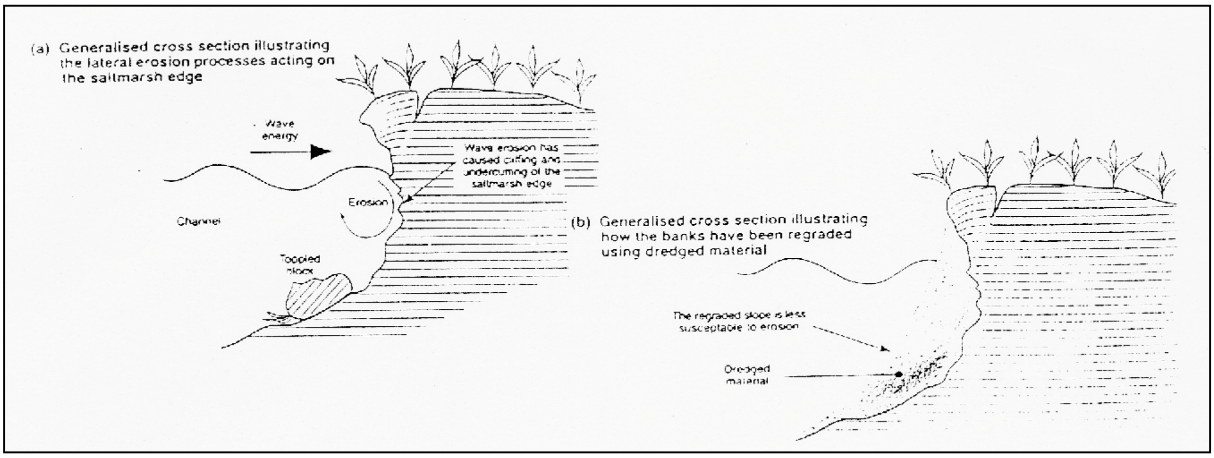

Figure 13.5 Diagram to show process of cliff regrading in Maldon Saltmarshes, Blackwater Estuary

13.3 NORTHEY ISLAND MANAGED RETREAT SCHEME, BLACKWATER ESTUARY

13.3.1 Site Description and Background

Northey Island lies in the inner Blackwater Estuary. The scheme at Northey Island was one of the first fully monitored marsh restoration schemes undertaken for flood defence purposes. The experimental site was reclaimed between 1843 and 1873 and was managed as a low productivity grazing marsh. In 1990, erosion of the saltmarsh fronting a 200m stretch of seawall was resulting in destabilisation of the wall structure, increasing the vulnerability of the flood embankment. The NRA (Environment Agency) therefore decided, with the support of English Nature and the agreement of the landowner (the National Trust), to undertake a trial managed retreat over an area of 0.8ha, rather than repair the existing embankment. The plan was implemented in August 1991. Figure 13.6 shows photographs taken at the managed retreat site five years post-breaching of flood defences.

13.3.2 Scheme Objectives

The objectives of the scheme were as follows:

- To undertake a trial managed retreat of the 0.8ha site for coast defence purposes; and
- To construct a secondary defence to protect the adjoining high ground against tidal erosion.

13.3.3 Methodology

The small size of the site, together with the relatively high elevation of the land, presented problems to the design of the scheme. The final design of the scheme was based upon data collected from topographic studies, tidal regime monitoring and modelling of the hydraulic conditions at the site.

The scheme was implemented in summer 1991 and involved the following components:

- An inner seawall was modified to provide a complete rear defence line (4.3m ODN);

- Original borrow dykes within the new saltmarsh area were infilled with material from the embankment to reduce edge effects to a minimum;

- The front seawall was lowered to the highest level of the new marsh (2.65m ODN, slightly higher than the weir crest of the open channel). This would slightly retard all ebb flows out of the site but would allow approximately 100 spring tides to flood over the embankment more quickly than they could through the open channel alone; and

- A 20m wide spillway was constructed in the embankment, located at the lowest point on the site which would allow free tidal flow into the site (approximately 150 tides per year) and complete evacuation of the flood water within one tidal cycle (Figure 13.6).

13.3.4 Monitoring

Prior to the implementation of the scheme, a full baseline survey of the site was carried out. This survey comprised:

- A topographic survey of the proposed site, fronting mudflats and surrounding land;
- The installation of a tide gauge; and
- Analysis of sediment characteristics of the surface and sub-surface sediments.

The existing vegetation of the site was also surveyed to provide a baseline for subsequent comparison. The results of the geomorphological survey allowed a number of predictions to be made regarding tidal flows and accretion rates. These were then incorporated into the engineering design and a recommended plan for implementation was drawn up.

The geomorphological monitoring has continued on a twice yearly basis for five years after implementation, with the botanical monitoring undertaken annually. No invertebrate and bird surveys have been carried out. However, observations during site visits indicate that the site supports typical saltmarsh/mudflat invertebrate communities, such as high densities of *Hydrobia ulvae* snails in drainage channels (Figure 13.6).

13.3.5 Maintenance

Monitoring of the site revealed a number of problems which have arisen during the course of the site development, requiring additional maintenance work to be undertaken. This maintenance included:

- The installation of a pipe to facilitate drainage from an adjacent field;
- The strengthening of the internal wall which was suffering some erosion from wind-waves internally generated within the site; and
- More complete infilling of the old borrow pits which were disrupting the flow of water across the site.

13.3.6 Results

The results of the monitoring programme for the various aspects of the scheme are discussed below.

Vegetation
The newly flooded site rapidly lost its cover of terrestrial grasses and halophytic vegetation began to colonise the surface sediments after only one growing season (including *Salicornia* spp., *Puccinellia maritima*, *Atriplex* spp., *Sueda maritima* and *Sueda vera*). Although the halophytic vegetation was initially slow to colonise, a good cover had been achieved after two years of tidal inundations. Observations made during a visit to the site in December 1996 (5 years post reclamation) indicated that the site was completely vegetated and that growth of *Spartina anglica* was increasing throughout the site (Figure 13.6).

Sedimentation
Regular monitoring of the surface topography of the site, using conventional EDM survey techniques, resulted in a valuable record of the manner in which sedimentation processes operate in a restored saltmarsh. Comparison of the mean surface elevations shows that no significant changes in surface elevation took place between 1991 and 1993. However, after this 2 year period, accretion rates began to accelerate (Table 13.2) giving statistically significant differences between successive surface elevations. These accretion rates were attributed to the colonisation of the marsh surface by *Salicornia*/ *Puccinelia*, a light vegetation cover which maximised deposition while preventing re-erosion. It is interesting to note that, once continuous accretion was initiated in 1993, summer accretion rates were much higher than winter rates probably due to the much denser vegetation cover during summer. However, during the period 1991 to 1993, summer accretion was more than offset by winter erosion. This was due to the presence of annual vegetation in this early stage of the marsh development so that the lack of any vegetation cover during winter allowed maximum wave re-entrainment of recently deposited sediment.

Table 13.2 Accretion over specified six month periods at Northey Island Retreat Site

Period	Accretion (cm)
Summer 1993	1.5
Winter 1993/1994	4.9
Summer 1994	1.5
Winter 1994/1995	2.1

Development of an adequate drainage network has been slowed by the lack of a physical connection between several areas of standing water and the main channel through the spillway. This has resulted in a large proportion of the site being permanently covered by standing water, preventing vegetation colonisation in these areas. These results also confirm experience from the United States which showed that relying on natural processes of erosion and accretion to establish drainage patterns took a long time, particularly where the sediment is consolidated, as on Northey Island (Haltiner and Williams, 1987). However, the small size of this site means that the water is able to reach all but the highest areas as a sheet flow on the rising tide, and therefore the lack of an adequate drainage network is less of a problem than on larger sites.

The surface topography data demonstrated that accretion rates in the site were highest at the entrance to the restored site but decreased thereafter to a zero accretion rate at 140m from the entrance. This offers some confirmation of a model of sediment inputs to marsh surfaces derived from experimental laboratory work (Pethick and Burd, 1995) (Section 4). The Northey Island data showed that an initial 100ppm suspended sediment concentration resulted in a 140m extinction distance for accretion with a *Salicornia* dominated vegetation community.

13.3.7 Possible Improvements

To avoid erosion of the secondary defence embankment it may be important to provide some form of internal bunds to reduce fetch areas prior to vegetation inception.

13.3.8 Reference Sources

Ian Black, *pers. comm.* 1996.

IECS, 1991, 1992, 1993b, 1994, 1995, 1996,

Burd, in prep.

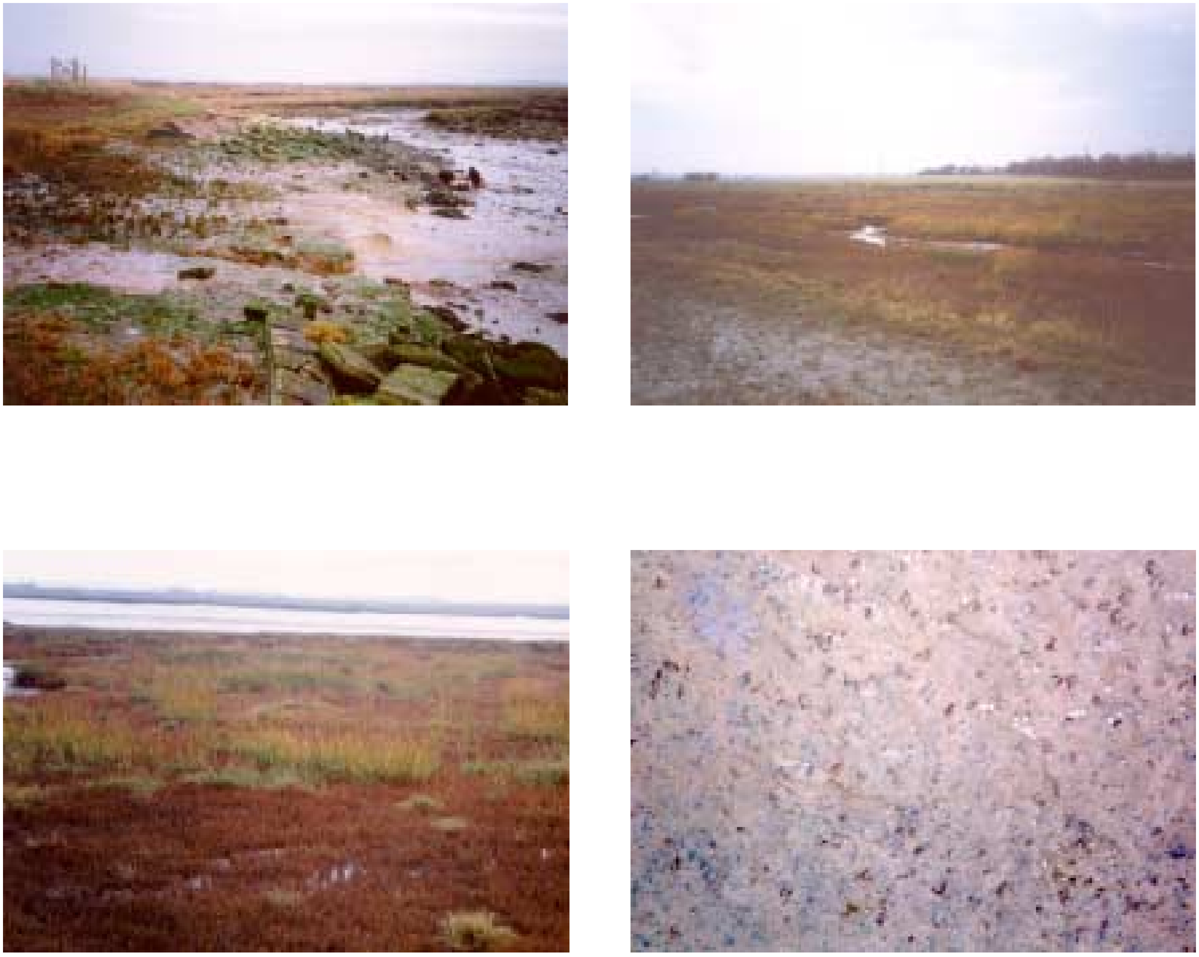

Figure 13.6 Northey Island managed retreat scheme, Blackwater Estuary (December 1996)

13.4 TOLLESBURY MANAGED RETREAT SCHEME, BLACKWATER ESTUARY

13.4.1 Site Description and Background

One of the most important of the managed retreat experiments in UK at the present time is that at Tollesbury Fleet, a tributary of the Blackwater Estuary in Essex. The experiment is managed by MAFF, although the site itself is owned by English Nature. The 24ha site exhibits a wide elevation range, increasing from -1m ODN (approximately half tide) to +3.4m ODN (High Water Spring Tides). Prior to managed retreat the site was arable farming land, part of which was wheat and part used as set aside with clover. Photographs of the managed retreat scheme 18 months following the breaching of the flood defences are shown in Figure 13.7. The site is now being considered as an experimental NNR.

13.4.2 Objectives

The objectives of the scheme are:

- To retreat the line of coastal defence;
- To restore saltmarsh habitat for conservation purposes by breaching the existing flood embankment; and
- To investigate the re-establishment of natural inter-tidal processes and habitat.

13.4.3 Methodology

Four options were originally considered in the design of this managed retreat scheme:

1) Total removal of the embankment;
2) Two breaches in the embankment;
3) One breach to the west of the site; and
4) One breach to the east of the site.

The choice of option 4, the provision of a breach, rather than total removal of the embankment was necessitated for economic rather than physical reasons. Modelling studies undertaken by HR Wallingford revealed that the double breach option would result in the development of a circular flow within the site. A single breach was designed in the east of the flood embankment with dimensions of 50m wide at the embankment crest with 1:5 slopes at the base at 0.8m ODN (Figure 13.7).

Various experimental trials have been undertaken within the site to investigate the effect of different preparations on the evolution of plant and animal communities. Plots throughout the site have received the following soil treatments:

- Crop stubble left standing after harvest;
- Crop cleared to the ground; and
- Crop removed and soil ploughed.

Prior to breaching the embankment, the plots were flooded via sluices in order to raise the salinity of the sediments. Certain experimental plots were then seeded with glasswort and seablight and others were planted with plugs from natural saltmarsh surrounding the site containing sea aster and sea lavender.

On 8 August 1995, a single breach was made in the embankment to provide tidal input to the site. A drainage ditch was cut in the base of this breach to 0.2m ODN to connect with the existing creek system. A herringbone drainage system was constructed (Figure 13.7).

Existing hedgerows and trees were not removed, but were left to die naturally as a result of public pressure. A considerable fetch builds up across the site and the hedgerows have formed natural wavebreaks. The strip of saltmarsh from the front of the embankment at the breach site was transplanted to the side of the creek (Figure 13.7). The longest stage in the planning process was the application for a footpath diversion.

13.4.4 Construction Problems

The original drainage sluice in the embankment was plugged with clay, however, this later burst. In hindsight the sluice would have been completely sealed or removed altogether.

13.4.5 Monitoring

Intensive monitoring of the site has been undertaken since initiation of the scheme and results are expected to be available in early 1997. Monitoring comprises:

- Sedimentation;
- Soil chemistry;
- Tidal and wave characteristics;
- Vegetation;
- Bird counts; and
- Invertebrate studies.

A further monitoring exercise has been initiated by the Environment Agency in the creeks external to the site.

13.4.6 Results

Although much monitoring and experimental work has been undertaken, results are still being collated and have yet to be published. Preliminary observations suggest that considerable morphological changes have taken place as a result of the increased discharge into the site. The old creek system which was dissected by the construction of the embankment is now thought to be beginning to reform. Natural drainage creeks are forming along tractor treads and constructed drainage ditches.

There has been some criticism of the provision of a breach rather than the total removal of the embankment. Although the remaining flood embankments are not to be maintained and will eventually fail this may take several decades. In the meantime the tidal flows within the site are constrained and the difference in elevation between the breach and the minimum surface of the site means that the lower parts of the site act as a sump at low tide. This in turn means that sedimentation within the site will tend to be concentrated within the 'sump' area and higher areas will receive commensurately less material.

Results of the vegetation trials of seeding and planting saltmarsh species have not yet been published. However, high rates of sedimentation in the lower plots, adjacent to the breach, have covered the transplanted and sown vegetation with sediment, which has greatly reduced the value of such experiments. Microalgal growth can clearly be seen across the site and macroalgae have been observed growing in the vicinity of the main drainage channel.

Bird surveys and observations indicate that the site is used by a variety of birds including black-tailed godwit, dunlin, redshank, grey plover, brent geese and two egrets. During a visit to the site in December 1996 there was evidence to suggest that brent geese might have been feeding on algal growth (*Enteromorpha*) in the vicinity of the breach.

13.4.7 Reference Source

Ian Black, *pers. comm.*(1996)

Figure 13.7 Tollesbury managed retreat scheme, Blackwater Estuary (December 1996)

13.5 ORPLANDS MARSH MANAGED RETREAT SCHEME, BLACKWATER ESTUARY

13.5.1 Site Description and Background

Orplands forms part of the St Lawrence Bay on the south shore of the Blackwater Estuary (Figure 13.8 and 13.9). As at Northey Island, the sea wall along a 2km frontage at Orplands had become destabilised as a result of erosion of the saltmarsh. Consequences of this erosion were that the seaward toe of the defences became undermined, with loss of the concrete revetment blocks through the increase in wave energy and overtopping, causing scour of the crest and backslope. A cost-benefit exercise was carried out on a range of options, with managed retreat being the preferred solution on economic and environmental grounds. The scheme was implemented in April 1995 to create 40ha of saltmarsh. The land is being managed for nature conservation as part of an agreement between landowners and MAFF under the habitat scheme. Photographs of the site 18 months post-breaching are shown in Figure 13.9.

13.5.2 Scheme Objectives

To restore saltmarsh yielding a natural defence that will provide the following advantages:

- The creation of a saline flood plain that will reduce the effect of storm tides;
- The creation of a new important high level marsh that will be of great value to both overwintering and summer breeding birds and for immature fish;
- The provision of a public access route for quiet recreation (by footpath diversion);
- The reduction of pollution levels (as with all marshes); and
- The reduction of public expenditure by saving £525000 on conventional flood defence techniques (this habitat cost £60000 to construct).

13.5.3 Methodology

The first stage of construction was to dig a new ditch along the landward edge of the site to ensure land drainage from adjacent areas, with the resulting spoil placed and shaped, but not compacted, immediately to seaward of the ditch to form a new footpath route. Flapped 300mm pipes allowed drainage of freshwater onto the new salting. Two new un-reveted clay counter walls were constructed to the north and south of the site to prevent flooding onto adjacent areas, with a third relict counter wall extended to divide the site into two compartments.

A dendritic network of eleven creeks was constructed across the site using a JCB digger. The creeks were connected to the old borrow pit running along the landward side of the original sea wall which on open breaching became a tidal channel (Figure 13.9). The design of the creeks was based on the morphology of natural saltmarsh creeks in the surrounding area, for example the spacing of creeks along the main channel, creek length and the angle at which creeks branch at the channel. Creek dimensions were designed on the specifications of the digging equipment and were thus consequently 1m deep and 1m wide (Figure 13.9). Random gentle meanders were made in the creeks to act as internal wave breaks, reducing tidal energy and scouring. The edges of the creeks were not graded to specific elevations, but were left with cliffed vertical sides to allow natural shape adjustments with coastal processes. Material from the creeks was placed 5m to one side to break up internally-generated waves.

In April 1995, the deteriorating Orplands seawall was breached in two places, enabling inundation by saline water of the coastal grassland behind (Figure 13.9). The breaches were made in the seawall of each compartment and were designed to accommodate tidal flows without substantial erosion. The material from the breaches was placed to landward serve as a buffer to waves entering through the breach. The old seawall was left to decay through natural erosion.

The tall grass vegetation was left in place at the rear of the field to die and decompose naturally with saline water intrusion, providing an input of organic material to the soil and a sediment trap for the finer silt materials. The sediments were allowed to undergo natural colonisation by saltmarsh vegetation. The surviving tall grasses at the fringe of the saltmarsh area were left to allow a transition of vegetation communities as the saltmarsh evolves.

13.5.4 Monitoring

A comprehensive monitoring programme is being carried out for the first five years of the site development with the following elements being covered:

- Vertical accretion/erosion on foreshore and salting;
- Lateral erosion/progradation rates;
- Physical and chemical characteristics of the sediment;
- Water velocities over the marsh on very high tides;
- Changes in saltmarsh morphology;
- Plant vigour and productivity; and
- Value of the vegetation to grazing birds and other secondary producers.

13.5.5 Results

No results from the monitoring have yet been published but early indications suggest that there has been an overall gain in sediment on the new marsh surface, although the extent that this has resulted from re-distribution of sediments from the wave buffer bunds is unclear. The current velocities through the breach have been found to be substantially greater than originally predicted from numerical models, although the breaches themselves have remained largely stable. Widespread colonisation by halophytic vegetation has already started to occur naturally at all levels of the marsh, although the original arable areas have a greater density of plant colonisation than the grassland areas (possibly due to their relative elevations) (Figure 13.9). The site is already being well used by invertebrates, birds and mammals, and has been the subject of widespread public approval.

13.5.6 Time Period

April 1995, development in progress.

13.5.7 Reference Sources

Mark Dixon, *pers. comm.* 1996.

Dixon and Weight, 1995.

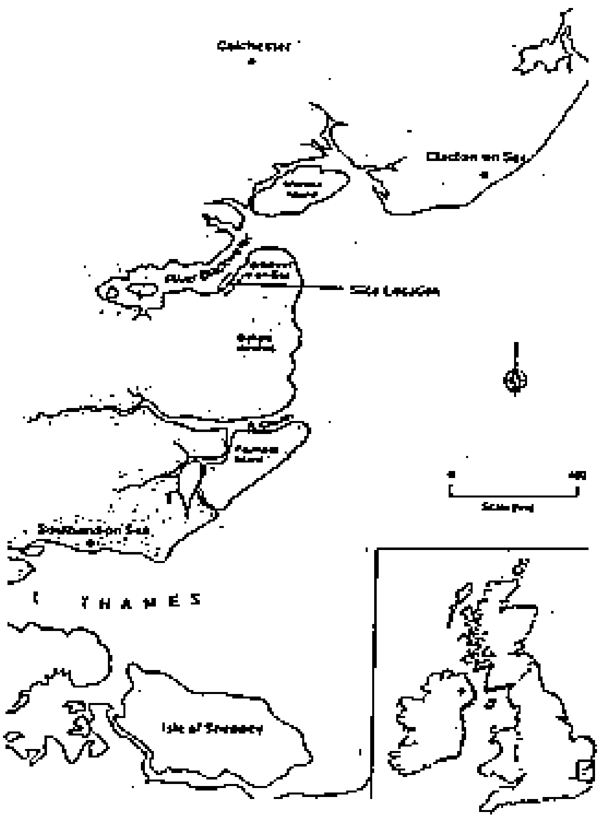

Figure 13.8 Location of Orplands Marsh managed retreat and Pewit Island intertidal recharge schemes, Blackwater Estuary

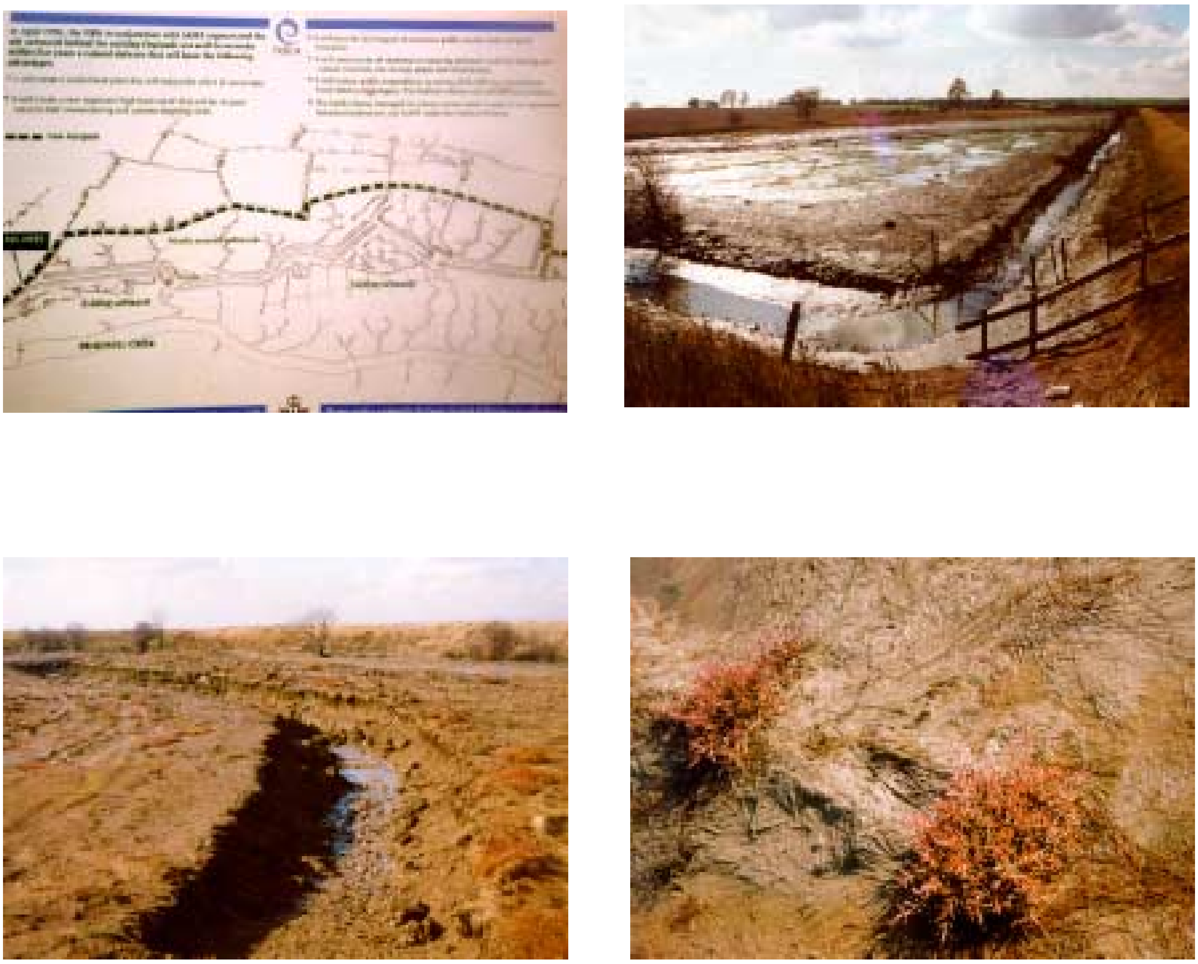

Figure 13.9 Orplands managed retreat scheme, Blackwater Estuary (October 1996)

13.6 ABBOT'S HALL (ST. LEONARDS) SALTMARSH RESTORATION SCHEME, BLACKWATER ESTUARY

13.6.1 Site Description and Background

Under the MAFF habitat creation scheme the owner of this 20ha area of arable land approached the NRA (Environment Agency) with a proposal to allow saltwater flooding and restoration of an area of saltmarsh. Abbot's Hall saltings are located on the northern bank of the Salcott Channel, a tributary of the Blackwater Estuary, Essex. Photographs of the scheme 18 months after construction are shown in Figure 13.10.

13.6.2 Scheme Objectives

The objective of the scheme is to regenerate saltmarsh habitat on arable land inside the flood defences in preparation for the future deterioration of the flood embankment and as a potential managed retreat site.

13.6.3 Methodology

The scheme was implemented in April 1996 by allowing saltwater to enter the site through two existing sluices (Figure 13.10), into a 2km network of creeks which were dug largely following contour lines and relic creeks. The bases of all creeks were dug to the same level as the sluices. Pipes have been attached to the two sluices that direct water across several creeks and mounds to a collecting pool in the centre of the site (Figure 13.10). The series of creek ditches and mounds have been designed to move water around the site. The pipe and sluice system was developed from a method first used at Horsey Island.

Because the sluices were low down in the tidal frame, and therefore vegetation propagules floating in the water column would be unlikely to enter the site naturally, a simple artificial seeding method was adopted. This method consisted of filling sacks with material gathered from the strandline of adjacent natural saltmarshes and strewing these by hand across the site, and has already resulted in colonisation by halophytic plants in the first growing season.

13.6.4 Monitoring

Small scale monitoring has been undertaken on water levels, vegetation and birds by MAFF under funding by ADAS.

13.6.5 Results

The creek network as constructed is already starting to change its configuration. In the first year the site is already well used by a wide range of breeding birds including redshank, oystercatcher, lapwing, wagtail and pipits, and an incoming rodent population has encouraged hunting by raptors. The site is also well used by a large number of hares. There are a range of proposals for experimental use of this site which will result in the manipulation of smaller sections for a variety of purposes, including use of the area as a small-scale analogy of an estuary system for hydrodynamic studies.

13.6.6 Scheme Improvements

It was found that although some areas of the site flood on every tide, others are rarely or never flooded. Work was undertaken in December 1996 to construct flaps on the inside of the sluice pipes at the collecting pool in order to retain water within the site over several tidal cycles (Figure 13.10). The objective of this modification was to increase water levels to cover the rest of the site and raise the water table.

13.6.7 Reference Sources

Mark Dixon, *pers. comm.*1996.

Ian Black, *pers. comm.* 1996.

Figure 13.10 Abbot's Hall saltmarsh restoration scheme, Blackwater Estuary (December, 1996)

13.7 BLAXTON MEADOW SALTMARSH RESTORATION SCHEME, SALTRAM (PLYM ESTUARY, DEVON)

13.7.1 Site Description and Background

In 1993 it was decided to recreate an area of saltmarsh under the MAFF Habitat Creation Scheme by allowing tidal flooding onto Blaxton Meadow in the Plym Estuary. A feasibility study was carried out in 1993 and the scheme was implemented in March 1995, with the capital works funded by the Environment Agency, area payments from MAFF and contribution of the land from the National Trust. There is a degree of freshwater discharge onto the site from the small catchment area behind.

13.7.2 Scheme Objectives

The aim of the scheme was to produce upper marsh plant communities and freshwater transitions, including a reedbed that could be used for water quality enhancement from a neighbouring sewage treatment plant.

13.7.3 Methodology

The scheme for flooding the site involved the use of tiered defences, with the front seawall largely retained apart from a spillway where the wall was lowered to a height of +2.4mOD. A series of tidal flap valves was also installed in the spillway to allow full evacuation of water from the site on the ebb tide.

Following problems of standing water on the site, it was decided to construct an embryonic creek network to allow the water to be drained from all areas. The network was designed by observing aerial photographs of natural saltmarsh systems to determine junction angles and creek densities, and fitting these to known ground contours on the site as far as possible. The creek network was excavated in December 1995.

13.7.4 Monitoring

Monitoring of the site has been undertaken, including sedimentation, tidal and wave characteristics, vegetation and bird counts. Although some monitoring of the site development has been established, it is hoped that this will be extended in the future.

13.7.5 Results

The scheme has had mixed success to date (i.e. one year of tidal flooding). There have been problems with vandalism of the tidal flaps which has caused significantly more flooding than predicted. Even for the period when the spillway structure was intact the site was also flooding more frequently than predicted; up to 48 inundations per year compared with the expected 18 inundations. Following the increased flooding onto the site through the vandalised tidal flap valves, additional water pressure was imposed on a headwall in the south west corner of the site, which subsequently collapsed. While the spillway structure was intact the site was flooding to the correct extent to produce the matrix of habitats anticipated in the feasibility study, with the water evacuating completely within one tidal cycle. However, despite the creek network there still appears to be ponding in the middle of the site (as in the case of Northey Island), possibly requiring additional work on the creeks to facilitate more complete drainage.

Some vertical erosion has occurred on the site immediately adjacent to the spillway where the water velocities are eroding the surface. Coarse sediments are being deposited in the area around the spillway and in the adjacent ditches. Compaction by construction vehicles has reduced erosion of the foreshore in front of the spillway, and thus hindered natural creek formation here.

After one year of tidal flooding, the frequently flooded areas are covered to a large extent by a mat of algae. On the higher ground, that is flooded less frequently, *Agrostis stolonifera* is dominant, with some *Juncus effusus*. It was hoped that the halophytic species that were originally present in the ditch behind the flood bank would provide propagules for colonisation of the rest of the site. However, erosion of this ditch appears to have contributed to removal of this vegetation, with the result that sources of the propagules are now much further distant, and colonisation may take longer than expected. On the flood bank *Spergularia media* and *Cochlearia danica* have colonised, and additional strandline species such as *Atriplex hastata* and *Beta maritima* are found in higher areas on the rest of the site. This suggests that further colonisation will occur when the flooding regime has become more stabilised.

It appears that use of the site by curlew is lower than expected, but this may be due to the abnormally cold weather of the winter of 1995/96. However, significantly more wildfowl (mainly shelduck and mallard) are using the site than previously. There has also been an increase in the number of predatory birds such as grey heron, little egret and kingfisher, suggesting the presence of a fish population in the ditches and possibly the standing water areas.

13.7.6 Possible Improvements

Further management measures have been proposed in order to enhance the chances of successful habitat creation. These include raising the effective level of the spillway by 0.1m with the boards which were to be used to prevent flooding in an emergency. The aim of this measure is to increase the influence of the small freshwater discharge and produce more freshwater transition plant communities. An interim method of plugging the pipes is to be used in the event of further vandalism in order to reduce the number of uncontrolled inundations before repairs can be made. Possibilities for enhancing natural colonisation of vegetation are being investigated, including the use of cattle on the site to break up the algal mat. A derogation on the Habitat Scheme has already been granted to allow light grazing by ponies on the eventual saltmarsh, but this will not be permitted until an appropriate community structure has been established.

13.7.7 Reference Source

Bell, 1996.

13.8 HORSEY ISLAND SALTMARSH RESTORATION AND INTERTIDAL RECHARGE SCHEMES, HAMFORD WATER

13.8.1 Saltmarsh Restoration Scheme: Site Description And Background

The Walton Backwater seawall on Horsey Island, Hamford Water, disintegrated in 1953 and was abandoned as there was no cost benefit in maintaining it. The area it was protecting, however, has extensive leisure, agricultural and conservation interest. A number of experimental saltmarsh restoration and recharge schemes have been undertaken at various sites on Horsey Island for flood defence and nature conservation purposes by the Environment Agency and English Nature. One scheme investigated saltmarsh restoration using a sluice and pipe system to reintroduce tidal inundation to a small area of arable land.

13.8.2 Saltmarsh Restoration Scheme: Scheme Objectives

The objective of this experimental scheme was to restore saltmarsh habitat on previously reclaimed arable land for flood defence and nature conservation purposes.

13.8.3 Saltmarsh Restoration Scheme: Methodology

This scheme adopted a simple approach to saltmarsh restoration by opening a sluice to allow intermittent flooding of the site in order to encourage warping up of the surface and colonisation by halophytic plant species. The methodology developed for this approach to saltmarsh restoration in this scheme has acted as a template for later schemes, including the Orplands scheme (see Section. 13.5).

13.8.4 Saltmarsh Restoration Scheme: Results

Colonisation of the site by saltmarsh vegetation was extremely slow here due to the relative level of the sluice in relation to the height of propagules floating in the water column. An improvement made in the Orplands scheme to overcome the slow rate of colonisation was the seeding of the site prior to inundation.

13.8.5 Intertidal Recharge Scheme: Site Description and Background

The Horsey Island experiment was the first trial estuarine intertidal recharge scheme undertaken in the UK in 1990. The site is located on the north east corner of Horsey Island.

13.8.6 Intertidal Recharge Scheme: Scheme Objectives

The objectives of the scheme were as follows:

- To recharge the intertidal area of Horsey Island with coarse dredged material to protect Walton Backwater from erosion and flooding;
- To conserve and enhance the saltmarsh habitat for wildlife; and
- To provide a beneficial use of dredged material.

13.8.7 Intertidal Recharge Scheme: Methodology

Investigations of waves, tidal currents and sediment transport processes were undertaken in order to determine a suitable recharge site in Hamford Water. Based on modelling studies and consultations with local communities, it was decided to use coarser recharge sediments than those naturally present at the recharge site for the following reasons:

- Coarse materials are more stable and less susceptible to erosion than fine sediments and would therefore be more resistant to the pronounced north to south sediment transport pathway and strong flood tide currents identified at the recharge site during modelling studies;

- This stability reduces the risk of material being moved offshore and lost from the recharge site, ensuring that sediment movement is onshore;

- The option of using fine sediments caused concern among local communities over the potential for the smothering of adjacent benthic communities. Damage to sensitive shellfish beds was of particular concern to the local fishing industry; and

- The impact of the potential redistribution of sediments on navigation was also considered.

A total of 18000m^3 of dredged material taken from nearby Harwich Harbour was sprayed on to the mid intertidal outside the abandoned seawall by rainbow discharge from a self-load, self-empty discharge vessel at high water on spring tides. The material was used to fill the gaps between a line of grounded barges that act as a wavebreak (Figure 13.12i). The recharge mixture of coarse sands, pebbles and grits were considerably coarser than the fairly uniform muds at the recharge sites.

13.8.8 Intertidal Recharge Scheme: Monitoring

Pre-recharge studies were made of the suitability of the dredged material and the Horsey Island recharge site including pollution studies, sediment grading curves, bathymetric studies, benthic surveys, and wave and current modelling.

All of the above mentioned surveys were repeated after sediment placement.

13.8.9 Intertidal Recharge Scheme: Results

Monitoring revealed a complete change in sediment character of the site due to recharge using coarser materials. Foreshore bathymetry changed slightly in localised areas. There was no increase in contaminants in the sediments at the site.

Benthic surveys revealed marked changes in the marine invertebrate communities colonising the sediments. Abundance of typical mud dwelling communities were initially reduced due to smothering. The altered composition of the sediments resulted in colonisation by species associated with coarser materials that were not found elsewhere on the site.

Currently, the Horsey Island intertidal recharge scheme is considered to be one of the most successful projects of its type undertaken in the UK to date. The dredged material has become "rapidly colonised" with benthic invertebrates, particularly king ragworm (*Nereis virens*), which support a thriving bass fishery and bird populations. The site is widely recognised for its ornithological importance. A new marsh habitat is now developing behind the recharged material.

13.8.10 Saltmarsh Recharge: Site Description and Background

The scheme site is located on the south-east corner of Horsey Island, near Walton-on-the-Naze, Essex. The site was suffering from vertical erosion of the saltmarsh. Work was undertaken by the contractor John Negus of PVW (now part of Westminster Dredging).

13.8.11 Saltmarsh Recharge: Scheme Objectives

To raise the level of the saltmarsh in order to combat vertical erosion:

- To ascertain the viability of spraying dredged silt directly onto the saltmarsh vegetation and to determine whether low saltmarsh community can be regenerated into high marsh communities; and
- To provide a beneficial use of fine dredged material.

13.8.12 Saltmarsh Recharge: Methodology

Half a load of dredged silt from Harwich Harbour was deposited onto a 0.5ha plot (above mean high water mark) of heavily grazed saltmarsh. The fine material was sprayed onto the saltmarsh from a direct pipe connected to a self-load suction dredger with the capacity for rainbow discharge (Figure 13.11). The spray extended 50m from the nozzle and the depth of the deposited spoil is not known.

13.8.13 Saltmarsh Recharge: Time Period

The work was carried out in August so that consolidation of the sediment could take place before the main release period for saltmarsh plant seeds and the arrival of overwintering bird species, and after the use of the saltmarsh by breeding birds.

13.8.14 Saltmarsh Recharge: Practical Problems

The nozzle discharging the silt was fixed so that the boat had to be moved in order to alter the direction of the spray. This proved to be difficult in a narrow creek.

13.8.15 Saltmarsh Recharge: Monitoring

None

13.8.16 Saltmarsh Recharge: Results

It was originally thought that most of the applied silt was washed off the sites over the first set of spring tides. However the actual amount of silt washed away is not known as there was no pre or post monitoring of surface levels. There was no apparent loss of plant cover however this does not imply that there would not have been if more sediment had remained on the marsh surface. Some sediment remained in depressions on the saltmarsh surface and was quickly covered by vegetation (it was thought to be colonisation by new plants rather than regrowth of original species).

13.8.17 Saltmarsh Recharge: Possible Improvements

More sediment may have been retained if it had been initially bunded e.g. hazel fencing or with rolls of coconut matting which may also have enhanced further accretion.

13.8.18 Reference Sources

IECS, 1993a

Dixon, 1992

Mark Dixon, *pers. comm.* 1996.

Ian Black, *pers. comm.* 1996.

Carpenter and Brampton, 1996.

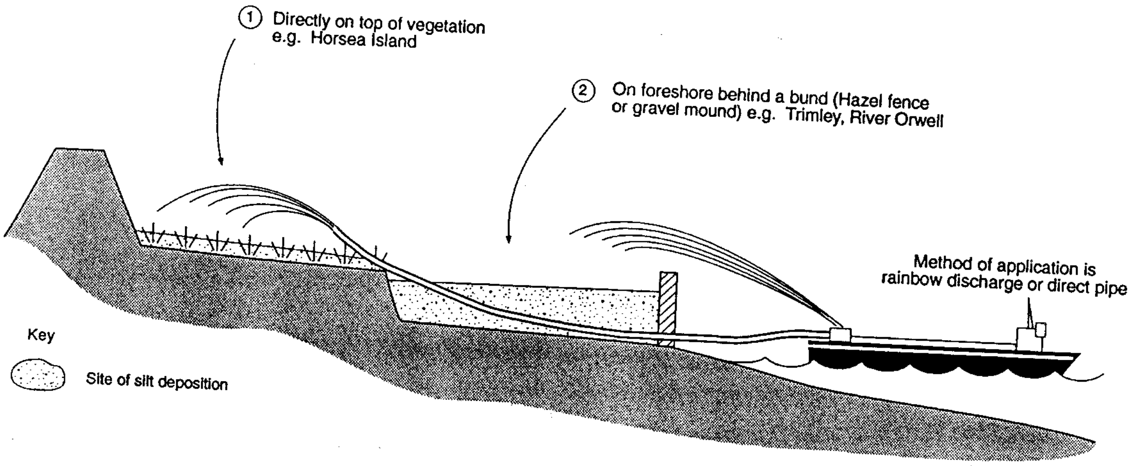

Figure 13.11 Schematic representation of intertidal recharge using fine dredged material at Horsey (Horsea) Island and Trimley Marshes (Carpenter and Brampton, 1996)

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Figure 13.12 Intertidal Recharge Schemes

i) Horsey Island, Hamford Water, and
ii) Pewet Island, Blackwater Estuary interitdal recharge schemes

13.9 PEWIT ISLAND, BLACKWATER ESTUARY INTERTIDAL RECHARGE SCHEME

13.9.1 Site Description and Background

Pewit Island is located offshore of Bradwell, Blackwater Estuary, Essex (Figures 13.8 and 13.13ii). The recharge site is located on the southern tip of the island. Pewet Island provides a natural wavebreak to a seawall that protects 67ha of land. This land's assets amount to millions of pounds. The erosion of the island would lead to large capital costs in the next 50 years as the seawall would only protect against a 1 in 15 year event if the island was lost. Trials have been underway by the Environment Agency to provide protection against erosion of the island by recharging intertidal flats with coarse dredged material. The contractor for the scheme was Westminster Dredging. Photographs of the site 20 months after the second application of dredged material are shown in Figure 13.13.

13.9.2 Scheme Objectives

To recharge the eroding foreshore of Pewet Island using coarse dredged material to combat further saltmarsh erosion and afford protection to the infrastructure and land behind it.

13.9.3 Methodology

A total of 5000m^3 of coarse dredged material was placed on the intertidal foreshore of the Island in two phases. Non-cohesive sediment was used with a wide grading curve (this is less expensive than a single aggregate size and offers more habitat diversity). The gravel was from maintenance dredging and was supplied free of charge by Harwich Haven Authority. A self-load suction dredger with the facility of rainbow discharge was used to place the material.

13.9.4 Time Period

The sediment was deposited in two phases; 2529m^3 in December 1992 and a further 2646m^3 in February 1995. It was carried out in the winter to avoid breeding bird season and recreational boating.

13.9.5 Monitoring

A monitoring programme was established that included both pre and post scheme monitoring.

Pre-scheme:

- The rate of erosion was determined by taking measurements from a time series of Ordnance Survey maps and 1:5000 aerial photographs;
- The tidal currents in the deposit area were established using a Valeport velocity meter at four stations taking readings in the time interval 1 hour before to 1 hour after high water on a spring and neap tide;
- The existing bathymetry was measured across an area 2km long and 0.5m wide around the southern tip of the island; and
- Topographic surveys of the nourishment site (the site was re-levelled after deposition and at six monthly intervals).

Post-scheme:

- Pre-scheme measurements are repeated every 6 months; and
- Bird use will be monitored for two years by English Nature.

13.9.6 Results

A schematic diagram of the movement of the mound of sand/gravel material up the intertidal is shown in Figure 13.14. Lateral erosion of the saltmarsh cliff has stopped in areas that are now protected by the sand/gravel bank. The bank provides nesting and roosting sites for birds. Nest sites for 20 pairs of little tern, 6 pairs of oystercatcher and 8 pairs of ringed plover have been recorded. An increase in the diversity of invertebrate species colonising the coarse materials has been recorded 18 months after recharge.

However, these coarse recharge sediments support a reduced invertebrate biomass and therefore there is a decrease in the potential food supply to waders. Some parts of the existing saltmarsh have been smothered due to roll back of the sand/gravel ridge over the edge of the saltmarsh (Figure 13.13). Fine materials are being deposited over the top of the coarser recharge materials in the lower intertidal (Figure 13.13).

The structure of fish communities has changed with flounder being replaced by bass and sole that favour the coarser sediments.

13.9.7 Possible Improvements

Roll-back of the gravel bank (and thus vegetation smothering) may have been prevented if the initial profile of the bank had been sufficiently high and broad crest to prevent wave overtopping.

13.9.8 Reference Sources

Mark Dixon, *pers. comm.* 1996.

Carpenter and Brampton, 1996.

Figure 13.13 Pewit Island intertidal recharge scheme, Blackwater Estuary (October, 1996)

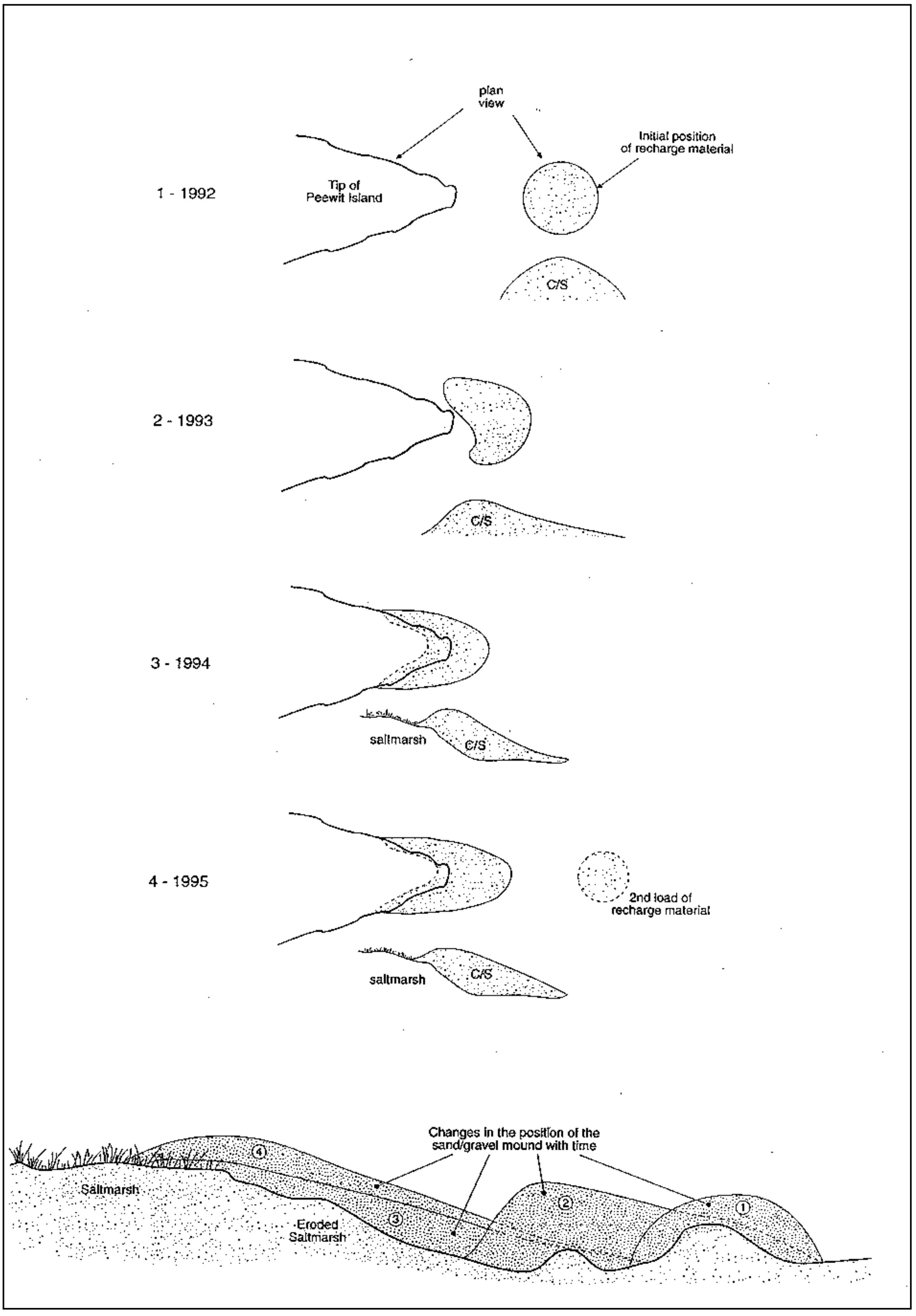

Figure 13.14 A schematic diagram of the movement of material following intertidal recharge at Pewit Island, Blackwater Estuary

13.10 TRIMLEY MARSHES INTERTIDAL RECHARGE SCHEME, ORWELL ESTUARY

13.10.1 Site Description and Background

Trimley Marshes are located on the east bank of the Orwell Estuary, Suffolk, near the Harwich Harbour which is situated at the confluence of Stour and Orwell Estuaries. These saltmarshes are experiencing erosion and in response the Environment Agency have funded intertidal recharge schemes which have been undertaken by Westminster Dredging.

13.10.2 Scheme Objectives

The aim of the intertidal recharge scheme in Trimley Marshes is to protect the shoreline from erosion from ship wash, whilst providing a beneficial use for dredged material.

13.10.3 Methodology

A total of 63000m^3 of dredged sands derived from Harwich Haven Authority have been placed in soft groynes perpendicular to the eroding shoreline (Figure 13.15). Dredged material was rainbowed directly onto the intertidal mudflats. Hopper placement was considered as an alternative technique.

Following the placement of the coarse materials it was planned to infill the area in between the groynes with fine grained silt dredged materials with the aim to provide a habitat of wildlife value.

Dredged materials consisting of fine muds and silts taken from Harwich Harbour were rainbowed onto the foreshore fronting Trimley marshes (Figure 13.11). Hazel fencing and straw bales were used to retain the material in place.

13.10.4 Reference Source

Mark Dixon, *pers. comm.* 1996.

Figure 13.15 Timeley and Parkstone Marshes intertidal recharge schemes, Stour and Orwell Estuaries

13.11 PARKSTONE MARSHES INTERTIDAL RECHARGE SCHEME, STOUR ESTUARY

13.11.1 Site Description and Background

Parkstone Marshes are located in the east of Copperas Bay on the south bank of the Stour Estuary, Essex (Figure 13.15). The site consists of intertidal mudflats backed by degraded saltmarsh. The intertidal recharge scheme was funded by the Environment Agency and work was contracted to Westminster dredging.

13.11.2 Scheme Objectives

- To prevent the erosion of the saltmarsh affording protection to the infrastructure behind.
- To restore the habitat for nature conservation.

13.11.3 Methodology

A total of 250000m^3 of dredged sands from Harwich Harbour were placed on the intertidal mudflats, raising them approximately 2m in height. The sediment was placed on the site by rainbow discharge.

13.11.4 Monitoring

Monitoring of the shore profile and sediments was undertaken one year pre-construction and continued for five years post-construction in the spring and autumn each year. Extensive ecological monitoring of the benthos was carried out.

13.11.5 Results

The scheme is considered to be highly successful as the erosion of the foreshore has been arrested and the wetland is naturally being restored. Within two years, a diverse benthic community has recolonised the dredged material. Replacement with coarser material causes a change in invertebrate species and communities which exhibit increased diversity, but reduced biomass.

13.11.6 Reference Source

Mark Dixon, *pers. comm.* 1996.

13.12 MEDWAY SALTMARSH REGENERATION SCHEME

13.12.1 Site Description and Background

A more recently undertaken intertidal recharge scheme was conducted in the summer of 1996 in the Medway Estuary. The recharge site is located on the intertidal flats around Bedlams Bottom in the Funton Creek (upper Stangate Creek), a tributary of the Medway Estuary. The experiment was jointly funded by Medway Ports, MAFF's Flood and Coastal Defence Division, the Environment Agency and English Nature.

13.12.2 Scheme Objectives

The objectives of the Medway experiment was to dispose of fine dredged material within an area of outstanding nature conservation interest. With the aim of retaining the dredgings within the active process zone in a manner that is not harmful to the environment.

13.12.3 Methodology

A total of 4000m^3 of fine dredged materials taken from Cadnam Basin were placed on the lower intertidal by split bottom barges and were left for natural hydraulic processes to gradually move it up the foreshore (trickle charge/feed). This approach is reported to enable the sediments to be redistributed within the intertidal system and promote the natural evolution of intertidal habitats.
Monitoring

Tracer studies were used to identify dredged sediment transport paths and deposition within the Medway and were undertaken by Environmental Tracing Systems Ltd. Fluorescent orange tracer with similar physical properties to the dredged sediments was used to track sediment movement for several weeks following recharge. The data from these studies gave an indication of the proportion of dredged material that was retained at the site.

13.12.4 Results

Early results from this experimental recharge scheme indicate that bottom dumping and trickle feeding is a success for relatively small infrequent volumes of material. Approximately 50% of the material was estimated to have been retained at the recharge site in Bedlams Bottom.

13.12.5 Reference Sources

J. Pethick. *pers. comm.* 1996.

14. CONCLUSIONS

Due to the wide variation in monitoring and evaluation techniques used, and the different nature of each of the creation /restoration projects, comparison of the successes of the various projects conducted by different organisations can be difficult. A successful creation scheme, in terms of the provision of wildlife habitat and populations, has been classified by the US Fish and Wildlife Service (1989) as one that has apparently accomplished a major part of its intended purpose. A scheme is considered marginally successful if it has accomplished a moderate part of its objectives and unsuccessful if it accomplished only a minor part.

The main components of habitat creation and establishment are summarised as follows.

14.1 PHYSICAL FEATURES AND PROCESSES WHICH SHAPE THE CREATED HABITAT

- **Tidal inundation** - the hydrodynamic properties of a particular location will ultimately control the volume of energy input into the site. The spatial distribution of tidal energy dissipation is a primary consideration in determining how the morphology of a particular site will evolve with time. The complexity of these systems derives from the fact that the changing morphological parameters produce feedback, which can result in the modification of the hydrodynamics within the system.

- **Wave climate** - the wave energy input will have an influence on the morphological form present at a particular site. The relative impact of waves on the morphology of a particular site will be largely dependent on the fetch characteristics and geometry of the site.

- **Periodicity** - The tide and wave energy inputs form a complex system of cyclic and episodic events. The relative impact of various events will be controlled by their magnitude and frequency of occurrence.

- **Sediment supply** - the morphological state at which a particular site exists will also be dependent on the sediment budget available. For instance a tidal creek may be designed in such a way as to provide conditions suitable for fine sediment deposition, yet if the material is not available in suspension, morphological evolution will be extremely slow. This problem may be overcome by the introduction of sediments into the system.

- **Area** - the final design of a particular scheme will be influenced by the spatial extent available.

- **Entrance area** - for creek formation, the environment within the intertidal will be dependent on the cross sectional area of the entrance. This controls the amount of tidal energy entering the system, as well as the sediment budget. Previous schemes have sought to control certain parameters at the mouth, for example, the reduction of wave energy to reduce the likelihood of wave induced erosion.

- **Topography/geology** - particular sites may be controlled by their topography or geology. Such site properties will impose constraints on the initial form and subsequent erodibility, so influencing morphological evolution.

- **Physical buffer zones -** the provision of space to allow for the expansion and adjustment of intertidal communities in response to changing hydrology, hydrological and other external forces, to avoid the loss of habitat area, and to allow for a certain amount of uncertainty inherent in the prediction of habitat development.

14.2 PHYSICAL PARAMETERS AFFECTING THE ESTABLISHMENT OF PRIMARY COLONISERS

For a newly created habitat, primary colonisers are the first organisms to establish. They include microbial, algal, invertebrate and plant species and are important for their role in stabilising the newly created habitat and for acting as precursors to the full development of the intertidal ecosystem.

- **Intertidal morphology -** includes aspects of topography, area, and gradient of intertidal profile. The morphological characteristics of a habitat creation scheme directly affect the physical processes occurring within a site. These in turn are of direct relevance to the ecological function and structure of the created habitat.

- **Elevation -** determines the frequency and duration of tidal inundation and exposure to wave action. This affects the level at which primary colonisers occur on the intertidal profile according to their tolerance to osmotic stress and the period they require without inundation for establishment.

- **Exposure -** occurs as a function of wave action, tidal currents and fetch. Exposure affects the level at which erosive processes occur on the intertidal profile and the level at which primary colonisers are able to establish.

- **Sediment composition -** includes particle size, nutrient and organic content. Habitat requirements of primary colonisers determine the species most likely to occur in a created habitat of given sediment characteristics.

- **Rates of sediment consolidation -** determines the rate at which primary colonisers can establish according to their requirements for substrate stability.

- **Water quality -** includes variation in salinity, levels of dissolved oxygen, water turbidity, organic content and pollutants. Species colonising newly created habitat will vary according to their tolerance and/or preference for the listed parameters.

- **Hydrology -** includes freshwater influences from precipitation, run-off and ground water influences affecting species that colonise the created habitat according to their tolerance of low and variable salinities.

- **Existing flora and fauna -** will determine the sources and types of species available for the natural recolonisation of newly created intertidal habitat.

14.3 ECOLOGICAL CHARACTERISTICS OF THE SITE NECESSARY FOR USE BY HIGHER CONSUMERS

- **Microbial and microalgal communities -** characterised by species diversity, productivity, distribution and rates of colonisation. These communities are important in the development of unconsolidated sediments, acting as precursors to saltmarsh development and are fundamental to the intertidal food chain.

- **Invertebrate fauna -** characterised by species diversity, distribution, productivity and rates of colonisation. The use of the created habitat as a food resource by birds and fish depends upon their prey preference in relation to the characteristics of the invertebrate communities present.

- **Vegetation -** characterised by species diversity, distribution, rates of colonisation, density of cover, and productivity. Vegetation is important for stabilising sediments, the provision of detritus, refuge habitat for intertidal fauna, nesting habitat for bird species.

- **Buffer zones -** defined as areas that act as zones of protection between the fragile newly created habitat and urban areas, providing a refuge for plants and animals between their preferred habitat and human activities.

- **Disturbance factors -** typified by visitor pressure, construction works, noise and light pollution. Such factors require minimisation or elimination to ensure maximum use of created habitat by higher consumers.

- **Environmental corridors -** characterised by availability of naturally occurring propagules (seeds, eggs and larvae) and primary colonisers. Defined as environmental linkages between created habitats and surrounding natural habitats, reducing the isolation of a newly created habitat and facilitating colonisation.

From the review and case studies it is clear that a great number of intertidal habitat creation schemes have been undertaken with varying degrees of success, and have included saltmarsh, intertidal mud and sand flats, lagoon systems, islands and tidal marsh systems. Failures have been attributed to poorly developed designs that have not considered all features necessary to achieve the final objectives. Whilst documented schemes have addressed certain components, few if any have considered the contributory affects of all the components at the design stage. Such a consideration is essential to establish a successful self-maintaining system. Given the information that is available on individual components, it is possible and necessary to consider all of the components together. Such consideration is, however, highly dependent on obtaining site-specific information.

Given the importance of the specific physical conditions of each site in determining the eventual ecology of the created habitat, pilot studies and demonstration schemes are useful in reducing uncertainties at specific locations, although time scales rarely permit such studies to proceed prior to the main project. In particular pilot studies might consider rates of sediment accretion, sediment consolidation, and microbial and invertebrate colonisation. These are features that have been poorly studied in previous habitat creation schemes. The information gathered from such pilot studies, in conjunction with a detailed design methodology, will enable suitable mitigation to be designed and implemented with confidence.

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APPENDICES

APPENDIX I

DEFINITION OF TERMS

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DEFINITION OF TERMS

Concepts and Definitions

The following definitions are based on those given in the Collins English Dictionary and Lewis (1990), unless otherwise stated.

- **Creation - to bring into being; to originate, design or invent**
  Creation is an action of man which converts a non-wetland habitat into a wetland habitat where there has not been one within recent history, a century for example. The term "non-wetland and wetland" in this definition from Lewis can be substituted to read "intertidal" for the purpose of this review. There are two types of created habitats, artificial and man-induced. Artificial habitat require the continuous or persistent activity of man for it to remain in existence, and without maintenance will return back to its original habitat type. Man-induced habitat creation is the more common type of scheme, whereby a habitat results from a single action by man and persists on its own as a self-perpetuating system. Wetlands created by multiple placements of dredged materials over a period of time are an acceptable synonym.

- **Restoration - to bring back to a former or normal condition, as by repairing or rebuilding**
  In contrast to creation, restoration refers to the return to a pre-existing condition. In the context of this report it refers to the return of a coastal habitat from a disturbed or totally altered condition to a previously existing natural condition, or altered condition, by man. Such human actions include fill removal, planting of vegetation, alteration of tidal channels or tidal regimes. According to Lewis (1990) it is not necessary to know the pre-existing conditions of the natural habitat, but only to know what habitat type was there, for example saltmarsh, and to return it to that same habitat type. Others consider that restoration does require a return to the exact pre-existing conditions and is therefore rarely achieved (Zedler, 1984).

- **Recharge - to charge again; Charge - to load or fill with required material**
  In the context of this review, recharge refers to the placement of sediment on the foreshore to enhance the natural ability of mudflats and marsh systems to respond to the controlling physical processes (NRA, 1995). By this definition, sediment recharge of intertidal areas can be viewed as a restoration process (i.e. a return to a pre-existing condition) and will be included within this term throughout this review. The recharge of intertidal areas with dredged material provides a beneficial use of the material (Paipai, 1995).

- **Mitigation - to make or become milder, less severe or less painful**
  Lewis (1990) defines mitigation as "the actual restoration, creation, or enhancement of wetlands to account for permitted wetland losses". For the purpose of this review the term mitigation is limited to the creation, restoration or recharge of coastal wetlands and more specifically intertidal habitat to account or compensate for intertidal losses. In the United States the concept of mitigation banking has developed, whereby wetland creation or restoration is undertaken expressly for the purpose of compensation for losses from future developments as part of a credit programme. Mitigation may be undertaken on a voluntary basis or may be led by regulations and licence/consent agreements.

- **Success - a favourable outcome**
  In general terms success is the achievement of established goals. With regard to habitat creation and restoration, success is the achievement of the specific objectives of a project, although this requires that success criteria are established prior to construction. It is interesting to note that a project may have failed to meet its set objectives, but still provide other values which may result in the failure of the project but the overall success of the habitat site. A discussion of the evaluation of success is presented in Section 2.4.

APPENDIX II

CASE STUDY SUMMARIES

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CASE STUDY SUMMARIES

Abbreviations used in the summaries

Approx.	Approximately
CSA	Continental Shelf Associates
EA	Environment Agency
FDOT	Florida Department of Transport
ha	Hectares
km	Kilometre
LW	Low Water
m	Metre
MLLW	Mean Lower Low Water
NAP	Nieuw Amsterdams Peil = Amsterdam Ordnance Datum
NERR	National Estuary Research Reserve
NVGD	National Geodetic Vertical Datum
TPA	Tampa Marine Institute
WDFW	Washington Department of Fish and Wildlife
WTRP	Winchester Tidelands Restoration Project

Reference No.	**1.**
Site name	**Blackwater Estuary, Various sites including West Mersea, Tollesbury and Old Hall Point**
Site type	**Intertidal Recharge**
Contractor	Westminster Dredging
Funding body	EA (Environment Agency)
Purpose	Flood/coastal defence and beneficial use.
Objectives of the scheme	To protect foreshore from erosion by recharging intertidal and providing a beneficial use for dredged material.
Description of site	Location: Various intertidal recharge schemes using coarse material have been undertaken in the Blackwater Estuary including: a) West Mersea, on the southern foreshore of Cobmarsh Island b) Old Hall Point, at the Mersea Quarters Spit at the mouth of Tollesbury Fleet
Methodology	a) 6500m^3 of dredged gravels was placed on the intertidal of the Cobmarsh Island, West Mersea. b) 6500m^3 of dredged gravels were pumped to saltmarsh (spring tide) level at Old Hall Point.
Technical details	a & b) Dredged material was rainbowed onto the intertidal and migrated landward by natural wave action. b) Material was transported in 8 dredger loads and construction took approximately 6 hours to complete
Maintenance/further development	b) The spit at Old Hall Point is now reported to be restored to its 1954 location and protection has been afforded to the Tollesbury marshes behind.
Reference source	Dixon, M. 1996. Foreshore Recharge (Explanatory Photographs). EA. M. Dixon, EA, pers. comm. 1996.

Reference No.	**2.**
Site name	**Blackwater Estuary, Essex, UK**
Site type	**Subtidal Recharge**
Contractor	Westminster Dredging
Funding body	EA
Purpose	Beneficial use and habitat creation.
Objectives of the scheme	To create artificial habitat for oysters whilst providing a beneficial use of dredged material.
Description of site	Location: Mid-channel of the Blackwater Estuary, near Mersea Marina, Essex, UK.
Methodology	1000m^3 of coarse gravel material was placed on the estuary bed to provide resistance and friction to water flow, forming an artificial habitat for oysters. Wild oyster populations colonised area and further stocking and management was undertaken by local fishermen.
Technical details	Dredged material was rainbowed onto the estuary floor and was supervised by oyster fishermen.
Scheme success/failure	Oyster populations are reported to be thriving.
Reference source	M. Dixon, EA, pers. comm. 1996.

Reference No.	**3.**
Site name	**Naze North, Essex, UK**
Site type	**Subtidal Recharge**
Contractor	Westminster Dredging
Funding body	EA
Purpose	Coastal Defence and beneficial use.
Objectives of the scheme	Creation of an offshore berm using dredged material to enhance defence against flooding.
Methodology	77000m^3 of dredged material was placed on the seabed to create an offshore berm.
Reference source	M.Dixon, EA, pers. comm., 1996.

Reference No.	**4.**
Site name	**Sheep Island, Maine, USA**
Site type	**Created Intertidal Mudflat**
Contractor	US Army Corps of Engineers
Funding body	US Army Corps of Engineers
Purpose	Beneficial use of dredged material.
Objectives of the scheme	To provide a beneficial use for dredged material by the creation of an intertidal mudflat around Sheep Island, Maine.
Description of site	Location: Sheep Island near Jonesport Harbour, Maine. Situated approx. 50 miles south of the US/Canada border. Area: 3 acres. Description: The island is protected from ocean swells and storm waves and is approx. 300m in length. Tidal range: Approx 12 feet.
Methodology	100000 cubic yards of silty sands derived from maintenance dredge placed on a 3 acre area of shallow subtidal sand and gravel surrounding the island.
Timescale	1988 (no other info)
Monitoring programme	Monitoring was initiated in Sept. '90, repeated Sept. '91 and August '92. Control site at Beals Island. Sediment: Grain size distribution and organic content. Ecology: Clams, baitworms and infaunal community structure.
Scheme success /failure	A diverse infaunal community had developed on the created intertidal mudflats within 2 years and abundance generally increased in the fourth year after construction. Differences between reference site attributed to sediment characteristics and seasonal variability of individual species. Clams and bait worm species of economic importance colonised in substantial numbers and sizable populations present after years.
Reference source	Ray GL, Clarke D, Wilber P, and Fredette TJ. 1994. Ecological evaluation of mud-flat habitats on the coast of Maine constructed of dredged material. Env. Effects of Dredging, Vol D-94-3, US Army Corps of Engineers, Waterways Exp. Station.

Reference No.	**5.**
Site name	**Beals Island, Jonesport**
Site type	**Created Intertidal Mudflat**
Purpose	Beneficial use of dredged material.
Description of site	Location: Beals Island, Jonesport, Maine, USA - 50 miles south of US/Canada border (see map). Description: Area protected from ocean swells and storm waves. Tidal range: 12 feet.
Timescale for scheme	Mid 1960s.
Monitoring programme	Monitoring was initiated in 1992. Sediment: Grain size analysis. Ecology: Soft shell clams, baitworms and infaunal community.
Scheme success/failure	Results led to further plans at Sheeps Island.
Reference Source	Ray GL, Clarke D, Wilber P, and Fredette TJ. 1994. Ecological evaluation of mud-flat habitats on the coast of Maine constructed of dredged material. Env. Effects of Dredging, Vol D-94-3, US Army Corps of Engineers, Waterways Exp. Station.

Reference No.	**6.**
Site name	**Black Rock Harbour, Connecticut, USA**
Site type	**Created Wetland / *Spartina* Saltmarsh**
Contractor	US Army Engineers
Funding body	US Army Corps of Engineers / Environmental Agency
Purpose	Beneficial use of contaminated dredged material, experimental.
Objectives of the scheme	To provide a beneficial use of contaminated dredged material by the creation of a saltmarsh habitat. Part of a study to evaluate contaminant mobility in alternative disposal sites (upland, aquatic and wetland).
Description of site	Location: Tounge Point, Connecticut. Area: 7,060m^2. HW depth: Approx. 0.3m.
Methodology	Dredged material was dumped over an area of 7,060m^2 and was excavated to reach the desired elevation. A weir was constructed to allow an interchange of tidal flow and at high tide the water depth was approx. 0.3m. Half the site was planted with native *Spartina alterniflora* transplanted from 650m^2 of nearby marshland and half planted with *Spartina* provided by a commercial producer.
Timescale	1983 (no more info)
Monitoring programme	Monitoring was undertaken between 1983-1989 and full ecosystem development at the site was to be evaluated in 1995. Saltmarsh ecology: Annual biomass surveys and contaminant analysis. Animal ecology: Species lists and contaminant analysis in snails.
Scheme success /failure	The saltmarsh creation scheme has been successful; plant growth was poor in the first year but vigorous four years post-construction and there is now a dense stand of vegetation. Transplanted natural *S. alterniflora* was slower to establish than "farmed" plants, however, 6 years post-construction biomass was greater in the latter. A breeding population of sandworms colonised the site 3 years after construction and subsequently fish, jelly fish, crabs and snails. Birds and mammals have been observed on the marshlands apparently feeding on these animals.
Reference source	Brandon DL, Lee CR, and Simmers JW. 1992. Long-term evaluation of plants and animals colonising contaminated estuarine dredged material placed in upland and wetland environments. Environmental Effects of Dredging, Vol D-92-4, US Army Corps of Engineers, Waterways Experiment Station. Simmers JW, Folsom BL, and Skogerboe JG. 1987. The Field Verification Programme - A progress Report on the confined Upland and Wetland Creation Disposal Alternatives. Vol D-87-3, US Army Corps of Engineers, Waterways Experiment Station.

Reference No.	**7.**
Site name	**Bolivar Peninsula, Texas, USA**
Site type	**Saltmarsh Creation**
Contractor	US Army Engineers
Funding body	N/A
Purpose	Beneficial use of dredged material, experimental
Objectives of the scheme	To demonstrate the feasibility of creating productive coastal wetland habitat on dredged material.
Description of site	Location: Bolivar Peninsula is approx. 15km northeast of Galveston, facing Galveston Bay on the North. Area: 4.5ha site
Methodology	Dredged material from nearby Intracoastal waterway was placed on the site, graded in to 3 different elevational tiers. Two *Spartina* spp., *S. alterniflora* and S. *patens*, planted as seeds and cuttings with 5 fertiliser treatments.
Technical details	Site preparation consisted of grading the dredged materials to a 0.7% slope at suitable elevations. A sandbag dyke was constructed to break high wave energies, with openings on either side to allow tidal flushing.
Timescale for scheme	1974 material dredged. 1975 material placement and site preparation. 1975-1976 construction of sandbank dyke. Summer 1976 plant cuttings. Spring 1977 sow seeds.
Monitoring programme	Monitoring of site was initiated in 1977 and in 1978/1979 comparisons were made with three nearby natural marshes. Soil and plant sample analysis, above- and below- ground biomass measurements and percentage cover estimates were made at sample points along three elevations; below HWM, HWM and above HWM.
Scheme success/failure	Rapid colonisation resulted in *Spartina* marsh habitat that fell within the normal variability in vegetation characteristics of natural local saltmarshes two years post construction. Plants spread to cover unplanted control plots which were also colonised by annual glass wort. Soil characteristics indicated that the site was young but maturing and nutrients were expected to reach equal levels to natural marshes 4-7 years post construction. Elevation is a critical factor in determining saltmarsh diversity and composition.
Reference source	Webb JW and Newling CJ. 1985. Comparison of Natural and man-made salt marshes in Galveston Bay complex, Texas. Wetlands 4, 75-86.

Reference No.	**8.**
Site name	**Sarahs Creek Constructed Marsh, Virginia, USA**
Site type	**Created Tidal Creek Saltmarsh**
Funding body	The Department of Resource Management and Policy, Virginia Institute of Marine Science, College of William and Mary.
Purpose	Experimental
Objectives of the scheme	To study the ecological conditions in a constructed tidal marsh and two natural reference tidal marshes.
Description of site	Location: Study site is located in Sarah=s Creek, a tributary of the York River, near Gloucester Point, Virginia, USA. Description: The constructed site is located in between two natural marshes and consists of a triangular plot of saltmarsh vegetation surrounding an intertidal channel with two interconnected branches. Area: 0.65ha; Channel depth: 1m; Tidal range: average is 0.75m.
Methodology	Saltmarsh was constructed by excavating an upland area and grading it to intertidal elevations. A channel was excavated to a depth of 1m below LW. One year old *Spartina alterniflora*, *S. Patens* and *Distichlis spicata* were planted.
Technical details	The channel was excavated to a depth of 1m below MLW. One year old green house reared saltmarsh plants were planted at 50 to 90 cm centres.
Monitoring programme	Two season monitoring strategy was initiated in May (Spring) and July (Summer) 1992 to compare the created marsh with two adjacent natural marshes on either side. Morphology: Area, volume, topography, (aerial photography and various software, e.g. Surfer and LI Contour V+). Tide range: Tide gauge data. Sediment: Total organic matter and carbon content analysed in 0-2cm and 14-16cm fractions (core samples) in three habitat types of each marsh (high, low and non-vegetated intertidal). Water quality: % salinity, temperature, dissolved oxygen (mg/l). Ecology: Vegetation: percent cover, stem density benthos: species abundance, diversity and equatability (grabs) zooplankton: identification and counts (micron mesh net).
Scheme success /failure	Similarities between constructed marsh (CM) and natural marshes (NM): water temperature, species composition, benthic community structure Differences between CM and NM: salinity (lower in CM), dissolved oxygen, organic carbon content, zoo plankton abundance, marsh surface utilisation, absence of *Ulva frutescens* and *Baccharis halimifolia* in CM, greater abundance of benthos in CM, greater fish and shellfish diversity in CM but less abundance, use of marshes by birds was less in CM, although wading birds showed preference for CM, fewer bird nesting sites in CM.
Reference source	Havens KJ, *et al*. 1995. An assessment of ecological conditions in a constructed tidal marsh and two natural reference tidal marshes in coastal Virginia. Ecol. Eng. 4 117-141.

Reference No.	**9.**
Site name	**Bodkin Island, Chesapeake Bay, Maryland, USA**
Site type	**Island Intertidal Habitat Restoration**
Contractor	US Army Engineer Waterways Experiment Station
Funding body	US Army Engineer District, Baltimore
Purpose	Restoration of bird habitat (Black Duck).
Objectives of the scheme	Primary objectives are reestablishment of Black Duck brood habitat, improvement and additions to nesting habitat, and overall island stability. Dredged material to be used to restore the island to 4.8 acres around the existing island.
Description of site	Location: Bodkin Island, nr. Kent Narrows, Chester River, Chesapeake Bay, USA Description: The island was formerly a peninsula at the turn of the century. Erosion has reduced the size of the island from 50 acres in 1847 to its current 0.94 acres. Area: 4.8 acres (19425m^3) Depth: -3 feet MLW Elevations: Permanent pools = -2 feet MLW Intertidal marsh = 0.5-1 foot MLW Nesting areas = 8-10 feet MLW
Methodology	45000 yd^3 of dredged material to be placed on the north side of the existing island and subsequently shaped and armoured to prevent erosion and over topping. Construction of the island will require a containment structure to minimise the loss of dredged material during placement. A small riprap or quarry spall is recommended for the containment structures. Upland/high marsh/low marsh gradation from the island crest down to tidal pools will provide shallow water for use by black duck hens and broods of ducklings. The restoration plan includes installation of 4 osprey nests. Stability of the island interior ensured by sizing the tidal creeks large enough to prevent erosion and by providing a rock-lined tidal inlet channel. Deposition in the tidal creeks is expected until a stable or equilibrium cross section is achieved.
Technical details	In addition to an offshore breakwater, a sill was constructed in the mouth of the channel at a depth of -0.5 feet to reduce wave action.
Timescale for scheme	Planning: March 1991 Placement of material: Unknown
Monitoring programme	None as yet. A vegetation plan has been developed that provides habitat and island stability. Planting is recommended in lieu of natural colonisation.
Scheme success /failure	Unknown
Reference source	Landin MC. 1991. Restoration of Black Duck intertidal habitat Bodkin Island, US Army Corps of Engineers, Wetlands Research, WRP-91-1. Maynord *et al* 1992. Design of Habitat Restoration using Dredged Material at Bodkin Island, Chesapeake Bay, Maryland. Wetlands Research Program Technical Report.

Reference No.	10.
Site name	**L'Ile de Bihlo, Loire Estuary, France**
Site type	**Island Creation**
Contractor	Les Techniciens du Port Autonome, Laboratoires d'hydrographique, CEA, University of Nantes, IFREMER
Funding body	Port Autonome de Nantes- St. Nazaire
Purpose	Beneficial use of dredged material.
Objectives of the scheme	To create artificial islands by hydraulic discharge of materials in the Loire estuary in order to stabilise the channel and provide wildlife habitat.
Description of site	Location: Loire Estuary between St. Nazaire and Donges. Volume: 7 million cubic metres of dredged material.
Methodology	Dredged material placed on tidal flats in the Loire estuary in 1977 during the deepening of the channel at the construction site of the methane terminal.
Timescale for scheme	Placement of material - 1977.
Monitoring programme	Monitoring prior to disposal - multidisciplinary survey to identify the various conditions necessary to the preservation of the environment (lateral mudflats). Post disposal monitoring was initiated in 1981 and continued through until 1987. Physical: Evolution of area in terms of hydrology (evolution of the currents, stability of the area), hydrosedimentation (topography, aerial photography, sediment sampling). Biological: Re-settlement and evolution of the populations, plant colonisation.
Reference source	Gallene B. 1991. Immersions de materiaux dragues dans l'estuaire de la Loire creation d'iles artificielles, PIANC 74, 26-36.

Reference No.	11.
Site name	**Ballona wetland, Southern California, USA**
Site type	**Saltmarsh Restoration**
Purpose	Wildlife habitat creation/flood control/water treatment
Objectives of the scheme	i) To create a diverse salt marsh system to provide a habitat for fish and wildlife, including sensitive, rare and endangered species. ii) To provide flood protection of existing and proposed developments. iii) To improve the quality of storm water run off and to contribute to the restoration of Santa Monica Bay.
Description of site	Location: Ballona wetland, Ballona Creek, in Santa Monica Bay, central Los Angeles California, USA. Description: Ballona wetland forms part of the Playa Vista Area of the Ballona Creek located approx. 900m upstream of the mouth. Area: Approx. 94ha
Reference source	Tsihrintzis VA, Vasarhrli GM and Lipa J. 1996. Ballona Wetland: a multi-objective salt marsh restoration plan. Proc. Inst. Civ. Eng. Wat., Marit & Energy 118, June, 131-144.

Reference No.	**12.**
Site name	**Hayward Area Recreation District Marsh**
Site type	**Habitat Restoration**
Purpose	Restoration of habitat for important bird life and saltmarsh harvest mouse.
Objectives of the scheme	(i) Restore habitat (feeding opportunities and shelter) for shorebirds and migratory waterfowl. (ii) Maintain healthy pickleweed on levees as habitat for the endangered saltmarsh harvest mouse.
Description of site	Location: Hayward Area Recreation District marsh is situated on the eastern shoreline of San Francisco Bay, California. Description: The site consists of 11 shallow ponds of various shapes and sizes. The ponds were used for salt production from as early as 1853 to the late 1940s. Following abandonment, tidal flow through a failed tide gate provided some circulation, and the levees between ponds developed a dense cover of pickleweed (Salicornia virginica) and alkali heath (Frankenia grandifolia). In 1986 the adjacent landowner blocked the ditch that provided tidal flow, and the ponds dried out. Area: 30ha. Elevation: Of pond bottoms - between 0.33 and 0.66m NGVD.
Methodology	Excavation of a new slough channel to connect the ponds in the marsh with the bay.
Technical details	Connections - three 36-in. culverts with slide-flap gates at the connection to the bay, two interior weirs, three interior 12-in. culverts, and three additional cuts in levees. The design was expected to create a zone between 0.73m and 0.79NGVD that would be inundated continuously for six days and then exposed continuously for six days.
Timescale for scheme	2 years
Monitoring programme	Sediment: Annual measurement of sediment deposition. Temp & Salinity: Weekly observations. Biological: Weekly observations of bird use and sampling of invertebrates. Others: One-time synoptic tide measurements in several ponds. Periodic inspection of water control structures two years after construction.
Scheme success /failure	The ponds support large populations of migratory waterfowl and shore birds.
Reference source	Coats R, Swanson M and Williams P. 1989. Hydrologic analysis for coastal wetland restoration. Environ. Mgmt. 13, 6, 715-727.

Reference No.	**13.**
Site name	**Tijuana River National Estuarine Research Reserve**
Site type	**Habitat Restoration**
Purpose	Habitat restoration for bird habitat (lightfooted clapper rail).
Objectives of the scheme	The primary objective was to protect and restore a productive cordgrass tidal saltmarsh. This required maintaining a continuously open tidal inlet by increasing the tidal prism in the estuary to pre-disturbance condition of 1852.
Description of site	Location: US/Mexico border. Description: Publicly owned coastal saltmarsh, estuary and upland habitats. The estuary has recently lost much of its tidal prism due to deliberate filling and accelerated sedimentation. This initiated occasional closure of the estuary mouth, beginning in 1963. In 1984 the closure lasted for 8.5 months until the barrier beach was breached mechanically. Large areas of lower marsh became extremely hypersaline, destroying cordgrass habitat for the endangered lightfooted clapper rail. By 1988 only a few pairs were nesting (compared to 21 pairs in 1985). Area: 260ha
Methodology	Extensive dredging was carried out to increase the tidal prism. Oneonta Slough was dredged in April 1987 and 23000m^3 was removed. The dredged spoil was used for sand replenishment on barrier beaches and dunes to retard storm overwash. River training berms were necessary to protect dredged areas from bedload sediment.
Timescale for scheme	Initiated in April 1987
Monitoring programme	Monitoring was initiated in April 1987. Sediment: Measurement of soil and water salinity. Ecology: Biological monitoring includes periodic sampling of vegetation, invertebrates, and fish. Hydrological: To test the effectiveness of the dredging in improving tidal circulation and ebb flow scouring of the tidal inlet.
Scheme success /failure	Results show substantial recovery of the cordgrass community since the mouth was reopened.
Reference source	Coats R, Swanson M and Williams P. 1989. Hydrologic analysis for coastal wetland restoration. Environ. Mgmt. 13, 6, 715-727.

Reference No.	**14.**
Site name	**Winchester Tidelands Restoration Project, South Slough (Coos) Estuary**
Site type	**Restoration of Tidal Wetland**
Funding body	South Slough National Estuary Research Reserve (NERR).
Purpose	Restoration of wetland for wildlife, experimental.
Objectives of the scheme	(i) To restore critical habitat for andromonous fish, migrating waterfowl, shorebirds, invertebrates, and mammals. (ii) To consider mechanisms to accelerate the rate of estuarine recovery.
Description of site	Location: Winchester Tidelands Restoration Project (WTRP) area of the South Slough Estuary. Description: Series of abandoned, subsided lands and freshwater drainage channels. Area: 30ha. Marsh elevation: 60-80cm below normal levels.
Methodology	Phase 1 : Passive restoration - tide gates and levees allowed to fail through age and lack of maintenance. Active restoration - dykes and levees were forcibly removed; elevations that replicate South Slough natural marsh surface elevations were constructed in a series of four "cells" within the WTRP site. Soil recovered during dyke destruction was used to raise the marsh base to target elevations. Dyke removal and cell construction took place sequentially (starting with a control cell with no elevation alteration) so that dyke material removed from low cells was used to construct cells at higher elevations.
Technical details	Natural marsh surface elevations in South Slough are: low marsh +1.4m NAVD; mid marsh +1.8m NAVD; high marsh +2.2m NAVD.
Timescale for scheme	Habitat assessment and pre-construction monitoring tasks - 1993/94. Phase 1 - Spring and Summer of 1995.
Monitoring programme	Pre-construction monitoring was initiated in 1993 and covered the project marsh sites, a series of off-site dyked and naturally breached marsh sites, and control sites located nearby. Several post-restoration (1995) research projects are proposed to track development of physical and biological processes during Phase 1 of the WTRP. Ecology: Proposed - experimental re-introduction of invertebrate species, assessments of secondary succession, the role of recruitment and colonization inhibitors in actively and passively restored dyked wetland.
Contact and Reference source	Mike Graybill, Manager, South Slough NERR. Rumrill, SS and Cornu, EC. 1995. South Slough Coastal Watershed Restoration. Restoration and Management Notes Vol 13 (1) 53-57. Rumrill, SS and Cornu, EC. 1993. Restoration of Estuarine Tidelands In South Slough National Research Reserve. Park Science Vol 13 (4) pp 3-6.

Reference No.	**15.**
Site name	**Salmon River Estuary, Oregon, USA**
Site type	**Saltmarsh Restoration**
Contractor	U.S. Forest Service
Funding body	U.S. Forest Service.
Purpose	Restoration to condition prior to the existing dykes and agricultural use.
Objectives of the scheme	To replace the pasture with what is presumed to be the areas original condition i.e.: high salt marsh dominated by tufted hairgrass (*Deschampsia caespitosa*), Baltic rush (*Juncus balticus*), and Pacific silverweed (*Potentilla pacifica*). To restore the pasture to marshes that are sufficiently high that they are seldom flooded by salt water in summer and that are deeply dissected by tidal creeks.
Description of site	Location: North shore of the Salmon River estuary, Oregon coast, USA. Description: Agricultural pasture enclosed by a dyke. Area: 52 acres.
Methodology	Partial breaching of the 5000 foot dyke enclosing the 52 acre pasture by the Forest Service in 1978. Restoration depended solely on the natural establishment of saltmarsh vegetation.
Timescale	September 1978
Monitoring programme	Monitoring was initiated in 1978 and continued for 10 years after dyke removal. Sediment: Sediment accretion and elevations measured across 47 cross-sections along 8 tidal creeks to determine the extent of creek erosion over the 10 year restoration period. Soil samples analysed for soil texture and organic content. Salinity: measured in late September 1988. Ecology: Intensive sampling system of permanent plots and collection of base-line data in the dyked pasture and in the adjacent high marsh "controls" . Vegetation change was monitored at 115 one-metre-square permanent plots along 20 transects and at 450 0.1-metre-square temporary plots, prior to dyke removal in 1978 and after breaching in 1979-84 and 1988.
Scheme success /failure	The scheme was a success for the following reasons: the restored salt marsh consists of typical Pacific Northwest salt marsh communities; tidal exchanges are complete; the creeks deepened from 20-60cm over the 10year restoration period due to scouring by tidal water and were able to provide habitat for juvenile fish; and the marsh is highly productive. The more rigorous restoration intent - of returning dyked pasture to its original high salt marsh condition - was not achieved (e.g. in 1988 surface elevation was about 35cm lower in the restored area than in the flanking controls due to subsidence of the dyked pasture from 1961 to 1978. The restored marsh surface was building up by a combination of sedimentation and soil swelling at a greater rate than the higher control areas. Recovery from subsidence is expected to take 5 decades or more).
Reference source	Frenkel RE and Morlan JC. 1991. Can We Restore Our Salt Marshes? Lessons from the Salmon River, Oregon. The Northwest Environmental Journal (7) pp119-135.

Reference No.	**16.**
Site name	**Union Slough Restoration Site**
Site type	**Restoration of Mudflat / Saltmarsh**
Contractor	Port of Everett
Funding body	Port of Everett
Purpose	Mitigation for lost littoral habitat functions due to port development.
Objectives of the scheme	To provide estuarine habitat and ecological functions to replace those lost to unavoidable impacts of the Ports development proposal. Mudflats and saltmarsh to be restored by breaching dykes to restore tidal circulation. Approximately 3 to 4 acres is expected to develop saltmarsh vegetation, about 4 acres will be excavated as drainage channels and the remainder will be mudflats.
Description of site	Location: Lower portion of Union Slough Description: The 29.2 acre site is owned by the Port of Everett. Approx 5.7 acres of isolated emergent (PEM) marsh and a 1.8 acre freshwater pond currently exist on the site. The remainder of the area is in dykes (2.8 acres), vegetated with an upland scrub-shrub community, and agricultural land (18.9 acres). Area: A minimum of 24 acres is to be restored.
Methodology	Breaching of dykes and the excavation of drainage channels. Planting of saltmarsh vegetation may be conducted after Year 3 and may be required after Year 5 to supplement natural colonization and to enhance the mudflat functions.
Timescale for scheme	Project planned in 1996, commencement date unknown.
Monitoring programme	An annual report will be prepared by the Port detailing all monitoring and inspections conducted in the previous year. Ecology: Monitoring of saltmarsh vegetation, juvenile salmonid use, juvenile salmonid prey base, and shorebird use will be conducted.
Maintenance	Primary maintenance activities anticipated are erosion control at the entrances to the wetland and on dykes or buttresses protecting the Interstate 5 embankment (see plan).
Scheme success /failure	N/A
Reference source	Port of Everett. 1996. Stage 1 Marine Terminal Improvements. Final Mitigation Plan.

Reference No.	**17.**
Site name	**Deepwater Slough Restoration Project**
Site type	**Restoration of Subtidal Habitat**
Funding body	Washington Department of Fish and Wildlife (WDFW)
Purpose	Provision of rearing and over-wintering fish habitat.
Objectives of the scheme	(i) Restoration of approximately 47 acres of distributary/subsidiary tidal river slough, at least 12.5 acres of blind channel, and restoration of a natural hydrologic regime to more than 250 acres of isolated wetland habitat. (ii) Reinforcement of existing dykes and construction of some new dykes to maintain cereal grain production. (iii) Opening of a dyked slough to provide additional passage for adult salmon to upstream spawning areas.
Description of site	Location: The farmed island segment of the Skagit Wildlife Area, Fir Island, Skagit County, Washington. Description: The study area includes the South Fork Skagit delta from the bifurcation of Freshwater and Steamboat Sloughs to Skagit Bay. Area: 435 acres.
Methodology	To open up the blocked Deepwater Slough of the South Fork Skagit River by removal of the blockage at the upper and lower ends of the slough and construction of approximately 25000 feet of levee along the sides of the slough.
Timescale for scheme	Unknown
Monitoring programme	Ecology: Waterfowl habitat and fish passage. Hydrology: Investigations to assess water flow and hydrologic changes.
Scheme success /failure	N/A
Reference source	US Department of Fish and Wildlife. 1996. Pre-Threshold Determination Scoping Notice. Deepwater Slough Restoration Project.

Reference No.	**18.**
Site name	**Sweetwater/Paradise Marsh Mitigation in San Diego County**
Site type	**Saltmarsh Restoration / Creation**
Funding body	California Department of Transportation (Caltrans)
Purpose	Mitigation project under the Endangered Species Act.
Objectives of the scheme	To create nesting habitat for the Light-Footed Clapper Rail and foraging area for the California Least Tern by enhancing the wetland area adjacent to the highway and flood control channel. Specific objectives : (i) Create appropriate elevations for low, middle, and high saltmarsh habitat. (ii) Revegetate graded sites with the appropriate saltmarsh plants. (iii) Improve tidal influence by creating tidal channels in saltmarsh segments. (iv) Increase habitat area for prey species of the clapper rail (crabs and other invertebrates) and least tern (fish) by creating mudflat and increasing deep channel area. (v) Salvage native saltmarsh plants from impacted areas for propagation and use in later stages of restoration. (vi) Convey approx. 300 acres of marshland and environmentally sensitive upland in perpetuity to public ownership for the preservation of endangered species.
Description of site	Location: Sweetwater/Paradise Creek marsh, west of I-5 at State Route 54 interchange, San Diego County, California. Description: 10ha of disturbed high saltmarsh changed to tidal channels and low marsh plus approximately 7ha of uplands to be converted to tidal channels and intertidal marsh. Area: Approx. 17ha.
Methodology	(i) The existing (degraded) saltmarsh was graded to create the appropriate topography for low and middle saltmarsh in the form of 8 islands. (ii) Deep channels were created surrounding the area to prevent intrusion by pests, and smaller tidal channels to facilitate tidal flushing within the marsh, enhance plant growth and, therefore, rail habitat. (iii) Both areas were opened to tidal flushing. (iv) Low-marsh vegetation transplants planted. (v) A tidal Spartina nursery and an irrigated middle-marsh plant nursery were created near the restoration site for salvaging plants removed from the project area.
Monitoring programme	Monitoring was initiated in 1989. Ecology: Monitoring of success of the planting, remedial planting as needed, and fencing areas that show signs of grazing. Fish were sampled periodically at 6 stations, 4 stations in the constructed marshes and 2 in the natural marsh. Benthic invertebrates were sampled periodically at 7 stations.
Scheme success /failure	The project had some success in establishing cordgrass, although the area with plants is less than the area planted. Present and future habitat value to the clapper rail is uncertain due to several problems incurred during the restoration process (see source for details).
Reference Source	Josselyn M, Zedler J, and Griswold T. 1990. Sweetwater/Paradise Marsh Mitigation in San Diego County. In: Wetland Creation and Restoration, the Status of the Science. Zedler, JB. 1996. A Scientific Perspective and Southern California Focus.

Reference No.	**19.**
Site name	**Drayton Harbour, Washington State, USA**
Site type	**Intertidal and Shallow Subtidal Habitat Creation**
Contractor	Port of Bellingham
Funding body	Port of Bellingham
Purpose	Mitigation for loss of mudflat habitat.
Objectives of the scheme	To use dredged material to create 15 acres of intertidal and shallow subtidal habitat that will support eelgrass.
Description of site	Location: Drayton Harbour. Area: 11.24 acres. Tidal range: Raise subtidal bed to low intertidal (-3MLLW).
Methodology	300000 cubic yards of dredged clay or sand to be placed in order to raise the subtidal bed to low intertidal (-3MLLW).
Reference Source	M. Vernice Santee, Washington State Department of Ecology, pers. comm. 1996.

Reference No.	**20.**
Site name	**Agua Hedionda Creek and Lagoon, California**
Site type	**Habitat Restoration / Creation**
Purpose	Mitigation for road construction
Objectives of the scheme	To create and enhance wetland habitats lost or impacted by road construction within the lagoon wetland boundaries.
Description of site	Location: Agua Hedionda Creek and Lagoon at Hidden Valley Road, Carlsbad, San Diego County, California. Description: Tidal saltmarsh, brackish ponds, freshwater marsh, riparian woodland, and upland transition. Area: 5.6ha of road fill, 7.5ha wetland enhancement, and over 70ha of land title transfer.
Methodology	Creation of openings along dykes to facilitate tidal flushing and lowering of peninsular fill areas to create additional saltmarsh habitat. Revegetation of the new intertidal area with saltmarsh species and revegetation of the transitional area with plant species suitable for wetland/upland interface to create a dense landscape buffer zone. Creation of a 0.8ha bird nesting island.
Scheme success /failure	The project involved the restoration of several different types of wetland throughout the lagoon, none of which can currently be considered successful. The overall impact of the mitigation on the lagoon ecosystem cannot be considered beneficial. For example, nearly 2 years after initial planting of the revegetation areas, nearly all of the plantings have failed to produce their specified habitat values, and "enhanced functional capacity of the overall lagoon" is not apparent.
Reference Source	Josselyn M, Zedler J, and Griswold T. 1990. Agua Hedionda Creek and Loaggon, San Diego County. In: Wetland Creation and Restoration, the Status of the Science.

Reference No.	**21.**
Site name	**Pacific Coast Terminals Saltmarsh Compensation**
Site type	**Habitat creation**
Purpose	Mitigation for the construction of a new storage tank facility at the Pacific Coast Terminals in British Columbia.
Objectives of the scheme	To meet the no-net-loss policy of the Canadian government. To restore habitat for fisheries.
Description of site	Location: Pacific Coast Terminals, British Columbia.
Methodology	Intertidal sediment dredged from the construction site was used as fill for the new marsh. Saltwort (*Glaux maritima*) was harvested from approved sites and transplanted as cores during May 1993.
Timescale for scheme	1991-1993.
Monitoring programme	Monitoring was initiated in 1991 and continued for one year. The monitoring was carried out at the mudflat, the compensation marsh, and an adjacent saltmarsh (as a control). Sediment: Data collected on soil density, pH and electrical conductivity. Analysis of fertility included measurements of total carbon and nitrogen and of amounts of available nitrate and ammonia. Ecology: Periodic observations of growth and photographs.
Scheme success /failure	The one year monitoring programme suggested that existing strategies for estuarine restoration may be ineffective for habitat of this type. Soils in the constructed marsh were coarser with a lower pH. At the end of the first growing season, plant growth was so poor that coverage was not determined. Successful early establishment of transplants was severely affected by intense grazing by Canadian geese.
Reference source	Zedler, JB. 1996. A Scientific Perspective and Southern California Focus. Pacific Estuarine Research Laboratory. Published by the Californian Sea Grant College System, University of California.

Reference No.	**22.**
Site name	**Archie Creek, Hillsborough County, Florida, USA**
Site type	**Saltmarsh Restoration**
Contractor	Plan prepared by Henley Environmental Services, Inc. in 1977. Carried out by Mangrove Systems Inc.
Funding body	Gardinier, Inc.
Purpose	Mitigation for construction of a pollution control system filling 1.5ha of tidal marsh
Objectives of the scheme	Restore tidal marsh with *S. alterniflora.*
Description of site	Location: Archie Creek, Hillsborough County. Description: Marsh/shore. Area: 1.82ha.
Methodology	Plugs of *S. alterniflora* were removed from an existing tidal marsh and placed into holes at 3-ft intervals in the planting area. Fertilizer was side dressed at each planting site after all the planting was completed.
Technical details	The plugs were 4 to 5 inches in diameter and were removed using standard post-hole diggers.
Timescale for scheme	Planting : April - June 1978.
Monitoring programme	Monitoring was carried out 6 months after planting and 1 year after planting. CSA carried out a field assessment in November 1984. Ecology: % survival.
Scheme success /failure	The scheme was successful. The site appeared to be a naturally functioning marsh. The area had several tidally inundated creeks and mudflat areas with slightly higher surrounding areas of *S. alterniflora*. A few small *A. germinans* seedlings had invaded a small area of sandy soil.
Reference source	Dial, RS and Deis, DR. 1986. Mitigation Options for Fish and Wildlife Resources Affected by Port and Other Water-dependent Developments in Tampa Bay, Florida. Continental Shelf Associates. Biological Report 86(6).

Reference No.	**23.**
Site name	**Fantasy Island and Spoil Island, Florida, USA**
Site type	**Habitat Creation**
Contractor	Tampa Marine Institute, Environmental Wetland Gardens, Inc.
Funding body	TPA
Purpose	Wetland mitigation for construction of shrimp boat docks.
Objectives of the scheme	Construct island using dredge material, stabilize existing island.
Description of site	Location: Fantasy Island and Spoil Island, located 2.5km south of Pendola Point in Hillsborough Bay. Description: Both islands were created in 1978 with dredged material. Area: 2ha
Methodology	Both islands were created in 1978 with dredged material. Fantasy Island was planted with 0.3 to 1.9m high transplants of *A. germinans* and *L. racemosa* (3:2 ratio). A total of 1,513 mangroves were placed in a 0.52ha area. Fantasy Island and most of the eastern shoreline of Spoil Island were planted with plugs of *S. alterniflora*. An area of 1.6ha was planted on 1m centres with 10000 plugs of *S. alterniflora*.
Technical details	The mangroves were removed by shovel and the root balls were wrapped in burlap.
Timescale for scheme	1978 - 1981.
Monitoring programme	Monitoring of mangrove 13 months and 5 years after planting, no formal monitoring of *S. alterniflora*. CSA carried out a field assessment in November 1984.
Scheme success /failure	The project showed that dredge spoil can effectively be used to create productive wetland. The project was not entirely successful in that a wetland habitat of comparable size to that which was lost was not created. Survival of mangrove transplants was approximately 55%. Fantasy Island is eroding on the southern shoreline and possibly expanding into a subtidal area to the south. it is doubtful that a mature mangrove community will develop without some structural modification to the site.
Reference source	Dial, RS and Deis, DR. 1986. Mitigation Options for Fish and Wildlife Resources Affected by Port and Other Water-dependent Developments in Tampa Bay, Florida. Continental Shelf Associates. Biological Report 86(6).

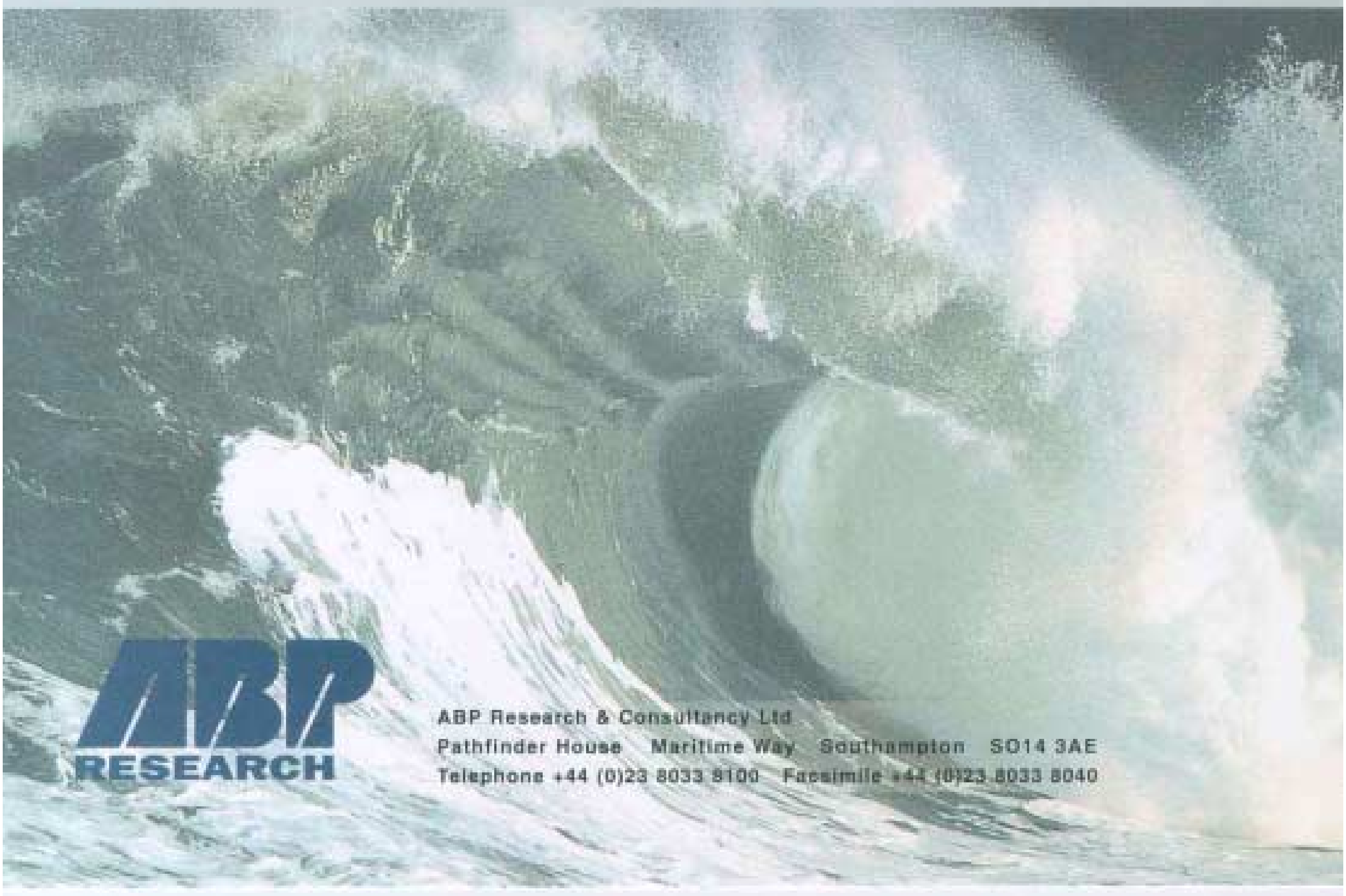
Reference No.	**24.**
Site name	**Wolf Creek/Apollo Beach, Tampa Bay, Florida, USA**
Site type	**Habitat Restoration**
Contractor	West Coast Engineering Corporation
Funding body	Frandorson Properties
Purpose	Enforcement action for illegal dredging of 110ha area of mangrove forest.
Objectives of the scheme	Lower elevation and plant with red mangrove (*Rhizophora mangle*).
Description of site	Location: Southwest of Apollo Beach to Wolf Creek. Description: Euryhaline/tidal creeks and marsh shore. Area: 1.6ha.
Methodology	*R. mangle* propagules planted in 30-ft wide bands spaced 150 ft apart, parallel to the shore. 25300 mangrove seedlings were planted over a 1.6ha area
Timescale for scheme	August 1974.
Monitoring programme	Monitoring was initiated 3 weeks after planting and continued periodically. CSA carried out a field assessment in November 1984. Ecology: % survival and growth rates and observations on other vegetation.
Scheme success /failure	The project demonstrated natural forestation within a wetland area in Tampa Bay. The restored area has not yet recovered however and appears to be more susceptible to freeze damage than the adjacent mangrove forests.
Reference source	Dial, RS and Deis, DR. 1986. Mitigation Options for Fish and Wildlife Resources Affected by Port and Other Water-dependent Developments in Tampa Bay, Florida. Continental Shelf Associates. Biological Report 86(6).

Reference No.	**25.**
Site name	**Branches Hammock, Manatee County, Florida, USA**
Site type	**Saltmarsh restoration**
Funding body	FDOT
Purpose	Mitigation for construction of roadway (Interstate 75) over needlerush marsh.
Objectives of the scheme	The construction required that temporary fill over 2.32ha of saltmarsh. The objective was therefore to restore the altered tidal creeks.
Description of site	Location: Branches Hammock, Manatee County. Description: Tidal creek connected to the Manatee River, surrounded by a tidally inundated *Juncus roemerianus* (needlerush) marsh. Area: 2.3ha.
Methodology	Areas prepared for revegetation by excavating the existing marsh sediment (peat, muck) to a minimum depth of 30 inches or through the entire depth of the sediment layer and stock piling the sediment. Once construction was complete the temporary fill roads were to be removed to 12 inches below the original marsh elevation. The stockpiled muck was to be used to return the affected area to the original elevation. 21000 plugs of *J. roemerianus* were planted on 1.2m centres with fertiliser.
Timescale for scheme	Revegetation - June and July 1980.
Monitoring programme	Monitoring was carried out 22 months after planting by the consultant and 4.5 years later by CSA.
Scheme success /failure	The project was partially successful in its goal of restoring 2.32ha of *J. roemerianus* marsh. None of the 1.4ha planted under the bridge spans survived and of the 0.92ha temporarily filled and later restored, only 50% was restored to a monospecific *J. roemerianus* marsh. The lack of success of *J. roemerianus* in the high marsh is attributable to the fact that the area was not restored to its previous elevations.
Reference source	Dial, RS and Deis, DR. 1986. Mitigation Options for Fish and Wildlife Resources Affected by Port and Other Water-dependent Developments in Tampa Bay, Florida. Continental Shelf Associates. Biological Report 86(6).

Reference No.	**26.**
Site name	**Weedon Island/Papy's Bayou, Pinellas County, Florida USA**
Site type	**Habitat Restoration**
Contractor	Sundown Construction Company
Funding body	Harbor Island Development, W. Langston Holland
Purpose	Mitigation for filling of wetland for uplands development.
Objectives of the scheme	Construct marsh by lowering the elevation of uplands.
Description of site	Area: 10ha
Methodology	The proposed wetland were excavated and to approximately MHW and planted with *S. alterniflora* on 0.9m centres. The plantings were purchased in pots from Environmental Wetland Nursery, Inc.
Timescale for scheme	August 1983.
Monitoring programme	Monitoring was carried out 1 year after planting to ensure a survival rate of at least 70% at the end of one growing season. CSA carried out a field assessment in November 1984. Ecology: Plant species composition and density.
Scheme success /failure	The *S. alterniflora* is viable and will probably coalesce into a continuous marsh. It is expected that mangrove seeds will colonize the area.
Reference source	Dial, RS and Deis, DR. 1986. Mitigation Options for Fish and Wildlife Resources Affected by Port and Other Water-dependent Developments in Tampa Bay, Florida. Continental Shelf Associates. Biological Report 86(6).

Reference No.	**27.**
Site name	**Upper Newport Bay**
Site type	**Created Intertidal Sand and Mudflats**
Purpose	Restoration of salt evaporation ponds and salt flats.
Objectives of the scheme	To restore silted up salt evaporation pans and salt flats.
Description of site	Location: The study site is located in Upper Newport Bay, southern California, USA. Area: 16.5ha
Methodology	The site was excavated to mid-tidal level.
Timescale for scheme	Constructed in 1982.
Scheme success /failure	In 1985, the site was extensively used by wintering waders.
Reference Source	Davidson NC and Evans PR. 1987. Habitat restoration and creation: its role and potential in the conservation of waders. Wader Study Group Bull. 49. Supplement : The Conservation of International Flyaway Populations of Waders. Davidson NC and Pienkowski MW (Eds).

Reference No.	**28.**
Site name	**Lauwerszeepolder, Waddenzee, Netherlands**
Site type	**Habitat Creation**
Purpose	Experimental - to study colonizing species and ecosystem development.
Objectives of the scheme	Study of the colonisation of the emergent sandflats.
Description of site	Location: Lauwerszeepolder, Waddenzee, Netherlands Description: After the storm surge of 1953 many embankments have been reconstructed along with fresh water lakes (to solve drainage problems). The new land created as a result consists of tidal flats and channels and sands which have been left for natural development of plant communities. Area: 7800ha Tidal range: Sandflats lie at about 95cm above NAP - 90cm (approx 90cm below mean sea level).
Timescale for scheme	Study carried out from 1969 to 1975
Monitoring programme	Monitoring was initiated in 1969 and continued to 1975. Sediment: Soil water conditions, silt content, grain size Ecology: Colonisation of *Salicornia* and *Atriplex* spp.
Scheme success /failure	Colonisation started with a sparse halophytic veg. consisting of *Salicornia dolichostachya*, *S. europaea*, *Suaeda maritima* and *Atriplex hastata*. In the first year the plant densities ranged from 0 to 300/ha reflecting the silt content of the sediments and the distance of the site from nearby saltmarshes.
Reference Source	Joenje W. 1979. Plant Succession and Nature Conservation of Newly Embanked Tidal Flats in the Lauwerszeepolder. In: Jefferies RL and Davy AJ (Eds) Ecological Processes in Coastal Environments. The First European Ecological Symposium and the 19th Symposium of the British Ecological Society.



ABP RESEARCH

ABP Research & Consultancy Ltd
Pathfinder House Maritime Way Southampton SO14 3AE
Telephone +44 (0)23 8033 8100 Facsimile +44 (0)23 8033 8040