The University of Hull

Hull, UK

The recreation of estuarine ecosystem: a case study at Paull Holme Strays, Humber Estuary, UK

being a Thesis submitted for the Degree of PhD in Biological Sciences from the University of Hull

by

Bryan Allen Swig

B.S. California Lutheran University
M.S. California State University, Northridge

Table Of Contents

_

List Of Figures

List of Tables

Acknowledgments

I am extremely grateful for the continuous support and encouragement from my supervisor Professor Mike Elliott, throughout my graduate studies. His considerable input on the study designs and drafts of the thesis in all stages of development has been invaluable. I deeply appreciate the time he spent with manuscripts, and the countless letters of recommendation. Dr. Krysia Mazik, Will Musk, Lucas Mander and Ollie Dawes provided support and advice along the way as well.

I am indebted to Jane Smith and Peter Mulder for providing the tireless assistance in the field where they braved the cold, wind, rain, snow and mud to lend a hand, I cannot thank them enough!

Parts of this study would not have been possible without the assistance of Lucas Mander, Paul Robertson and the staff in the Institute of Estuarine & Coastal Studies who provided aerial photos and assistance in carrying out ornithogical studies and habitat mapping. I also have to thank the Environment Agency for providing me with access to the study sites.

I extend a special thanks to my wife Bailey, and Andrea Huvard giving me the motivation I needed to finish writing up this thesis. I also have to thank Andrea for help with editing and general support throughout the years.

Abstract

Habitat restoration can partially compensate for the extensive loss of coastal wetlands, which are transitional habitats that form the barriers between seas and oceans and the terrestrial environment. Coastal wetlands historically due to their location have either been altered by flooding or draining to suit human needs. Coastal wetlands when left unaltered function as water filters, flood buffers, nurseries for juvenile nekton and as wildlife habitat for birds and animals. Creation of costal wetland habitat with intact ecological functions remains a challenging task. Paull Holme Strays (PHS) is an example of a restored coastal wetland on the Humber Estuary, UK. This site was restored to natural coastal wetland in 2003, and is an effort in the restoration of both mud flat and salt marsh. To evaluate immediate restoration success of the restored wetland at PHS, the development of the abiotic [environmental] and biotic parameters (benthic invertebrate community development, shore bird assemblages and fish assemblages) at PHS were monitored. for the 3 years following restoration and compared to adjacent unaltered mudflats and salt marshes. In addition to the field surveys experimental studies on the colonization of halophytes within the salt mars were conducted. The study indicates that an abiotic factor such as sediment accretion is occurring at a rapid rate in the northwest of the restoration site. Grain size and organic content have remained constant and are at levels comparable to those in a natural reference habitat. The study also indicates that there has been an immediate biotic response to restoration, as both the biomass and species richness of invertebrates using the restoration site have increased since monitoring began, however levels are still lower than those of the natural reference habitat. Shorebird species diversity and densities were frequently as high in the restored habitat of PHS as in reference natural habitat between 2004 and 2005, although each species preferred specific sites. The examination of fish assemblages between 2004 and 2006 revealed that densities and species diversity were as high or higher in the restored habitat of PHS in comparison to reference unaltered habitats. The experimental studies of halophyte colonization, aimed to determine whether colonization could be expedited and showed that doing nothing to compacted farm soils prior to reinstatement of the flooding provided the best for initial colonization. These filed surveys that were conducted show that although restoration of PHS provided adequate habitat for some wetland species immediately, it takes decades for restoration sites to develop completely. Experimental manipulations allow evaluation of ecological functions and can supplement structural assessments of restoration success. This work indicates management measures, which will allow the development of new restoration sties to occur more rapidly, allowing for costal wetland function to start sooner.

Chapter 1

Introduction

1.1. Estuarine wetland characteristics and their nature conservation interests

Estuary's are defined as the areas of wetland that form at the margins of the land and the sea with an input of freshwater (Elliott and McLusky, 2002). They are partially enclosed bodies of water, often have a soft tidal shore and its surroundings, are open to saline water from the sea and receiving fresh water from rivers, land run-off or seepages. Tidal flats are a major part of estuarine wetland ecosystems. They vary from soft muds and saltmarshs in the sheltered inner parts of estuaries to firm sand flats in the outer parts. Mudflats are important as they support large numbers of invertebrates such as crustaceans, molluscs and annelids on which fish and many national and international waterfowl feed (Cearreta *et al.*, 2002). Higher up the intertidal zone occur saltmarshes, which occur in the intertidal zones of moderate to low energy shorelines along estuaries. They have ecological value in primary production, nutrient cycling, as habitat for fish, birds and other wildlife and in stabilizing shorelines (Hughes and Paramor, 1999; van der Wal *et al.* 2000; Dierschke and Bairlein 2004). Tidal flats are vulnerable to progressive land claim, bait and shellfish collecting, and the impacts of waste discharge and pollution (Pethick, 2002; McLusky and Elliott, 2004). Maintaining healthy tidal flats is crucial to estuarine conservation since this habitat is the source of much of the richness of the estuarine ecosystem (ABP Research, 1998). If healthy tidal flats are lost it can and will lead to loss of aesthetic qualities, and of economic opportunity (Bolam *et al.* 2002). Economic opportunity loss can be seen as a loss in potential profit made from taking advantage of the services that tidal flats provide. But more importantly it will lead to a reduction in biodiversity and losses in biodiversity can be seen in the reduction of services that the ecosystem is able to provide humanity (Snelgrove, 1999). Typically, the inner, sheltered parts of estuaries are muddy and the outer parts are sandy. On the upper tidal levels, characteristic salt marsh vegetation develops (Pethick, 2002).

The widespread discharge of effluents entering estuaries has led to them being polluted, and has contributed to the perception by society of estuaries being wastelands. However, over the years as people have become more aware of nature and its conservation, efforts have been made to reduce the amount of pollution allowed to enter the estuaries through effluents by treating this wastewater prior to its entrance into the estuaries (McLusky and Elliott, 2004)

1.2. Loss of tidal flats and saltmarsh

Recently, large losses of wetlands have been reported globally, including the US (Phillips, 1986; Zedler, 1996; Schwimmer, 2001), Spain (Castillo *et al.* 2000), England (Burd, 1992; Cooper *et al.,* 2001) and the Netherlands (Dijkema, 1987; Cox *et al.,* 2003). These losses have been attributed to several factors associated with human development and climate change (Goodwin *et al.,* 2001; Adam, 2002). In England, much attention has been given to the combined effect of historic land claim and current sea-level rise on saltmarsh erosion. In times of relative sea level rise (i.e. a combination of land subsidence and sea-level rise) both tidal flats and saltmarshes must increase in elevation to avoid being ‘ecologically drowned’. This has generated the assumption that these habitats must move inland in order to adjust to the rising sea level. In situations where land claim has resulted in the construction of seawalls close to the mean high water level (a common feature in the UK), tidal flats and saltmarshes will not be able to migrate further inland. This phenomenon is called ‘coastal squeeze’ (Doody, 2004). The situation however, is much more complex because the vertical range of a saltmarsh is not only controlled by sea level, but also by tidal range and sediment supply, whereas the horizontal extent is mainly affected by the local wind-wave climate (Allen, 2000). Whether or not a saltmarsh will be sustained in periods of relative sea-level rise depends on several, locally varying, factors. Nevertheless, in an attempt to halt any further losses and to promote salt-marsh development, the UK government has adopted a policy of managed realignment, which is the breaching or removal of coastal defense structures to allow tidal flats and saltmarshes to develop on formerly reclaimed land (Anon, 1992).

1.3 Managed realignment

The notion of managed realignment to restore wetlands is practiced in various countries, albeit under a different names and driven by different incentives. In the US for example, the ‘no net loss of wetlands’ policy serves as a major incentive for mitigation projects in which the area of wetland habitat being damaged or lost is re-created elsewhere (Zedler, 1996). The breaching of sea defenses is only one option amongst several mitigation measures (Zedler, 2001). Along the Dutch and German Wadden Sea coast, the option of de-embankment of summerpolders is currently pursued because of increased awareness of nature conservation issues together with the diminished value of reclaimed land for agricultural exploitation (Wolff, 1992; Alhorn and Kunz, 2002). The saltmarshes in this area have mainly developed as a result of artificial reclamation works (Dijkema *et al.,* 1980), which involved the construction of sedimentation fields surrounded by brushwood to enhance sediment accretion, followed by the excavation of ditches to improve drainage and accelerate plant establishment (Bakker *et al.,* 2002). Once continuous vegetation had developed, these salt marshes were embanked by a low summerdike that would protect the newly claimed land from flooding in summer but not during storm surges or high tides in winter (Bakker *et al.,* 2002; Alhorn and Kunz, 2002). The so-called summerpolders were mainly used for cattle grazing and hay making.

In the United Kingdom, large areas of high salt marsh have been claimed successively in the past (Davidson *et al.,* 1991) and mainly used as freshwater grazing marsh (Doody, 1992). When techniques for protection against tidal flooding improved, several grazing marshes as well as newly enclosed land were transformed into arable fields (Williams and Hall, 1987; Doody, 1992) and seawalls were built further seaward. Nowadays, the diminishing value of arable land together with an increasing cost of seawall maintenance form the primary motives for managed realignment. In addition, the UK has a legal requirement to compensate for salt-marsh loss in order to comply with national and international policies on the conservation of wetland habitat (Pethick, 2002).

The concept of managed realignment or de-embankment to re-create wetlands is not new. In the past, accidental breaches in seawalls have occurred as a result of storm surges and, when left unrepaired, the newly inundated land often became transformed into salt marsh (Burd *et al.,* 1994). A total area of *c*. 2700 ha of embanked summerpolders can be found in the Wadden Sea, with 1000 ha in the Netherlands and 1700 ha in Germany. The total area of land claimed in the UK is 89,000 ha (Davidson *et al.,* 1991), but here the embankment often functions as the main sea defense restricting the potential for de-embankment.

1.4. Tidal flat and saltmarsh restoration

Success of tidal flat and saltmarsh restoration is judged on whether or not a self-sustaining community is established that attempts to recreate a natural ecosystem (Beeby, 1993). When creating tidal flats and saltmarsh using realignment techniques the starting point is considerably different from what would naturally happen with sea level rise (French, 2006). Under natural sea level rise adjustment, tidal flats and saltmarsh slowly move landward and upward. But with managed realignment this landward movement occurs all at once. Therefore it is important to consider whether it is ever possible to achieve a fully naturally functioning tidal flat/ saltmarsh ecosystem given the non-naturalness of the starting point (French, 2006). Additionally, time is an important issue to keep in mind when measuring restoration success as recruitment of species tends to be successional with lower levels of the food chain needing to become established as food sources, and for vegetation to become established for fish habitat (Weinstein *et al.,* 2005). Therefore, when trying to establish ecological goals of a restoration site there has to be clear objectives and a realistic appraisal of what is possible (Simenstad *et al*., 2006). The decisions may have economic and social dimensions to them therefore the following must be considered:

1) Our capacity to restore the site: is it realistic to believe that we can restore the site or are their physical and/ or chemical constraints that are preventing this?
2) Financial constraints on the reclamation process.
3) Local needs, ecological, social, or economical: land reclamation/ restoration may not be done because a valued plant or animal community has been established on the site.
4) Pollution of the surrounding areas.

Successful restoration of plant communities depends on the availability of target species, the ability of the species to reach a target area and the presence of suitable environmental conditions that allow the species to germinate and establish (Wolters *et al.,* 2005). The establishment of algae, plankton and plants is vital for a successful restoration as they are ultimately responsible for primary production in estuary ecosystems (ABP Research and Consulting Ltd, 1998). Some research done by Hughes and Paramor (2004) has suggested that in the south east of England biological factors can dominate the establishment of saltmarsh, stating that grazing by both the ragworm (Nereis diversicolor) and the mud snail (*Hydrobia*) lead to the failure of marsh development. There have been few studies on fauna colonization of restored habitats as the primary aim of managed realignment is the restoration of saltmarsh (French, 2006), which stabilizes sediment, thus reducing erosion. Studies involving fauna have largely related to restored wetland utilization by fish (Colclough *et al*., 2005) and birds (Atkinson *et al.,* 2004; and Mander *et al*., 2007), with little research having been carried out on invertebrate colonisation (French, 2006). This has lead to a lack of information on macrofaunal colonisation and community development initially following habitat restoration.

1.5. The Humber estuary and Paull Holme Strays

The Humber estuary is of high economic and social value with ports, chemical industries, oil refineries and power generation plants, together with a population of over 300,000 on its banks and 12 million inhabitants within the whole catchment (Edwards and Winn, 2006). Additionally the estuary is of great ecological importance, it has been designated a European Marine Site, SPA (Special Protection Area), cSAC (Candidate Special Area of Conservation), SSSI (Site of Special Scientific Interest) and Ramsar site. Therefore, whilst flood defence is a priority it must be carried out in compliance with the requirements of the European Habitats Directives and UK Habitats regulations 1994 (Edwards and Winn, 2006).

Paull Holme Strays, breached in 2003, was the first managed realignment site on the Humber estuary and, given the importance of nature conservation on the Humber, has the dual role of coastal protection and habitat restoration. The site covers 90 hectares and is one of twelve potential managed realignment sites within the Humber catchment (Environment Agency, 2005). Paull Holme Strays is located on the north bank of the Humber estuary, approximately 10 km to the east of Hull and 35 km from the North Sea at Spurn Point, in the middle region of the estuary.

1.6. Aims, Objectives and Hypothesis

In the previous sections the incentives for tidal flat and saltmarsh restoration, the evaluation of success and a framework summarizing the factors that are involved in tidal flat and saltmarsh restoration have been presented. Based on this information I propose the following conceptual diagram (fig 1.1) this diagram shows the theorized interlinking's between processes that occur within a restoration site. The following questions will be addressed in this thesis:

1. What are the initial physical parameters physical of a restoration site and how do they relate to biological functioning? (Chapter 2)

2. How can we assess actual dispersal (i.e. the deposition of seeds at the soil surface) of salt-marsh species by tidal water? (Chapter 3)

3. Which salt-marsh species are dispersed into a restoration site after de-embankment and what is the relationship with the established vegetation? (Chapter 3).

4. How does the vegetation develop after de-embankment? (Chapter 3)

5. What invertebrates initially colonise a restoration site? (Chapter 4)

6. How does the invertebrate community develop and change initially following de-embankment? (Chapter 4)

7. How do birds used restoration sites and do restoration sites add to the carrying capacity of the larger environment? (Chapter 5)

8. What fish initially colonise a restoration site? (Chapter 6)

9. How does the fish community develop and change initially following de-embankment? (Chapter 6)

10. Do fish diet and invertebrate community within the restoration site match? (Chapter 6)

11. Finally, Chapter 7 will set out what the different chapters have contributed to our understanding of wetland restoration, development and the evaluation of restoration success and what the implications for future restoration are.

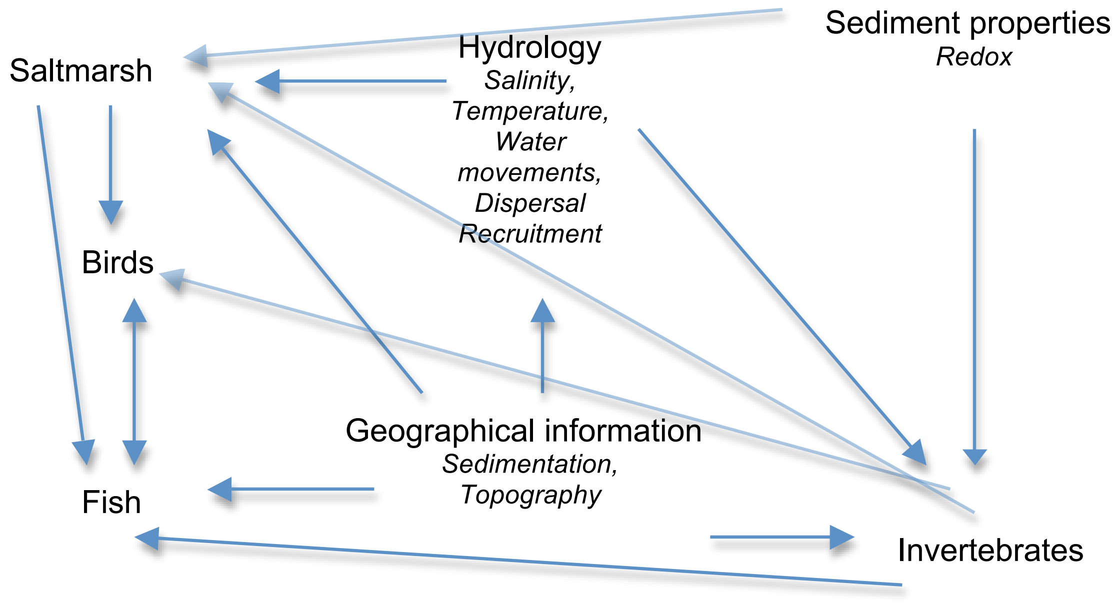


Fig 1.1: Conceptual diagram showing the interlinking between the process that occur with the restoration of an estuarine wetland habitat.

Chapter 2

Physical properties of the restored Wetland at Paull Holme Strays

2.1 Introduction

A macrostructural view of the restored habitat at Paull Holme Strays (fig 2.1) provides a basic framework for understanding the inter-relationships that drive the various ecosystems found within. At this basic level, habitats can be identified in terms of topographical and/ or bathymetric criteria. Allowing for the establishment of intertidal, subtidal, benthic and pelagic zones. This is important, as each habitat is unique and specific in the pressures that it exudes; which is reflected in the structure and functioning of organisms found within (Levinton, 2001).

Galatowitsch and Van der Valk (1996) stated that the definitive test of success for restored wetlands is how closely they function like natural wetlands. Currently, the ability of a restored wetland to replace natural wetlands function is a topic of considerable debate (Zedler and Callaway, 1999; Stolt *et al*., 2000; Hunter and Faulkner 2001; Spencer *et al*., 2001). Physical properties of restored wetlands have almost always been shown to differ from natural wetlands (Verhoeven *et al*., 2001). For example, a restored wetland will usually have a lower soil organic matter content than a natural wetland (Confer and Niering, 1992; Bishel-Machung *et al*., 1996; Galatowisch and Van der Valk, 1996; Shaffer and Ernst, 1999; Whittecar and Daniels, 1999; Campbell *et al*., 2002). Such soil conditions can lead to low growth and survival of colonizing species. As a result of the low organic matter content, the microbial communities of the soil in restored wetlands are considered less viable than those of natural wetlands (Duncan and Groffman, 1994).

As sediments are the physical foundation of every wetland ecosystem, and developing biotic communities are dependent upon them for many vital resources, like nitrogen cycling. An integration of the sediments with the plant and animal communities provides the structure for the many functions associated with wetland ecosystems (Brinson, 1993; Mitsch and Goselink, 1993; Brinson and Rheinhardt, 1996). Many of these functions, such as traps for sediment, sinks for various non-point source pollutants and zones for dentifrication of nitrate-laden ground water, are difficult to measure (Stolt *et al*., 2000). In the place of direct measurements, sediment and landscape properties can be measured and then related to the potential of the wetland function in one or more of these capacities (Maltby, 1987; Kentula *et al*., 1992). Sediment properties such as the presence of redoxomorphic features, and organic matter distribution, reflect the health of a wetland environment (Bishel-Machung *et al*., 1996).

It is widely accepted that the hydrology and hydrologic regime of wetlands is a major if not principal factor affecting the structure and functions of wetlands (Cole *et al.,* 1997). Hydrology affects the structure of both borrow pit basins, and mudflats, through the control of the deposition of sediments (Winter, 1999). Local geologic conditions can affect surface–groundwater exchange in wetlands (Winter, 1999). Hydroperiod is related to detritus processing, decomposition rates are slower and organic matter accumulates in longer hydroperiod areas of a wetland (Barlocher *et al*., 1978; Kirkman *et* *al.,* 2000).

As very few restored wetland projects take any consideration of physical characteristics (Shaffer and Ernst 1999), additional data on the development of physical properties in restored wetlands is needed to indicate whether they are actually proceeding toward natural wetland ecosystem functioning and if they will be able to replace the functions performed by the impacted wetlands (Spencer *et al.,* 2001). Furthermore, physical characteristics are useful parameters for comparison across sites, as some can be easily measured in the field or laboratory (Bishel-Machung *et al.,* 1996).

For this investigation, I chose to observe the hydrology, measure accretion/ erosion rates, temperature and ph of the borrow pits, soil particle size and organic matter/ water content shown to be: (1) an excellent indicator of soil quality; (2) well correlated with a number of important wetland processes such as respiration (Howard and Howard 1993) and denitrification (Brettar and Hofle 2002).

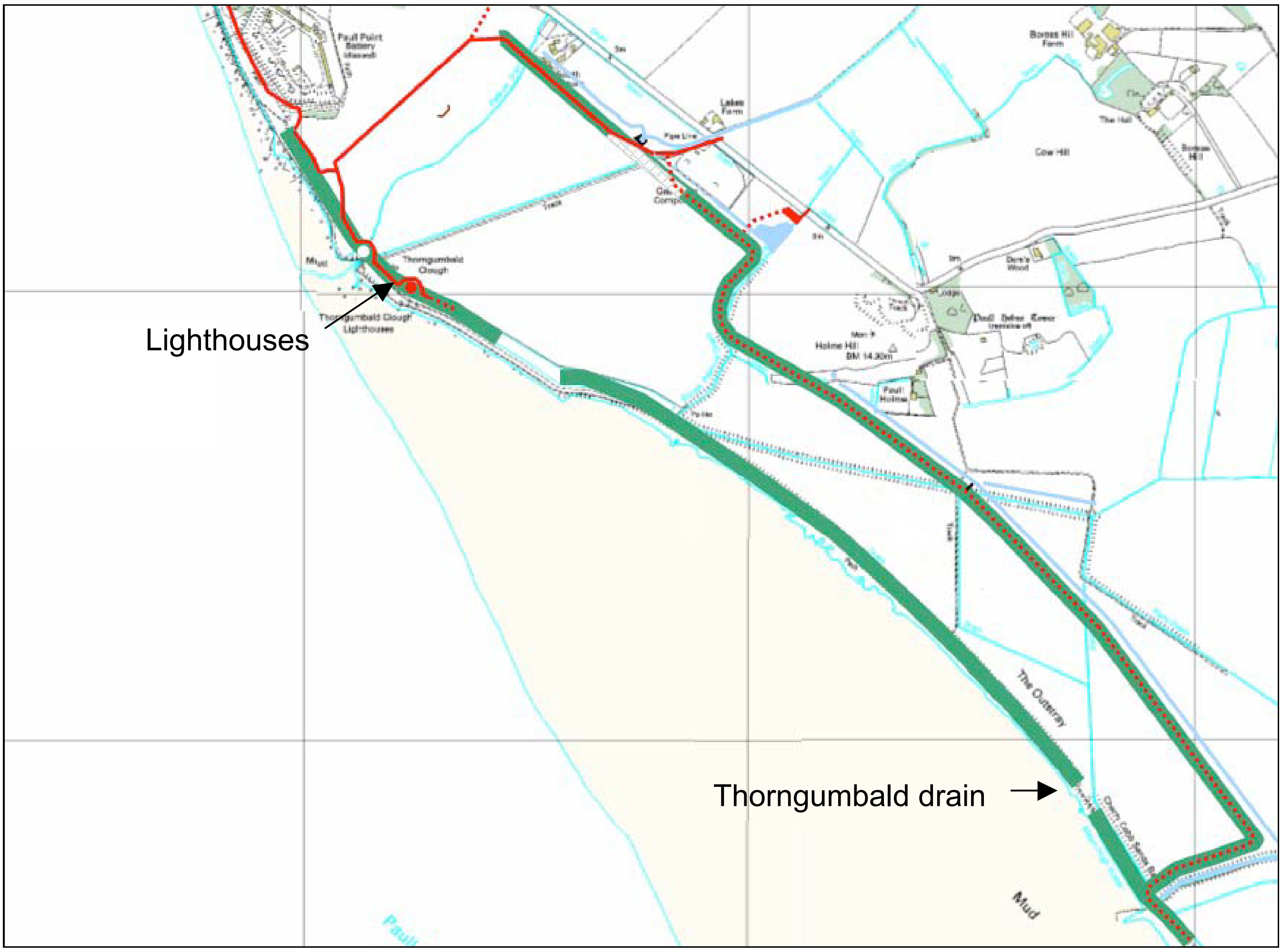


Fig. 2.1: Restoration site on the middle Humber estuary at Paull Holme Strays. The map shows both the old flood defence wall (green line) and the realigned wall (green line with red dots in it), it also shows the access roads (red lines) also shown are the location of the lighthouses and Thorngumbald drain. ** Map modified from IECS map of PHS

2.2. Materials and methods

2.2.1. Hydrographic mapping

A base map of the managed realignment site was constructed by walking the perimeter of the site at the bottom of the sea wall, marking each change in orientation of the embankment upon a GPS. This map was then used to describe the pattern of tidal inundation at the managed realignment site. A tidal cycle with a high water height of 6.7m above chart datum (falling in the middle of the spring neap cycle) was used as an approximation of mean high water, based on almanac tidal predictions (Featherstone and Lee-Elliott, 2003). High water times for each day of study were determined using online predictions (Easytide, 2004). Preliminary observations showed that tidal inundation through the eastern breach did not occur on a 6.7m tide, therefore mapping the inundation of the site could be completed through mapping the inundation at the main breach at the west end of PHS. Mapping was done on two consecutive days. Mapping took place from two vantage points on top of the flood defence wall on opposite sites of the western area of the site; this was done to provide a complete view of tidal action from within the site. Maps were drawn every 30 minutes starting at high water and ending at low water

2.2.2. Topographical survey

38 transects were set-up at 70-metre intervals along the Northwest-Southeast axis of the site. Elevation sampling points were defined using a UTM grid with a 50m resolution, which were located by GPS. Sampling points were approximately 71m apart according to Pythagorean geometrical theorem (Weinstein, 1999). Transects were of different lengths, due to changes in the width of the site. In addition to these sampling points there were a number of supplementary points taken in order to improve the accuracy of the topography of the site. The latter included taking measurements along the inside of the site perimeter along the bottom of the sea walls.

To integrate the elevation measurements to the geodetic OS network, comprising elevations above Ordnance Datum (OD), the measurements were all made in relation to the OS benchmark immediately to the north of the site. The height of the benchmark is 2.246m OD (Garbutt, 2004).

Measurements were recorded using a Horizon 2024 theodolite and an ‘E-staff’. Elevations were made using the line of collimation method (Bannister and Raymond, 1975; Bannister et al, 1998; Clancy, 1991). This technique requires that sets of measurements be taken relative to a virtual horizon plane of constant elevation (Fig. 2.2.), until distance or terrain make the E-staff no longer visible through the scope. When this occurs the position of the scope is reset, using a ‘fore sight’ before the scope is moved and a ‘back sight’ to the same E-staff lactation after the scope is moved. This establishes a new horizontal plane, and a new set of ‘intermediate sights’ can be made that can be related back to the previous sets of measurements.

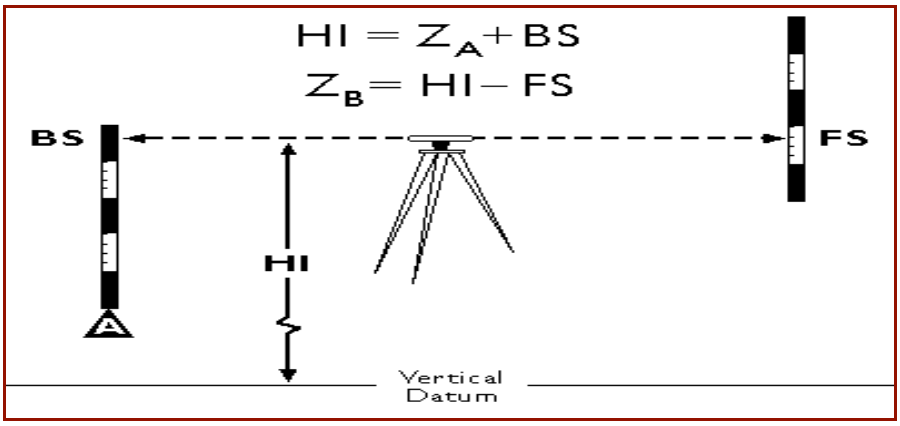


Fig. 2. 2: Line of collimation survey method. A leveling instrument is positioned midway between a point at which the ground elevation is known (point A) and a point whose elevation is to be measured (B). The height of the instrument above the datum elevation is HI. The surveyor first reads a backsight measurement (BS) off of a leveling rod held by his trusty assistant over the benchmark at A. The height of the instrument can be calculated as the sum of the known elevation at the benchmark (Z_A) and the backsight height (BS). The assistant then moves the rod to point B. The surveyor rotates the telescope 180°, then reads a foresight (FS) off the rod at B. The elevation at B (Z_B) can then be calculated as the difference between the height of the instrument (HI) and the foresight height (FS). **Image adapted from Wolf & Brinker, 1994

2.2.3. Accretion/ Erosion Monitoring

Pairs of bamboo canes were placed 150 cm apart and pushed into the sediment to a depth to ensure their stability, and their ability to withstand displacement by the currents. The heights of the two canes were adjusted with the aid of a spirit level, so that the tops of the canes across the top were level.

Many of these accretion/ erosion stations were the same that were used in a previous study done by Boyes and Mazik (2004) where a total of 8 monitoring stations were positioned within the managed realignment site provide measurements of change at the high, middle and low shore (Fig. 2.3). Monitoring took place between December of 2003 and June of 2005. With data collected prior to March of 2004 came from Boyes and Mazik (2004).

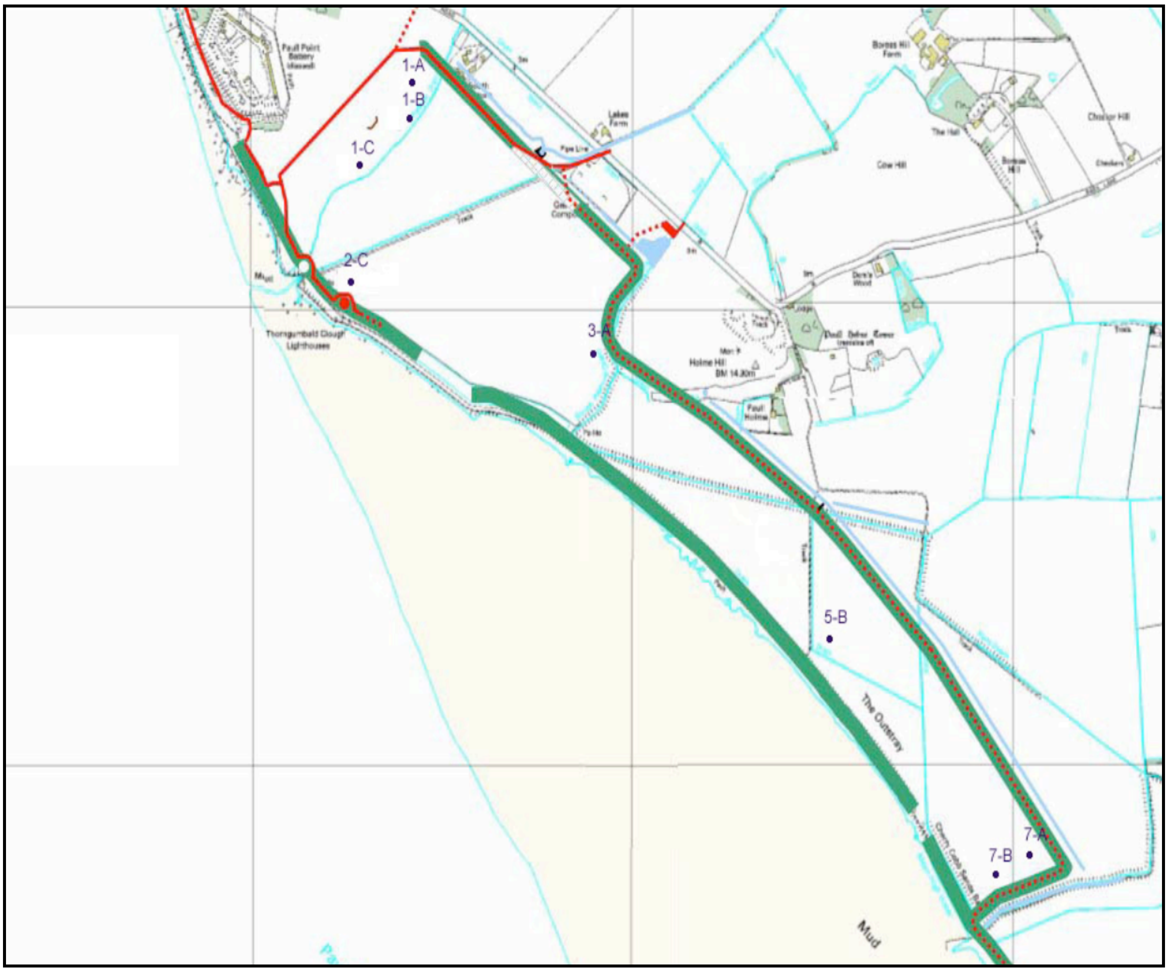


Fig. 2.3: Accretion/ erosion monitoring stations from within the restoration site at Paull Holme Strays. ** Map modified from IECS map of PHS

Ranwell (1964) methodology for calculating rates of change in accretion and erosion was used. Measurements of the elevation of the sediment relative to the tops of the canes were taken at one-month intervals. At each station a cane was placed horizontally across the top the pair of vertical canes, and held in place with 'Blu- tak'. Five measurements were then taken between the distances from the mudflat surface to the horizontal cane at 25 cm intervals between the two vertical canes. Measurements were made no closer than 25 cm to the parallel canes, due to the potential of scouring around each cane, which could distort the accretion/ erosion rates determined across the centre of the station (Boyes and Mazik, 2004). The five measurements were used to compute a single mean value and standard error (SE) for each station, for each monitoring occasion.

2.2.4. Water quality measurements

Measurements of water quality of the four major water bodies at the site were taken monthly from June 2005 to July of 2005 (Fig. 2.4) at low water at one point. Measurements for temperature and pH were taken using a pH probe by HANNA. The pH meter was calibrated before each sampling occasion using standard pH solutions. Temperature and pH measurements were taken after a period of one minute, this allowed the probe adjust to the physo- chemical conditions at each site. Measurements for salinity were taken using a Bellingham and Stanly handheld refractometer. Before each sampling occasion the refractometer was calibrated to zero using distilled water, and the prism was rinsed with distilled water between each measurement.

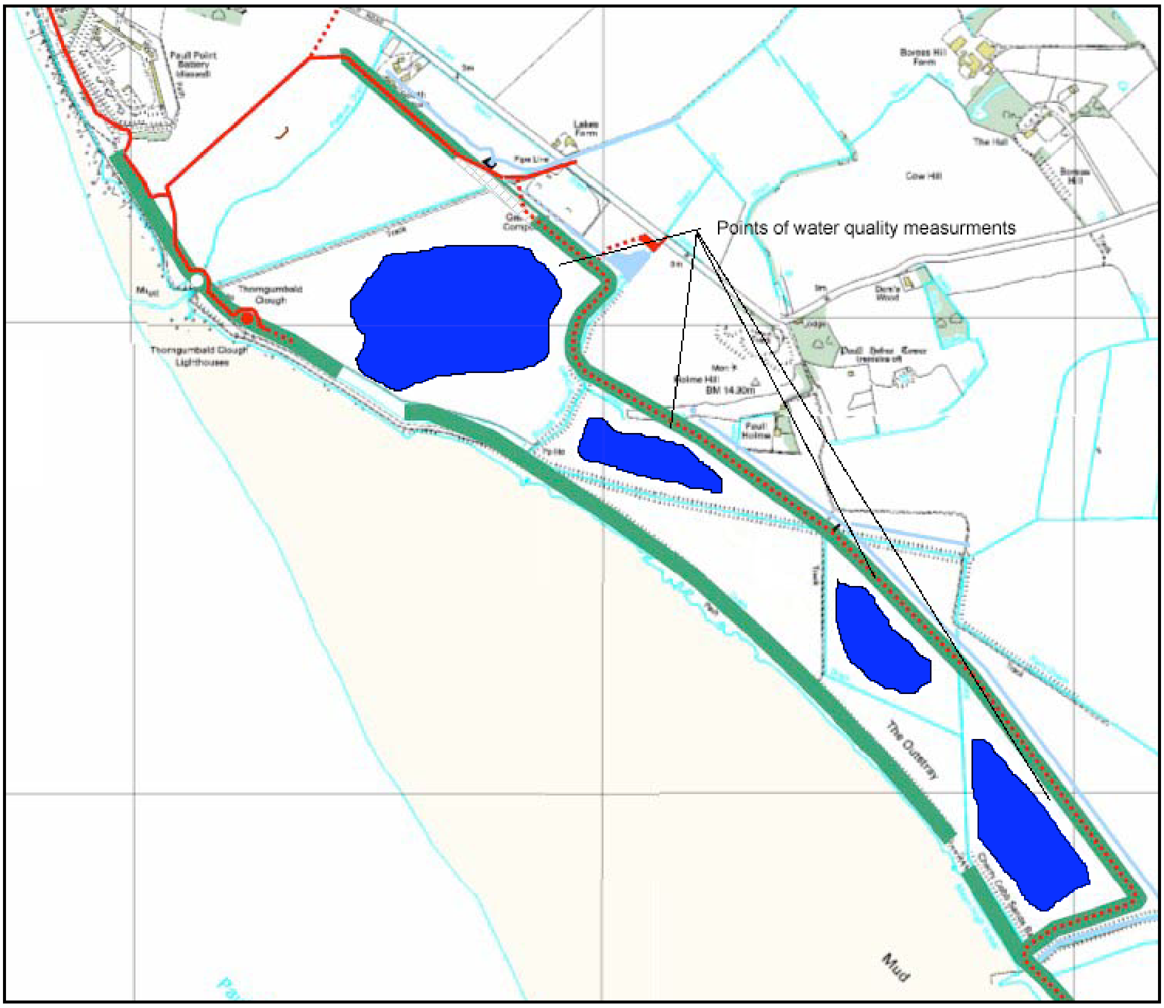


Fig. 2.4: Locations of water quality measurements from within the restoration site at Paull Holme Strays. ** Map modified from IECS map of PHS

2.2.5. Redox measurements

Redox measurements were taken from12 stations (Fig. 2.5) within the restored site every centimetre to a depth of 5cm, at low water from November 2004 to July 2005. These sites correspond to sites that were also used to look at benthos colonization (chapter 4). Measurements were taken using a HANNA HI 991002 meter and probe. Prior to each sampling occasion the probe was cleaned with HANNA HI-7061 probe cleaning solution and then stored in HANNA HI- 70300M storage solution. At each site for each depth the probe was allowed to stabilize to the physio- chemical parameters prior to any readings being made. Between each station the probe was cleaned with distilled water.

Fig. 2.5: Redox monitoring stations at Paull Holme Strays. Redox measurements were taken every cm to a depth of 5 cm. ** Map modified from IECS map of PHS

2.2.6. Sediment Particle size analysis, Water and Organic content

Sediment samples were collected monthly from July 2004 to September 2005 at 14 stations through out the site (Fig. 2.6). Sediment samples were taken at low water from the top 2 cm of undisturbed surface material with a plastic scoop. Sediment was collected at least 1cm away from any surface disturbance.

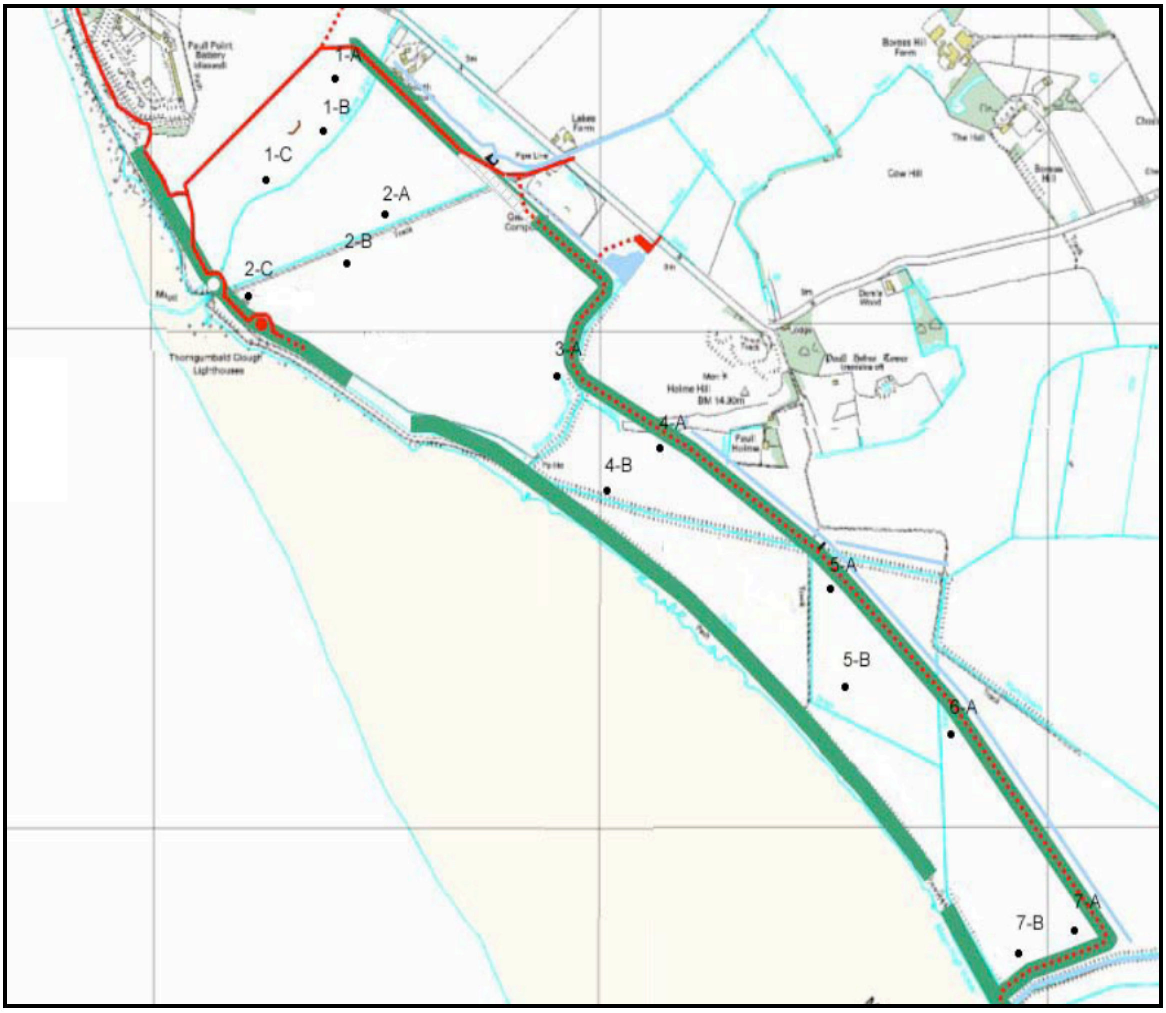


Fig. 2.6: Sediment Particle size, Water and Organic content monitoring stations within the restoration site Paull Holme Strays. Sediment samples were collected from the top 2cm. ** Map modified from IECS map of PHS

Samples were placed into labelled plastic bags and then stored at approximately 4° C by placing them on wet ice inside of an ice chest until returned to the laboratory. Once in the lab they were stored at 4° C until analysis. Samples were thawed and sieved to check for particles of a size of 1mm or larger. If found all particles of a diameter of 1mm or larger were removed and analyzed using the sieve method. Samples and parts of samples that had no sediments larger than 1mm were transferred into 100 ml glass beakers and homogenized by vigorously stirring them with a glass rod for 1 minute to prevent bias due to settlement. Once homogenized, approximately 1 to 2 grams of each sample was placed into a 1000 ml beaker along with 800 ml of 80° C water. Water of 80° C was used as it reduces the solubility of gases, which result in the formation of bubbles. Bubbles are a problem as they are measured with the sample and are counted as particles-biasing the results. Samples were then analyzed using laser diffraction with a Malvern Mastersizer hydro 2000 MU, which calculated the grain size of the sediments in (PHI).

50 grams of each sediment sample was taken for water and organic content analysis. Samples were placed in pre- weighed crucibles that had been cleaned and dried in an oven set at 85° C. Samples were then weighed ±0.001g and placed in an oven set at 85° C for 24 hours. The dried samples were then reweighed, and the change in sediment mass i.e. water content was expressed as a percentage of weight loss after 24 hours in an 85° C oven. The samples were put into a muffle furnace set to 475° C; once the furnace reached 475° C the samples were kept at that temperature for 4

hours (Buchanan, 1984). After 4 hours the samples were allowed to cool to a temperature of 80° C before being removed from the furnace. They were then reweighed to determine the change in sediment mass and the difference i.e. the organic content which was expressed as a percentage of weight loss on ignition (% LOI).

2.3. Results

2.3.1. Topography

The difference between the highest and lowest elevation from within the site was 9.3 m. The difference between the elevation of the lower limit of the developing saltmarsh and the developing mudflats of the site was 2.7 m. There was a difference of 8.5 m between the upper and lower areas on the west end of the site and a difference of 1.7 m on the east end of the site. Elevations above OD are displayed graphically in Fig. 2.7. Areas of Fig. 2.7 displayed in red and orange are never flooded, while areas in yellow are only flooded on tides with a high water of 6.7 m or greater, and areas in green and blue are flooded with every high water.

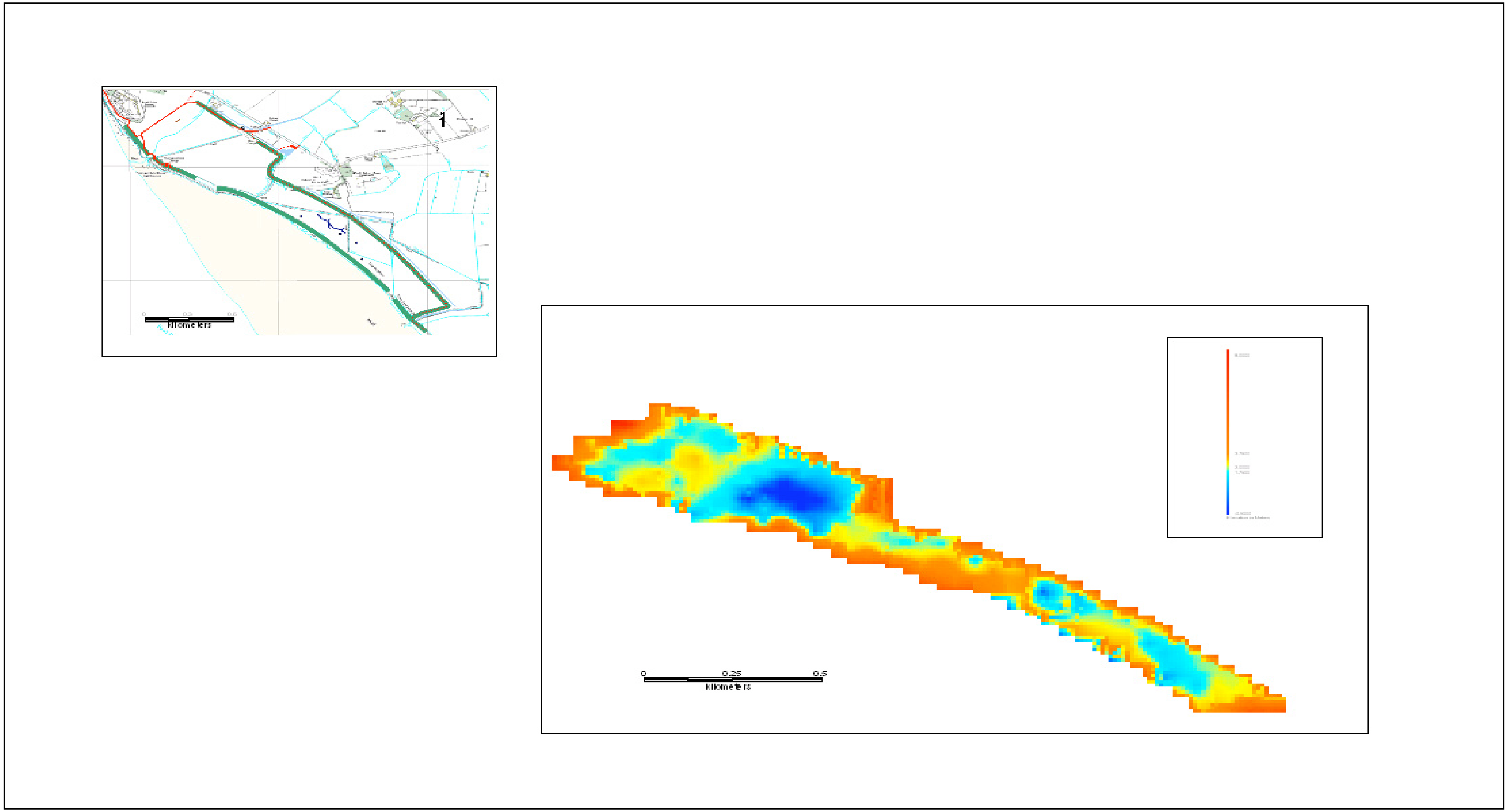


Fig. 2.7. Contour map of Paull Holme Strays restoration wetland.

2.3.2. Hydrographic mapping

Paull Holme Strays experiences a semi- diurnal tidal regime with a mean tidal range of 2.7 metres and a spring tidal range of 5.8 metres. All tidal exchange occurs through the two breaches that have been cut into the existing flood defence wall, with a relatively simple flood- ebb pattern. Water is forced into the restoration site through the two breaches in the existing floodwall with the flooding tide, once the water has entered through one of the two breaches it then slowly dissipates through the restoration site by flooding the lowest lying areas first. Once the lowest lying areas flood and as water continues to enter the site higher laying areas begin to flood. However, the lack of a secondary system of smaller channels limits the circulation into the mid- and high- marsh plain on neap tides.

An estimation of the area of inundation at stages throughout the tidal cycle was made (Fig. 2.8). The sequence indicates that the tidal curve is sigmoid with slow change occurring just before and after times of high and low tide are separated by periods of rapid change. This suggest that draining of the tidal prism is a constant process and that there is no low water slack period, as soon as the tide has finished ebbing it then starts to flood again.

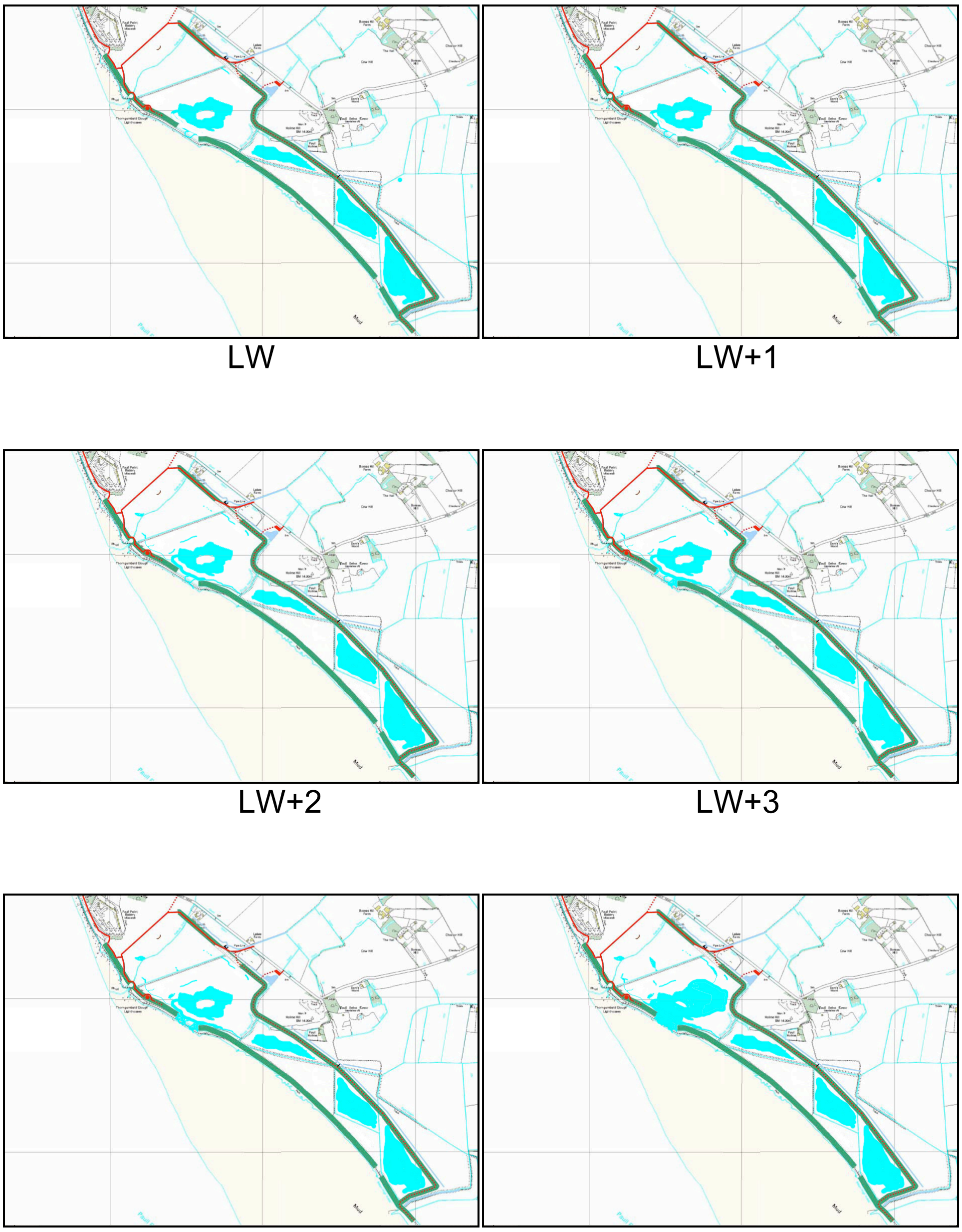
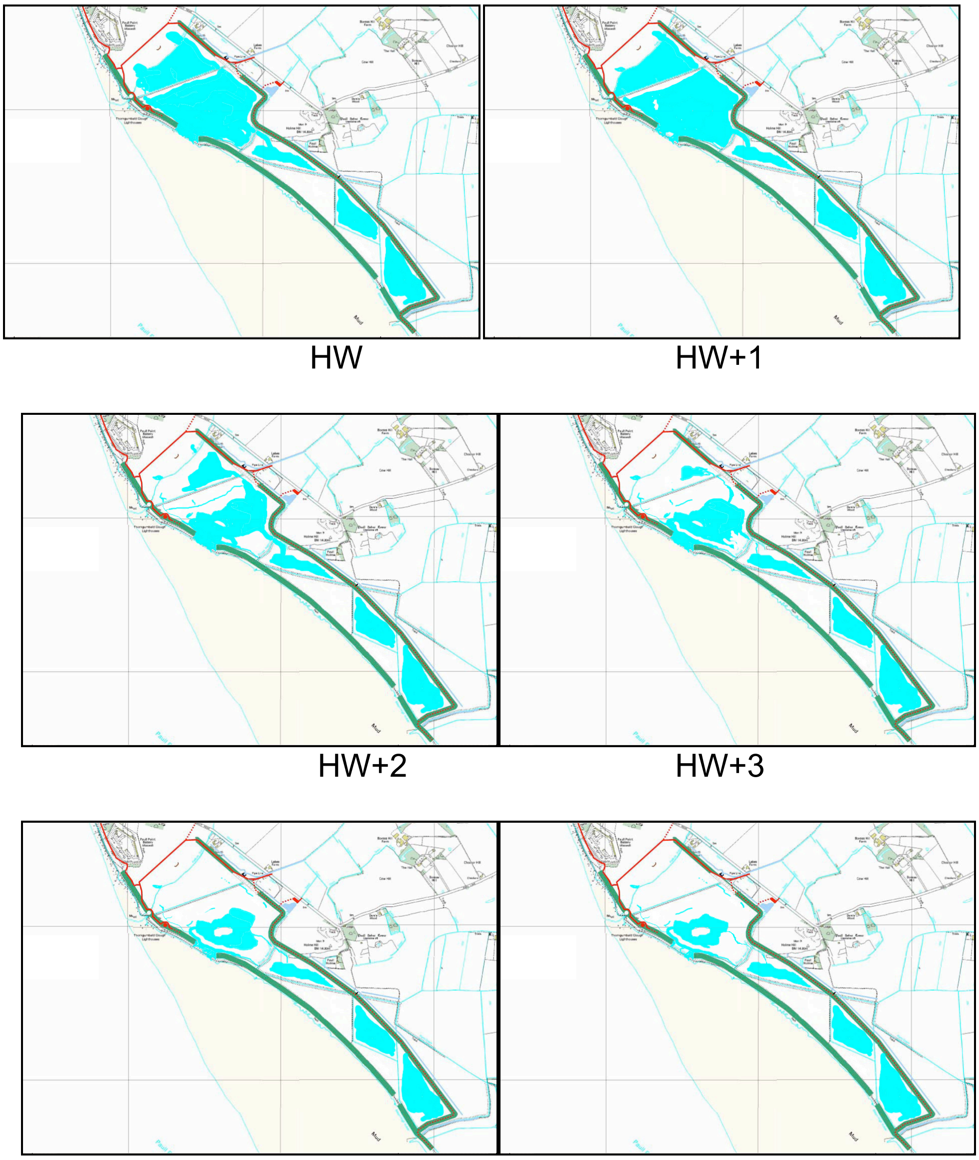

LW
LW+1
LW+2
LW+3
LW+4
LW+5

Fig. 2.8: Inundation of the Paull Holme Strays based on a mean high water tide of 6.7 m. Maps start in the upper left at Low water and work up in hour increments to Low water + five hours.



HW HW+1

HW+2 HW+3

HW+4 HW+5

Fig. 2.8: continued. Maps start in the upper left at High water and work up in hour increments to High water + five hours.

2.3.3. Sediment Particle size analysis, water and organic content

None of the sites monitored had gravel present in as part of there sediment composition (Fig.2.9). Therefore sediments were composed of just a mixture of sand and silt/ clay. Sand content ranged from a minimum of 6% to a maximum of 78% of the sediment composition based upon site, and silt/

clay ranged from a minimum of 83% to a maximum of 93% of the sediment composition based upon site (Table 2.1). A Kruskal-Wallis test showed that there was a statistically significant difference in grain size (PHI) (χ^2= 38.899; d.f. = 24; p< .05), percent sand (χ^2= 42.637, d.f. = 24; p< .05), and percent silt/ clay (χ^2=42.568; d.f. = 24; p< .05) of the samples based upon site, however due to a lack of degrees of freedom it was not possible to determine where the differences lie.

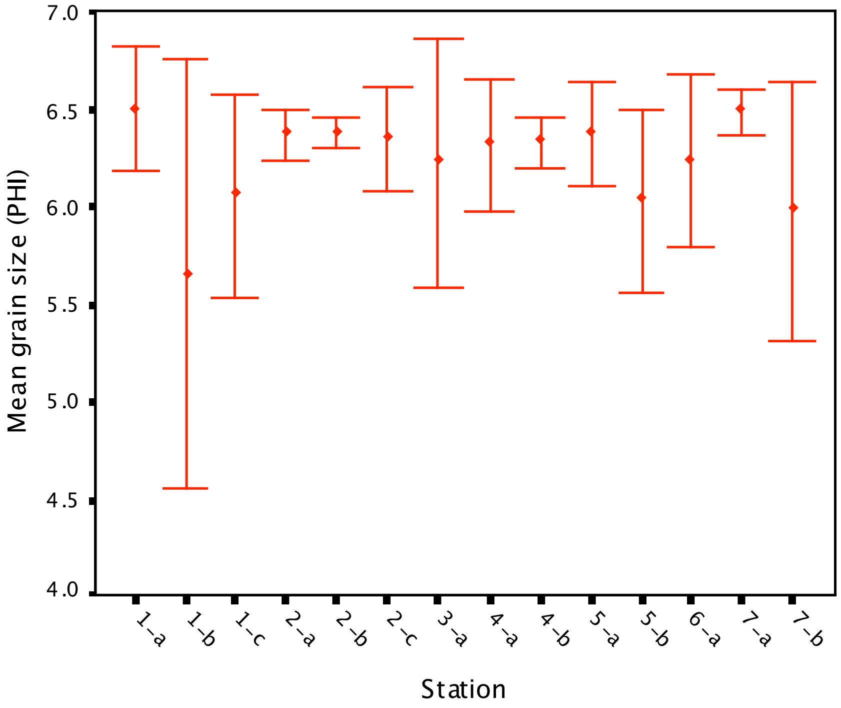


Fig. 2.9: Mean sediment grain size of the top 2mm of sediment (± SD) from sampling stations within Paull Holme Strays. Data collected monthly between July 2004 and September 2005.

Sites contained on average between 9 and 12 % organic matter. A one-way ANOVA, together with Tukey's test was performed to test the differences in organic content based upon station as the data met the assumptions of the parametric method based upon tests for normality and homogeneity of variance. The analysis showed a significant difference in the amount of organic content based upon site (F= 2.828; d.f.= 11,99; p< .01), stations 1-A and 1-B had significantly higher organic content than station 6-A (Fig. 2.10).

Table 2.1: N, Percent sand and silt (SE) from the top 2cm of sediment at each site. Data collected monthly between July 2004 and September 2005.

Site	N	% Sand	% Silt
1-a	13	6.52 (2.00)	93.47 (2.00)
1-b	13	19.14 (7.50)	80.85 (7.50)
1-c	13	14.66 (4.24)	85.34 (4.24)
2-a	4	78.34 (31.80)	68.20 (23.31
2-b	4	33.15 (25.69)	66.84 (25.69)
2-c	13	17.50 (9.35)	82.50 (9.35)
3-a	14	16.40 (7.20)	83.59 (7.20)
4-a	14	7.85 (2.54)	92.15 (2.54)
4-b	14	16.94 (9.36)	83.05 (9.36)
5-a	14	15.80 (8.20)	84.19 (8.20)
5-b	14	12.90 (4.76)	87.09 (4.76)
6-a	14	14.94 (7.36)	85.05 (7.36)
7-a	14	7.10 (1.02)	92.92 (1.03)
7-b	14	16.89 (6.14)	83.10 (6.14)

The water content of the sediments taken from each sample is shown in Fig. 2.11. The amount of water varied by station ranging from 50% on the west end to 30% on the east end. A one-way ANOVA together with a Tukey's test showed a significant difference in the amount of water present based upon site (F= 7.304; d.f.= 11, 88; $p < 0.01$). Site 5-b on the east end of the site had significantly less water content ($p < 0.05$) than stations 4-a, 1-b, 3-a, 2-c, 1-a and 1-c. Station 6-a had significantly less water content ($p < 0.05$) than stations 3-a, 2-c, 1-a and 1-c. Site 7-a had significantly less water content ($p < 0.05$) than stations 2-c, 1-a and 1-c. Station 7-b has significantly less water content ($p < 0.05$) than station 1-c. Fig. 2.12, displays the homogeneous subsets found.

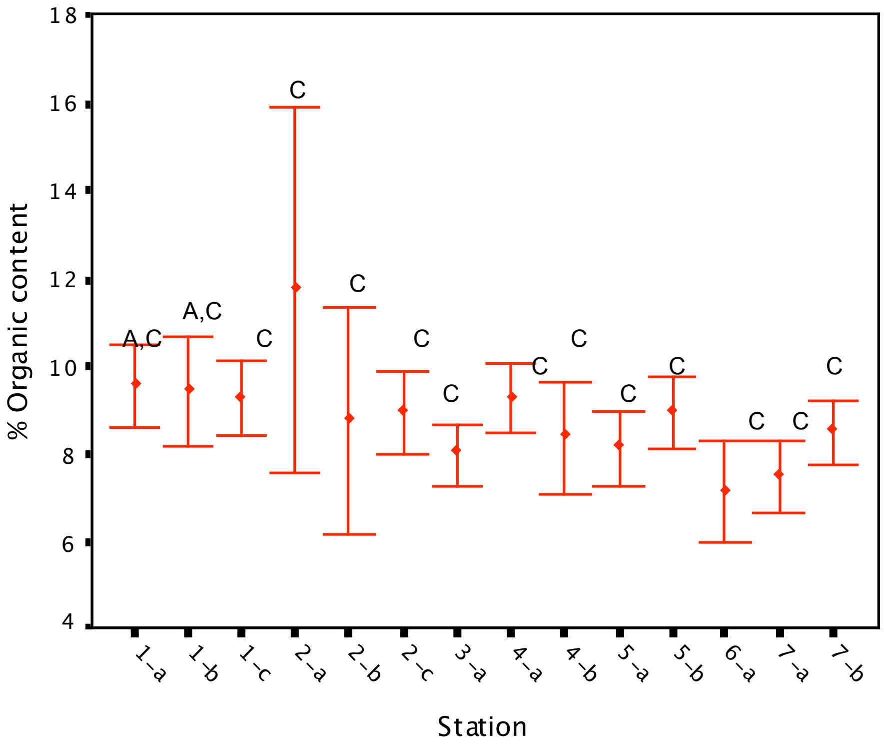


Fig. 2.10: Mean percent organic content of the top 2mm of sediment (± SD) from sampling stations within Paull Holme Strays. Letters indicate a significant difference in the % organic content (p< 0.05, Tukeys test). Data collected monthly between July 2004 and September 2005.

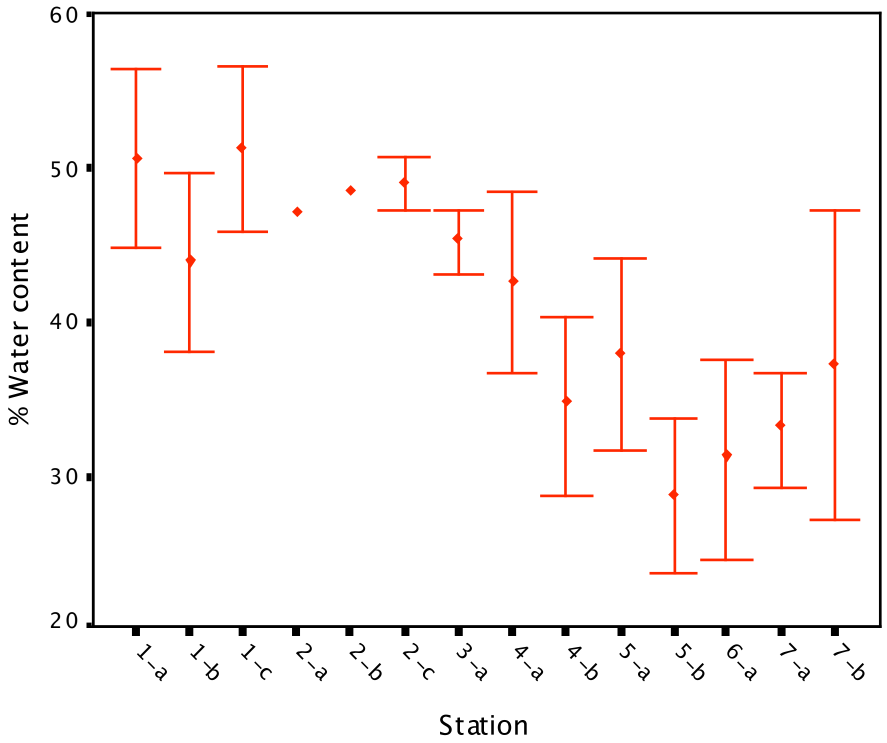
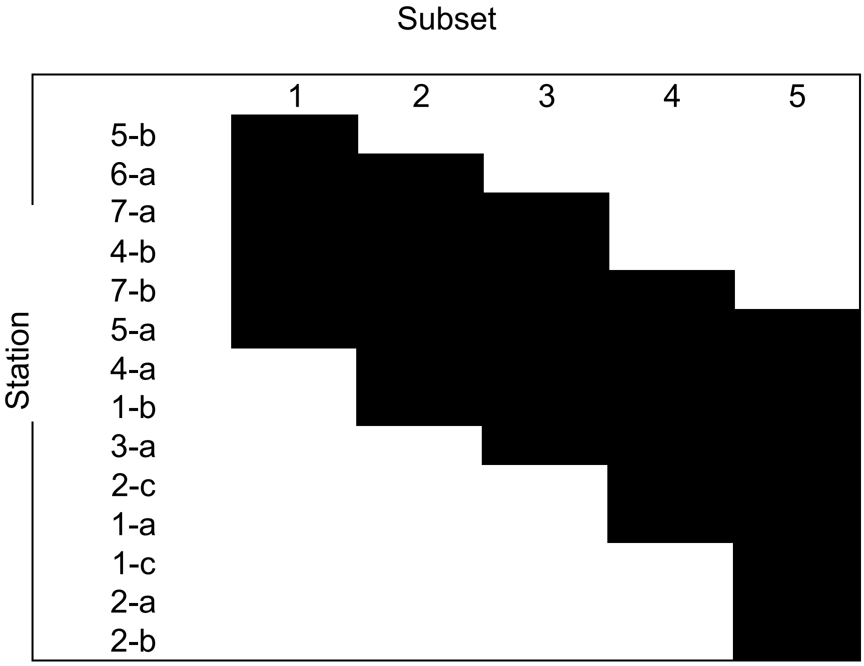


Fig. 2.11: Mean percent water content of the top 2mm of sediment (± SD) from sampling stations within Paull Holme Strays. Data collected monthly between July 2004 and September 2005.



Station	1	2	3	4	5
5-b	■				
6-a	■	■			
7-a	■	■	■		
4-b	■	■	■		
7-b	■	■	■	■	
5-a	■	■	■	■	■
4-a		■	■	■	■
1-b		■	■	■	■
3-a			■	■	■
2-c				■	■
1-a				■	■
1-c					■
2-a					■
2-b					■

Fig. 2.12: Homogenous subsets of similarity of water content between the 14 stations examined within Paull Holme Strays.

2.3.4. Redox potential

Redox potential ranged from oxidizing to reducing conditions (aerobic to anaerobic sediments) (Fig. 2.13). At sites on the western end where inundation is more consistent and frequent the redox potential discontinuity (RPD) was at a lower depth than on the east end of the restoration site (Fig. 2.14). However, a one-way ANOVA showed that a difference in the RPD between sites was not significant (F= 1.86; d.f. = 10; 68, p>.05). A plot of the Eh_4 (mV) (redox potential at 4 cm) (Fig. 2.15) showed that sites on the west end of the restoration site had reduced and anaerobic soils at a depth of 4cm while sites on the east end had slightly oxidized and aerobic soils. A one-way ANOVA showed that differences in Eh_4 (mV) based upon site were significant (F= 2.776; d.f. = 11, 78; p< .05). A Tukey's test showed that sites 1-a and 1-c where inundation was consistent and frequent had reduced soils and were significantly different (p< .05) than the soils at site 6-a which had oxidized soils due to less frequent inundation. Seasonal patterns of variation in redox potential were observed with summer months having more reduced soils however; the results were not significant as tested with a non-parametric Kruskal-Wallis test (χ^2= 7.617, d.f. = 3; p> .05).

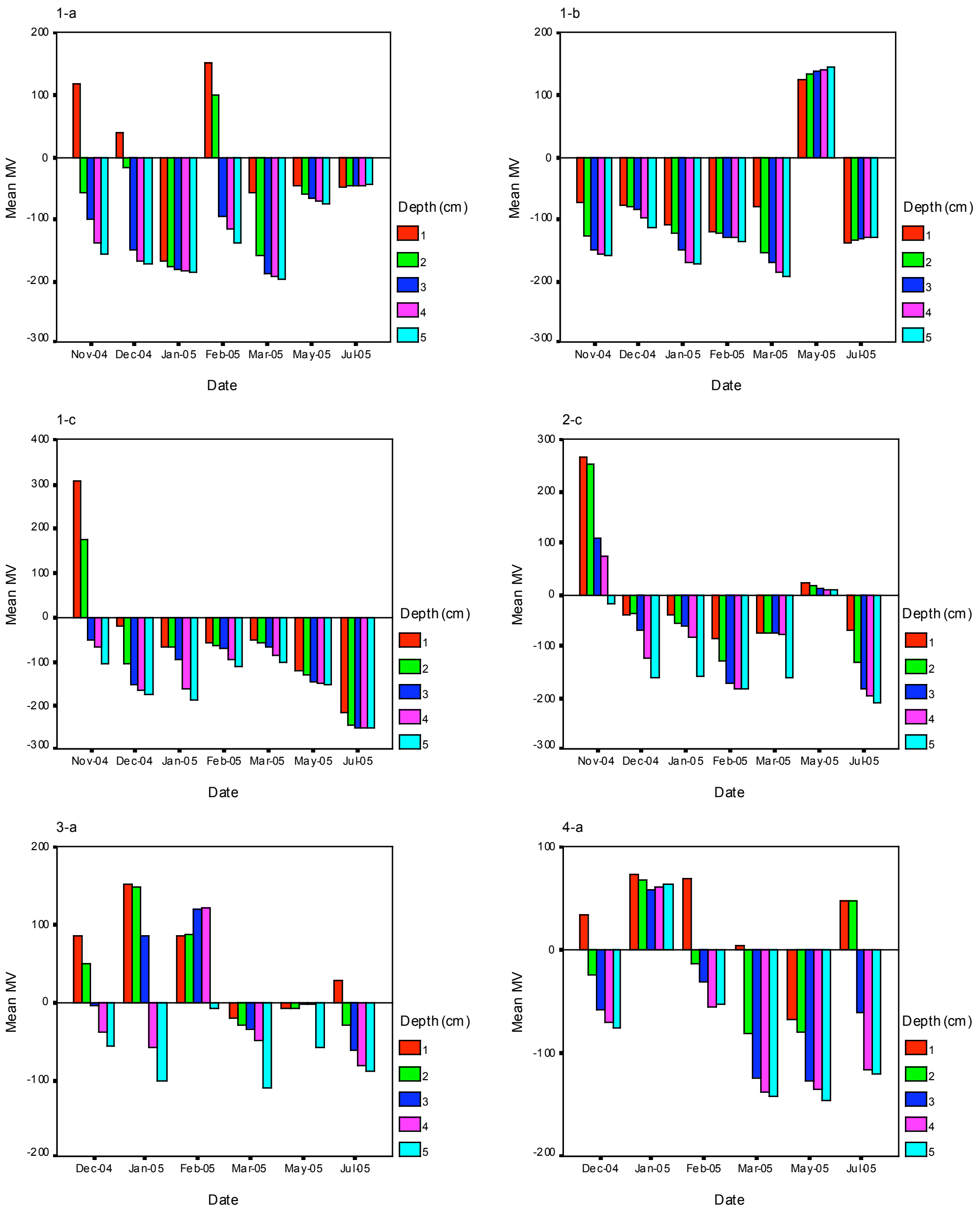


Fig. 2.13: Redox potential at 1-cm, 2-cm, 3-cm, 4-cm and 5-cm depths for multiple sites in the constructed wetland at Paull Holme Strays. Data was collected between November 2004 and July 2005.

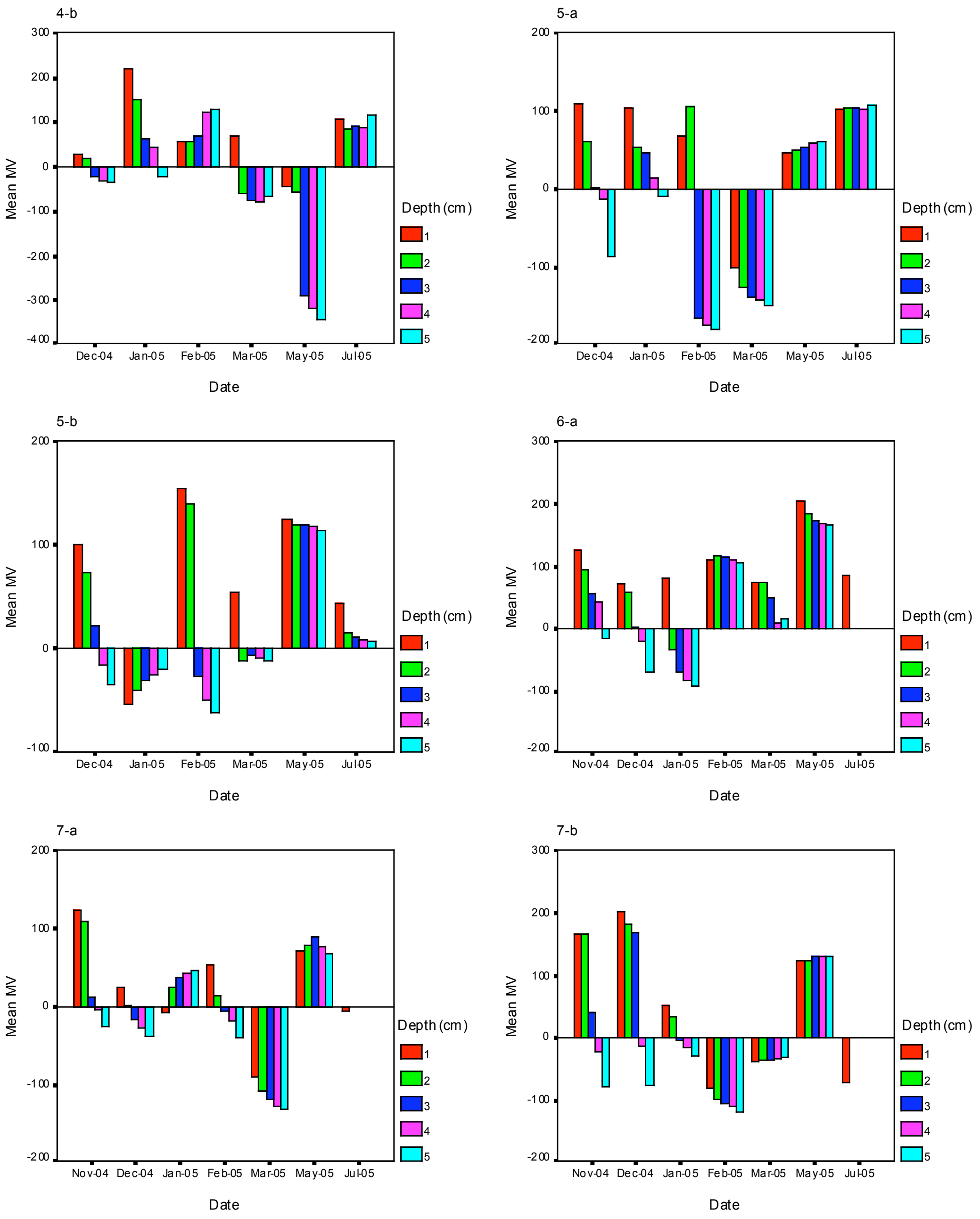


Fig. 2.13: Continued.

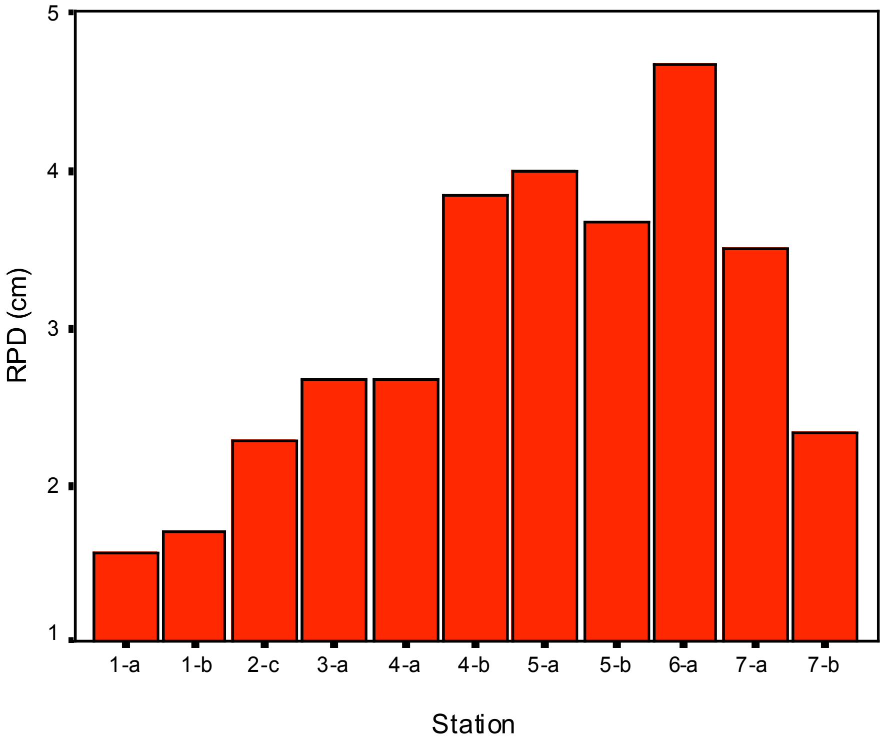


Fig 2.14: The redox potential discontinuity (RPD) based upon station. Data was collected between November 2004 and July 2005.

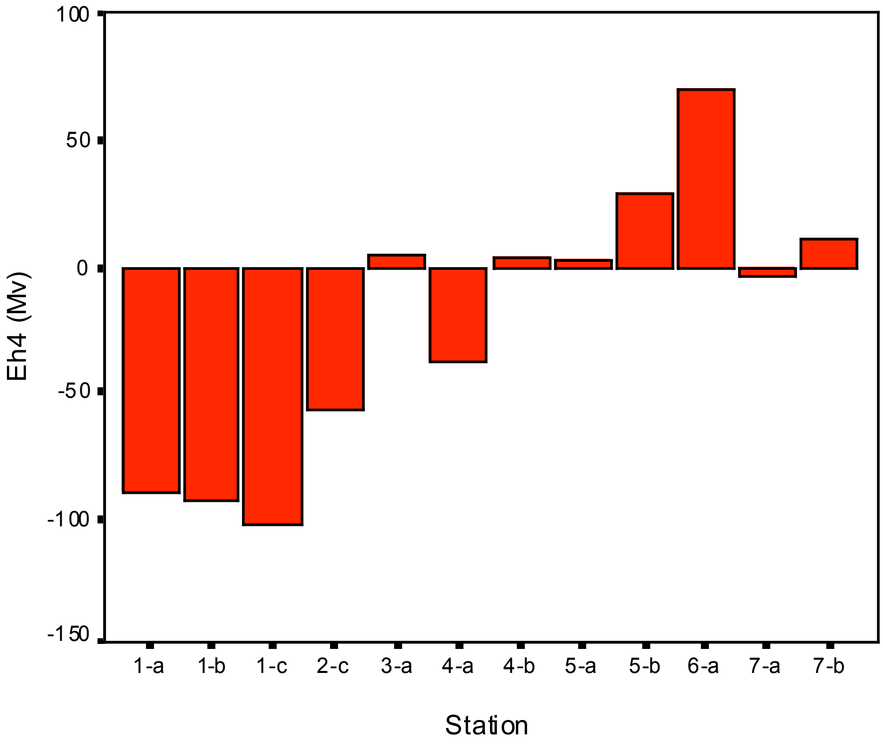


Fig 2.15: Eh_4 (mV) (redox potential at 4 cm) based upon station. Data was collected between November 2004 and July 2005.

2.3.5. Accretion/ Erosion

Mean total accretion/ erosion rates at the eight stations as recorded over the monitoring period can be seen in Fig. 2.16. The overall trend presented by the data is one of accretion. Fig. 2.16 shows that the higher levels of accretion are occurring in the far west of the site (stations 1-A, 1-B), however a high level of accretion can also be seen at station 7-A (Table 2.2).

A one-way ANOVA test, together with Tukey's test was performed on the accretion/ erosion rates by station. This showed no significant differences in the accretion/ erosion rates between stations for all occasions when measurements were taken (F= 2.003; d.f. = 1, 6; $p>0.05$).

Table 2.2: Summary of measured accretion rates across the site from west to east.

Accretion Station	Location within site (East = E, West = W)	Mean change in elevation (mm)	Mean daily accretion rate (mm per day)	Net accretion (A) or Net erosion (E)
1-a	W	68	0.16	A
1-b	W	53	0.14	A
1-c	W	16	0.04	A
2-c	W	43	0.11	A
3-a	W	23	0.05	A
5-b	E	20	0.10	A
7-a	E	52	0.13	A
7-b	E	9	0.05	A

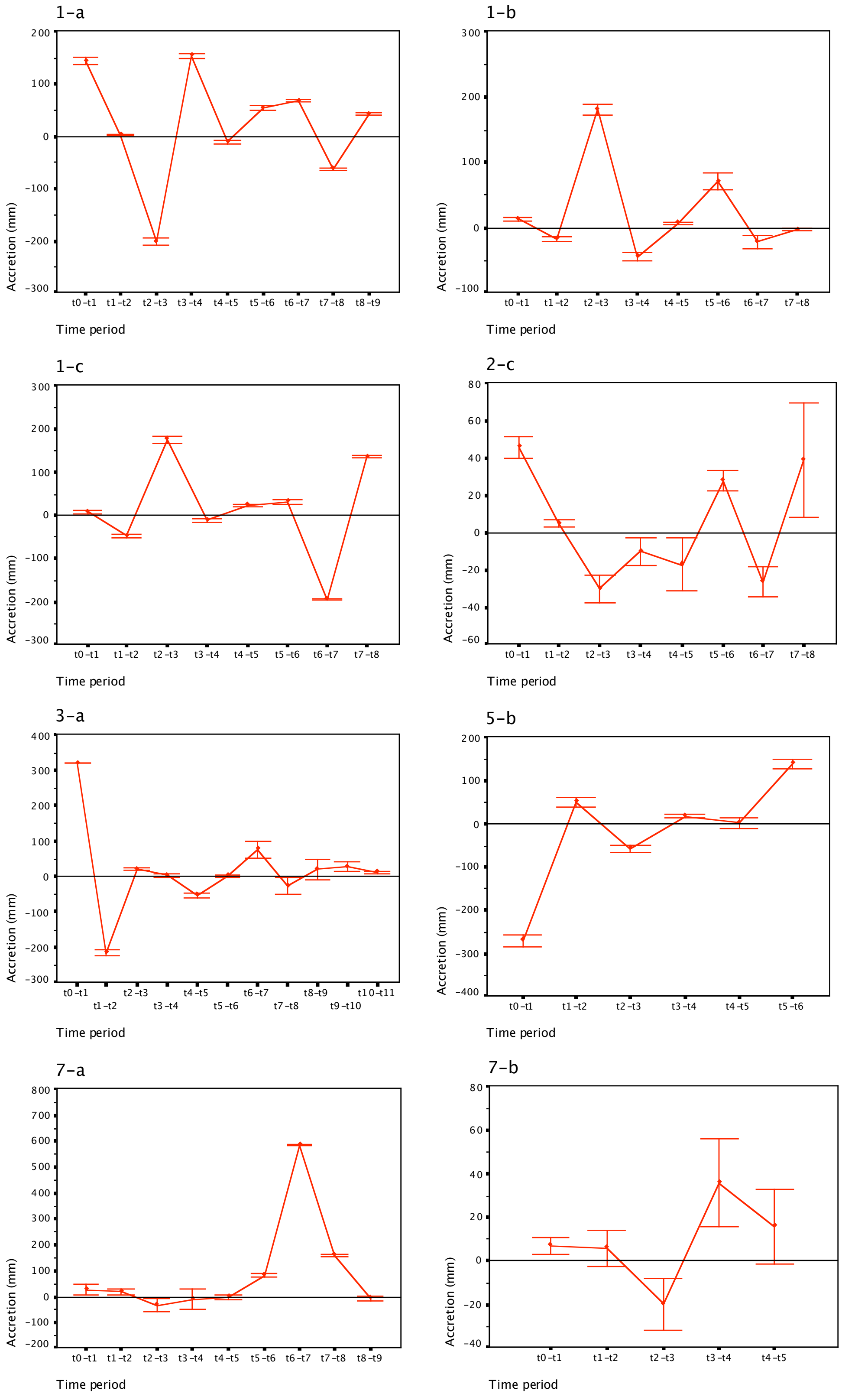


Fig. 2.16: Mean change in elevation (Accretion/ Erosion) in millimetres (± SD) over time by station. Time period represents the number of months sampled. Samples were collected from December 2003 and June 2005. Stations are organized from the Northwest to Southeast.

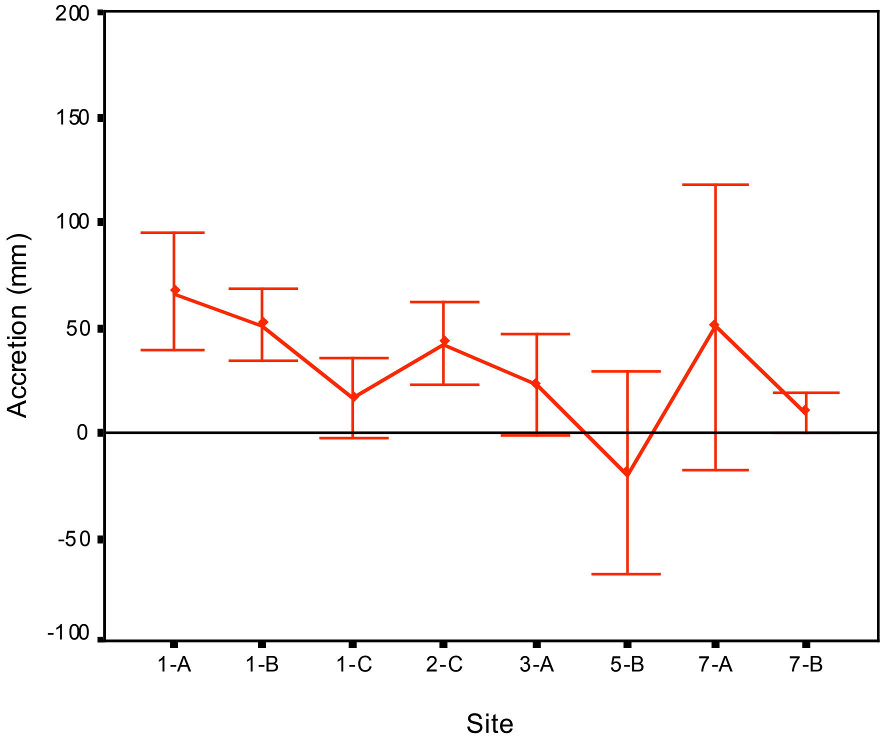


Fig. 2.17: Mean change in elevation (accretion/ erosion) (± SD) at stations from Northeast to Southwest from December 2003 to June 2005.

2.3.6. Water quality

Temperature (°C), pH and salinity were measured in the four borrow pits located within the site on a monthly basis between June 2004 and July 2005. A Friedman's non- parametric test showed no significant difference between the locations in relation to water temperature. However, Friedman's non- parametric test showed changes in water temperature due to seasonal effects were statistically significant (Fig. 2.18) (χ^2= 43.423, d.f. = 11; p<0.01), with a 15°C temperature drop in winter.

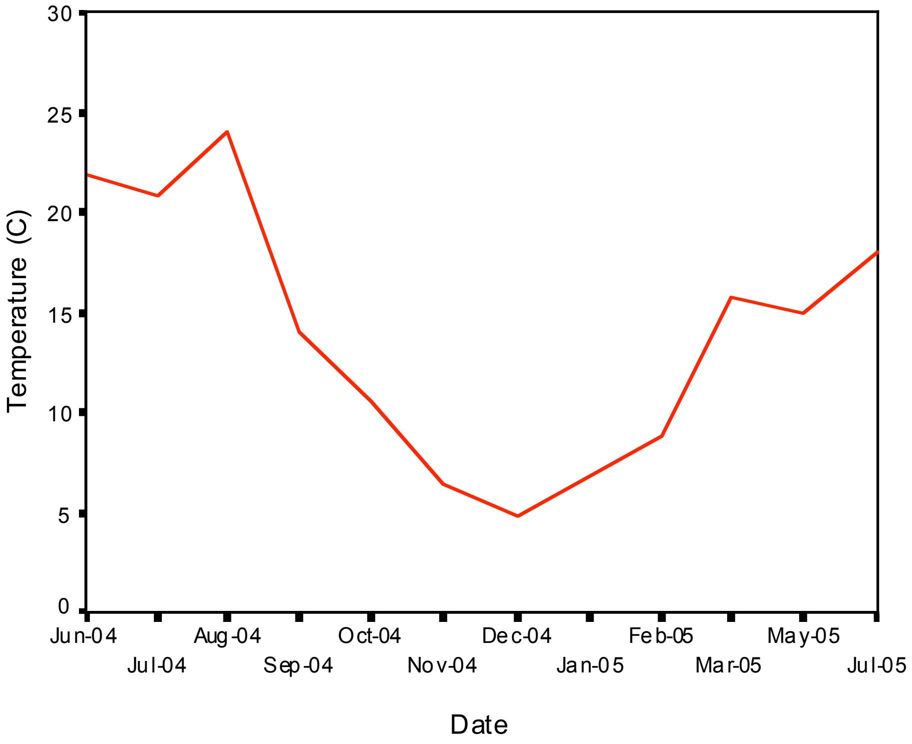


Fig. 2.18: Average monthly temperature of the water within PHS (restoration site) from June 2004 to July 2005.

Borrow pit location did not have a significant influence on the pH, although there were statistically significant changes with season (Fig. 2.19) (χ^2= 33.269; d.f. = 11; p<0.01). The changes in pH point towards a trend of higher values in the summer and lower ones in the winter.

The salinity measured at each of the borrow pits showed no significant differences. But the salinity was statistically significantly different based upon date (Fig. 2.20)(χ^2= 33.187; d.f. = 11; p<0.01). Unlike the temperature and pH that showed changes based upon season, salinity remained constant except for the month of August 2004 where it was much lower. This variation could have been due to the fact that it was very stormy conditions off shore that month, with a lot of rainfall, thus allowing more fresh water than normal to enter the site, which in turn diluted the salinity.

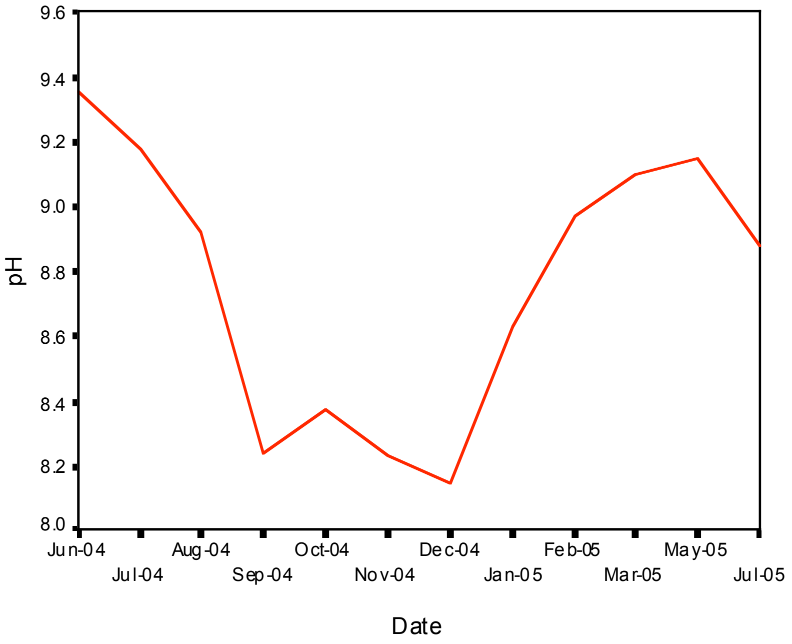


Fig. 2.19: Average monthly pH of the water within PHS (restoration site) from June 2004 to July 2005.

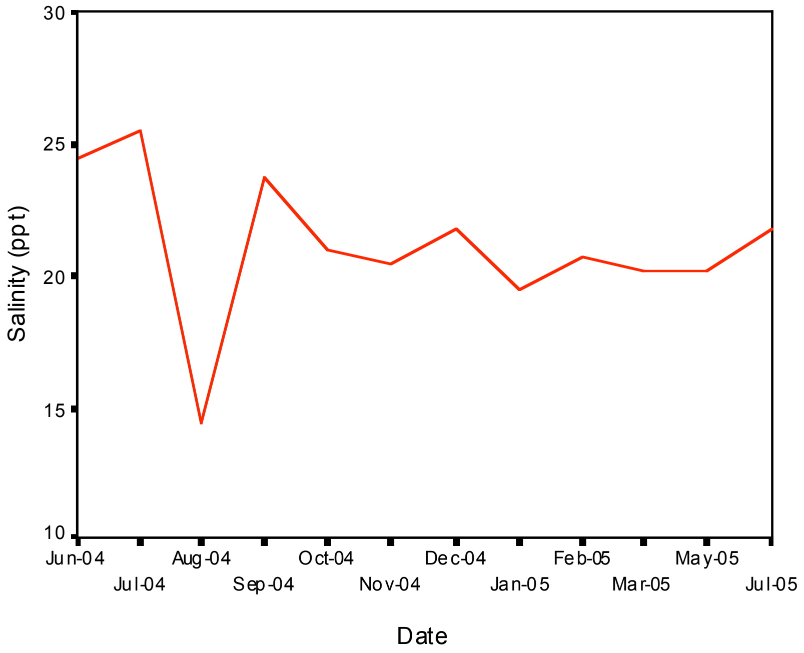


Fig. 2.20: Average monthly salinity of the water within PHS (restoration site) from June 2004 to July 2005.

2.4. Discussion

Sediments are complex assemblages of inorganic and organic material that act as the basis for the development of biological functioning within a restoration site. Any given sediment is a function of parent material acted upon by organisms and climate over time (Jenny, 1961).

Differences found in organic matter across the site were seen and could have been due to in part to input and accumulation rates of organic materials. Differences could have also been due to small particles having bigger surface area to volume ratios than large particles, thus allowing small particles allowing to build up organic matter more quickly than large particles; fine sediments (small particles) are also less oxygenated (and therefore more reduced) than coarse sediments (large particles) therefore the organic matter does not break down as quickly; fine sediments will also have larger infaunal populations and perhaps more microphytobenthos and these will increase the organic content.

Redox potential discontinuity of the substratum indicated that that anaerobic processes such as denitrification are possible. However, organic carbon is necessary to drive the microbial mediated denitrification process. Redox potential discontinuity near the surface is characteristic of flooded sediments (Ponnamperuna, 1972) and has been shown to be a limiting factor in plant colonization/ growth (Howes *et al*., 1981; Mendelssohn, 1981). Tidal flooding most likely influences many other important soil conditions (Ponnamperuna, 1972; Bertness and Ellison, 1987).

An aim of the Environment Agency in the creation of Paull Holme Strays is that the 90 hectares will create 45 ha of mudflat and 45 ha of saltmarsh. The success of the managed realignment site in developing such habitats will depend, amongst other things, on the availability of sediments, there ablity to withstand the erosive actions of the waves and the tidal currents, while allowing sediment accretion to occur at a rate at least equal to local rates of relative sea- level rise (Watts *et al*., 2003). Even without the subsidence experienced at many sites (El- Sayed, 1996), this calls for 6 mm year $^{-1}$ of accretion to keep pace with current sea level rise. This rate of accretion has been far surpassed at the restoration site were the average rate of accretion was . 27 mm year^{-1}.

Compensation wetlands are created to replace natural wetlands that are lost as a result of human activities. The debate continues on whether created wetlands are similar to natural wetlands in their characteristics, environment and function. This study of a constructed wetland indicated that initial sediment properties such as percent sand, clay, and carbon are important in the function and health of a constructed wetland and are within the range found in natural wetlands. The range in partial size distribution in wetlands is dependent on sediment deposition and the variety of depositional energies in an alluvial wetland (Solt *et al*., 2000). When the mean hydro-period is high, levels of organic matter are much higher (Aust, 1991). Low levels of organic matter may limit processes such as denitrification (Solt *et al*., 2000). Some plants may be growth limited in constructed sites because of this. In the case of Paull Holme Strays organic matter made up an average of 10% of the sediment.

In tidal wetlands, the site topography creates the particular hydrologic regime that determines vegetation type and establishment. Tidal wetlands morphologically evolve towards a dynamic equilibrium state in response to the particular combination of hydrologic and sediment regimes (Myrick and Leopold, 1963; Pestrong, 1965). Depositional processes tend to create the marsh plain at an elevation of approximately MHW and a tidal drainage network with channel density, order, and cross- sectional characteristics dependent primarily on the area of the wetland and the local tidal characteristics (Haltiner and Williams, 1987; Coats *et al*., 1995). When either the local or regional external physical processes (such as sediment supply) or the internal conditions (such as marsh plain elevation or channel characteristics) are altered, the wetland will respond by evolving a new equilibrium, via the processes of accretion or erosion.

Animal burrowing and building, channelling during flooding, and sediment deposition and erosion aid in varying the topography of a wetland, this micro relief helps establish biodiversity in wetlands by providing areas of high and low habitat for various plants and animals (Golet *et al*., 1993). Sediment properties such as nutrient levels and pH may be affected by subtle changes in relief (Paratley and Fahey, 1986; Beatty, 1984).

Little direct consideration has been given to ground water conditions in intertidal work (Haltiner *et al*., 1997). While it’s widely recognized that the shallow low groundwater zone creates the actual hydrologic regime experienced by the plant roots, it has been accepted that providing adequate tidal circulation would create the proper ground water and sediment moisture conditions for wetland plants. However, water levels that are too high will result in habitat that will not support target vegetation or animals as the site tends to be under water more than predicted. If this is the case, it is difficult to remedy once wetland vegetation is established without disturbing the wetland again. If on the other hand water levels are to lower than expected and the major functions of the habitat are likely to be wildlife habitat, flood water storage, and sediment trapping, studies by (Solt *ed al*., 2000) have shown that there may not be major differences between constructed and reference sites.

Results of this study look at physical parameters of a constructed wetland and, although the conditions examined were not a direct measurement of wetland function, each of the parameters measured served as a proxy for examining functionality.

Chapter 3

Saltmarsh assemblages of the restored tidal wetlands at Paull Holme Strays

3.1. Introduction

Saltmarshes are plastic coastal features, shaped by the interaction of water, sediments and vegetation. For stability they require protection from high-energy waves, therefore they are usually restricted to comparatively sheltered locations in five main physiographic situations: in estuaries, in saline lagoons, behind barrier islands, at the heads of sea lochs, and on beach plains. (Wiegert *et al.*, 1981).

The patterns by which plant communities become established on saltmarshes are directly related to the abiotic factors of the habitat, particularly elevation in relation to tidal level, which is generally regarded as the most important environmental factor affecting the establishment of saltmarsh species (Snow and Vince 1984), with soil salinity (Snow and Vince, 1984), waterlogging (Cooper *et al.,*2001), wave action (Wiehe, 1935) or soil aeration (Armstrong *et al.,* 1985) as the main determinants of the seaward limit of species and interspecific competition for light (Ungar, 1998) and/ or nutrients (Levine *et al.,* 1998) as the main determinants for the landward limit of species (Fig. 3.1).

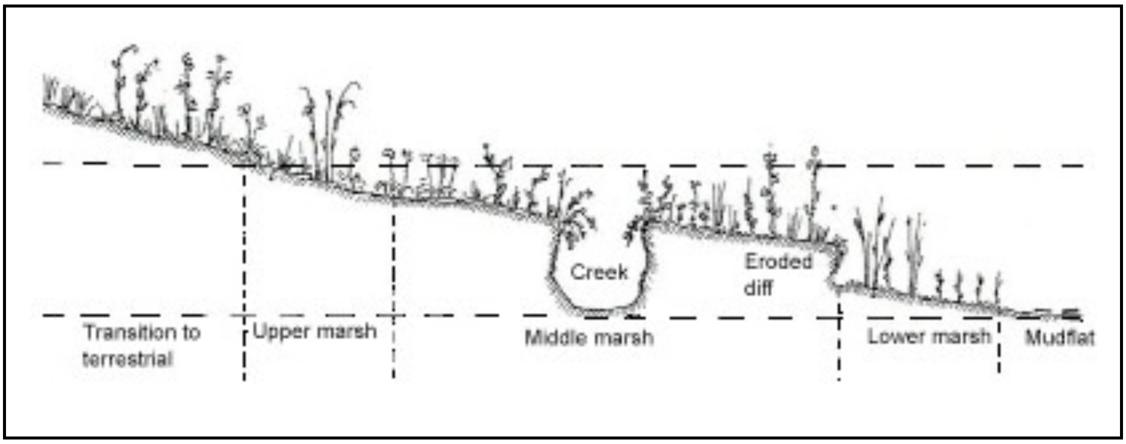


Fig. 3.1: Tidal zonation of saltmarsh vegetation. As sea level rises, the plant communities migrate inland to higher elevations. MHW=mean high water and MLW=mean low water.

There is a trend of, saltmarsh restoration projects to focus on the restoration of the abiotic environment (Wolters *et al.,* 2005) but recent studies have shown that availability and dispersal of the target species may be a bottleneck for successful restoration (Bischoff, 2002; Bissels *et al.,* 2004; Wolters *et al.,* 2005). The addition of seeds or plant material containing the target species greatly enhanced the establishment of those species in inland grassland communities (Pywell *et al.,* 2002), floodplain grass-lands (Hölzel and Otte, 2003) and salt marshes (Rand, 2002). Many restoration schemes, however, minimize human intervention and the target species have to come in spontaneously.

There are several sources from which the target species can colonize a restoration site. The fastest establishment is expected when the species are still present in the community species pool defined as the established vegetation and belowground seed bank of the target area (Zobel *et al.,* 1998). In the case of managed-realignment sites, where seawalls are breached in order to restore tidal inundation to previously reclaimed land, the presence of salt-marsh species in the community species pool is highly unlikely as most sites have been embanked and exploited for decades to centuries, resulting in the disappearance of halophytic species from the established vegetation. Presence in the soil seed bank is also unlikely, as the majority of salt-marsh species do not build up a long-term persistent seed bank (Wolters and Bakker, 2002). Hence, the target species have to disperse into the restoration site from a local species pool (i.e. adjacent salt marsh) or regional species pool (i.e. all salt marshes within a biogeographically uniform region) (Zobel *et al.,* 1998). A review of salt-marsh restoration at different de-embankment sites in northwest Europe showed that between 48 and 100% of the species present in the local species pool established in the restoration sites within 1 to 13 years after de-embankment. When compared to the regional species pool, only 26 to 64 % of the species established in the restoration sites (Wolters *et al.,* 2005).

Armstrong *et al.* (1985) identified target species for the reestablishment of saltmarsh on the East coast of England at Welwick. They have stated that Cord grass has dominates the low marsh with low frequencies of *Salicornia europaea* and *Suaeda maritima* and *Puccinellia maritima* upon elevated areas. This association approximately corresponds to the NVC association SM6. Above the *Spartina* sp. sward, the prevalence of *Puccinellia maritima* increases to establish a transitional low-middle marsh SM10 association. As elevation increases further *Aster tripolium* co-exists with *Puccinellia maritima*, establishing a low-middle marsh community that corresponds most closely to a species-poor variant of the *Puccinellia maritima* dominate subcommunity SM13. Rodwell (2000) primarily identifies this association as a lower marsh community, in accordance with its typical presence in West coast zonations, but recognizes the divergence in East coast communities, where reduced levels of exposure can generally facilitate distribution of floral associations to higher elevations than are typical of the West coast. Armstrong *et al.* (1985) identify a middle marsh and an upper marsh association at the higher elevations at Welwick. One of these associations is similar to the diverse NVC middle marsh community SM17, with significant presence of *Limonium vulgare*, *Armeria maritima*, *Triglochin maritima*, *Plantago maritima* and *Festuca rubra*. The middle marsh community at Welwick presents two issues. Armstrong *et al.* (1985) do not make reference to a significant presence of *Artemisia maritima*, identified by Rodwell (2000) as the dominant characteristic species of the SM17 community. This suggests that some qualification of the identification may be necessary. Armstrong *et al.* (1985) relate the community to the GSM umbrella of Chapman (1974), but the overly generic character of the description tends to accommodate associations of greater divergence than that for which the NVC categories typically aim to provide. This suggests that the GSM could represent a composite of more than one NVC community. It can therefore also be concluded that the middle marsh at Welwick represents a subcommunity of SM17, or a composite of SM17 and SM22, that requires further investigation. The upper marsh association evident at Welwick is the couch grass association SM24, dominated by *Agropyron pungens*, often to the extent that a monoculture of stiff clumps is formed

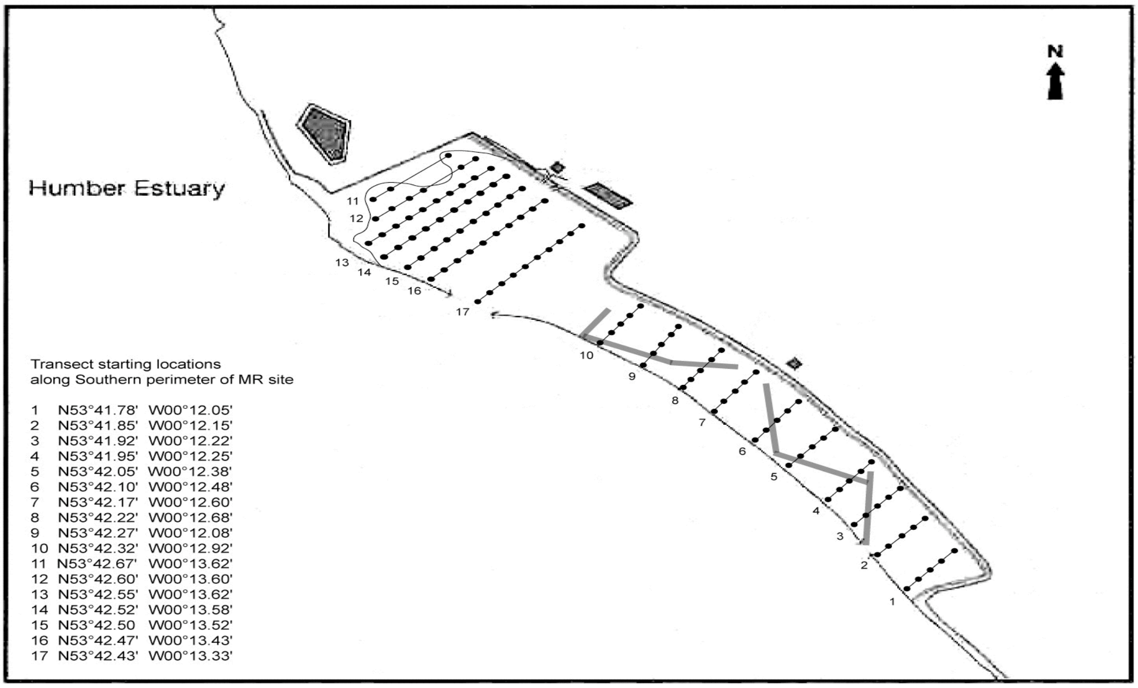
(Rodwell, 2000). This association often represents the termination of saltmarsh zonation.

The aim of this chapter is to evaluate the process of salt-marsh restoration by (i) identifying and mapping the number of species colonizing the site within the first three years of breaching, (ii) analyzing spatial and temporal patterns in species abundance and (iii) determining whether an actual salt-marsh community has developed immediately following breaching. In addition, we investigate how the preparation of a site prior to breach effects the establishment of species target species from the local and regional species pools.

3.2. Materials and methods

3.2.1. Floral surveys

Systematic sampling using belt transects (Offwell Wetland Survey, 2004) were used for base habitat mapping as the considerable heterogeneity of the distribution of vegetation across the site indicated that random sampling would not provide data representative of the site, as the site is composed of both saltmarsh and mud flat habitat. Seventeen transects were laid out to assess distribution of habitat types across the site (Fig. 3.2) at in late July of 2004 and 2005. The transects were placed 200 metres apart in the east of the site, and 100 meters apart in the west where the spatial heterogeneity of habitats was judged to increase. Transects through the central area of the site were established indirectly by estimation from on top of the sea wall (the permanent water body dominating this area precluded the use of formal transect methodology). Commencing from the bottom of the sea wall at the south of the site, 1m^2 quadrats were sampled at 60-meter intervals along a south west-northeast axis for each transect. The percent cover of vegetation (each plant species), dead terrestrial plants, exposed sediment and open water was recorded within each quadrate, using visual estimates. Visual estimates were carried out by myself and a handful of volunteers, prior to a survey all volunteers were trained in how to make visual estimates. Vegetation identifications were made with reference to Clapham *et al.* (1987), and through the illustrations and descriptions of Burd (1994), Hubbard (1984), Blamey and Wilson (1989), and Rose (1981).



Transect starting locations
along Southern perimeter of MR site

1	N53°41.78'	W00°12.05'
2	N53°41.85'	W00°12.15'
3	N53°41.92'	W00°12.22'
4	N53°41.95'	W00°12.25'
5	N53°42.05'	W00°12.38'
6	N53°42.10'	W00°12.48'
7	N53°42.17'	W00°12.60'
8	N53°42.22'	W00°12.68'
9	N53°42.27'	W00°12.08'
10	N53°42.32'	W00°12.92'
11	N53°42.67'	W00°13.62'
12	N53°42.60'	W00°13.60'
13	N53°42.55'	W00°13.62'
14	N53°42.52'	W00°13.58'
15	N53°42.50	W00°13.52'
16	N53°42.47'	W00°13.43'
17	N53°42.43'	W00°13.33'

Fig. 3.2: Transects used for the mapping of vegetation at Paull Holme Strays in July of 2004 and 2005.

3.2.2. Seed dispersal

Seed traps were made from 35 x 45 cm polyethylene Astroturf mats. Each mat contained ≈ 1300 tufts with a diameter of 0.5 cm at the base and consisting of eight 1.5 cm high shoots. The mats were fastened to the salt-marsh surface at the four corners by 15 cm long galvanized steel spikes. A plastic bag was placed under the Astroturf mats to avoid seeds from the soils surface adhering to the bottom of the mats (Wolters *et al.*, 2004). All vegetation taller than 20 cm that was within 2 m of the mats was cut back to the soil surface. Trimming was done in an effort to eliminate potential seed collection bias by plants in close proximity to the collection mats.

Three replicate Astroturf mats were placed at each of the 2 sampling stations. Station one within the realignment site at Paull Holme Strays, is located in the centre of the realignment at an elevation of 3.1 m (Fig. 3.3). The established vegetation at this site consisted of *Salicornia*, *Aster* and *Elytrigia*, and represents the average development of saltmarsh within the realignment site. Station two was within the natural saltmarsh habitat at Cherry Cob Sands, (which is the marsh adjacent to Paull Holme Strays). Mats at Cherry Cobb Sands were placed in the middle marsh, as this habitat is representative of the habitat aimed to be achieved at the realignment site.

The Astroturf mats were collected and replace every month starting in September 2005, with the final collection in March 2006. Upon collection each mat was placed in an individual bag, and transported back to the lab to be rinsed. Rinsing was done by standing the mats upright in a 0.5mm mesh sieve and directing a jet of water from the top of the mat to the bottom. Depending on the amount of accumulated sediment, it took between 5 and 15 minutes to completely rinse a mat. When necessary, the material collected on the sieve was washed gently to remove any surplus of fine sediment, which would pass through the 0.5 mm mesh sieve. The remaining material was then spread in a thin layer (< 5mm) on a tray filled with sterile potting soil. The samples were then stored in a dark cold room at 4°C for 1 month to allow for cold stratification. After this period the samples were transferred to a glasshouse (25° C day temperature, 15° C night temperature, 15 hours light, automatic watering for 1 min twice a day). For every 10 trays, a control tray without sample was placed to check for seeds blown into the greenhouse. Emerging seedlings and tillering stolons were identified, counted and removed as soon as possible, or in cases where there was difficulty with identification they were transferred to flowerpots in case flowering was needed for identification.

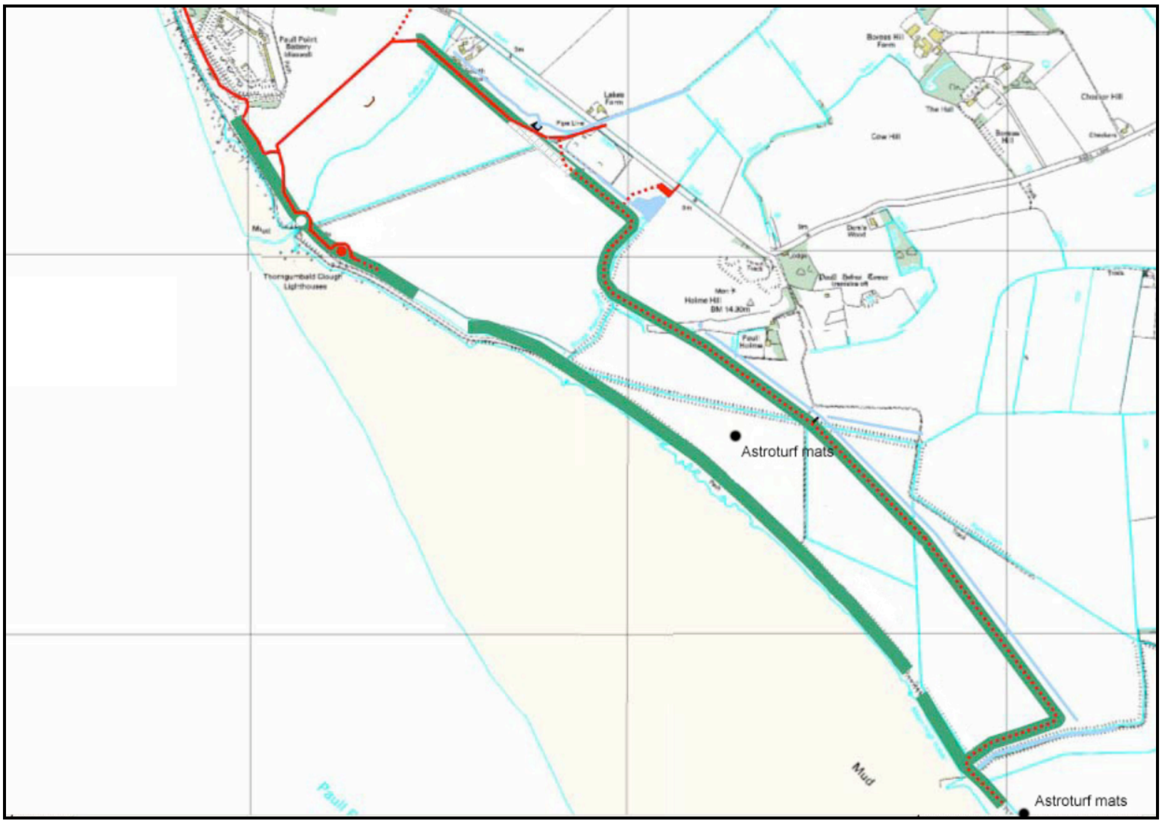


Fig. 3.3: Location of Astroturf mats used to look at seed dispersal at Paull Holme Strays and the adjacent marsh Cherry Cobb Sands. Between September 2005 and March 2006. ** Map modified from IECS map of PHS

3.2.3. Disturbance effects on marsh vegetation development

A 9 m^2 was measured and marked out using bamboo canes at three different locations within the restoration site at Paull Holme Strays (Fig 3.4). These locations were chosen as they represent areas of low middle and high marsh. A 1 by 1 meter grid was then laid and marked in each of the 9 m^2 squares. Marsh disturbance was examined with the use of two different experimental conditions and one control treatment. Experimental conditions were 1) cut all of the existing vegetation to the soil surface, 2) cut all of the existing vegetation to the soil surface and remove from the treatment area, then using a spade till the soil to a depth of 1 foot. Each treatment was carried out in triplicate at three different locations within the restoration site (Table 3.1).

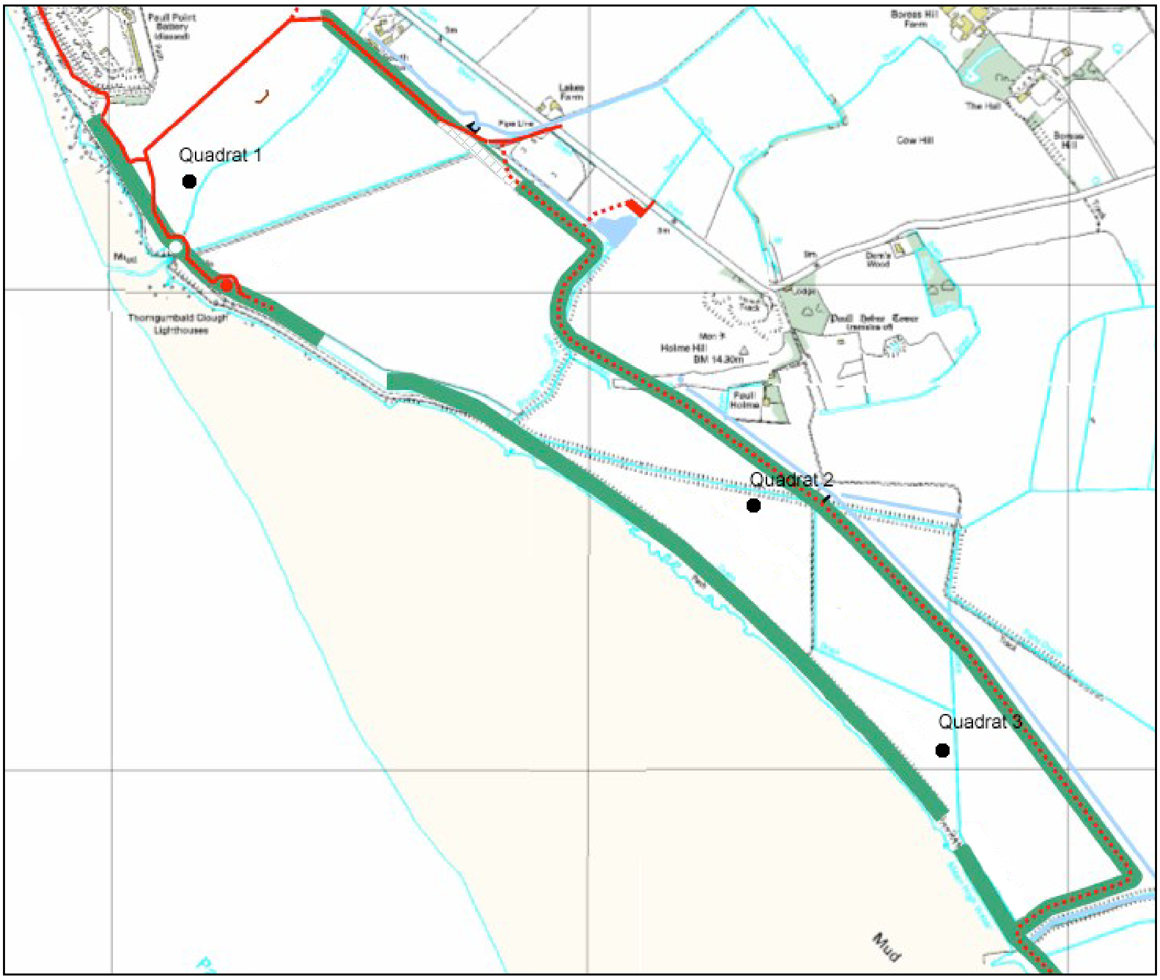


Fig. 3.4: Location of Quadrates used to look at soil treatments effects on salt marsh development in the restoration site between March and August 2005. ** Map modified from IECS map of PHS

Table 3. 1: Location of treatments within each quadrat: A= Control, B= Existing vegetation mowed down, and C= Soil tilled.

	Quadrat 1	
C	A	B
B	C	A
A	B	C

	Quadrat 2	
A	B	C
C	B	A
A	C	B

	Quadrat 3	
C	B	A
B	A	C
C	B	A

Experimental conditions were prepared in the winter of 2004/ 2005 with monitoring commencing in March of 2005 through monthly surveys until the end of the growing season (for peak biomass) in August 2005. During each sampling occasion the number of species present and percent cover of each species was recorded.

3.2.4. Data Analysis

Quadrat composition data were collated and analyzed using MAVIS (CEH, 2000), to compare relative abundance of species in quadrate samples against NVC reference associations (Rodwell, 2000). MAVIS identifies potential matches and allocates percentage scores on the basis of the closeness of each match but it remains necessary for the operator to assess the output, and gauge the most appropriate match through reference to the classification system in question. Differences in saltmarsh abundance between years were examined using a Wilcoxon Signed Ranks Test. A one-way ANOVA was used to test for differences in the number of diaspores between restoration and natural marshes. A Sorensen similarity index was used to look at relationships between diaspores and established vegetation. A Kruskal Wallis test was used too examine relationships between soil disturbance and saltmarsh colonization.

3.3. Results

3.3.1. Floral survey

11 halophyte species were observed within the restoration site. The sampled species comprised *Salicornia europaea, Aster tripolium, Spergularia marina, Puccinellia distans, Puccinellia maritima, Atriplex prostrata, Phragmites australis, Agropyron pungens, Spartina anglica, Chamaemelum nobile* and *Arrhenatherum elatius* (Table 3.2). No significant difference was found in the number of quadrats that contained halophytes between 2004 and 2005 ($Z= -1.34$; $p> 0.05$). There was also no significant difference found in the percent cover by halophytes within restoration site between 2004 and 2005 ($Z = 1.34$; $p> 0.05$). In 2004 halophytes covered 16.38 percent of the quadrats measured and in 2005 they covered 19.20 percent of the quadrats measured..

Table 3.2: Abundance of halophyte species found at the restoration site in 2004 and 2005. The table shows the number of quadrats that each species was found in and mean % cover by those species.

	2004		2005	
Species	Number of quadrats	mean% cover	Number of quadrats	mean% cover
Spartina anglica	0	0	3	0.08
Phragmites australis	2	0.01	1	0.01
Puccinellia spp.	7	0.12	2	0.42
Agropyron pungens	16	12.37	6	0.58
Aster tripolium	6	0.04	8	0.69
Atriplex prostrata	19	3.69	19	5.19
Salicornia europaea	1	0.01	3	0.04
Spergularia marina	3	0.11	9	1.83
Chamaemelum nobile	1	0.03	0	0
Arrhenatherum elatius	0	0	11	10.36

For data collected in 2004, MAVIS identified 5 associations that could be directly related to existing NVC associations: natural grassland MG1; upper-marsh Sm12b, SM24, SM23; and brackish- water/ fresh water swamp associations S18b (Fig. 3.5). In 2005 MAVIS identified 6 associations that could be directly related to existing NVC associations: natural grass land association MG1; the lower marsh associations SM6; the middle marsh association SM23; and the upper marsh or swamp associations SM24, SM12b, S4a and S21b; and brackish- water/ fresh water swamp associations S18b (Fig. 3.5).

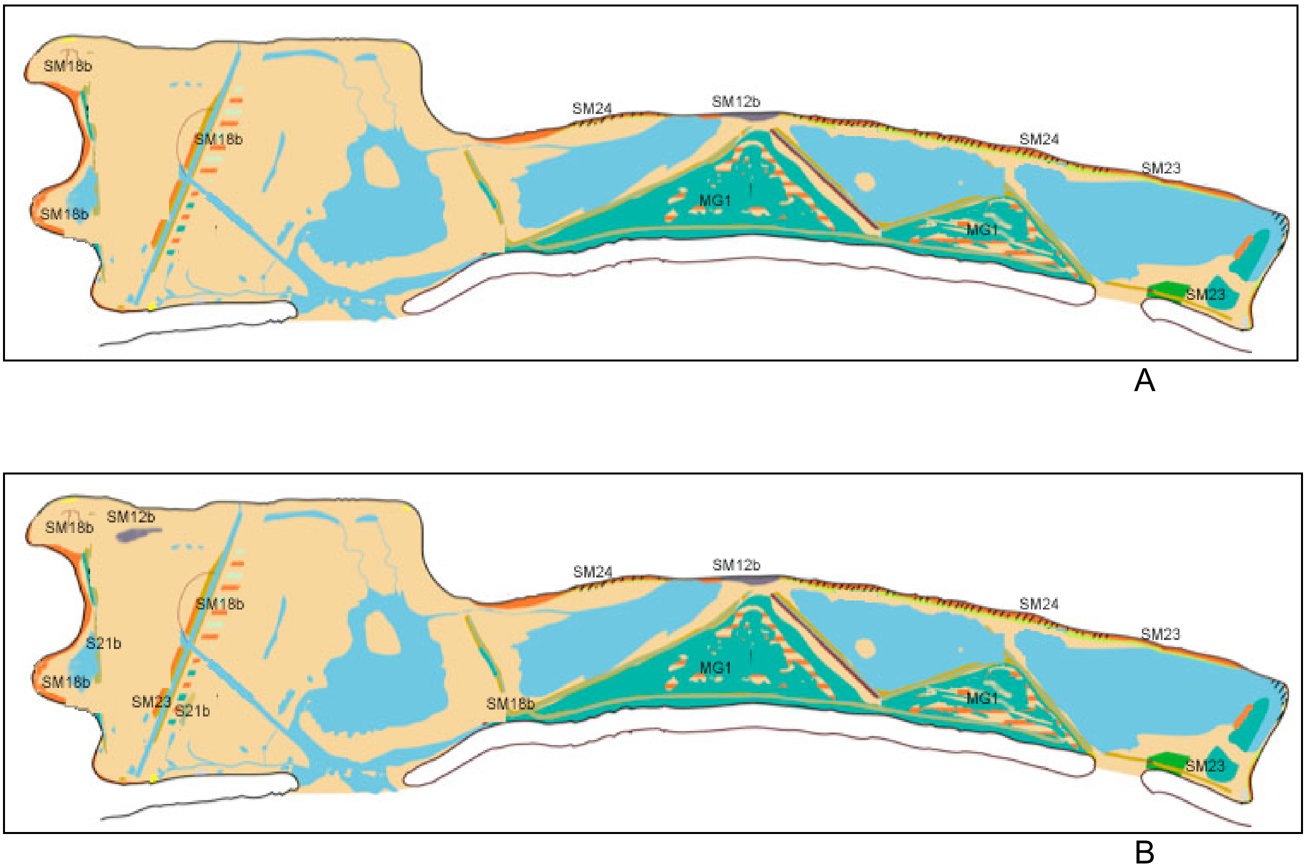


Fig.3.5: Habitat maps derived from ground surveys, including floral associations identified by MAVIS (CEH, 2000). Map A shows floral development up to 2004 and map B shows floral development up to 2005.

3.3.2. Seed dispersal

From September 2005 to March 2006 a total of 5 species (*Atriplex prostrata, Aster tripolium, Spergularia* sp., *Sueda maritima* and *Artemisia maritima*) were trapped on the Astroturf mats, of which all were saltmarsh species. Many of these species were present in very low numbers at any location at any one time (Table 3.3).

Table 3.3: Species dominance of seeds trapped by Astroturf mats for individual sampling stations, stations are separated by location and date.

Oct-05

PHS

	A	% Dom	Cum%
Atriplex prostrata	21	13	13
Aster tripolium	6	4	17
Spergularia sp.	129	83	100
Sueda maritima	0	0	100
Artemisia maritima	0	0	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	0	0	0
Aster tripolium	0	0	0
Spergularia sp.	0	0	0
Sueda maritima	0	0	0
Artemisia maritima	0	0	0

Nov-05

PHS

	A	% Dom	Cum%
Atriplex prostrata	27	30	30
Aster tripolium	30	34	64
Spergularia sp.	32	36	100
Sueda maritima	0	0	100
Artemisia maritima	0	0	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	2	33	33
Aster tripolium	0	0	33
Spergularia sp.	4	67	100
Sueda maritima	0	0	100
Artemisia maritima	0	0	100

Dec-05

PHS

	A	% Dom	Cum%
Atriplex prostrata	34	76	76
Aster tripolium	3	7	82
Spergularia sp.	4	9	91
Sueda maritima	0	0	91
Artemisia maritima	4	9	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	78	86	86
Aster tripolium	11	12	98
Spergularia sp.	0	0	98
Sueda maritima	0	0	98
Artemisia maritima	2	2	100

Table 3.3: Continued

Jan-06

PHS

	A	% Dom	Cum%
Atriplex prostrata	80	81	81
Aster tripolium	4	4	85
Spergularia sp.	13	13	98
Sueda maritima	0	0	98
Artemisia maritima	2	2	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	70	75	75
Aster tripolium	17	18	94
Spergularia sp.	0	0	94
Sueda maritima	0	0	94
Artemisia maritima	6	6	100

Feb-06

PHS

	A	% Dom	Cum%
Atriplex prostrata	38	69	69
Aster tripolium	13	24	93
Spergularia sp.	0	0	93
Sueda maritima	2	4	96
Artemisia maritima	2	4	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	2	20	20
Aster tripolium	0	0	20
Spergularia sp.	2	20	40
Sueda maritima	2	20	60
Artemisia maritima	4	40	100

Mar-06

PHS

	A	% Dom	Cum%
Atriplex prostrata	6	60	60
Aster tripodlum	0	0	60
Spergularia sp.	4	40	100
Sueda maritima	0	0	100
Artemisia maritima	0	0	100

Cherry

	A	% Dom	Cum%
Atriplex prostrata	6	75	75
Aster tripolium	2	25	100
Spergularia sp.	0	0	100
Sueda maritima	0	0	100
Artemisia maritima	0	0	100

Temporal differences in the number of trapped diaspores of all saltmarsh species indicated that the main dispersal period took place between November and January in both the restoration and the adjacent marsh (Fig. 3.6). Some differences between species could be found with *Aster tripolium* and *Spergularia* sp. starting to be dispersed earlier than *Atriplex prostrata* (Fig. 3.7).

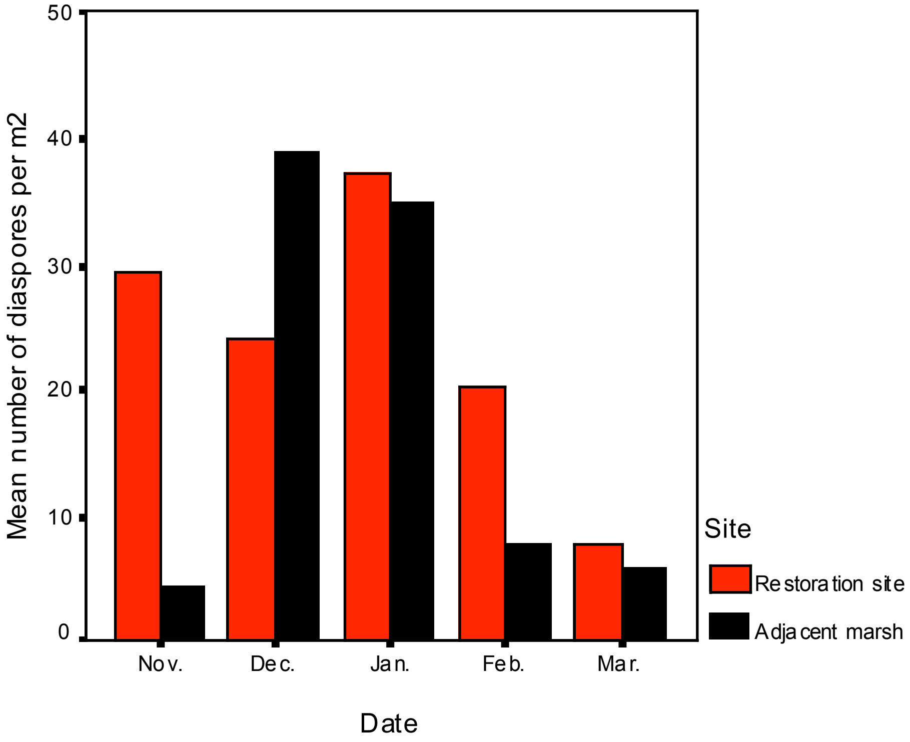


Fig. 3.6: Temporal pattern of the distribution of diaspores from halophytes captured on Astroturf mats per m^2 for the restoration site (Paull Holme Strays) and adjacent marsh (Cherry Cobb).

.

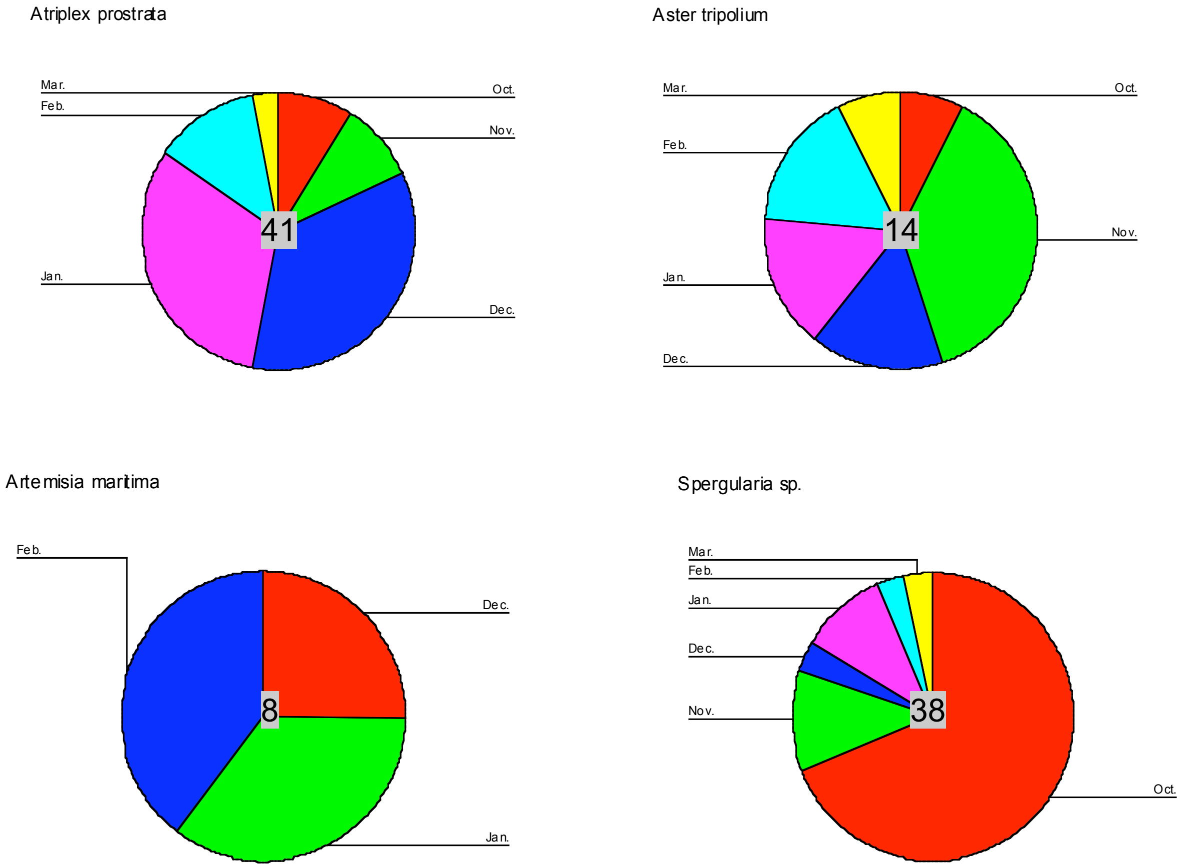


Fig. 3.7: Temporal pattern in number of diaspores trapped in all locations combined. The number of diaspores trapped per m^2 in each period is expressed as a percentage of the sum of the means for the eight locations (number in the circle) over the entire sampling period for each species. Different periods are indicated by the first three letters of the month in which the seed traps were collected, starting from October (Oct.) to March (Mar.). Note that for the period from September to October only PHS was sampled.

A one-way ANOVA was used to test for differences in the number of diaspores between the restoration site and the adjacent marsh for each of the 5 salt-marsh species (Fig. 3.8). Diaspores of *Atriplex prosrtata, Aster tripolium,* *Spergularia* sp. and *Artemisia maritima* were all trapped in the same abundances regardless of site (Table 3.4). *Sueda maritima* was not found in enough abundance to analyse statistically but it was found at both the restoration site and the adjacent marsh.

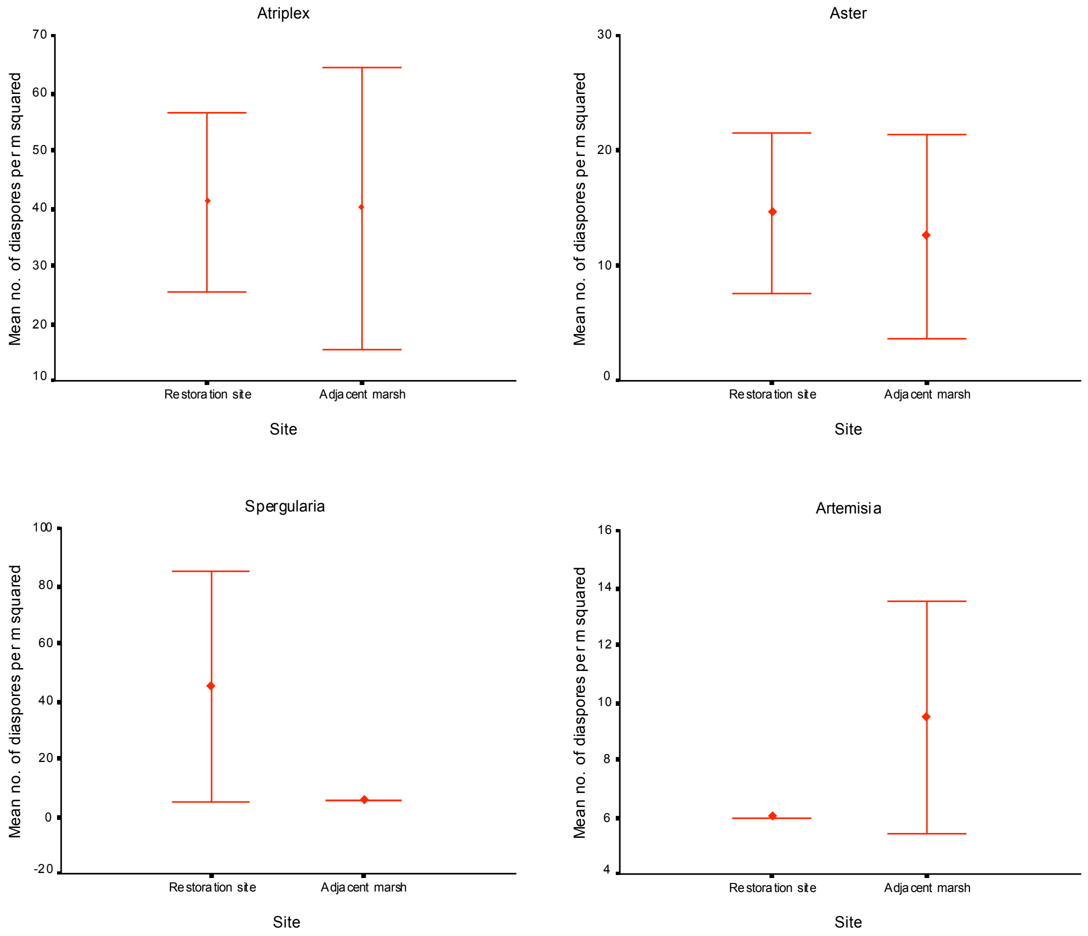


Fig. 3.8: Cumulative mean number of diaspores per m^2 (± SD) by species, trapped between October 2005 and March 2006. Note differences in *y*- axis scale for different species.

Table 3.4: ANOVA results for species of diaspores trapped on astro turf mats in restoration and adjacent natural marsh sites.

	f	df
Atriplex prostrata	0.008*	1,25
Aster tripolium	0.137*	1,17
Spergularia sp.	0.916*	1,13
Artemisia maritima	3.00*	1,6

* $p < 0.01$

Of the 11 salt-marsh species recorded in the established vegetation in the restoration site, 5 were trapped in the Astroturf mats (Fig. 3.9), resulting in a similarity index of 0.38 (Table 3.5). A higher similarity in species composition (0.45) was found between the diaspores and vegetation of the adjacent marsh surface. The same similarity (0.45) was calculated for the diaspores in the restoration site compared with the vegetation of the adjacent marsh (Table 3.5). Interestingly, diaspores of *Suaeda maritima* and *Artemisia maritima* were trapped in both the restoration and adjacent marsh but neither was recorded in the established vegetation of the restoration or adjacent marsh.

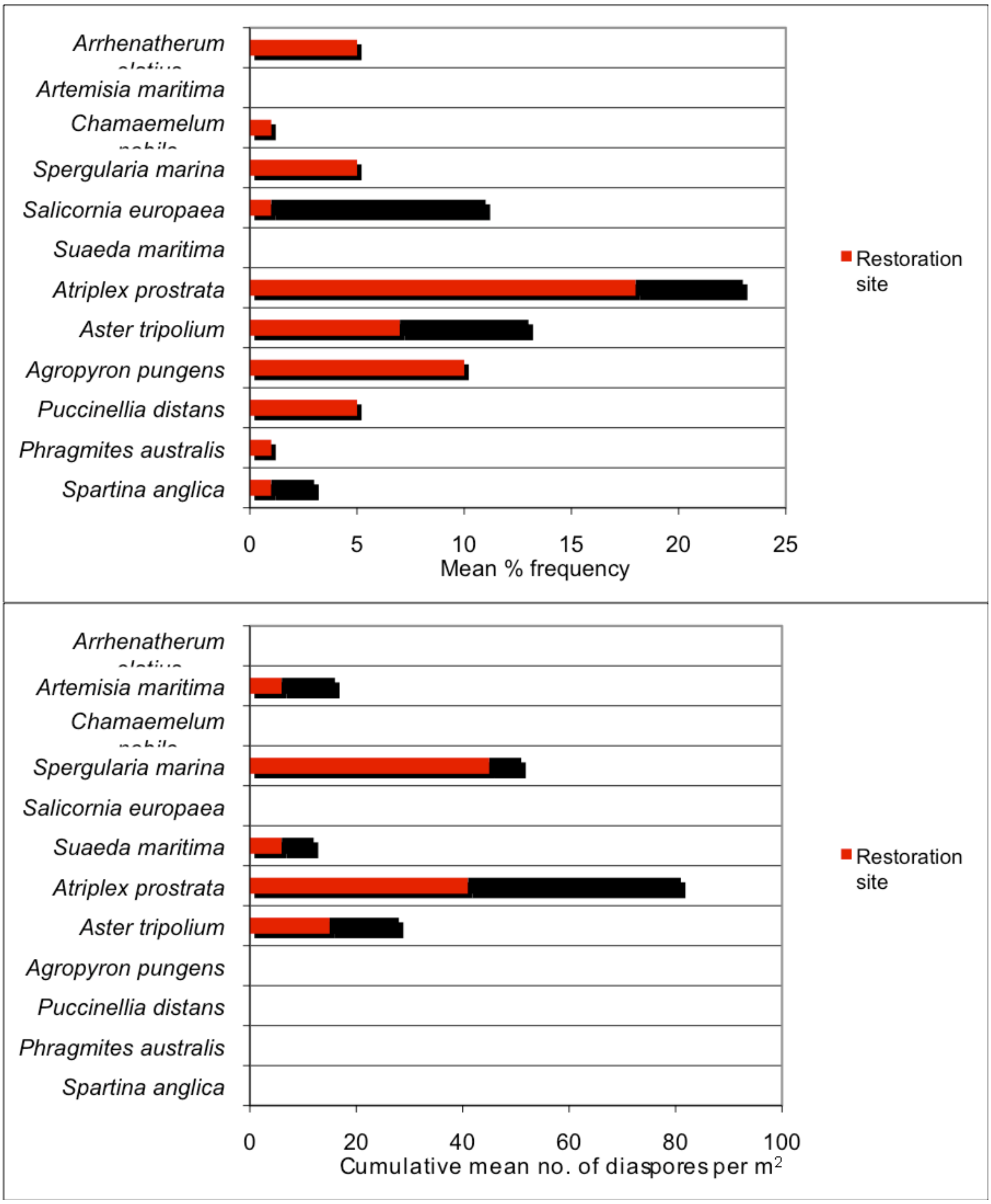


Fig. 3.9: Comparison between vegetation and species trapped on astro turf for the restoration site and adjacent marsh. Data for % frequency of halophytes in the adjacent marsh was provided by IECS.

Table 3.5: Sorensen similarity indices comparing established vegetation and diaspores trapped on Astroturf® in the restoration site (Paull Holme Strays) and the adjacent marsh surface (Cherry Cobb).

	Astroturf	
Vegetation	PHS	Cherry Cobb
PHS	0.38	0.38
Cherry Cobb	0.45	0.45

3.3.3. Disturbance effects on marsh vegetation development

From March 2005 to August 2005 a total of 5 species were found growing in three disturbance test areas. They were *Atriplex prostrata*, *Spergularia marina*, *Puccinellia* sp., *Phragmites australis* and *Spartina anglica*. *Spartina anglica* was only found growing at site 3 in treatment B for two months with a percent cover of 1 %.

There were no differences in number of species found based upon treatment (χ^2 = 4.23; d.f.= 2; p>0.05). An average of 2 species was found at each of the treatment types (Fig. 3.10).

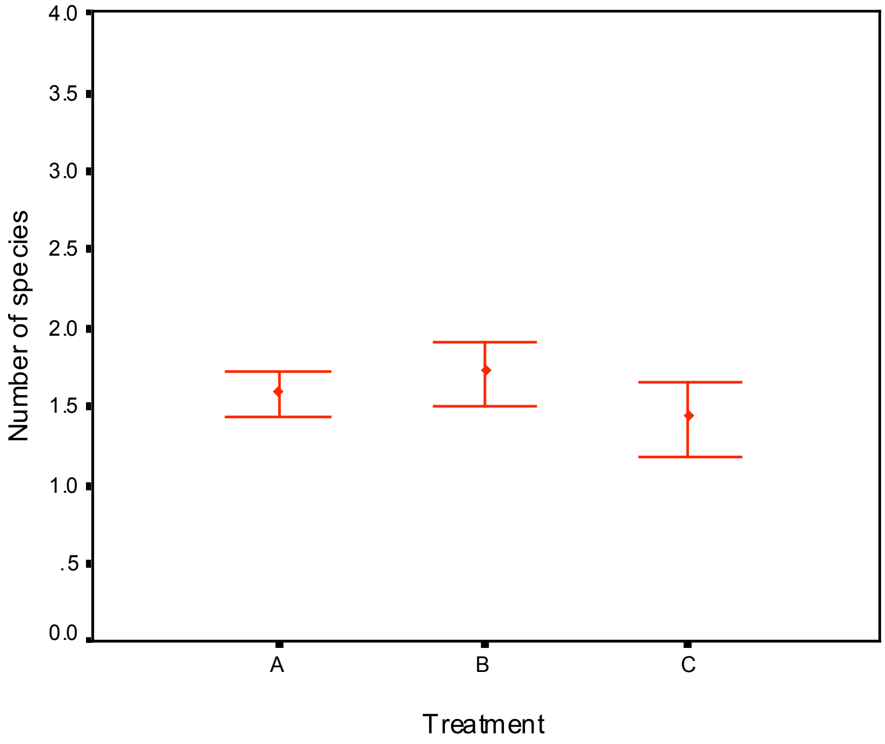


Fig. 3.10: Mean number of species found based upon treatment type (± SD). Treatment A= Control, B= Existing vegetation mowed down, and C= Soil tilled.

A Kruskal Wallis test showed a significant difference in the total percent cover based upon treatment type ($\chi^2 = 17.49$; d.f. = 2; $p<0.01$). A Games-Howell post hoc test then showed that there is no differences in the percent cover between treatments A and B, which both had a significantly higher percent plant cover than treatment C (Fig. 3.11). Both *Atriplex prostrata* and *Spergularia marina* showed significant differences in percent cover based upon treatment type ($\chi^2 = 8.88$; d.f. = 2; $p<0.01$), ($\chi^2 = 14.39$; d.f. = 2; $p<0.01$). A Games- Howell post hoc test showed that in each case there was no difference in the percent cover between treatments A and B, which both had significantly higher percent plant cover than that found in treatment C.

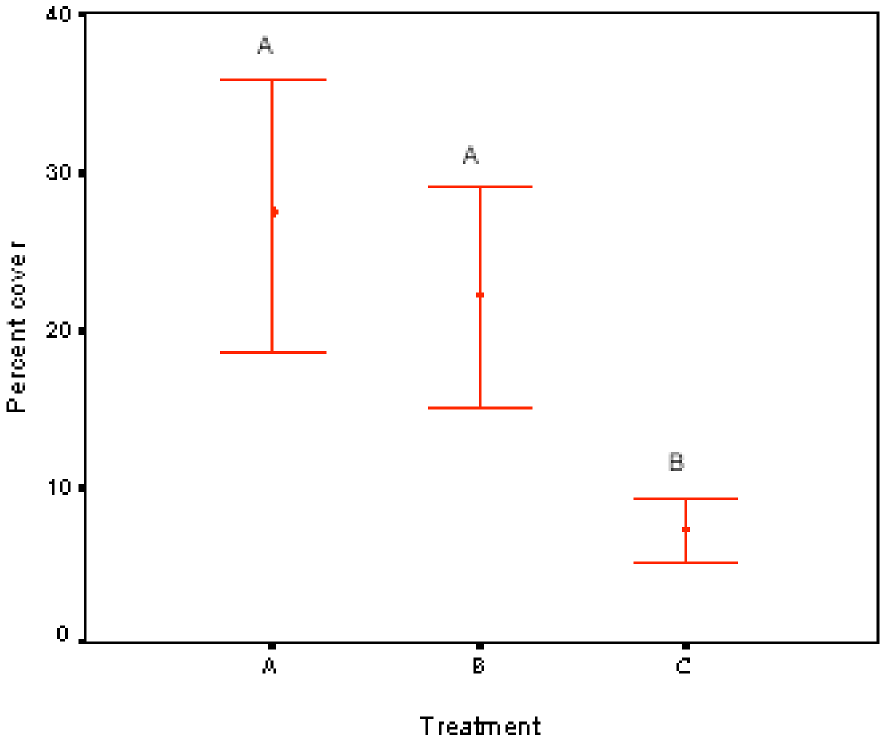


Fig. 3.11: Percent plant cover based upon treatment type (± SD).Treatment A= Control, B= Existing vegetation mowed down, and C= Soil tilled Letters indicate a significant difference in the percent plant cover based upon a Games-Howell test.

3.4. Discussion

Reintroduction of proper hydrology is critical to restoration of wetland plant communities (Lowry, 1990). Fortunately, most marsh plants can tolerate relatively large fluctuations in water level (Kusler and Kentula, 1990) and, for that reason; marshes are among the easiest wetlands to restore (Tiner, 1995). The simple structure and rapid maturation of marsh vegetation communities and the presence of native seed stocks also contribute to rapid and relatively successful restorations (Kusler and Kentula, 1990).

Marshes are often disproportionately represented in restoration and creation efforts (Esselink *ed al.*, 2000). In addition, other wetland types (e.g., forested wetland) are often converted to marsh for purposes of wildlife habitat enhancement (Golet, 1986); this is at the expense of wildlife dependent on the former wetland type.

Former marshes that have been drained and cultivated are relatively easy to restore, particularly if they are small and drained by open ditches (Tiner, 1995). Because natural seed banks may be viable for centuries, planting is not required; restoration of such sites simply involves plugging the ditches. Tile-drained marshes may be more difficult and costly to restore, because you must block or remove subsurface soil drains and in doing so you can damage your natural seed banks making it necessary to obtain seeds from other sources. Marshes that have been drained for agriculture in Rhode Island generally have been drained using open ditches Tiner (1995), but such restoration opportunities are scarce in the UK. Wetland basins with large watersheds are also more difficult to restore because they may require water- and erosion-control measures to prevent washouts (Tiner, 1995). Tiner (1995) also recommended that tilling sites before restoring hydrology to hasten recolonization by hydrophytes in the existing seed bank. However, Brown (1999) found that restored sites with disturbed substrata ended up as monotypic stands of cattail, providing less value for wetland birds. Results of soil disturbance experiments in this study showed that when substrata was disturbed monotypic stands of *Atriplex prostrata* and *Spergularia marina* colonized which supports the results of Brown (1999).

The software/program MAVIS is typically used to look at developed habitats in a state of equilibrium, whereas the associations evident at PHS are typically comprised of developing and / or pioneer vegetation yet to reach a discernible equilibrium status. The habitat map was therefore constructed with reference to NVC associations thought to be the best fit. However, it is recognized that subsequent floral development may produce associations that can be more closely related to established classifications — for example, the aforementioned *A. prostrata* association (SM18b) may represent a precursor to dominance by *E. atherica* SM24, with which *A. prostrata* is often associated, as already evidenced by the upper saltmarsh development in the south east of the site.

Most existing terrestrial vegetation in intertidal areas was killed following the reintroduction of tidal cycles. The declines of mildly salt-tolerant species coincide with the colonization of important salt-marsh species. This could be seen with the disappearance of NVC association MG1 between 2004 and 2005.This was similar to vegetation responses seen in other marshes where tidal flow has been reintroduced (Beeftink, 1979; Frenkel and Morlan, 1991).

Data from 2005 suggests that cover from some of the opportunistic plants (e.g. *Arrhenatherum elatius*) may be declining slowly as *Spergularia marina* increases slowly. It must be recognized that change is likely to occur over time, even in reference sites (Zedler *et al*., 1992, Warren and Niering, 1993). Although saltmarsh vegetation has been re-established, the total ratio of vegetated ground to unvegetated ground was 0.72 two years after breaching with vegetation occupying around 25% of the site, compared to approximately 40% covered by dead vegetation.

Having determined the distribution of salt marsh present, the habitat maps (Fig. 3.5) were compared with the previous prediction of climax equilibria that could be supported on the site 1 year after breaching, as predicted prior to inundation by Halcrow Group Ltd, 2002 (Fig. 3.12). The predictions are based on the original expected rates of accretion and the resulting elevations after 1 year although the dynamic, co-dependent nature of accretion rate, elevation development, and saltmarsh development indicate that any predictions are subject to a notable error as the findings are highly variable. This can be seen by comparing the extent of upper saltmarsh in the original predictions (Fig. 3.12) to that currently evident. Upper saltmarsh currently occupies a much greater area than that represented in the original predictions, where it is restricted to the proximity of the seawall. Conversely, middle marsh presently occupies a very restricted range in the south west of the site, in contrast to the extensive areas indicated in the original predictions (Halcrow Group Ltd, 2002)(Fig. 3.12).

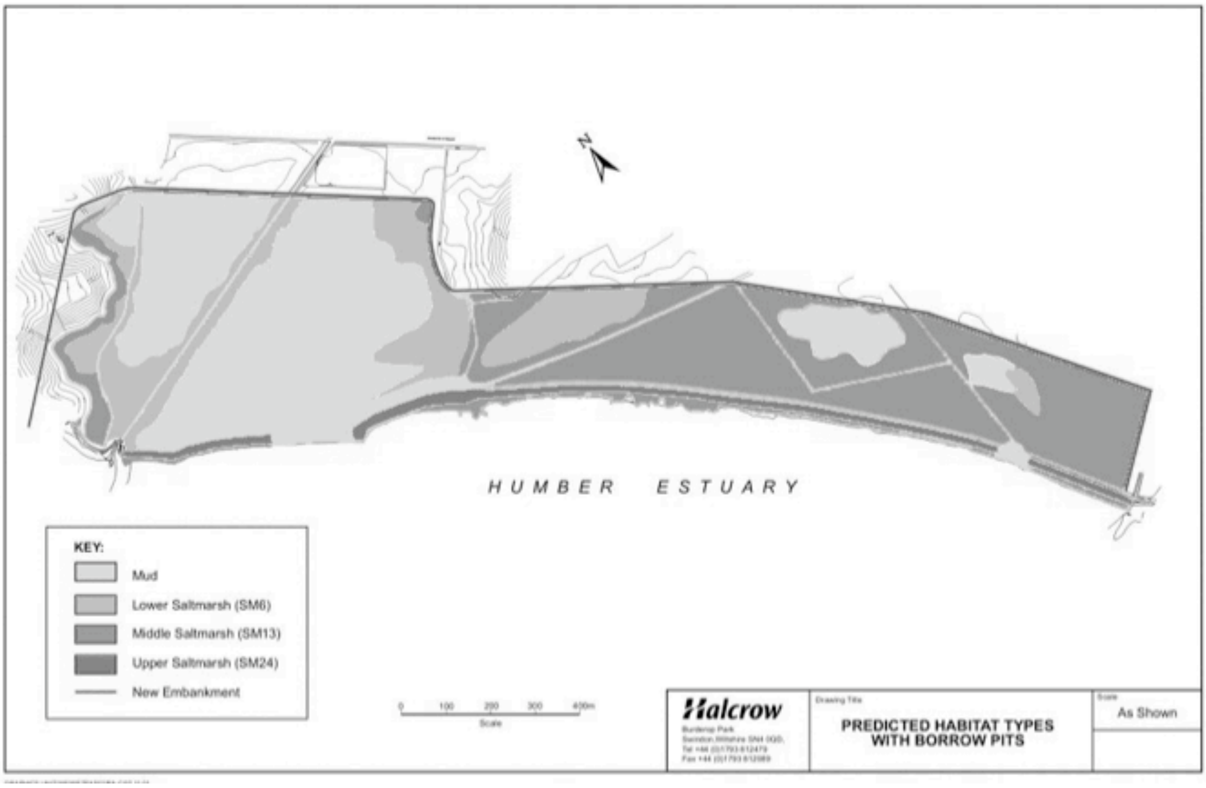


Fig.3.12: Predictions of potential salt marsh development 1 year after breaching.
Note: colour has been amended from Halcorow Group Ltd (2002).

In this study, a total of 5 species of salt-marsh species colonized onto the Astroturf mats, although only four with more than 8 diaspores per m^2. It could be argued that certain species might be more efficiently trapped than others because of morphological differences between their seeds e.g. seeds from *Salicornia europaea* are small and smooth while seeds from *Aster tripolium* are ridged and textured. However, in an experiment specifically conducted to test the seed retaining efficiency of Astroturf mats, no differences were found for three morphologically different species (Wolters *et al*., 2004). It should be noted that the number of diaspores retained by the Astroturf mats was the net result of deposition and removal of diaspores by tidal water over the monthly replacement series. The results therefore represent the minimum total number of diasprores dispersed during the sampling period. Dispersion by wind and birds could have also contributed to the total number of diaspores trapped in this study. Huiskes *et al*. (1995) observed that a considerable number of propagules transported along the bottom of the water column with the flood tide returned to the same station with the ebb tide. However, this was not true of the propagules floating on the surface of the water column, suggesting various dispersal modes between species with dissimilar floating characteristics. Two of the most abundant species trapped on the Astroturf mats were among the same caught in standing nets by Huiskes *et al*. (1995), *Aster* and *Spergularia*. These species are characterized by relatively short seed floatation times of a few hours to a few days (Koutstaal, Markusse and De Munck, 1987).

One of the aims of this study was to examine the temporal dispersal patterns in order to provide information on the appropriate timing of breaches for managed realignment. Results show a peak in diaspore dispersal between November and January, which agrees with results from Hutchings and Russell (1989) and Wolters *et al*. (2005). Therefore, in order to take full advantage of the peak dispersal of salt-marsh species in the first year after breaching, it is recommended that breaching of sea defences occur no later than early September.

For most species the number of diaspores trapped was representative of their abundance in the vegetation. Exceptions were *Spartina anglica, Phragmites australis*, *Puccinellia distans*, *Agropyrum pungence*, and *Salicornia europea* of which less than 1 diaspores per m^2 were trapped during the entire sampling period, in spite of this they occurred with a frequency of 2%, 2%, 5%, 10%, and 2% respectively in the vegetation of the restoration site.

Another interesting observation is of the trapping of *Artemisia martima* and *Suaeda maritima* in both the restoration and adjacent marsh, as neither of these species was present in the established salt marsh of either site. It is possible that they were transported over longer distances by tidal water, or deposited on to the Astroturf mats by birds.

It is concluded that in general the overall results indicate a predominantly local dispersal of salt-marsh species, despite the potential for long distance transport and wide distribution by tidal water.

Chapter 4

Benthic invertebrate community structure in the restored wetlands at Paull Holme Strays

4.1. Introduction

In estuarine wetlands benthic invertebrates are used to assess the development of secondary production and food web support following restoration (Craft *ed al.,* 2003; Talley and Levin, 1999; Zedler, 1996). Many studies have indicated that benthic invertebrate community composition is slow to develop, requiring years to achieve equivalence to natural wetlands (Craft and Sacco, 2003; Minello and Webb, 1997; Scatolini and Zedler, 1996). The distribution of benthic invertebrates is based on a number of factors, including water temperature, water depth and the nature of the seabed (Sanders, 1968). Additionally it is thought that the slow rate of colonization by benthic invertebrates is attributed to the absence of a planktonic dispersal stage (Levin *et al.,* 1996) and the low organic matter content of constructed wetland soils (Craft, 2000; Broome *et al.,* 2001).

Work has been done on the patterns of distribution of fauna in mudflats within saltmarshes at relatively broad scales (Branch and Branch, 1980; Wells, 1986), as well as in mudflats as a single habitat (McLusky and Elliott, 2004; Little, 2000). Although fauna distribution patterns may be related to features of the habitat (Underwood and Barrett, 1990), distribution patterns are often explained by variation in tidal inundation (McMahon and Britton, 1985). Recent studies have emphasized small-scale patchiness in abundances of invertebrates within and among habitats. This has been reported for organisms living on the intertidal (Archambault and Bourget, 1996; Benedetti-Cecchi and Cinelli, 1996; Thompson *et al.*, 1996; Underwood and Chapman, 1996, 1998), subtidal (Chapman *et al.*, 1995; Laponite and Bourget, 1999) and on mudflats (Thrush *et al.*, 1994; Mermillod-Blondin *et al.*, 2003).

Patterns of variation of benthos in soft sediments (such as mudflats) can be related to characteristics of the sediments themselves, e.g. grain- size (Whitlack, 1981), sorting (Gray, 1974) or organic content (Elliott *et al.,* 1998).

There is considerable evidence that, whilst biota may respond to properties of the sediments (Ford *et al.*, 1999), they also cause significant changes in such properties. Biota can change porosity and permeability (Meadows and Tait, 1989), grain size, water content, organic content, and the erosion threshold of the sediments (Austen *et al.*, 1999; de Brouwer *et al.*, 2000; Tolhurst *et al.*, 2003). These changes are vary complex, with a single species often having multiple antagonistic and synergistic effects (Anderson, 2001; de Deckere *et al.*, 2001). These are dependent upon other biological and physical properties or processes, resulting in seemingly idiosyncratic responses (Defew *et al.*, 2002). Despite knowledge of such variation, little attempt has been made to quantify this variability adequately, even in broad terms, so our understanding of these complex interactions remains poor (Black *et al.*, 2002). Even without making any assumptions about causative relationships, the implication is that if they are measured at the relevant scales, bio-dependent properties of the sediment should, in broad terms, correlate with the structure of the macrobenthos (and vice versa) as has been shown for meiofauna (e.g. Decho and Fleeger, 1988; Pinckney and Sandulli, 1990).

In spite of the importance of benthic invertebrates as an indicator of the structural and functional equivalence of a wetland following restoration, little is know about community development (Craft and Sacco, 2003), however Talley and Levin (1999); and Mazik *et al.* (2007) have done some preliminary studies on macrofaunal succession within restored wetlands. Given the importance of benthic invertebrates as prey for a variety of avian and aquatic consumers (e.g., nekton, waders and wildfowl), there is especially a need to assess their value to the local ecosystem. As the newly developed intertidal mudflats are one of the more visible features of the managed realignment site at Paull Holme Strays and the site has just been created there has been little previous assessment of its biological resources. Knowledge of the benthic invertebrates living within the muddy sediments of this system is especially poor.

The aim of this chapter is to describe the initial invertebrate colonization and development of the macrofaunal communities, between 2004 and 2005, within the site and to provide comparison with the communities with the established mudflats adjacent to the restored wetland.

4.2. Materials and methods

4.2.1. Field methods

Sediment cores (10cm diameter, i.e. 78.5 cm^2 surface area, and a depth of 10cm) were taken from 14 randomly selected stations from within the restoration site, and 11 just outside the restoration site on the existing natural mudflat (Fig. 4.1). Three replicate samples were taken from 12 of the 14 stations within the restoration site monthly between August 2004 and September 2005. Three replicate samples were only taken from the other 2 stations within the restoration area (stations 2- A and 2- B) and the all of the stations on the existing natural mudflat annually (October 2004 and September 2005) as access to them is limited to hovercraft (Fig. 4.1). Samples were then placed into containers, which were labelled externally giving the station name, date and subsample number (i.e. 1 of 2, 2 of 2, etc.), if applicable. A label bearing the same information was also placed inside the containers with the infaunal samples. This label was written in pencil on a paper of a quality suitable for wet labels. After the containers were labelled, they were filled with mud cores. Care was taken to make sure that no container was filled over approximately the 40 percent mark with a core. Once the core was placed into the in container, they were sealed and transported back to the laboratory. Once in the lab samples were sieved through a 500 μm sieve. The sieve residue was then preserved in a 4% formaldehyde solution. Samples were left for at least 24 hours to allow for preservation to take place.

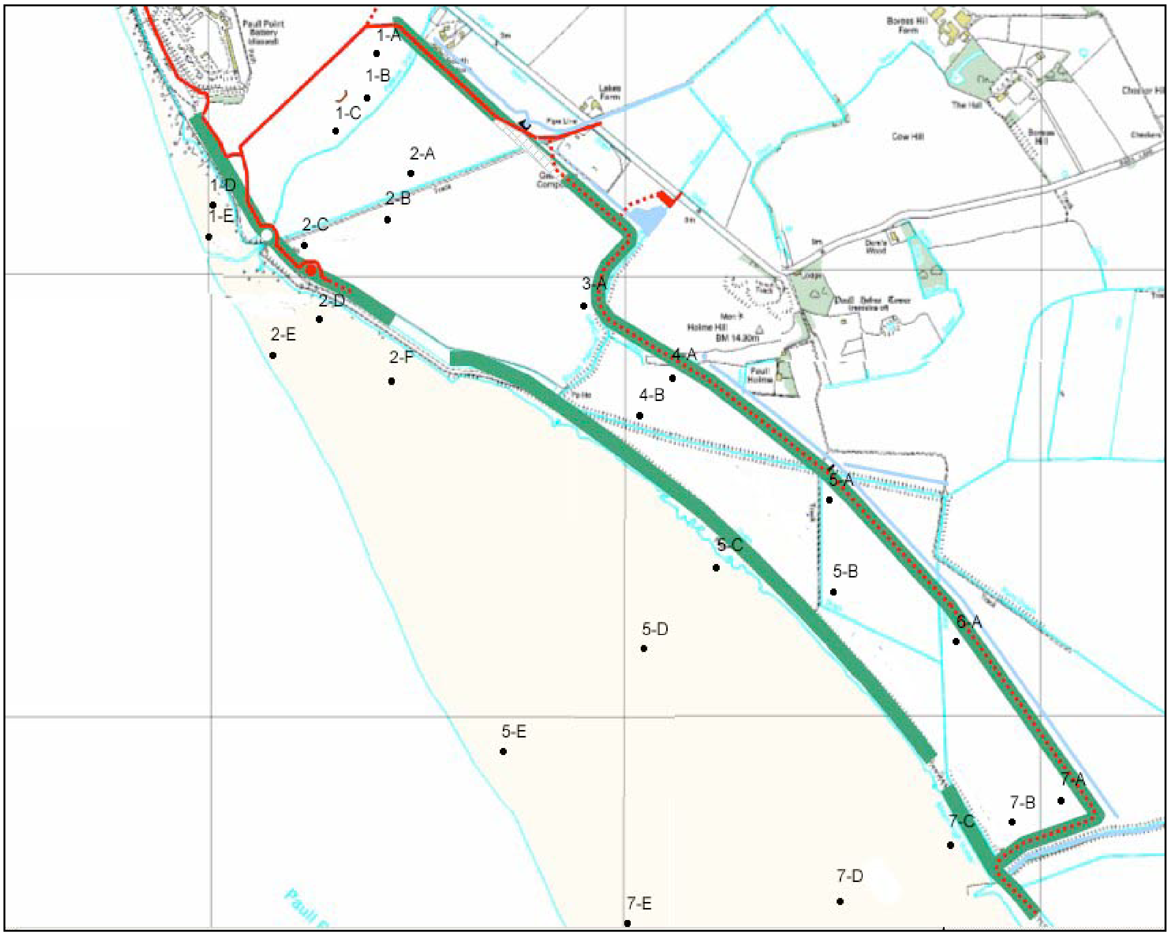


Fig. 4.1: Benthos collection stations at Paull Holme Strays. The green wall indicates the area of the restoration site. Stations 2- A, 2- B, 1- D, 1- E, 2- D, 2- E, 2- F, 5- C, 5- D, 5- E, 7- D, and 7- E are stations limited to hovercraft access. All stations were monitored between August 2004 and September 2005. ** Map modified from IECS map of PHS

4.2.2 Laboratory methods

At the time of analysis samples were again sieved through a 500 μm sieve before sorting. The sieve residue was placed then placed into a large white tray. Once in the tray samples were covered with water. At this point all animals were identified to the lowest taxonomic level possible (in most cases species level) using a binocular microscope. Abundance was then recorded as the number of individuals m^{-2}. The biomass for each species was then recorded as wet (tissue dry) weight. Species were placed onto blotting paper for 30 seconds, to allow for the absorption of preservative into the blotting paper, following this samples were placed onto a precision balance and a weight was taken in grams. Biomass measurements included all identifiable fragments and were made to ± 0.0001g. Both of these parameters were calculated as total abundance and biomass for individual species and for the whole community.

4.2.3 Data Analysis

The data that were collected during this study between August 2004 and September 2005 were compared to samples taken at a reference marsh (Cherry Cobb Sands) in 2000, 2001 and 2002 by the Environment Agency (Fig. 4.2). The relative composition of the species, in terms of abundance and biomass was expressed as a percentage of the total and calculated using mean data for each site per month. The mean biomass ratio (B/A) value was calculated by taking the total biomass for each site per month and dividing it by the total abundance from the same site for the same month. The abundance ratio (A/S) value was calculated by taking the total abundance for each site per month and dividing it by the number of species from the same site for the same month. The Shannon Weiner diversity index (H’) and Pielous index of evenness (J’) were also calculated.

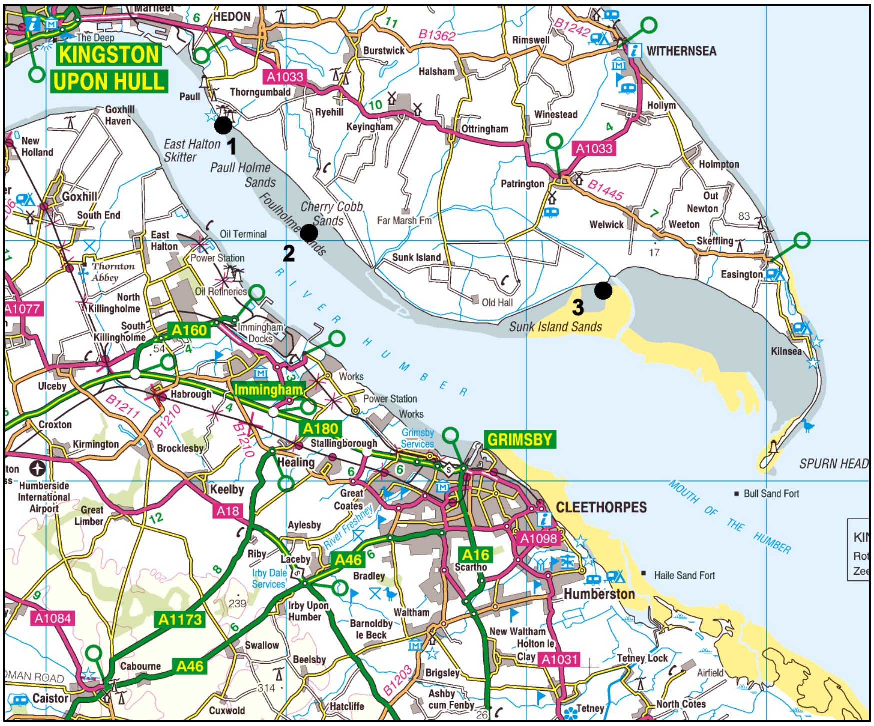


Fig. 4.2: Study sites along the Humber estuary. 1) Paull Holme Strays (sampled during this study August 2004 to September 2005), 2) Cherry Cobb sands (sampled by the Environment Agency in 2000, 2001, and 2002). 3 km separate Paull Holme Strays and Cherry Cobb sands. ** Map modified from googlemaps.com

The data were tested for homogeneity of variance (Levene's test) and statistical testing of site parameters was done using a one-way ANOVA followed by a post hoc comparison of means (using a Tukey's HSD). In cases where homogeneity of variance could not be achieved through transformation of the data, a Kruskal- Wallis test was used, followed by the Games-Howell test, which assumes unequal variance. Cherry Cobb Sands was not included in seasonal analysis of as the data was not collected at the same time as the data in this study and would have biased the results. A two-way ANOVA could have been used to simultaneously test for between site and between season differences, but the size of the data site combined the inequality of variances, prevented the generation of any easily interpretable results.

4.3. Results

The distribution of species and the abundance of those species inside of the restoration site at Paull Holme Strays appears to be related to tidal inundation (see chapter 2 for Hydrographic mapping) (Table 4.1). Stations that are infrequently inundated have lower abundances and diversity. Stations affected by this phenomenon are those adjacent to the new sea wall in the eastern part of he site and those in the far north western corner, where sedimentation has take place at a high rate, resulting in mudflats that are only able to be covered by spring tides. The mean number of species ranged from 2.7 (station 6-A) to 11.3 at station 3-A, directly opposite the western breach. A comparatively high value 7.7 was recorded at station 7-B, adjacent to the eastern breach (Table 4.1). The maximum mean abundance was recorded from station 6-A (24023 individuals/ m^{-2}). A minimum abundance value of 484 (individuals/ m^{-2}) was recorded at station 2-A. Shannon Weiner diversity (H') ranged from 0.2 (station 6-A) to 2.1 (station 5-B), (Table 4.1). Pielou's evenness (J') ranged from 0.1 to 0.9. Biomass ranged from 0.2 $g.m^{-2}$ (station 2-A) to 5.0 $g.m^{-2}$ at station 2-B, which is adjacent to the flood creek.

Table 4.1: Biological parameters (means) for all stations sampled inside and outside of Paull Holme Strays between August 2004 and September 2005 and Cherry Cobb sands sampled 2000, 2001 and 2002. Stations are separated by location.

INSIDE

	n	S	A	B	H'	J'	A/S	B/A
1-A	27	5.7	794.4	0.3	1.5	0.8	140.2	0.000
1-B	27	5.3	631.4	0.4	1.4	0.7	118.4	0.001
1-C	30	7.0	682.3	3.6	1.7	0.8	97.5	0.005
2-A	6	4.3	483.9	0.2	1.3	0.7	111.7	0.000
2-B	6	9.0	702.5	5.0	1.7	0.7	78.1	0.007
2-C	30	9.3	544.3	2.3	1.9	0.9	58.3	0.004
3-A	30	11.3	898.2	3.5	1.6	0.6	79.3	0.004
4-A	27	6.7	1655.9	2.7	1.3	0.6	248.4	0.002
4-B	27	4.3	682.0	0.9	1.4	0.8	157.4	0.001
5-A	30	6.0	1043.8	0.2	1.5	0.7	174.0	0.000
5-B	30	7.0	537.7	1.2	2.1	0.9	76.8	0.002
6-A	30	2.7	2402.7	0.5	0.2	0.1	901.0	0.000
7-A	30	6.0	952.1	0.5	1.5	0.8	158.7	0.001
7-B	30	7.7	1486.8	0.3	1.4	0.6	193.9	0.000

OUTSIDE

	n	S	A	B	H'	J'	A/S	B/A
1-D	6	10.3	907.5	12.6	2.2	0.8	87.8	0.014
1-E	6	7.3	837.5	0.5	1.3	0.5	114.2	0.001
2-D	6	11.7	3091.6	5.0	1.6	0.6	265.0	0.002
2-E	6	7.7	484.0	0.6	2.2	0.9	63.1	0.001
2-F	6	13.0	1434.1	8.9	2.3	0.8	110.3	0.006
5-C	6	14.0	1993.3	20.2	2.1	0.8	142.4	0.010
5-D	6	15.3	1760.5	8.1	2.3	0.8	114.8	0.005
5-E	6	10.3	1796.6	1.2	1.5	0.5	173.9	0.001
7-C	6	10.0	1079.6	8.1	1.8	0.7	108.0	0.007
7-D	6	12.7	2731.8	8.9	1.6	0.6	215.7	0.003
7-E	6	8.7	356.9	7.4	2.1	0.5	41.2	0.021

REFERENCE

	n	S	A	B	H'	J'	A/S	B/A
CHERRY	9	12.3	8190.1	N/A	1.5	0.5	668.6	N/A

*n= number of samples; S= number of species; A= total abundance / m^2; B= total biomass (g m^{-2}); H'= Shannon Weiner diversity; J'= Pielou's evenness; A/S= abundance ratio; B/A= biomass ratio.

In the natural habitat (outside the managed realignment site), patterns in the biological parameters were generally a function of shore height (Pearson Correlation= 0.59; $p < 0.05$) (Fig. 4.3), with the maximum number of species being recorded from the upper and mid shore stations, 2-F, 5-C and 5-D (Table 4.1). Mean abundance values ranged from 357-individuals/ m^{-2} at station 7-E (lower shore) to 3092-individuals/ m^{-2} at station 2-D (mid shore). Similarly, the maximum biomass was recorded from the upper/ mid shore at station 5-C (Table 4.1). In contrast both diversity values (H') and evenness (J') showed no clear spatial patterns.

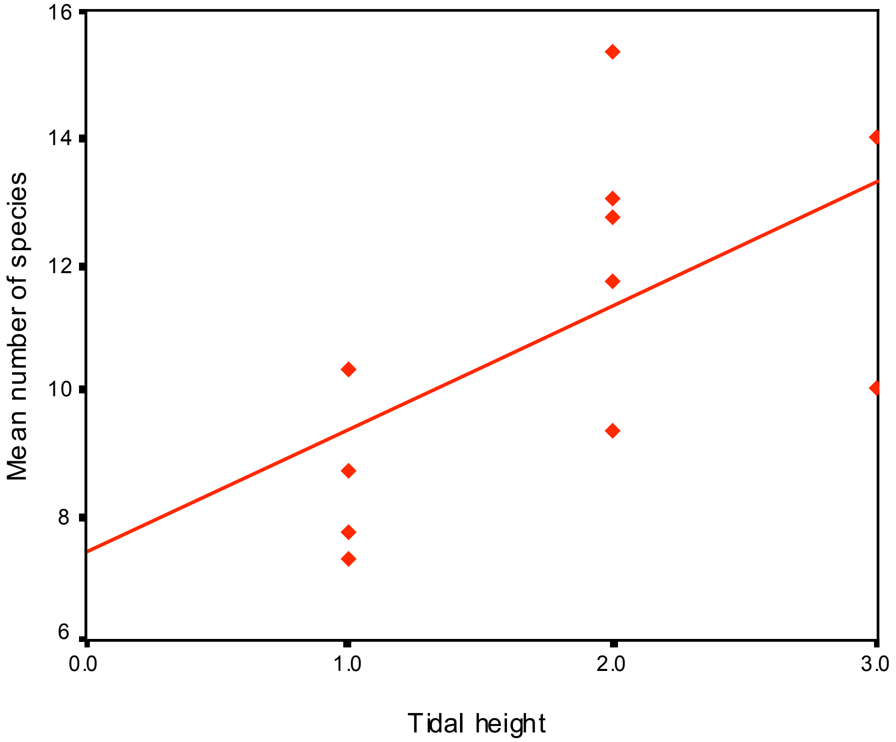


Fig. 4.3 Scattergram showing the biological correlation between the mean number of species and tidal height where 1 is low tide, 2 is mid tide and 3 is high tide. From the natural habitat outside of the Paull Holme Strays managed realignment site.

The restoration site and the natural site had abundance ratio values (A/S) that indicate high numbers of organisms represented by a small number of species. In the same way, the low biomass ratio values are indicative of a community composed of large numbers of small-bodied organisms (Pearson and Rosenberg*,* 1978).

Table 4.2: Kruskal- Wallis results for comparisons in biological parameters from inside and outside of Paull Holme Strays restoration site.

Indices	χ^2	d.f.
S	221.39*	25
A	172.34*	25
B	192.16*	24
H'	151.13*	24
J'	63.96*	24
A/S	116.24*	25
B/A	160.86*	25

*p 0< .01

Comparison between stations inside the restoration site, and those in the natural sites showed significant differences between all of the biological parameters (Table 4.2). The mean number of species found varied by station ranging from 12.3 at the reference site (Cherry Cobb Sands) to 10.8 in the natural site to 6.6 in the restoration site (Fig. 4.4). A Kruskal- Wallis test together with a Games-Howell test showed a significant difference in the number of species present based upon site (χ^2= 221.39; d.f. = 25; $p< 0.01$). Stations 6-a, 4-b, 5-a, 7-a and1-a, (all located in the restoration site had significantly fewer species ($p< 0.05$) than stations 2-a, 5-b, 1-b, 4-a, 1-c, 7-b. 3-a, 2-b, 1-e, 2-e, 1-d, 5-e, 7-e, 7-c, 2-d, Cherry Cobb Sands, 7-d, 2-f, 5-c and 5-d. Stations 2-a, 5-b, 1-b, 4-a, 1-c, 7-b, 2-c, 3-a, 2-b and 1-e had significantly fewer species ($p< 0.05$) than stations, 2-e, 1-d, 5-e, 7-e, 7-c, 2-d, Cherry Cobb Sands, 7-d, 2-f, 5-c and 5-d. Stations 1-c, 7-b, 2-c, 3-a, 2-b, 1-e and 2-e had significantly fewer species (p<

0.05) than stations 1-d, 5-e, 7-e, 7-c, 2-d, Cherry Cobb Sands, 7-d, 2-f, 5-c and 5-d. Stations 1-e, 2-e, 1-d, 5-e, 7-e and 7-c had significantly fewer species ($p < 0.05$) than stations 2-d, Cherry Cobb Sands, 7-d, 2-f, 5-c and 5-d. Stations 1-d, 5-e, 7-e, 7-c, 2-d and Cherry Cobb Sands had significantly fewer species ($p < 0.05$) than stations 7-d, 2-f, 5-c and 5-d. Stations 5-e, 7-e, 7-c, 2-d, Cherry Cobb Sands and 7-d had significantly fewer species ($p < 0.05$) than stations 2-f, 5-c and 5-d. Stations 7-e, 7-c, 2-d, Cherry Cobb Sands, 7-d, 2-f and 5-c had significantly fewer species ($p < 0.05$) than station 5-d. Figure 4.5 displays the homogeneous subsets found.

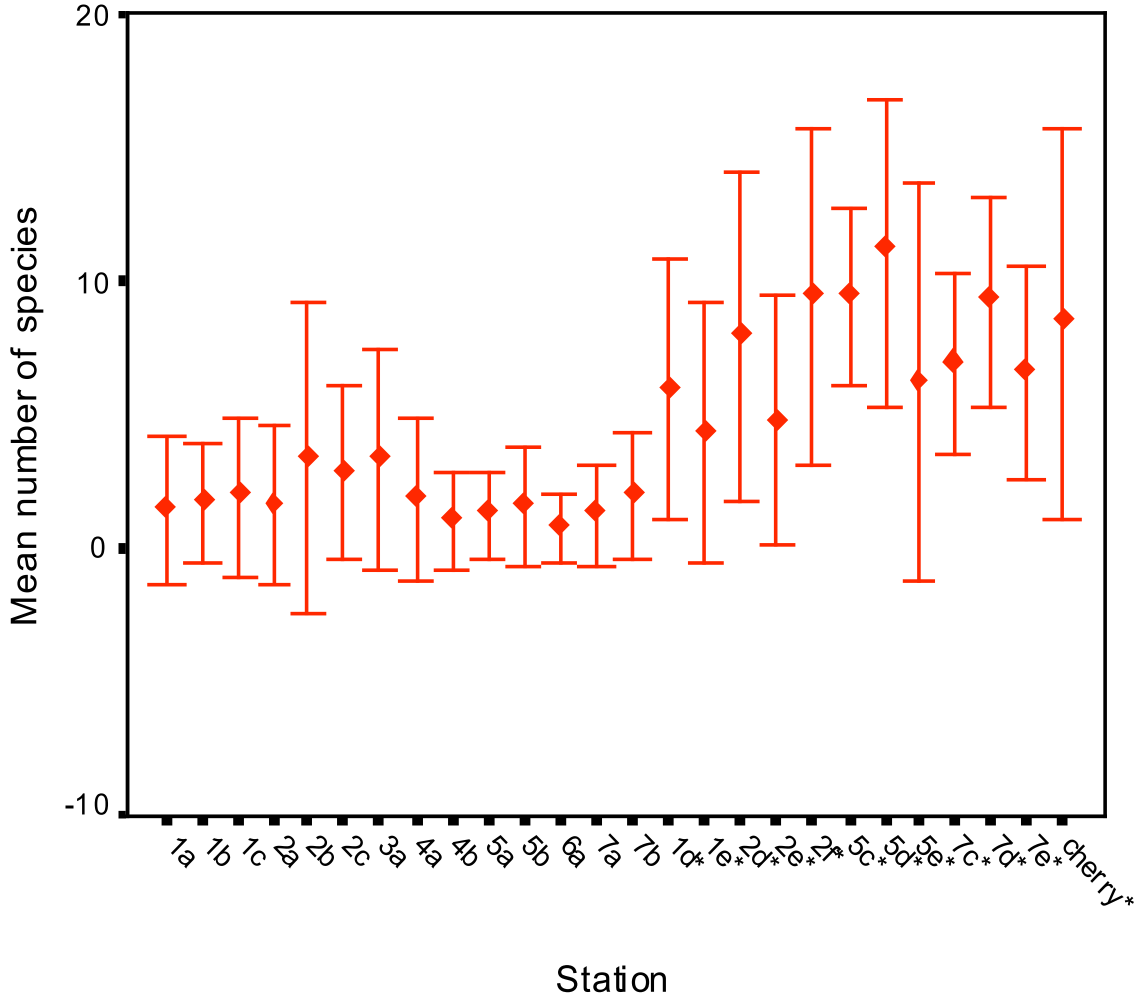
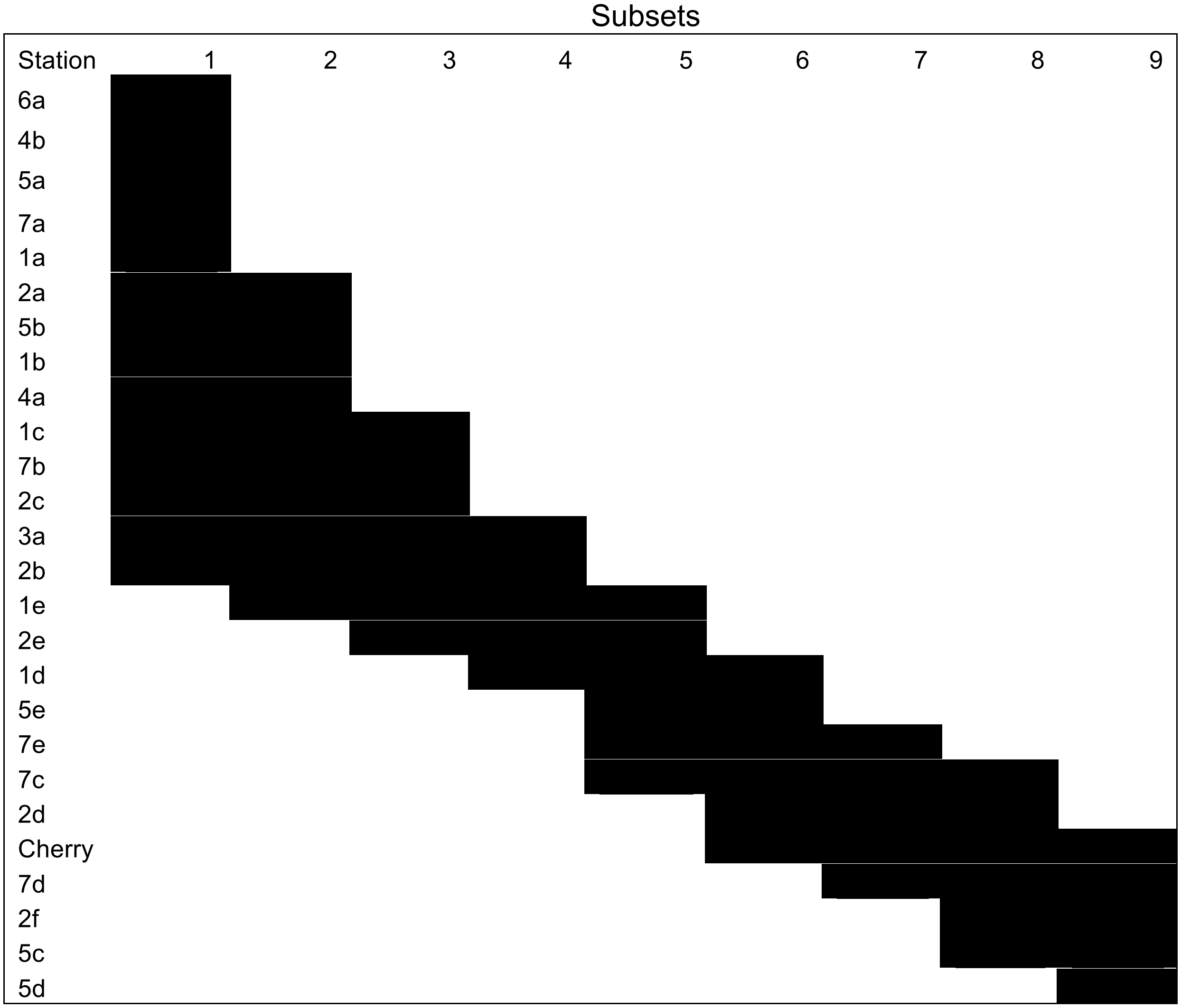


Fig. 4.4: Mean number of species sampled inside and outside of Paull Holme Strays restoration site between August 2004 and September 2005, and Cherry Cobb sands sampled 2000, 2001 and 2002 (± SD). * Indicates sample stations located on the natural mudflat.



Subsets

Station	1	2	3	4	5	6	7	8	9
6a	■								
4b	■								
5a	■								
7a	■								
1a	■								
2a	■	■							
5b	■	■							
1b	■	■							
4a	■	■							
1c	■	■	■						
7b	■	■	■						
2c	■	■	■						
3a	■	■	■	■					
2b	■	■	■	■					
1e		■	■	■	■				
2e			■	■	■				
1d				■	■	■			
5e					■	■			
7e					■	■	■		
7c					■	■	■	■	
2d						■	■	■	
Cherry						■	■	■	■
7d							■	■	■
2f								■	■
5c								■	■
5d									■

Fig. 4.5: Homogenous subsets of similarity between the 26 stations examined in 2000, 2001, 20002, 2004- 2005 relating to the number of species present.

Biomass varied by station ranging from 6.98 g m^{-2} in the natural site to 1.54 g m^{-2} in the restoration site (Fig. 4.6). A Kruskal- Wallis test together with a Games-Howell test showed a significant difference in the biomass of the benthic invertebrates based upon site (χ^2= 192.16; d.f. = 24; p< 0.01). Stations 5-a, 2-a, 6-a, 1-a, 7-b, 1-b, 4-b, 5-b, 1-e, 2-e, 4-a, 2-c, 7-a, 5-e, 1-c, 3-a and 2-b had significantly lower biomass values ($p < 0.05$) than stations 2-d, 7-e, 7-c, 1-d, 7-d, 2-f, 5-d, and 5-c. Stations 2-e, 4-a, 2-c, 7-a, 5-e, 1-c, 3-a, 2-b and 2-d had significantly lower biomass values ($p < 0.05$) than stations 7-e, 7-c, 1-d, 7-d, 2-f, 5-d, and 5-c. Stations 2-b, 2-d and 7-e had significantly lower biomass values ($p < 0.05$) than stations 7-c, 1-d, 7-d, 2-f, 5-d, and 5-c. Stations 2-d, 7-e, 7-c and 1-d had significantly lower biomass values ($p < 0.05$) than 7-d, 2-f, 5-d, and 5-c. Stations 7-e, 7-c, 1-d and 7-d had significantly lower biomass values ($p < 0.05$) than stations 2-f, 5-d, and 5-c. Stations 7-c, 1-d, 7-d, 2-f and 5-d had significantly lower biomass values ($p < 0.05$) than station 5-c. Figure 4.7 displays the homogeneous subsets found.

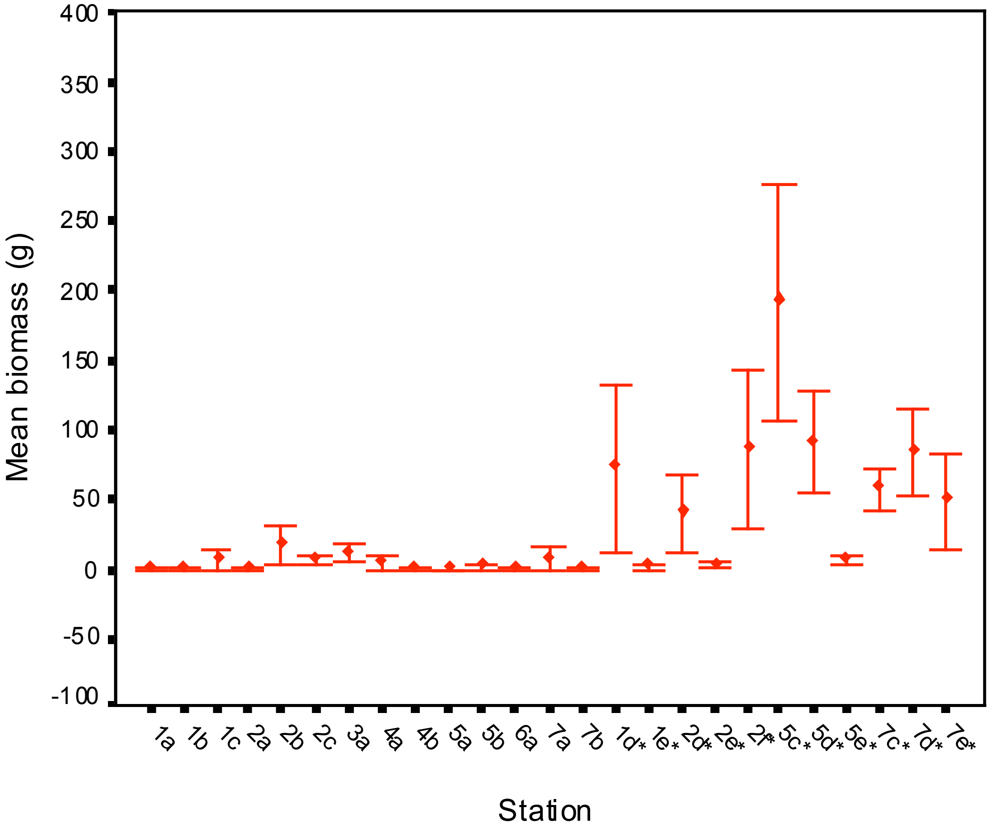


Fig. 4.6: Mean biomass in grams of invertebrates sampled inside and outside of Paull Holme Strays restoration site between August 2004 and September 2005 (± SD). * Indicates sample stations located on the natural mudflat.

	Subsets						
STATION	1	2	3	4	5	6	7
5a	■						
2a	■						
6a	■						
1a	■						
7b	■						
1b	■						
4b	■						
5b	■						
1e	■						
2e	■	■					
4a	■	■					
2c	■	■					
7a	■	■					
5e	■	■					
1c	■	■					
3a	■	■					
2b	■	■	■				
2d		■	■	■			
7e			■	■	■		
7c				■	■	■	
1d				■	■	■	
7d					■	■	
2f						■	
5d						■	
5c							■

Fig. 4.7: Homogenous subsets of similarity between the 25 stations examined between August 2004 and September 2005 relating to the biomass of the benthic invertebrates found.

The Mean Shannon Weiner diversity index (H') varied by station ranging from 1.5 at the reference site (Cherry Cobb Sands) to 1.75 in the natural site to 1.3 in the restoration site (Fig. 4.8). A Kruskal- Wallis test together with a Games-Howell test showed a significant difference in the number of species present based upon site (χ^2= 151.13; d.f. = 24; p< 0.01). Stations 6-a, 4,a, 7-a, 1-a, 4-b, 1-b, 5-a, 1-c, 5-b, 2-a and 7-b (all in the restoration site) had significantly less diversity (p< 0.05) than stations 1-e, 3-a, 2-c, 2-b, 5-e, 2-d, 7-c, 2-e, 7-e, 7-d, 1-d, 5-c, 5-d and 2-f. Stations 4,a, 7-a, 1-a, 4-b, 1-b, 5-a, 1-c, 5-b, 2-a, 7-b, 1-e, 3-a and 2-c had significantly less diversity ($p < 0.05$) than stations 2-b, 5-e, 2-d, 7-c, 2-e, 7-e, 7-d, 1-d, 5-c, 5-d and 2-f. Stations 1-c, 5-b, 2-a, 7-b, 1-e, 3-a, 2-c 2-b, 5-e and 2-d and 7-c had significantly less diversity ($p < 0.05$) than stations 7-c, 2-e, 7-e, 7-d, 1-d, 5-c, 5-d and 2-f. Stations 5-b, 2-a, 7-b, 1-e, 3-a, 2-c 2-b, 5-e, and 7-c had significantly less diversity ($p < 0.05$) than stations 2-e, 7-e, 7-d, 1-d, 5-c, 5-d and 2-f. Stations 1-e, 3-a, 2-c 2-b, 5-e, 2-d, 7-c, 2-e, 7-e and7-d had significantly less diversity ($p < 0.05$) than stations 1-d, 5-c, 5-d and 2-f. Stations 2-c 2-b, 5-e, 2-d, 7-c, 2-e, 7-e, 7-d, 1-d and 5-c had significantly less diversity ($p < 0.05$) than stations 5-d and 2-f. Stations 2-b, 5-e, 2-d, 7-c, 2-e, 7-e, 7-d, 1-d, 5-c and 5-d had significantly less diversity ($p < 0.05$) than station 2-f. Figure 4.9 displays the homogeneous subsets found.

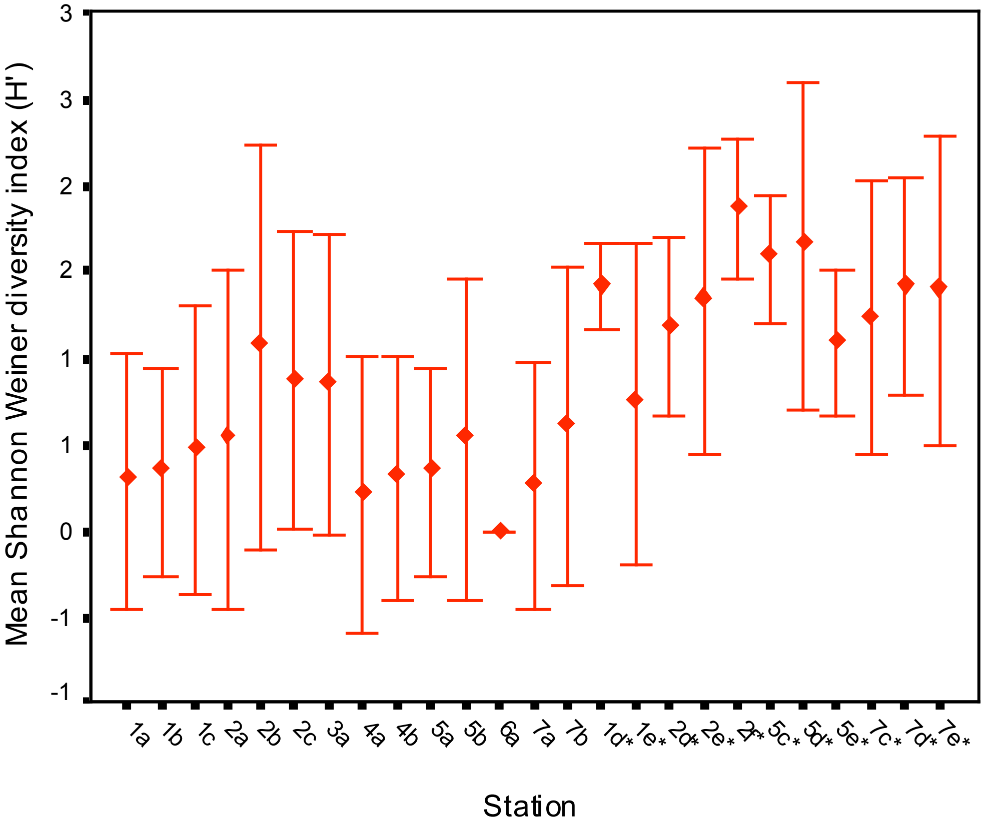
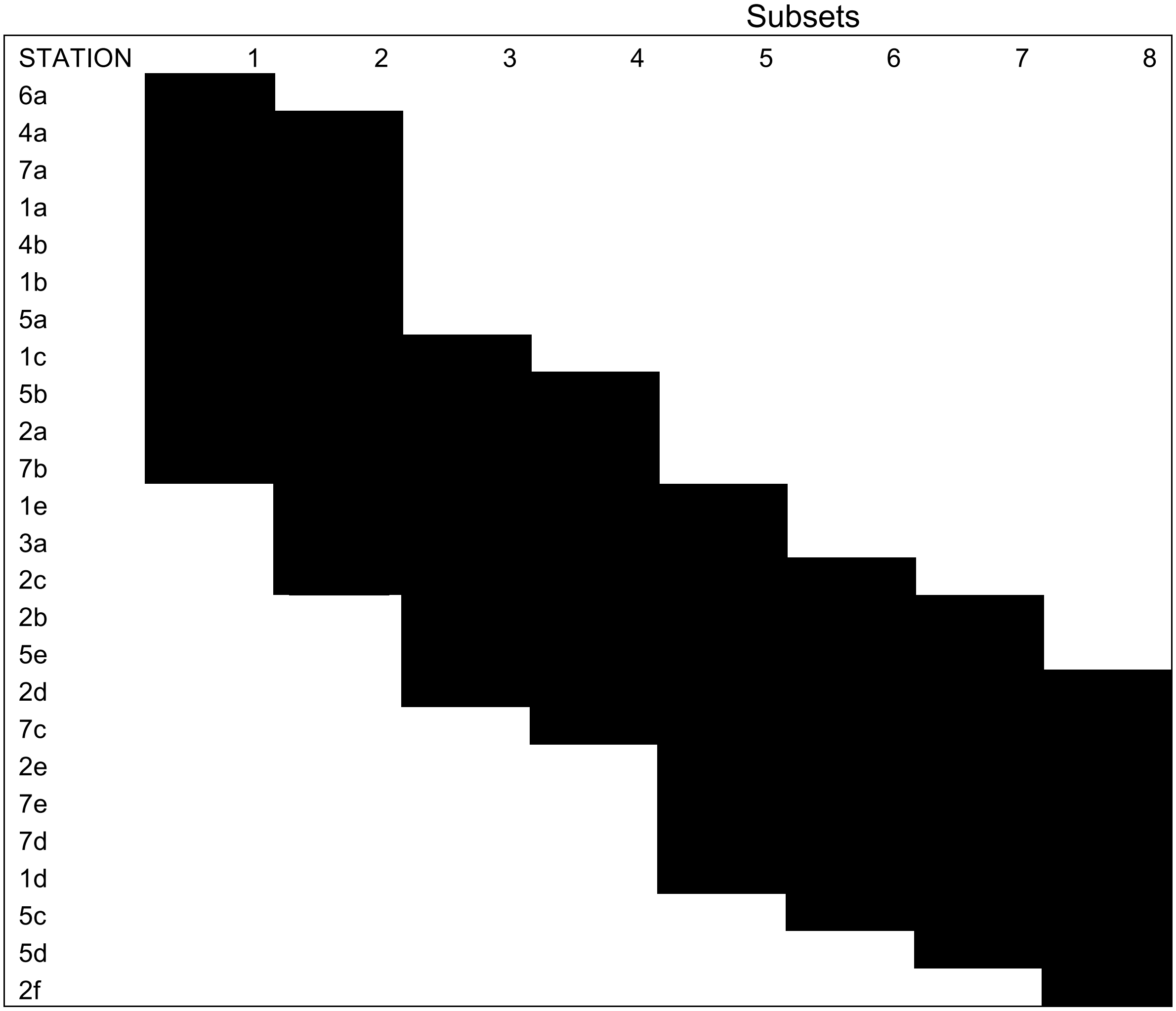


Fig. 4.8: Mean Shannon Weiner diversity index (H') sampled inside and outside of Paull Holme Strays between August 2004 and September 2005 (± SD). * Indicates sample stations located on the natural mudflat.



	Subsets							
STATION	1	2	3	4	5	6	7	8
6a	■							
4a	■	■						
7a	■	■						
1a	■	■						
4b	■	■						
1b	■	■						
5a	■	■						
1c	■	■	■					
5b	■	■	■	■				
2a	■	■	■	■				
7b	■	■	■	■				
1e		■	■	■	■			
3a		■	■	■	■			
2c		■	■	■	■	■		
2b			■	■	■	■	■	
5e			■	■	■	■	■	
2d			■	■	■	■	■	■
7c				■	■	■	■	■
2e					■	■	■	■
7e					■	■	■	■
7d					■	■	■	■
1d					■	■	■	■
5c						■	■	■
5d							■	■
2f								■

Fig. 4.9: Homogenous subsets of similarity between the 25 stations examined between August 2004 and September 2005 relating to the Shannon Weiner diversity index (H’).

Pielou’s evenness (J’) varied by station ranging from 0.5 at the reference site (Cherry Cobb Sands) to 0.68 in the natural site to 0.69 in the restoration site (Fig. 4.10). A Kruskal- Wallis test together with a Games-Howell test showed a significant difference in the number of species present based upon site (χ^2= 63.96; d.f. = 24; $p< 0.01$). Stations 6-a, 4-a, 7-a, 1-a, 4-b, 1-c, 5-a, 2-a, 1-e, 5-b, 7-b, 2-d and 3-a had significantly ($p < 0.05$) lower evenness values than stations 7-c, 7-d, 5-d, 2-b, 5-c, 2-c, 1-d, 5-e, 2-e, 7-e and 2-f.

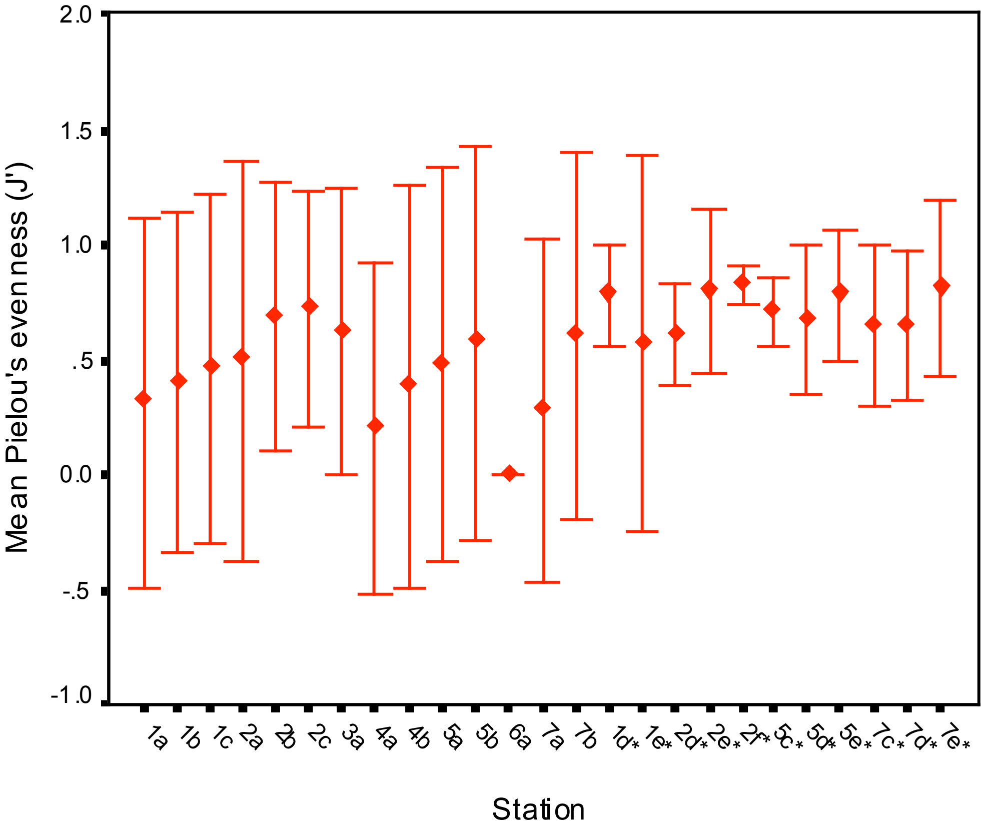


Fig. 4.10: Mean Pielou's evenness (J') sampled inside and outside of Paull Holme Strays between August 2004 and September 2005 (± SD). * Indicates sample stations located on the natural mudflat.

Abundance varied by station ranging from 8190 per m^{-2} in the reference site (Cherry Cobb Sands) to1496 per m^2 in the natural site to 963 per m^2 in the restoration site (Fig. 4.11). A Kruskal- Wallis test together with a Games-Howell test showed a significant difference in abundance based upon site ($\chi^2 = 172.34$; d.f. = 25; $p < 0.01$). Stations in the restoration and natural habitat had significantly fewer individuals per m^2 ($p < 0.05$) than the reference site (Cherry Cobb Sands).

Abundance was slightly higher outside of the site although the difference was not significant

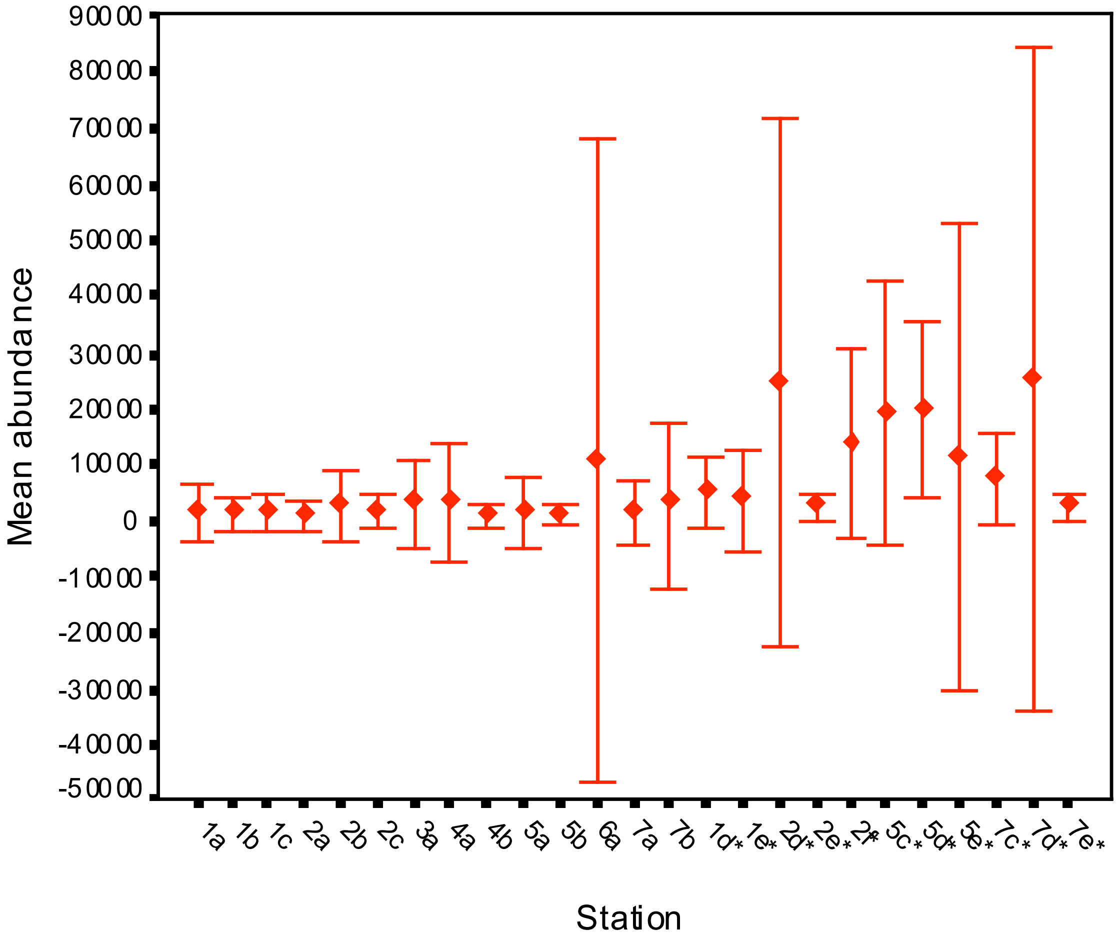


Fig. 4.11: Mean abundance per m^2 sampled inside and outside of Paull Holme Strays between August 2004 and September 2005 (± SD). * Indicates sample stations located on the natural mudflat.

Both the Abundance ratio and the biomass ratio varied significantly base upon site (Table 4.2), however a post hoc Games-Howell test was unable to detect where those differences lie.

A total of 31species were recorded from the survey areas as a whole, 26 of which were present inside the restoration site. Overall, over the monitoring period the benthic community inside of the site was dominated by *Isotomidae* and *Paranais litoralis*, both representing 68.3% of the abundance (Table 4.3). These taxa together with *Tubificoides benedii*, Tuplidae, Nematoda, *Hydrobia ulvae* and *Hediste diversicolor*, accounted for 89% of the abundance over the monitoring period. There were very low numbers of *Macoma balthica*, *Corophium volutator* and spionidae, which are all common inhabitants of this part of the Humber estuary (Mazik, 2004). The biomass in the restored site was heavily dominated by *Hediste diversicolor* (50%), which together with *Macoma balthica* (28%) accounted for 78% of the biomass (Table 4.4). These taxa together with *Abra tenuis* and *Arenicola marina* accounted for 89% of the biomass (Table 4.4).

Table 4.3: Species dominance in terms of abundance for sites inside (collectively), outside (collectively) Paull Holme restoration site and the natural site Cherry Cobb sands.

INSIDE				**OUTSIDE**				**CHERRYCOBB SANDS**			
	Total Abundance	% Dom	Cum %		Total Abundance	% Dom	Cum %		Total Abundance	% Dom	Cum %
Abra tenuis	59	0.9	0.9	*Abra tenuis*	1100	7.0	7.0	*Abra tenuis*	1159	1.2	1.2
Acarina	92	1.5	2.4	*Acarina*	0	0.0	7.0	*Acarina*	0	0.0	1.2
Aphelochaeta marioni	9	0.1	2.6	*Aphelochaeta marioni*	0	0.0	7.0	*Aphelochaeta marioni*	0	0.0	1.2
Arenicola marina	45	0.7	3.3	*Arenicola marina*	0	0.0	7.0	*Arenicola marina*	0	0.0	1.2
Capitella sp.	9	0.1	3.4	*Capitella sp.*	0	0.0	7.0	*Capitella sp.*	0	0.0	1.2
Carcinus maenas	9	0.1	3.6	*Carcinus maenas*	0	0.0	7.0	*Carcinus maenas*	0	0.0	1.2
Chone/ Jasmineria	9	0.1	3.7	*Chone/ Jasmineria*	0	0.0	7.0	*Chone/ Jasmineria*	0	0.0	1.2
Copepoda	18	0.3	4.0	*Copepoda*	46	0.3	7.3	*Copepoda*	0	0.0	1.2
Corophium volutator	64	1.0	5.0	*Corophium volutator*	75	0.5	7.8	*Corophium volutator*	169	0.2	1.4
Cyathura carinata	0	0.0	5.0	*Cyathura carinata*	1331	8.5	16.3	*Cyathura carinata*	255	0.3	1.6
Enchytraeidae sp. indet.	106	1.7	6.7	*Enchytraeidae sp. indet.*	4109	26.3	42.6	*Enchytraeidae sp. indet.*	34596	35.9	37.6
Etone flava/ longa	0	0.0	6.7	*Etone flava/ longa*	163	1.0	43.6	*Etone flava/ longa*	254	0.3	37.8
Hediste diversicolor	293	4.7	11.4	*Hediste diversicolor*	960	6.1	49.8	*Hediste diversicolor*	1053	1.1	38.9
Heterochaeta costata	9	0.1	11.6	*Heterochaeta costata*	52	0.3	50.1	*Heterochaeta costata*	0	0.0	38.9
Hydrobia ulvae	141	2.3	13.8	*Hydrobia ulvae*	458	2.9	53.0	*Hydrobia ulvae*	302	0.3	39.2
Idoteidae	0	0.0	13.8	*Idoteidae*	0	0.0	53.0	*Idoteidae*	0	0.0	39.2
Isotomidae	3355	53.7	67.5	*Isotomidae*	154	1.0	54.0	*Isotomidae*	0	0.0	39.2
Juv. Tellinacea sp.	27	0.4	67.9	*Juv. Tellinacea sp.*	220	1.4	55.4	*Juv. Tellinacea sp.*	0	0.0	39.2
Macoma balthica	79	1.3	69.2	*Macoma balthica*	817	5.2	60.7	*Macoma balthica*	4670	4.8	44.1
Manayunkia aestuarina	18	0.3	69.5	*Manayunkia aestuarina*	702	4.5	65.2	*Manayunkia aestuarina*	3872	4.0	48.1
Nematoda	389	6.2	75.7	*Nematoda*	2962	18.9	84.1	*Nematoda*	39844	41.4	89.5
Nephtys hombergii	0	0.0	75.7	*Nephtys hombergii*	77	0.5	84.6	*Nephtys hombergii*	127	0.1	89.6
Nitocra	0	0.0	75.7	*Nitocra*	0	0.0	84.6	*Nitocra*	0	0.0	89.6
Paranais litoralis	632	10.1	85.8	*Paranais litoralis*	58	0.4	85.0	*Paranais litoralis*	254	0.3	89.9
Pygospio elegans	44	0.7	86.5	*Pygospio elegans*	653	4.2	89.1	*Pygospio elegans*	1698	1.8	91.6
Sabellidae sp.	27	0.4	86.9	*Sabellidae sp.*	0	0.0	89.1	*Sabellidae sp.*	0	0.0	91.6
Scrobicularia plana	27	0.4	87.4	*Scrobicularia plana*	64	0.4	89.5	*Scrobicularia plana*	458	0.5	92.1
Streblospio shrubsolii	9	0.1	87.5	*Streblospio shrubsolii*	435	2.8	92.3	*Streblospio shrubsolii*	127	0.1	92.2
Tuplidae	336	5.4	92.9	*Tuplidae*	100	0.6	93.0	*Tuplidae*	0	0.0	92.2
Tubificoides benedii	427	6.8	99.7	*Tubificoides benedii*	1076	6.9	99.9	*Tubificoides benedii*	7490	7.8	100.0
Tubificoides pseudogaster	18	0.3	100.0	*Tubificoides pseudogaster*	23	0.1	100.0	*Tubificoides pseudogaster*	0	0.0	100.0

Table 4.4: Species biomass for sites inside (collectively) and outside (collectively) Paull Holme Strays restoration site from 2004 to 2005.

INSIDE				OUTSIDE			
	Total Biomass	% Dom	Cum %		Total Biomass	% Dom	Cum %
Abra tenuis	2.77	7.1	7.1	*Abra tenuis*	17.8	10.2	10.2
Acarina	0.013	0.0	7.1	*Acarina*	0.000	0.0	10.2
Aphelochaeta marioni	0.013	0.0	7.2	*Aphelochaeta marioni*	0.000	0.0	10.2
Arenicola marina	1.67	4.3	11.4	*Arenicola marina*	0.000	0.0	10.2
Capitella sp.	0.013	0.0	11.5	*Capitella sp.*	0.000	0.0	10.2
Carcinus maenas	0.346	0.9	12.4	*Carcinus maenas*	0.000	0.0	10.2
Chone/ Jasmineria	0.013	0.0	12.4	*Chone/ Jasmineria*	0.000	0.0	10.2
Copepoda	0.022	0.1	12.5	*Copepoda*	0.013	0.0	10.2
Corophium volutator	0.866	2.2	14.7	*Corophium volutator*	0.603	0.3	10.6
Cyathura carinata	0.000	0.0	14.7	*Cyathura carinata*	1.11	0.6	11.2
Enchytraeidae sp. indet.	0.024	0.1	14.7	*Enchytraeidae sp. indet.*	0.141	0.1	11.3
Etone flava/ longa	0.000	0.0	14.7	*Etone flava/ longa*	0.595	0.3	11.6
Hediste diversicolor	19.4	49.9	64.6	*Hediste diversicolor*	31.3	18.0	29.6
Heterochaeta costata	0.013	0.0	64.6	*Heterochaeta costata*	0.013	0.0	29.6
Hydrobia ulvae	0.730	1.9	66.5	*Hydrobia ulvae*	1.54	0.9	30.5
Idoteidae	0.013	0.0	66.6	*Idoteidae*	0.000	0.0	30.5
Isotomidae	0.123	0.3	66.9	*Isotomidae*	0.013	0.0	30.5
Juv. Tellinacea sp.	0.038	0.1	67.0	*Juv. Tellinacea sp.*	0.100	0.1	30.6
Macoma balthica	10.9	27.9	94.9	*Macoma balthica*	24.3	13.9	44.5
Manayunkia aestuarina	0.013	0.0	94.9	*Manayunkia aestuarina*	0.014	0.0	44.5
Nematoda	0.025	0.1	95.0	*Nematoda*	0.018	0.0	44.5
Nephtys hombergii	0.000	0.0	95.0	*Nephtys hombergii*	1.88	1.1	45.6
Nitocra	0.035	0.1	95.0	*Nitocra*	0.000	0.0	45.6
Paranais litoralis	0.253	0.6	95.7	*Paranais litoralis*	0.013	0.0	45.6
Pygospio elegans	0.393	1.0	96.7	*Pygospio elegans*	0.617	0.4	46.0
Sabellidae sp.	0.153	0.4	97.1	*Sabellidae sp.*	0.000	0.0	46.0
Scrobicularia plana	0.013	0.0	97.1	*Scrobicularia plana*	92.5	53.1	99.0
Streblospio shrubsolii	0.013	0.0	97.2	*Streblospio shrubsolii*	0.061	0.0	99.1
Tuplidae	0.989	2.5	99.7	*Tuplidae*	0.831	0.5	99.5
Tubificoides benedii	0.103	0.3	100.0	*Tubificoides benedii*	0.772	0.4	100.0
Tubificoides pseudogaster	0.013	0.0	100.0	*Tubificoides pseudogaster*	0.013	0.0	100.0

Hediste diversicolor, *Hydrobia ulvae* and Enchytraeid worms generally dominated communities at stations adjacent to the new sea wall in the Southeast of the restoration site (Table 4.5). The terrestrial organisms Isotomidae and Tuplidae were also present at multiple stations. The dominant species at stations

1-A and 1-B are *H. ulavae*, *T. benedii* and terrestrial organisms, while stations 4-A and 4-B were almost entirely composed of terrestrial organisms (Table4.5).

Table 4.5: Species dominance for individual sampling stations from inside and outside Paull Holme Strays restoration site, stations are separated by location and were sampled from August 2004- September 2005.

RESTORED HABITAT

1-A	A	% Dom	Cum %
Abra tenuis	127	3.07	3.08
Acarina	191	4.62	7.70
Hydrobia ulvae	255	6.18	13.9
Isotomidae	1974	47.8	61.7
Nematoda	424	10.3	71.9
Tuplidae	369	8.94	80.9
Tubificoides benedii	789	19.1	100.0

1-B	A	% Dom	Cum %
Acarina	127	4.47	4.47
Arenicola marina	382	13.4	17.9
Hediste diversicolor	127	4.47	22.4
Hydrobia ulvae	127	4.47	26.9
Isotomidae	1592	56.0	82.7
Nematoda	254	8.94	91.8
Tuplidae	233	8.12	100.0

1-C	A	% Dom	Cum %
Acarina	127	2.73	2.73
Hediste diversicolor	1104	23.8	26.5
Hydrobia ulvae	127	2.74	29.2
Isotomidae	649	13.9	43.2
Macoma baltica	127	2.73	45.9
Nematoda	709	15.3	61.2
Tuplidae	179	3.85	65.1
Tubificoides benedii	1624	34.9	100.0

2-A	A	% Dom	Cum %
Acarina	339	13.9	13.9
Isotomidae	1443	59.1	72.9
Juv. Tellinacea sp.	127	5.19	78.1
Macoma baltica	127	5.19	83.3
Nematoda	127	5.19	88.5
Tuplidae	280	11.5	100.0

2-B	A	% Dom	Cum %
Carcinus maenas	127	1.795	1.80
Enchytraeidae sp. indet.	382	5.39	7.19
Hediste diversicolor	1146	16.2	23.4
Hydrobia ulvae	127	1.79	25.2
Isotomidae	3439	48.6	73.8
Juv. Tellinacea sp.	255	3.60	77.4
Macoma baltica	178	2.52	79.9
Manayunkia aestuarina	127	1.79	81.7
Nematoda	828	11.7	93.4
Paranais litoralis	212	2.99	96.4
Strebiospio shrubsolii	127	1.79	98.2
Tubificoides benedii	127	1.79	100.0

2-C	A	% Dom	Cum %
Hediste diversicolor	205	5.37	5.37
Heterochaeta costata	127	3.33	8.70
Isotomidae	445	11.7	20.4
Macoma baltica	145	3.80	24.2
Nematoda	1019	26.7	50.9
Pygospio elegans	127	3.33	54.2
Scrobicularia plana	382	10.0	64.2
Tuplidae	275	7.21	71.4
Tubificoides benedii	1091	28.6	100.0

Table 4.5:
Continued.

3-A	A	% Dom	Cum %
Aphelochacta marioni	127	1.01	1.01
Capitella sp.	127	1.01	2.02
Chone/ Jasmineria	127	1.01	3.04
Corophium volutator	127	1.01	4.05
Enchytraeidae sp. indet.	127	1.01	5.06
Hydrobia ulvae	410	3.27	8.33
Isotomidae	483	3.85	12.2
Macoma baltica	277	2.21	14.4
Manayunkia aestuarina	127	1.01	15.4
Nematoda	1126	8.97	24.4
Paranais litoralis	7579	60.4	84.8
Pygospio elegans	484	3.86	88.6
Sabellidae sp.	382	3.05	91.7
Tuplidae	185	1.47	93.2
Tubificoides benedii	859	6.85	100.0

4-A	A	% Dom	Cum %
Corophium volutator	127	2.16	2.16
Enchytraeidae sp. indet.	212	3.61	5.77
Hediste diversicolor	212	3.61	9.38
Hydrobia ulvae	169	2.88	12.3
Isotomidae	3708	63.1	75.4
Paranais litoralis	594	10.1	85.5
Tuplidae	727	12.37	97.8
Tubificoides benedii	127	2.16	100.0

4-B	A	% Dom	Cum %
Acarina	127	5.807	5.81
Enchytraeidae sp. indet.	127	5.807	11.6
Hydrobia ulvae	127	5.807	17.4
Isotomidae	624	28.532	45.9
Tuplidae	991	45.313	91.3
Tubificoides benedii	191	8.733	100.0

5-A	A	% Dom	Cum %
Acarina	127	3.18	3.18
Arenicola marina	127	3.18	6.37
Copopoda	255	6.39	12.8
Hediste diversicolor	127	3.18	15.9
Hydrobia ulvae	191	4.78	20.7
Isotomidae	2388	59.9	80.6
Nematoda	127	3.18	83.8
Paranais litoralis	255	6.39	90.2
Tuplidae	190	4.76	94.9
Tubificoides benedii	203	5.09	100.0

5-B	A	% Dom	Cum %
Abra tenius	700	19.9	19.9
Acarina	127	3.60	23.5
Arenicola marina	127	3.60	27.1
Corophium volutator	382	10.8	37.9
Hediste diversicolor	127	3.60	41.5
Hydrobia ulvae	127	3.60	45.1
Isotomidae	798	22.6	67.7
Macoma baltica	127	3.60	71.4
Nematoda	191	5.42	76.8
Tuplidae	564	16.0	92.8
Tubificoides benedii	255	7.23	100.0

6-A	A	% Dom	Cum %
Corophium volutator	255	1.06	1.06
Hediste diversicolor	127	0.53	1.58
Isotomidae	23374	96.9	98.5
Tuplidae	127	0.527	99.0
Tubificoides benedii	233	0.966	100.0

Table 4.5:
Continued.

7-A	A	% Dom	Cum %
Enchytraeidae sp. indet.	510	12.9	12.9
Hediste diversicolor	796	20.1	33.1
Hydrobia ulvae	191	4.83	37.9
Isotomidae	1862	47.1	84.9
Nematoda	127	3.21	88.2
Tuplidae	339	8.58	96.8
Tubificoides benedii	127	3.21	100.0

7-B	A	% Dom	Cum %
Acarina	127	1.99	1.99
Enchytraeidae sp. indet.	127	1.99	3.97
Hediste diversicolor	127	1.99	5.96
Hydrobia ulvae	127	1.99	7.94
Isotomidae	4196	65.6	73.5
Macoma baltica	127	1.99	75.5
Nematoda	509	7.96	83.5
Paranais litoralis	212	3.31	86.8
Tuplidae	240	3.75	90.5
Tubificoides benedii	350	5.47	96.0
Tubificoides pseudogaster	255	3.99	100.0

NATURAL HABITAT

1-D	A	% Dom	Cum %
Corophium volutator	700	7.71	7.71
Enchytraeidae sp. indet.	127	1.39	9.11
Hediste diversicolor	721	7.95	17.1
Heterochaeta costata	127	1.39	18.5
Hydrobia ulvae	446	4.92	23.4
Juv. Tellinacea sp.	191	2.10	25.5
Macoma baltica	1299	14.3	39.8
Manayunkia aestuarina	127	1.39	41.2
Nematoda	1931	21.3	62.5
Paranais litoralis	127	1.39	63.9
Pygospio elegans	127	1.39	65.3
Scrobicularia plana	191	2.10	67.4
Strebiospio shrubsolii	828	9.12	76.5
Tubificoides benedii	2006	22.1	98.6
Tubificoides pseudogaster	127	1.39	100.0

1-E	A	% Dom	Cum %
Enchytraeidae sp. indet.	191	3.15	3.15
Etone flava/ longa	127	2.09	5.24
Hydrobia ulvae	127	2.09	7.33
Isotomidae	169	2.79	10.1
Juv. Tellinacea sp.	127	2.09	12.2
Macoma baltica	424	6.99	19.2
Nematoda	4203	69.3	88.5
Nephtys hombergii	127	2.09	90.6
Paranais litoralis	127	2.09	92.7
Scrobicularia plana	127	2.09	94.8
Tubificoides benedii	191	3.15	97.9
Tubificoides pseudogaster	127	2.09	100.0

Table 4.5:
Continued.

2-D	A	% Dom	Cum %
Abra tenius	796	2.84	2.84
Cyathura carinata	13396	47.8	50.7
Etone flava/ longa	222	0.792	51.5
Hediste diversicolor	976	3.48	54.9
Hydrobia ulvae	828	2.96	57.9
Juv. Tellinacea sp.	382	1.36	59.3
Macoma baltica	509	1.82	61.1
Manayunkia aestuarina	1401	5.00	66.1
Nematoda	7303	26.1	92.1
Paranais litoralis	127	0.45	92.6
Strebiospio shrubsolii	1273	4.54	97.1
Tuplidae	254	0.907	98.0
Tubificoides benedii	551	1.97	100.0

2-E	A	% Dom	Cum %
Abra tenius	127	3.14	3.14
Enchytraeidae sp. indet.	339	8.37	11.5
Etone flava/ longa	254	6.27	17.8
Hediste diversicolor	127	3.14	20.9
Hydrobia ulvae	127	3.14	24.0
Isotomidae	382	9.43	33.5
Macoma baltica	541	13.4	46.8
Nematoda	1210	29.9	76.7
Nephtys hombergii	255	6.30	82.9
Paranais litoralis	127	3.14	86.1
Strebiospio shrubsolii	339	8.37	94.5
Tubificoides benedii	223	5.50	100.0

2-F	A	% Dom	Cum %
Abra tenius	2216	12.9	12.9
Cyathura carinata	190	1.11	14.0
Enchytraeidae sp. indet.	1995	11.6	25.7
Etone flava/ longa	169	0.986	26.7
Hediste diversicolor	815	4.75	31.4
Heterochaeta costata	127	0.741	32.2
Hydrobia ulvae	866	5.05	37.2
Juv. Tellinacea sp.	700	4.08	41.3
Macoma baltica	1477	8.62	49.9
Manayunkia aestuarina	255	1.49	51.4
Nematoda	4178	24.4	75.8
Pygospio elegans	2777	16.2	91.9
Scrobicularia plana	127	0.74	92.7
Strebiospio shrubsolii	191	1.11	93.8
Tuplidae	127	0.741	94.6
Tubificoides benedii	934	5.45	100.0

5-C	A	% Dom	Cum %
Abra tenius	4033	17.0	17.0
Copopoda	127	0.536	17.6
Enchytraeidae sp. indet.	2866	12.1	29.7
Etone flava/ longa	169	0.714	30.4
Hediste diversicolor	3099	13.1	43.5
Heterochaeta costata	127	0.536	44.0
Hydrobia ulvae	636	2.69	46.7
Macoma baltica	658	2.78	49.5
Manayunkia aestuarina	2675	11.3	60.8
Nematoda	2292	9.68	70.4
Pygospio elegans	127	0.536	70.9
Scrobicularia plana	127	0.536	71.5
Strebiospio shrubsolii	993	4.19	75.7
Tuplidae	127	0.536	76.2
Tubificoides benedii	5626	23.8	100.0

Table 4.5:
Continued.

5-D	A	% Dom	Cum %
Abra tenius	3163	14.5	14.5
Copopoda	382	1.75	16.3
Cyathura carinata	636	2.91	19.2
Enchytraeidae sp. indet.	5859	26.9	46.1
Etone flava/ longa	127	0.583	46.6
Hediste diversicolor	1486	6.82	53.5
Hydrobia ulvae	318	1.46	54.9
Isotomidae	127	0.583	55.5
Juv. Tellinacea sp.	764	3.51	59.0
Macoma baltica	1125	5.16	64.2
Manayunkia aestuarina	790	3.62	67.8
Nematoda	3864	17.7	85.5
Pygospio elegans	1401	6.43	91.9
Strebiospio shrubsolii	721	3.31	95.3
Tuplidae	169	0.775	96.0
Tubificoides benedii	866	3.97	100.0

5-E	A	% Dom	Cum %
Abra tenius	127	0.575	0.575
Corophium volutator	127	0.575	1.15
Enchytraeidae sp. indet.	13587	61.5	62.7
Etone flava/ longa	255	1.15	63.9
Hediste diversicolor	318	1.44	65.3
Hydrobia ulvae	127	0.575	65.9
Isotomidae	382	1.73	67.6
Juv. Tellinacea sp.	255	1.15	68.8
Macoma baltica	917	4.15	72.9
Manayunkia aestuarina	1783	8.07	80.9
Nematoda	2441	11.1	92.0
Nephtys hombergii	190	0.861	92.9
Paranais litoralis	127	0.575	93.5
Pygospio elegans	679	3.08	96.5
Strebiospio shrubsolii	191	0.865	97.4
Tuplidae	127	0.575	97.9
Tubificoides benedii	445	2.02	100.0

7-C	A	% Dom	Cum %
Abra tenius	255	2.88	2.88
Enchytraeidae sp. indet.	3927	44.37	47.3
Etone flava/ longa	127	1.44	48.7
Hediste diversicolor	1974	22.3	70.9
Heterochaeta costata	191	2.16	73.2
Hydrobia ulvae	191	2.16	75.3
Isotomidae	255	2.88	78.2
Macoma baltica	382	4.32	82.5
Manayunkia aestuarina	127	1.44	83.9
Nematoda	679	7.67	91.6
Pygospio elegans	127	1.44	93.1
Tuplidae	169	1.91	94.9
Tubificoides benedii	446	5.04	100.0

7-D	A	% Dom	Cum %
Abra tenius	1044	3.74	3.74
Cyathura carinata	424	1.52	5.26
Enchytraeidae sp. indet.	16304	58.36	63.6
Etone flava/ longa	127	0.455	64.1
Hediste diversicolor	913	3.27	67.3
Hydrobia ulvae	1114	3.99	71.3
Isotomidae	382	1.37	72.7
Macoma baltica	531	1.90	74.6
Manayunkia aestuarina	560	2.00	76.6
Nematoda	4203	15.0	91.7
Nephtys hombergii	127	0.455	92.1
Pygospio elegans	1613	5.77	97.9
Strebiospio shrubsolii	127	0.455	98.3
Tuplidae	127	0.455	98.8
Tubificoides benedii	339	1.21	100.0

Table 4.5:
Continued.

7-E	A	% Dom	Cum %
Abra tenius	339	10.3	10.3
Enchytraeidae sp. indet.	4	0.122	10.4
Etone flava/ longa	212	6.45	16.9
Hediste diversicolor	127	3.86	20.8
Hydrobia ulvae	255	7.76	28.5
Macoma baltica	1125	34.2	62.7
Nematoda	280	8.52	71.3
Nephtys hombergii	148	4.50	75.8
Pygospio elegans	331	10.1	85.8
Scrobicularia plana	127	3.86	89.7
Strebiospio shrubsolii	127	3.86	93.6
Tubificoides benedii	212	6.45	100.0

CHERRY	A	% Dom	Cum %
Abra tenius	1159	1.20	1.20
Corophium volutator	169	0.175	1.38
Cyathura carinata	255	0.265	1.64
Enchytraeidae sp. indet.	34596	35.92	37.6
Etone flava/ longa	254	0.264	37.8
Hediste diversicolor	1053	1.09	38.9
Hydrobia ulvae	302	0.314	39.2
Macoma baltica	4670	4.85	44.1
Manayunkia aestuarina	3872	4.02	48.1
Nematoda	39844	41.4	89.5
Nephtys hombergii	127	0.132	89.6
Paranais litoralis	254	0.264	89.9
Pygospio elegans	1698	1.76	91.6
Scrobicularia plana	458	0.475	92.1
Strebiospio shrubsolii	127	0.132	92.2
Tubificoides benedii	7490	7.78	100.0

In contrast, the community (as a whole) on the natural mudflats was dominated by enchytraeid oligochaete worms and nematodes, which collectively accounted for 61% of the abundance. Other key species were *Tubificoides benedii*, *Manayunkia aestuarina* (Table 4.6). The biomass was dominated by *H. diversicolor* (34%), which together with the bivalves *M. baltica* and *A. tenius*, account for (64%) of the biomass (Table 4.4). *T. benedii* and *M. aestuarina* accounted for less than 1% of the biomass.

There was also a clear spatial pattern in the community type at the natural site. Low shore communities are characterized by nematodes, enchytraeid worms and *T. benedii*, with *M. baltica*, *H. ulavae*, *P. elegans* and *S. shrubsolli* found at all stations (Table 4.6). The communities in the upper and mid shore differed in that *H. diversicolor* was more abundant and a lower number of bivalves were recorded than from the lower shore.

Table 4.6: Key species in terms of abundance total number of animals found by year inside Paull Holme Strays managed restoration site (collectively), outside Paul Home Strays managed restoration site (collectively) Paul Home Strays managed restoration site and the natural site Cherry Cobb sands.

	INSIDE 2004			**OUTSIDE 2004**	
	Total Abundance	% Dom		Total Abundance	% Dom
Paranais litoralis	3235	30.3	*Enchytraeidae sp. indet.*	6521	36.7
Isotomidae	1413	13.2	*Nematoda*	2117	11.9
Abra tenius	1274	11.9	*Tubificoides benedii*	1589	8.9
Tubificoides benedii	1069	10.0	*Hediste diversicolor*	1441	8.1
Nematoda	840	7.9	*Abra tenius*	843	4.7
			Macoma baltica	755	4.2
			Manayunkia aestuarina	751	4.2

	INSIDE 2005			**OUTSIDE 2005**	
	Total Abundance	% Dom		Total Abundance	% Dom
Isotomidae	3997	41.9	*Enchytraeidae sp. indet.*	6427	27.0
Nematoda	831	8.7	*Nematoda*	3717	15.6
Hediste diversicolor	581	6.1	*Abra tenius*	3674	15.4
Tuplidae	433	4.5	*Pygospio elegans*	1435	6.0
Pygospio elegans	382	4.0	*Tubificoides benedii*	1266	5.3
Sabellidae sp.	382	4.0	*Hediste diversicolor*	1197	5.0
Tubificoides benedii	369	3.9	*Manayunkia aestuarina*	1042	4.3
Paranais litoralis	318	3.3	*Macoma baltica*	875	3.6
Corophium volutator	305	3.2	*Hydrobia ulvae*	750	3.2
Enchytraeidae sp. indet.	297	3.1	*Juv. Tellinacea sp.*	636	2.7

	CHERRYCOBB SANDS 2002	
	Total Abundance	% Dom
Enchytraeidae sp. indet.	34596	35.9
Macoma baltica	4670	4.8
Manayunkia aestuarina	3872	4.0
Nematoda	39844	41.4
Tubificoides benedii	7490	7.8

There has been a general trend of increase in the number of species at stations in the reference site since 2004 (Fig. 4.12). Station 1-a, 1-b showed a significant difference in the number of species based upon month (Table 4.7). There were significantly fewer species found in May ‘05 and July ‘05 ($p< 0.05$) than were found in October ‘04 and January ‘05 at station 1-a (Fig. 4.13). May ‘05 had significantly fewer species than November ‘04 ($p< 0.05$) for station 1-b (Fig. 4.13). At station 2-a August ’04 had significantly fewer species compared to October ’04 ($p< 0.05$), which had significantly fewer species than September ’05 ($p< 0.05$) (Fig. 4.12) There were significantly fewer species present in August ’04 and October ’04 than in September ’05 ($p< 0.05$) at station 2-b (Fig. 4.13). There were significantly fewer species found in August ’04 than in November ’04 (p, 0.05) (Fig. 4.13). For station 3-a August ’04 had fewer species than February ’05, March ’05 and July ’05 ($p< 0.05$), which had, were species than October ’04 ($p< 0.05$) (Fig. 4.13). At station 4-a there were significantly more species found in October ’04 than any other month ($p< 0.05$) (Fig. 4.13). Station 6-a saw significantly fewer species ($p< 0.05$) in March ’05 and May ’05 than in October ’04, November ’04, December ’04, February ’05, July ’05 and September ’05, and October ’04 and November ’04 had significantly fewer species ($p< 0.05$) than December ’04, January ’05, July ’05 and September ’05 (Fig. 4.13).

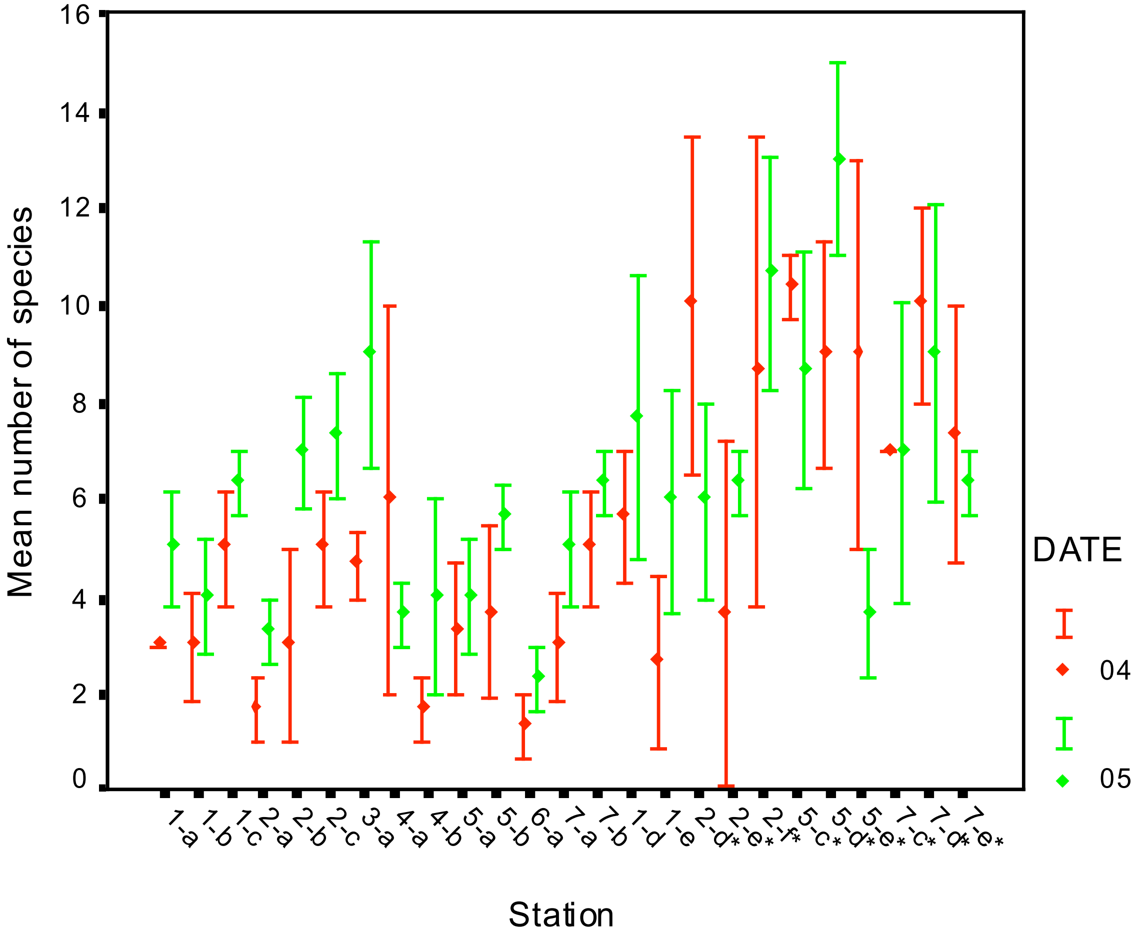


Fig. 4.12: Temporal changes in the number of species present (mean ± SD), between 2004 and 2005 for all stations sampled, inside and outside of Paull Holme Strays restoration site from August 2004- September 2005. * Indicates sample stations located on the natural mudflat.

Table 4.7: One-way ANOVA results from comparisons of species richness per m^2 found per month by station within Paull Holme Strays restoration site from August 2004- September 2005.

Station	F	d.f.
1-A	4.305*	8,16
1-B	3.03**	8,18
1-C	1.01	8,18
2-A	37.5**	2,6
2-B	15.05**	2,6
2-C	3.17*	8,19
3-A	5.93**	9,18
4-A	13.13**	8,17
4-B	1.44	8,19
5-A	1.21	9,21
5-B	1.72	8,17
6-A	9.68**	8,17
7-A	2.01	9,20
7-B	2.74	8,18

* $p < 0.01$; ** $p < 0.05$

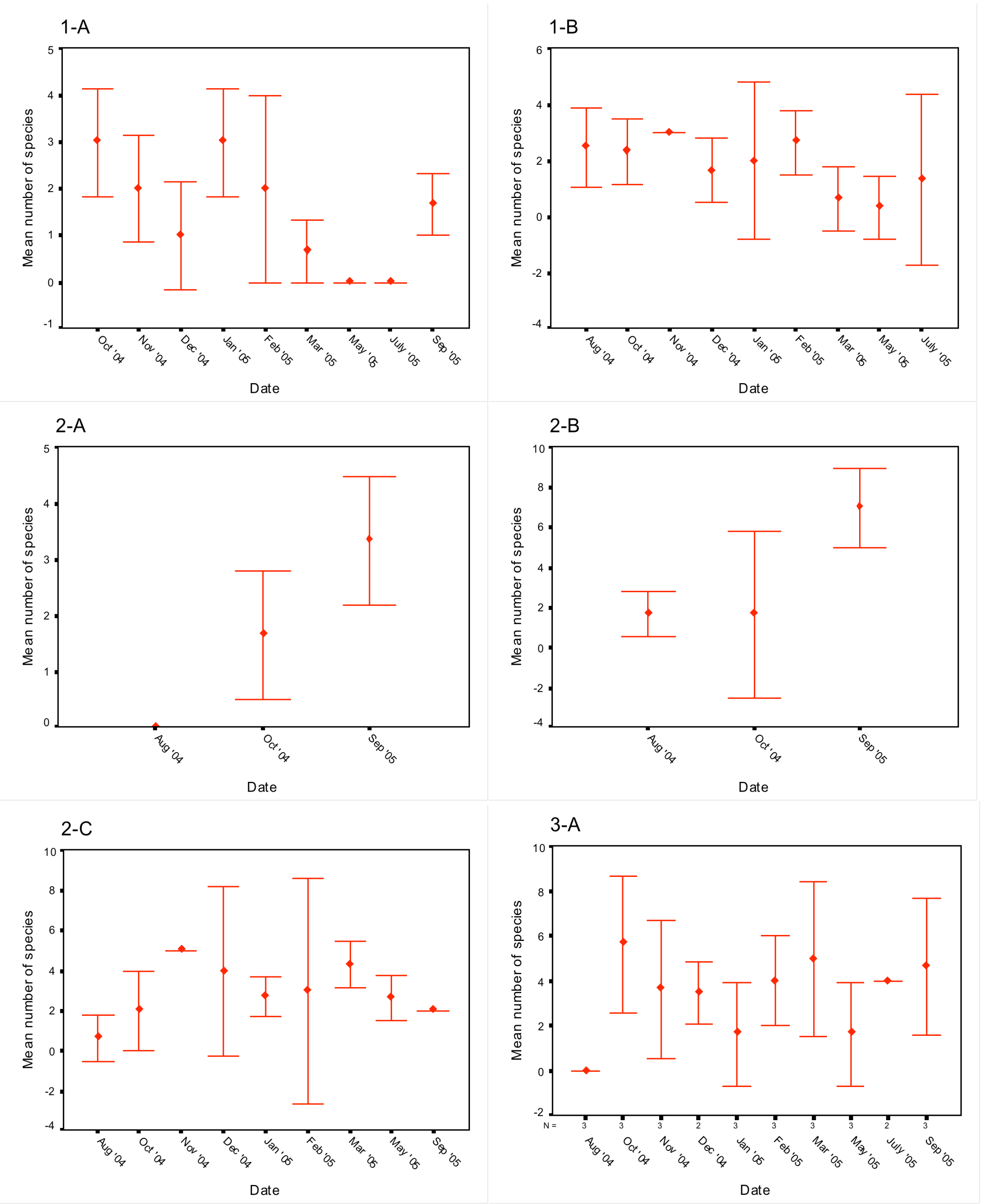


Fig. 4.13: Temporal changes in species richness (mean ± SD) based upon station from inside Paull Holme Strays restoration site. Please note differences in the x- axis based upon station.

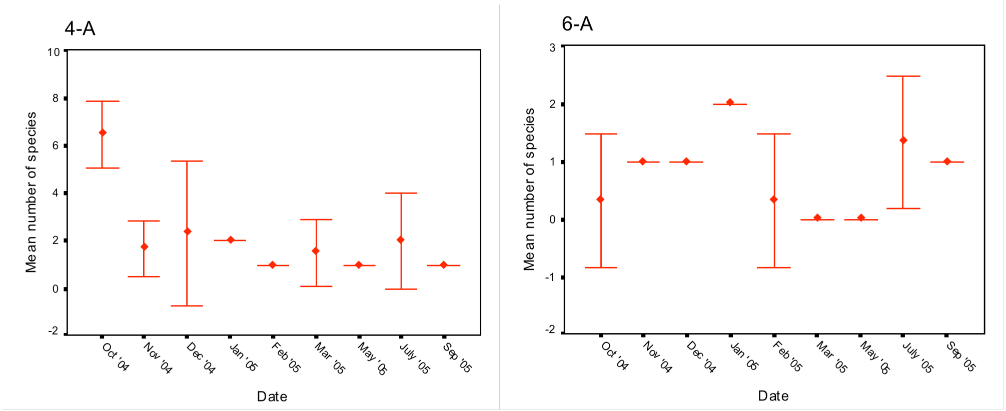


Fig. 4.13: continued

Additionally, within the restoration site significant temporal changes have been seen in abundance (Table 4.8). Station 2-c had significantly ($p < 0.05$) fewer individuals per m^2 in August ’04 and October ’04 than in December ’04, January ’05, February ’05, March ’05, May ’05 and September ’05, which all had significantly ($p < 0.05$) fewer individuals per m^2 than November ’04 (Fig. 4.14). At station 3-a October ’04 had significantly ($p < 0.05$) more individuals per m^2 than any other month sampled (Fig. 4.14). In May ’05 there were significantly ($p < 0.05$) more individuals per m^2 than any other month sampled at station 4-a (Fig 4.14). At station 4-b January ’05 had significantly ($p < 0.05$) more individuals per m^2 than any other month sampled (Fig. 4.14). July ’05 had significantly ($p < 0.05$) more individuals per m^2 than any other month sampled (Fig. 4.14). At station 6-a there were significantly ($p < 0.05$) more individuals per m^2 in September ’05 than any other month sampled. At station 7-b there were significantly ($p < 0.05$) more individuals per m^2 in September ’05 than any other month sampled (Fig. 4.14).

Table 4.8: One-way ANOVA results from comparisons of abundance (individuals per m^2) found per month by station within Paull Holme Strays restoration site between August 2004 and September 2005.

Station	F	d.f.
1-A	0.91	8,16
1-B	1.62	8,18
1-C	1.07	8,18
2-A	4	2,6
2-B	0.88	2,6
2-C	7.15**	8,19
3-A	6.61**	9,18
4-A	3.74*	8,17
4-B	8.56**	8,19
5-A	12.19**	9,21
5-B	0.77	8,17
6-A	13.47**	8,17
7-A	0.81	9,20
7-B	9.33**	8,18

* $p < 0.01$; ** $p<0.05$

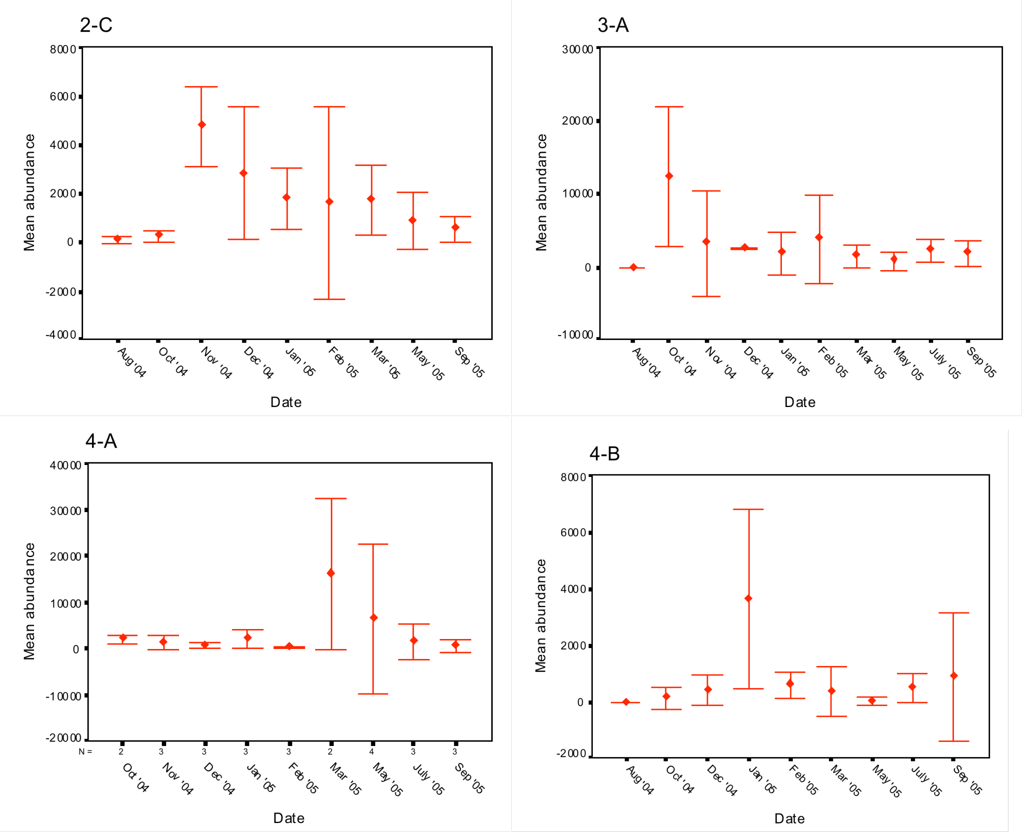


Fig. 4.14: Temporal changes in the abundance (individuals per m^2) (mean ± SD) based upon station from inside Paull Holme Strays restoration site. Please note differences in the x- axis.

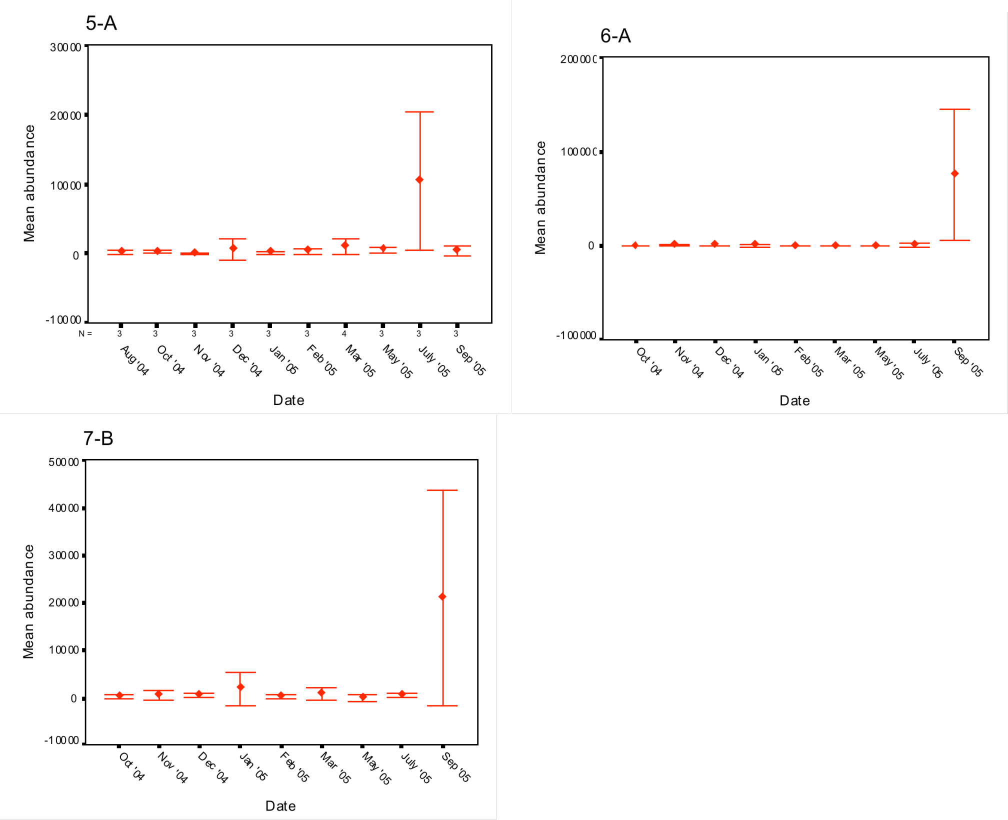


Fig. 4.14: Continued

Within the restoration site significant temporal changes were seen in biomass (Table 4.9). Biomass was significantly higher ($p < 0.05$) in August ’04 than in any other month at station 1-b (Fig. 4.15). At station 1-c July ’05 had significantly ($p < 0.05$) more biomass than any other month sampled (Fig. 4.15). Biomass was significantly ($p < 0.05$) lower in August ’04 and October ’04 than any other month sampled at station 2-b (Fig. 4.15). January ’05 had significantly ($p < 0.05$) more biomass than any other month at station 4-b (Fig. 4.15). Station 6-a had significantly ($p < 0.05$) more biomass in September ’05 than any other month

(Fig. 4.5). September ’05 also saw a significantly ($p < 0.05$) higher biomass at station 7-a when compared to the other months sampled (Fig. 4.15).

Table 4.9: One-way ANOVA results from comparisons of biomass (total weight of individuals m^{-2}) found per month by station within Paull Holme Strays restoration site between August 2004 and September 2005.

Station	F	d.f
1-A	1.94	8,16
1-B	3.55**	8,18
1-C	263.57*	8,18
2-A	2.89	2,6
2-B	7.3*	2,6
2-C	0.66	8,19
3-A	1.08	9,18
4-A	1.23	8,17
4-B	42.05	8,19
5-A	0.5	9,20
5-B	1.07	8,17
6-A	5.32**	8,16
7-A	18.37**	9,20
7-B	2.1	8,18

* $p < 0.01$; ** $p < 0.05$

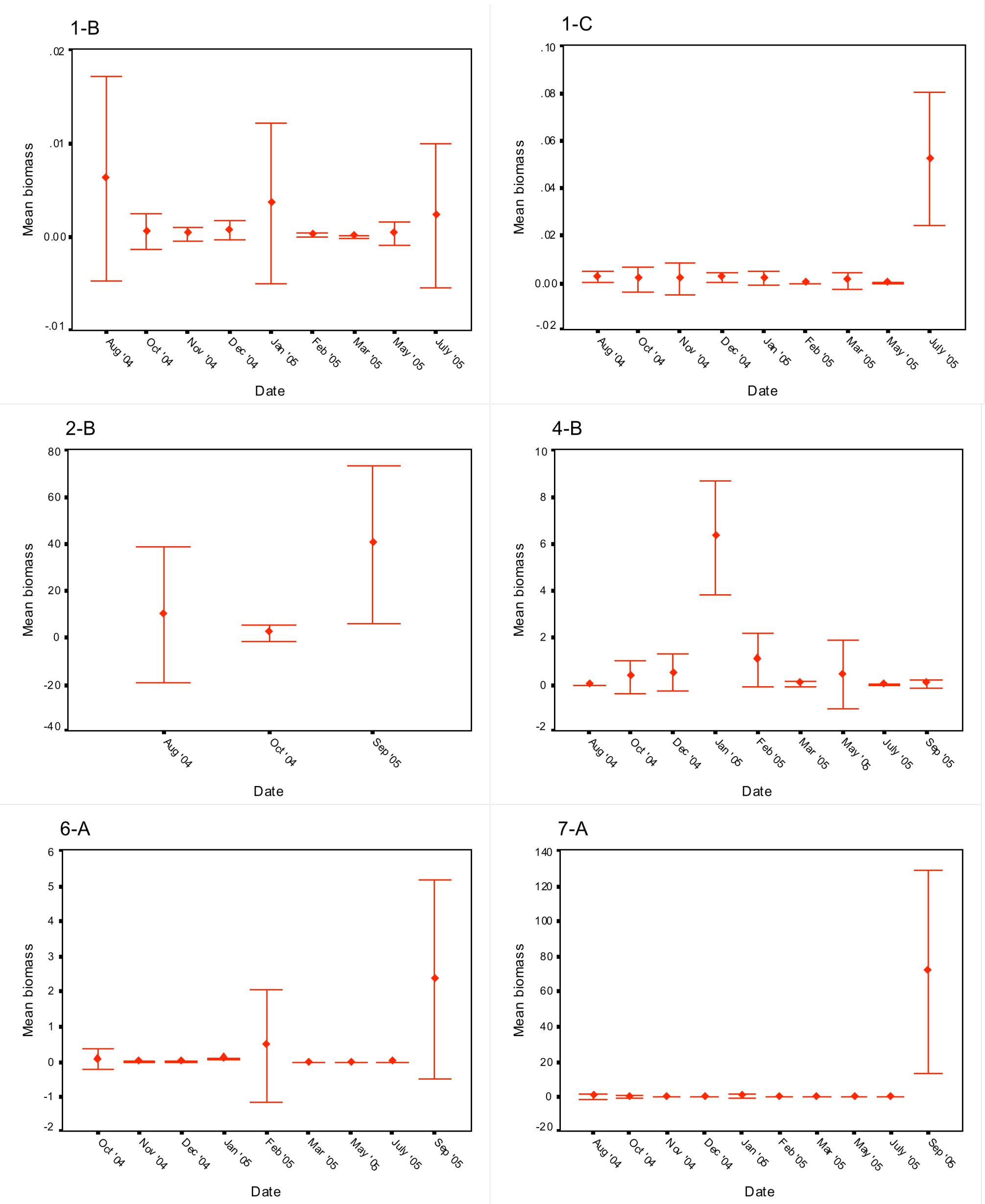


Fig. 4.15: Temporal changes in the biomass (total weight of individuals m^{-2}) (mean ± SD) based upon station from inside Paull Holme Strays restoration site between August 2004 and September 2005. Please note differences in the x-axis.

In the natural habitat, a significant difference was seen in abundance at station 2-e (Table 4.10) were significantly more individuals per m^2 were present in September ’05 ($p< 0.05$). Additionally, there were significant differences in biomass at stations 2-f, 5-d, 7-d and 7-e (Table 4.11). Stations 2-f, 5-d, and 7-d all had significantly more biomass in September ‘05 ($p< 0.05$), while station 7-e had significantly more biomass in October ’04 ($p< 0.05$). Maximum species richness was found at stations in the mid in upper shore in all years sampled, however there was no significant difference found in species richness in any station between month or year (Table 4.12). Considering stations in the restoration site and the natural habitat collectively, there was a significant difference in the number of species found between 2004 and 2005 based upon a Wilcoxon sing rank test ($z= -17.76$, $p< 0.01$) (Fig. 4.16).

Table 4.10: One-way ANOVA results from comparisons of abundance (individuals m^{-2}) found per month by station within the natural habitat (outside of the restoration site at Paull Holme Strays), between August 2004 and September 2005.

Station	F	d.f
1-D	0.72	4,2
1-E	6.16	1,4
2-D	0.43	1,4
2-E	13.46*	1,4
2-F	4.33	1,4
5-C	0.51	1,4
5-D	1.35	1,4
5-E	1.14	1,4
7-C	1.06	1,4
7-D	2.82	1,4
7-E	3.63	1,4

* $p< 0.01$; ** $p<0.05$

Table 4.11: One-way ANOVA results from comparisons of biomass (total weight of individuals m^{-2}) found per month by station within the natural habitat (outside of the restoration site at Paull Holme Strays), between August 2004 and September 2005.

Station	F	d.f.
1-D	10.9	4,2
1-E	0.54	1,4
2-D	1.5	1,4
2-E	1.01	1,4
2-F	17.91*	1,4
5-C	0.33	1,4
5-D	245.57**	1,4
5-E	0.17	1,4
7-C	0.07	1,4
7-D	18.65*	1,4
7-E	34.62**	1,4

* $p < 0.01$; ** $p < 0.05$

Table 4.12: One-way ANOVA results from comparisons of species richness per m^2 found per month by station within the natural habitat (outside of the restoration site at Paull Holme Strays), between August 2004 and September 2005.

Station	F	d.f.
1-D	8.5	4,2
1-E	5.26	1,4
2-D	4	1,4
2-E	1.75	1,4
2-F	0.79	1,4
5-C	1.78	1,4
5-D	8.8	1,4
5-E	6.4	1,4
7-C	1	1,4
7-D	0.14	1,4
7-E	0.14	1,4

* $p < 0.01$; ** $p < 0.05$

Numerically Communities could be divided into two major groups - those inside the restoration site and those found in natural habitat. Furthermore, the communities in the natural habitat could be separated by shore height with the mid and upper shore being the most species rich. Enchytraeidae, nematodes, *Hydrobia ulavae* and *Hedistie diversicolor* dominated the mid and upper shore of the natural community. In the lower shore nematodes and Enchytraediae were still dominant but the importance of bivalves increased (Table 4.5). In comparison the community in the restored habitat has changed considerably over the sampling period, with the most noticeable difference being the decrease in *Paranais litoralis*. In 2004 it accounted for 30% of the total species abundance and in 2005 it only accounted for 3% of the total species abundance (Table 4.6).

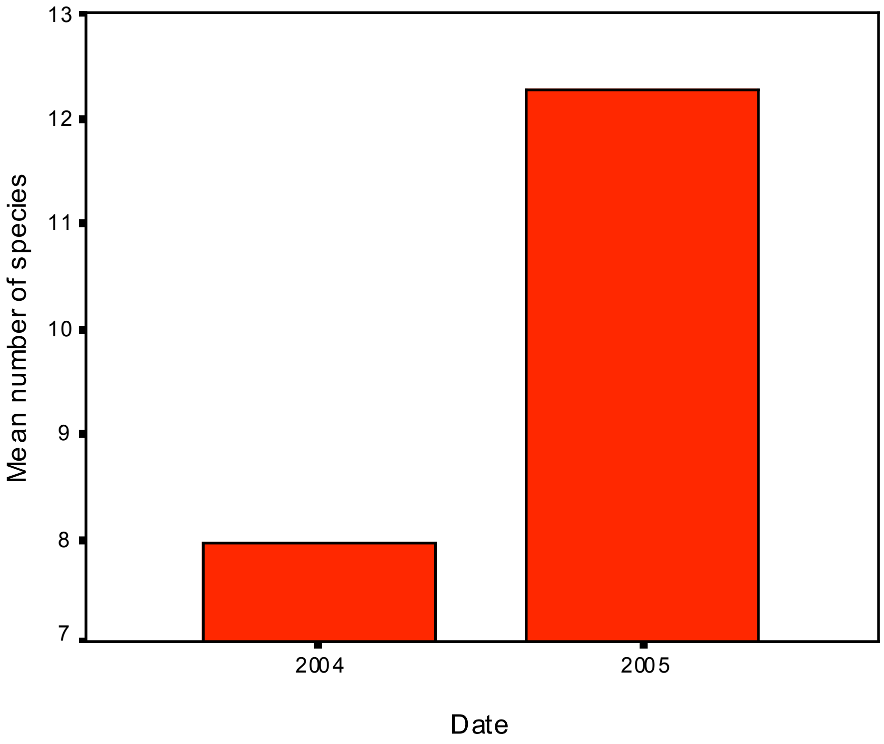


Fig. 4.16: Temporal changes in the mean number of species present between 2004 and 2005 for all stations sampled both inside and outside the restoration site at Paull Holme Strays.

4.4. Discussion

The macrofaunal communities outside of the managed realignment site and at Cherry Cobb can be considered typical of the middle estuarine area, being composed of a large number of relatively small organisms of a very few species. The intertidal communities of this area of the Humber estuary have been studied by various authors (Allen and Elliott, 1999; Mortimer *et al.*, 1999; Allen, 2000; Mazik and Elliott, 2000; Widdows *et al*., 2000) with findings being broadly consistent to those of the present study, in terms of species composition and abundances (Table 4.3). While the macrofaunal communities within the realignment site are still considered to be in the early stages of development with the number of species, abundance, biomass and diversity remaining low though 2005.

Warwick (1990) stated that one, or a few large species, each represented by a few individuals characterize unstressed or stable environments. Whilst these species are rarely the numerical dominants in marine or estuarine communities, they are dominant in terms of the biomass. In contrast, stressed communities are characterized by high numbers of short lived *r*- strategists with small body size, a high reproductive capacity and a variable population size. The infaunal communities on the established mudflats have remained relatively stable (Table 4.13) over the sampling period when looking at species composition, number of species, abundance and diversity. The communities of the developing mudflats on the restoration site on the other hand, have changed considerably over the past two years they started out highly unstable. With time the species richness, total biomass and total abundance have all increased gradually. It is important to note that there was a large decrease in the number of *Paranais litoralis* between 2004 and 2005, which should have led to a decrease in the total abundance in 2005 but due to a large influx of terrestrial organisms this was not seen.

Table 4.13: Coefficients of variation for biological parameters for each station sampled both inside and outside Paull Holme Strays restoration site. Stations are separated by location.

INSIDE

	S	A	B
1-A	93%	216%	165%
1-B	63%	132%	225%
1-C	75%	116%	257%
2-A	89%	168%	112%
2-B	84%	133%	119%
2-C	54%	93%	105%
2-D	37%	95%	84%
2-E	47%	52%	74%
2-F	32%	61%	81%
3-A	59%	129%	126%
4-A	78%	172%	280%
4-B	81%	163%	198%
5-A	62%	233%	168%
5-B	68%	110%	255%
6-A	74%	268%	228%
7-A	71%	225%	328%
7-B	59%	240%	200%

OUTSIDE

	S	A	B
1-D	40%	60%	9%
1-E	55%	127%	100%
2-D	37%	95%	84%
2-E	47%	52%	74%
5-C	16%	62%	54%
5-D	18%	182%	56%
7-C	23%	54%	33%
7-D	21%	116%	465%
7-E	28%	50%	83%

REFERENCE

	S	A	B
CHERRY	37%	111%	NA

*S= number of species; A= total abundance/ m^2; B= biomass (g m$^{2)}$

Paranais litoralis is an opportunistic species with a swimming adult dispersal phase and is often an early colonizer of disturbed or newly formed habitats (Talley and Levin, 1999; Levington and Kurdziel, 2000; Stocks and Grassle, 2003). *Hediste diversicolor* lives for about two years, and dies once its spawns (monotelic). *Hediste* larvae are free swimming, but are said to not enter the plankton, instead they remain near the substratum to avoid being swept out to sea. Bartels-Hardege and Zeeck (1990) have stated that spawning in the wild occurs in the spring. *Corophium volutator* live for approximately 6 months. Females carry young in a brood pouch until they are ready to be released. In one season a female may produce as many as 5 broods, and as the young are liberated they form small side burrows off of the parental one until they are large enough to inhabit their own (Budd, 2007; Little, 2000). *Macoma balthica* is an active suspension feeding species, that is a biodestabilizer (Widdows, 2000). They breed annually (iteroparous) with breeding occurring from spring to fall (Little, 2000; Caddy, 1967). *Hydrobia ulvae* is a short-lived species with only one breeding period (semelparous). At the end of their life they produce approximately 40 eggs that hatch into planktonic veligers, before settling and metamorphosing into snails (Jackson, 2000; Little, 2000).

Talley and Levin (1999) found that *P. litoralis* characterizes the early successional stages of community development whereas enchytraeid and tubificid oligochaetes are more characteristic of older, established communities, such as those found outside of the managed realignment site. This is consistent with the findings of this study. Prenda and Gallardo (1992) also noted *P. litoralis* is predominantly found in the sediments with high silt and organic content, subjected to low current speeds. This may, in part, explain the distribution of *P.*

litoralis within the managed realignment site where it was found in the upper shore area and the eastern lower shore area.

Community structure inside the restoration site has also changed considerably over time with characteristic estuarine species such as *Hediste diversicolor* and *Hydrobia ulvae* increasing in abundance and becoming more dominant. As this shift in species occurs the restoration site is becoming more and more like the established mudflats, resulting in the habitats within the restoration site becoming more stable. However, terrestrial organisms are still present in high numbers at most stations inside the managed realignment site. There does not appear to be any temporal trends in their distribution. Perhaps this is because many of the sampling stations are in close proximity to the floodwalls, thus making it likely that that they move between the mudflats and their terrestrial habitat during low tide.

Colonization appears to be the most rapid close to the breach sites (Table 4.14) in the East and West where the communities most closely resemble those on the upper shore outside of the site and Cherry Cobb. This includes stations 2-B, 2-C and 3-A in the west and station 7-B, which is directly opposite the eastern breach. Since a large borrow pit was present directly opposite the western breach (Fig. 2. 6), a large area in which maximum colonization and community development may be taking place could not be sampled [in the past]. But by summer of 2005, the rate of accretion in the north west of this site has increased significantly (Fig. 2.15), filling in this borrow pit and allowing parts of it to be exposed mudflat at high water. During sampling it has been noted that at low water birds congregate on this newly exposed mudflat for feeding. As a result characterization of the benthic communities in this area is recommended.

Table 4.14: Mean rate of colonization of benthic invertebrates per month for sampling stations within the restoration site. Rate of colonization equals the mean number of individuals added to each station per m^2.

	Rate of colonization
1-A	-469
1-B	3
1-C	106
2-A	404
2-B	1380
2-C	21
3-A	-67
4-A	-175
4-B	19
5-A	28
5-B	69
6-A	9522
7-A	142
7-B	2580

Community development and the distribution of certain species is related to tidal inundation and, to a lesser extent accretion with colonization being greatest in areas of moderate accretion but being limited in areas of high accretion (Mazik, 2006). However this is an assumption based upon accretion and tidal inundation data collected from a limited number of sites (7) in 2005/ 2006 by Smith (2007). Smith (2007) used a Spearman's rank correlation to show significant positive correlations between tidal inundation and the number of species and total abundance. Additionally, the distribution of nematodes, Spionidae, Enchytraeidae, *Macoma balthica* and *Hediste diversicolor* were positively correlated with tidal inundation. Due to sedimentation being so rapid in the northwestern area of the restoration site (Fig 2.15), the mudflat in this area has built up to such an extent that inundation does not occur on every tide. It is not surprising then, that species typical of this area of the Humber estuary (Enchytraeidae, *H. diversicolor*) are present in only low numbers.

Despite monitoring community development since the spring after breaching, the community inside of the restoration site is not fully stable or fully developed when compared to the communities of the natural sites; abundance and diversity are also considered to be low when compared to these established mudflats. In comparison to other studies, colonization and establishment as shown by univariate indices has been slow. For example, Bolam and Whomersley (2005) studied colonization of dredged material used for mudflat enhancement at three sites in the south east of England. They suggested that, following defaunation, low diversity mudflat communities composed largely of opportunistic species could become established within 3-12 months as shown by the community univariate properties. A community structure comparable to that prior to defaunation or to that of a reference site could be achieved within two years. On the other hand, Levien *et al*. (1996) found that, as a whole, macrofaunal communities on artificially created saltmarsh could regain early successional characteristics by around four years after initial colonization. The rate of infaunal colonization was found to be greatest in the lower marsh/ upper mudflat, which is comparable to the results of this study (Table 4.14). While macrofaunal densities and diversity resembled those of natural marsh relatively quickly, community structure did not. This was attributed to differences in sediment characteristics. In addition, Craft *et al.* (1999) stated that restoration of ecological features such as secondary production, habitat and species diversity generally does not occur within the first decade. But, results of this study have shown a significant increase in species richness over the first two years of habitat development, which suggests that it will not take decades for community structure and secondary production to occur.

Chaper 5

Abundance and behaviour of wader and waterfowl assemblages in the restored tidal wetlands of Paull Holme Strays

5.1. Introduction

Coastal wetlands are unique habitats that serve numerous ecological roles. Annual use of coastal wetlands by tens of thousands of shorebirds has demonstrated the importance of these habitats as foraging and roosting areas. The fulfilment of both structural (e.g. shorebird density, species diversity) and functional (e.g. behaviour) roles is a critical aspect of the ecological and aesthetic values of these habitats (Millennium Ecosystem Assessment, 2005; Zedler, 1996).

Extensive global loss of wetland reduces the integrity of migratory flyways and the quality of wintering habitat (Zedler, 1996). As with many estuaries, the Humber estuary's wetlands have been reduced in area significantly over the past 2000 years for agriculture and urban development (French, 2006; McLusky and Elliott 2004; Humber Estuary and Coast, 1994). Consequently, wetland restoration has become an important tool to stop the damage to ecosystem functions resulting from habitat loss, although the creation of functional habitat and the assessment of restoration success remain challenging tasks (Zedler and Lindig-Cisneros, 2000; Elliott *et al*., 2007). The highly mobile nature of birds facilitates rapid colonization of restored habitats (Hemesath and Dinsmore, 1993;

Passell, 2000). However, bird density and diversity, though simple and frequently used means to assess some structural attributes of restored sites (e.g., LaGrange and Dinsmore, 1989; Ratti *et al*., 2001), does not necessarily represent the quality of the ecological functions of the restored habitat. High abundance may stem from the presence of opportunistic species (e.g., gulls); species that thrive in degraded or edge habitats, while wetland species, often targets of restoration, remain absent from the restored sites (Melven and Webb, 1998). When assessing habitat restoration for shorebirds, measurements of species richness within this specific target group may provide a more accurate indication of restoration success. Additionally, success can be measured by looking at the specific needs of a target species or assemblage. When looking at the needs of a target species or assemblage both habitat type and prey availability/ quality (diversity and size) need to be taken into account. Each target species/ assemblage has specific needs so by looking at a matrix of physical and biological parameters of a restoration site one can assign a value based on the needs of a target species or assemblage.

Direct observations of wetland bird behaviour can yield a broader picture of bird use of restored sites by incorporating functional attributes such foraging and other activities into the assessment of restoration success. However, these critical aspects of bird responses to habitat restoration are rarely documented (Brusati *et al*., 2001). In addition, habitat selection by wetland birds varies among vegetated, tidal flat and submerged habitat types based on wetland bird activity: foraging typically takes place on tidal flats, whereas stands of vegetation at higher elevations are usually used for roosts (Warnock and Takekawa, 1995). Thus, the evaluation of wetland bird use of different habitat types for feeding, roosting, breeding, may be a better indicator of the functional quality of restored sites.

Understanding how to establish functional habitat for wetland birds requires identification of habitat characteristics that influence bird density, diversity, and behaviour (George and Zack, 2001). Wetland birds tend to prefer habitats with open water or mudflats (Warnock and Takekawa, 1995; Long and Ralph, 2001) or areas with diverse habitat types (Craig and Beal, 1992). Restored sites my initially provide large expanses of unvegetated habitat that attracts wetland birds, although this open space often diminishes over time as plants become established. In addition, the degree of human impact within the larger landscape matrix is often negatively related to bird density and diversity (Craig and Beal, 1992; Fairbairn and Dinsmore, 20001; George and Zack, 2001; Marzluff and Ewing, 2001); however, restored sites are frequently established at the edge of natural habitats or in disturbed areas in close proximity to man-made structures.

The aims of this chapter are to determine whether wetland bird density, diversity, behaviour and habitat use were similar between reference and the restored mudflats at Paull Holme Strays and to evaluate if, after Paull Holme Strays was breached, the wetland bird population on the existing mud flat remained consistent. It is hypothesised that the restoration of the wetland habitat at Paull Holme Strays will increases the numbers of wetland birds using the middle Humber estuary.

5.2. Materials and methods

5.2.1. Field Methods

Surveys were conducted over a ~300 ha wetland area in the middle region of the Humber Estuary (Fig. 5.1). A restored site (Paull Holme Strays) (Fig. 5.2) was created in 2003 from a disturbed open space that had been blocked off to tidal flow and farmed until 2002. The site was reopened to tidal flow in 2003. In 2003, a ~60 ha survey plot was designated that encompassed most of the area of the restored site. Survey plots of approximately the same size were established in two reference locations one adjacent to the restored site on the existing mudflat and the other to the west of the restored site at Saltend (Fig. 5.1). These reference sites were chosen for their similarity in habitat type to the restored site. All three sites are similar in that they are all mudflats within 1.5 km of one and other and they all experience the same tidal regime.

Fig. 5.1: Saltend to Paull Holme Strays on the North side of the middle Humber Estuary. The distance between Paull Holme Strays and Saltend is 1.5 KM.
*Map modified from gogleMaps.com

Surveys for bird abundance and behaviour were performed monthly from January 2002 to December 2002 at the reference site located just outside of the restored habitat at Paull Holme Strays by IECS (Institute of Estuarine and Coastal Studies, University of Hull, UK). This data was used to look to consider if the restoration of Paull Holme Strays increased the middle Humber estuary's bird carrying capacity or if it caused a shift in the location of the birds.

Paull Holme Strays was monitored for bird abundance and behaviour at both restored and reference sites from June 2004 to June 2005 as was the reference site Saltend (the reference site at Saltend was monitored by IECS). All surveys were conducted using a Kowa TSN3 spotting telescope with 25x and 60x eyepieces, and Alpina 10x binoculars from a distance greater than 50 m to minimize bird disturbance. A series of 3 counts undertaken on each survey date, at approximately low, mid and high water at the reference and restored sites at Paull Holme Strays, while surveys at the reference site at Paull Holme Strays prior to the breach and at Saltend consisted of a single count at of peak abundance (maximum number of birds on the site) over the tidal cycle. Each survey consisted of a single scan of the study area to document the density, behaviour and habitat use of all waders and waterfowl. Behaviour was separated into two categories: foraging (both searching and handling food) and roosting (Brusati *et al*., 2001). Birds flying over the site and birds that arrived on the site during a survey were not included in the censuses.

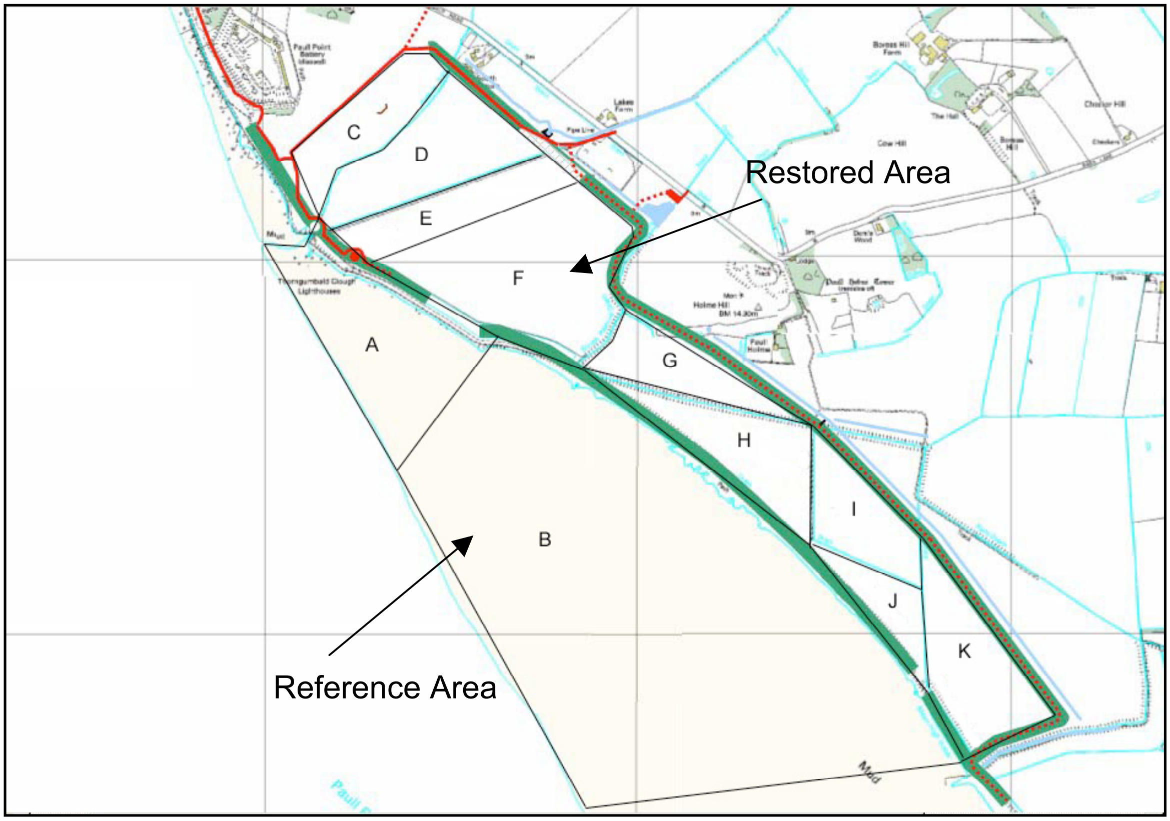


Fig. 5.2: Zones for ornithological surveys at Paull Holme Strays. Zones A and B are mudflats that existed prior to the breach in 2003. Zones C through K are all within the managed realignment site. ** Map modified from IECS map of PHS

5.2.2 Data Analysis

All birds observed using a site during a count were included in the calculation of the Shannon- Wiener species diversity index (H'), which is sensitive to changes in rare species, where:

$H' = \sum(p_i)(\ln p_i)$

and p_i is the proportion of the bird community belonging to the *i*th species (Krebs 1994). Species evenness, E (the equitability of abundance among species) was calculated for each site, where:

E= average H' per site/ H'_{max}.

The maximum species diversity is given by $H'_{max} = \ln S$, where S was the total number of species observed (Krebs, 1994). Species richness is the total number individuals of species observed in each site. For detailed analyses of behaviour and abundance, the study focussed on the most common taxa: Shelduck (*Tadorna tadorna*), Teal (*Anas crecca*), Mallard (*Anas platyrhynchos*), Golden Plover (*Pluvialis apricaria*), Redshank (*Tringa tetanus*), Dunlin (*Calidris alpina*), and Curlew (*Numenius arquata*).

Species diversity and the density (birds/ ha) of these species were analyzed using one-way ANOVAs, where the factor was reference vs. restored status time was used as the replicate as only one count was carried out per sampling occasion (Brawley *ed al.,* 1998; Armitage, 2003). A Levenes's test was used to test for homogeneity of variance and in cases where homogeneity of variance could not be achieved through transformation of the data; a Kruskal-Wallis test was used.

Behaviour was analysed separately for each species with two-factor ANOVAs, where the dependent variable was the number of birds observed engaging in each activity, and the independent variables were site and tidal height. When individual factors were found to be significant interactions were tested for by a Tukey's HSD test.

5.3. Results

A total of 25 species and maximum density of 1793 individuals per hectare was detected when looking at all of the monitoring stations collectively between June 2004 and June 2005. Between 16 and 23 species were recorded at each of the four monitoring stations (Paull Holme Strays restored habitat, Saltend, Paull Holme Strays existing habitat, Paull Holme Strays existing habitat in 2002) with the greatest species richness (23 species) occurring in the restored habitat at Paull Holme Strays (Table 5.1). Of those 23 species golden plover make up 39% of the abundance per hectare (Fig. 5.3). Wader and waterfowl species diversity was highest in the reference habitat at Paull Holme Strays post breach H’ = 2.51 (Fig. 5.4). Both species diversity and richness increased in this reference area post breach.

Table 5.1: Mean abundance of waders and waterfowl (number per ha) (SE) in each site monitored. Restored Paull, Post-breach reference Paull and Saltend were monitored from June 2004 to June 2005. Pre-breach Paull was monitored from January 2002 to December 2002.

	Restored Paull	Post-breach reference Paull	Pre-breach reference Paull	Salt-end
Avocet	.53 (.07)	.01 (.01)	-	.21 (.1)
Recurvirostra avosetta				
Black-headed gull	1.37 (.44)	-	1.00 (.82)	-
Larus ridibundus				
Bar-tailed godwit	.04 (.01)	-	.04 (.01)	-
Limosa lapponica				
Black-tailed godwit	1.90 (.62)	.11 (.03)	.13 (.10)	1.00 (.41)
Limosa limosa				
Common gull	.15 (.05)	.08 (.01)	.08 (.06)	-
Larus canus				
Common sandpiper	-	-	-	.01 (.005)
Actitis hypoleucos				
Cormorant	.08 (.01)	.02 (.003)	-	-
Phalacrocorax carbo				
Curlew	.51 (.14)	.30 (.08)	.42 (.06)	2.40 (.37)
Numenius arquata				
Dunlin	1.10 (.58)	1.60 (.86)	4.20 (1.29)	4.20 (.88)
Calidris alpina				
Greater black-backed gull	-	-	.086 (.07)	-
Larus marinus				
Golden plover	26.68 (13.53)	14.60 (7.09)	41.06 (17.89)	42.58 (24.48)
Pluvialis apricaria				
Greenshank	.05 (.01)	.22 (.21)	-	-
Tringa nebularia				

Table 5-1: Continued.

	Restored Paull	Post-breach reference Paull	Pre-breach reference Paull	Salt-end
Grey Heron *Ardea cinerea*	.03 (.006)	.02 (.01)	-	-
Grey Plover *Pluvialis squatarola*	17.55 (17.04)	.29 (.22)	.15 (.15)	.26 (.23)
Knot *Calidris canutus*	.69 (.65)	1.74 (1.15)	.44 (.34)	.32 (.11)
Lapwing *Vanellus vanellus*	7.20 (5.37)	1.78 (.74)	-	13.51 (4.37)
Mallard *Anas platyrhynchos*	.2 (.06)	.24 (.07)	.34 (.12)	.77 (.17)
Mute swan *Cygnus olor*	.04 (.02)	-	-	-
Oystercatcher *Haematopus ostralegus*	.09 (.04)	.11 (.04)	.04 (.01)	.05 (.02)
Redshank *Tringa erythropus*	.38 (.09)	.75 (.18)	1.15 (.59)	3.92 (.82)
Ringed Plover *Charadrius dubius*	.36 (.17)	.07 (.02)	.08 (.04)	1.04 (.18)
Shelduck *Tadorna tadorna*	1.14 (.23)	.58 (.17)	.34 (.06)	.82 (.19)
Teal *Anas crecca*	2.14 (.65)	-	-	2.17 (.73)
Turnstone *Arenaria interpres*	.05 (.01)	.03 (.01)	.02 (.01)	.02 (.006)
Wigeon *Anas penelope*	.84 (.50)	.06 (.01)	-	-

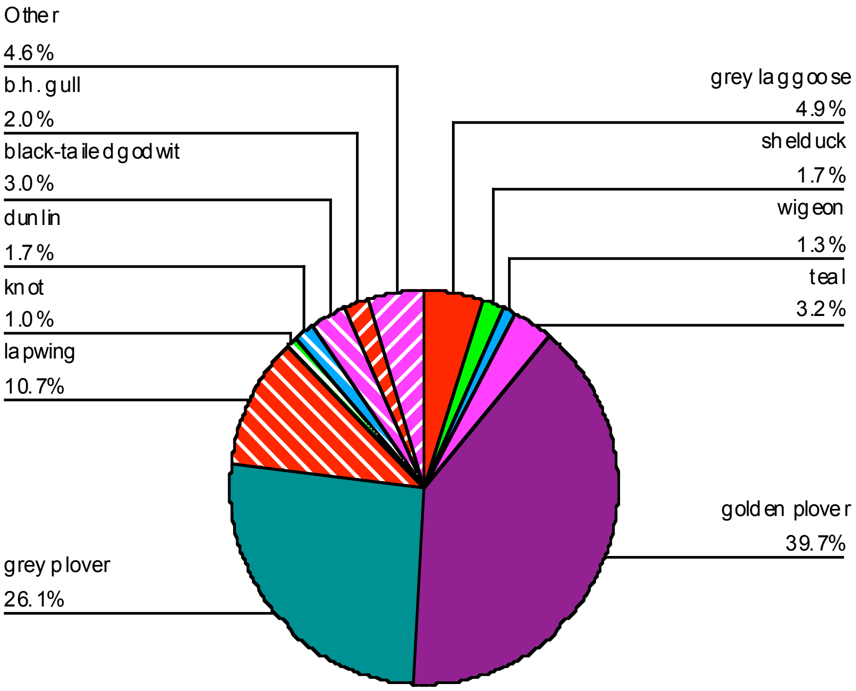


Fig 5.3: Pie graph showing the percent abundance per hectare of key species found inside the restoration site at Paull Holme Strays between June 2004 and June 2005.

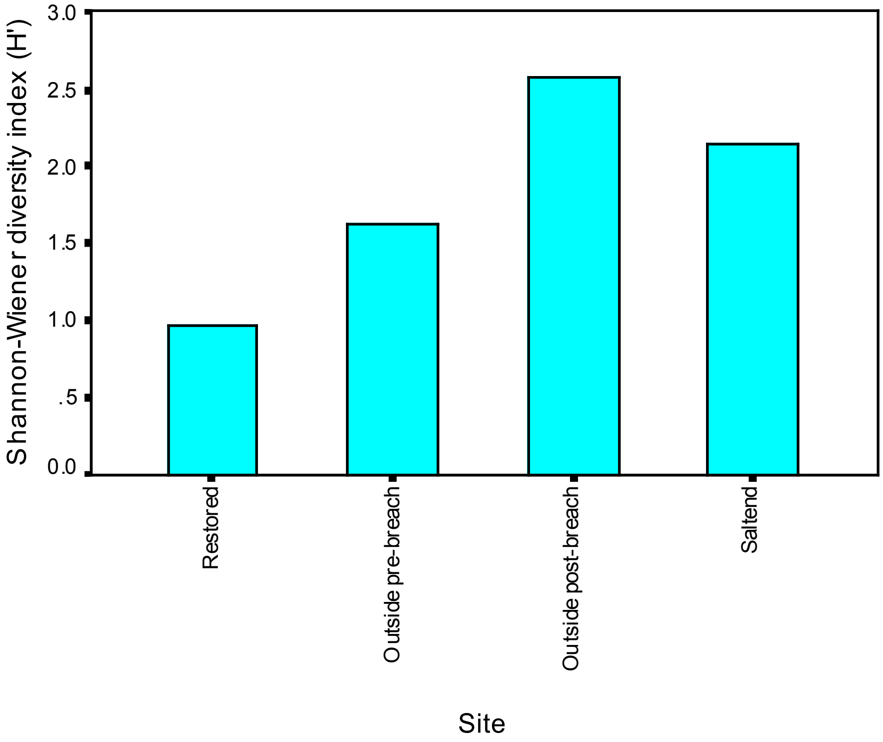


Fig. 5.4: Shannon-Wiener species diversity index for waders and waterfowl in reference and restored sites. The restored habitat at Paull Holme Strays, Post-breach reference habitat at Paull Holme Strays and Saltend were monitored from June 2004 to June 2005. The pre-breach habitat at Paull was monitored from January 2002 to December 2002. Higher values represent higher diversity.

There are seasonal trends in bird use of the restoration site at Paull Holme Strays with peak numbers of birds using the site in winter (Fig. 5.5). The trends in bird use of the restoration site follow those of the natural habitats monitored again with peak bird use occurring in winter (Fig. 5.5), however in the natural habitat there is a summer time peak due to a large influx of golden plover

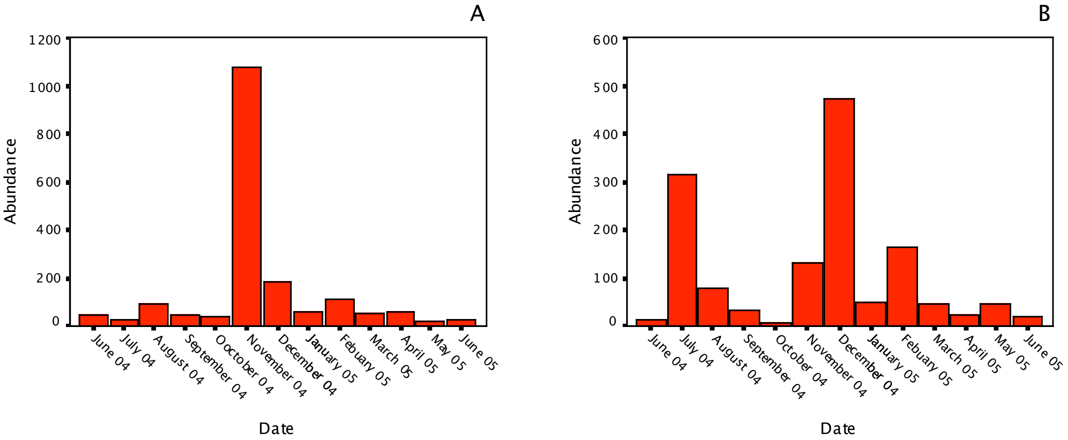


Fig. 5.5: Mean abundance of birds per month as monitored from June 2004 to June 2005, A) from the restoration site at Paull Holme Strays and B) from the natural habitat adjacent to the restoration site at Paull Holme Strays.

Differences in the abundance of waders and waterfowl at the four monitoring stations (Paull Holme Strays restored habitat, Saltend, Paull Holme Strays existing habitat, Paull Holme Strays existing habitat in 2002) were not significant (f= 2.04; d.f = 3, 616; p> 0.05). However when the key species used to look at behaviour at each of the four monitoring stations were looked at Golden Plover (f= 0.47; d.f.= 3,28; p> 0.05) and Teal (f= 0.49; d.f.= 3,16; p> 0.05) showed that both location and restoration status did not effect their density. Curlew (f= 22.80; d.f.= 3,47; p< 0.01), Dunlin (f= 7.07; d.f.= 3,39; p< 0.01), Mallard (f= 5.29; d.f.= 3,41; p< 0.01), Redshank (f= 10.31; d.f.= 3,42; p< 0.01) and Shelduck (f= 3.23; d.f.= 3,46; p< 0.05) all had significantly different densities based upon location, showing that restoration status/ habitat quality can effect density (Fig. 5.6). Curlew, Dunlin, Mallard and Redshank densities were all highest in the reference site at Saltend. Shelduck densities at the reference site at Paull prior to the breach were lower than at all of the other reference sites and the restored site.

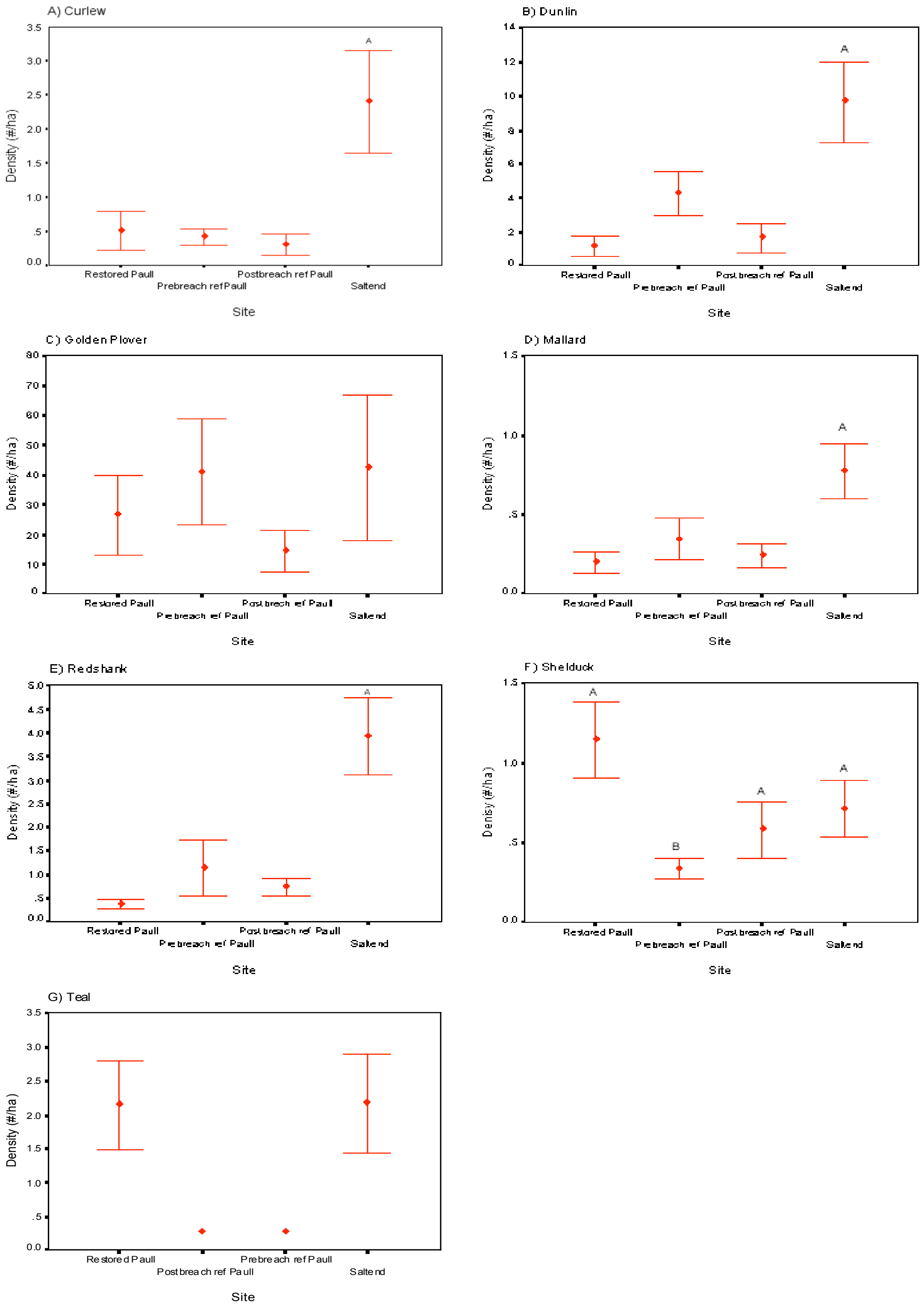


Fig. 5.6: Wader and waterfowl densities at the restored habitat at Paull Holme Strays, Post-breach reference habitat at Paull Holme Strays and Saltend, which were monitored from June 2004 to June 2005 and the pre-breach habitat at Paull which was monitored from January 2002 to December 2002. Note different y- axes. Letters indicate a significant difference in density p<0.05 as determined by a Tukey's HSD.

Of the two behaviours looked at feeding was the most common activity observed in the monitoring stations (Paull Holme Strays restored habitat and Paull Holme Strays existing habitat) with over 60% of Curlew, Dunlin, Redshank, Shelduck and Teal foraging when observed between June 2004 and June 2005 (Table 5.2). Tidal state on its own had no effect on the behaviour of the Shelduck, Teal, Mallard, Golden Plover, Redshank, Dunlin and Curlew (Table 5.2). But the behaviour of Shelduck, Teal, Golden Plover and Dunlin all varied by site (Fig. 5.7). Shelduck were found feed significantly more of the time ($p < 0.05$) when inside of the restoration site at Paull Holme Strays. Teal were found to spend the majority of their time feeding ($p < 0.05$) regardless of location, showing that restoration status/ habitat quality does not affect their behaviour. Golden Plover were found to split their time between roosting and feeding but significantly fewer ($p < 0.05$) Golden Plover were found feeding in the restoration site at Paull Holme Strays. Of the sites monitored Dunlin were found to spend significantly ($p < 0.05$) more time roosting in the restored habitat at Paull Holme Strays. Curlew behaviour was influence by a combination of site and tidal height (Fig. 5.8). Both Shelduck and Teal showed a preference for feeding regardless of restoration status. Golden Plover showed a preference for roosting regardless of restoration status. Dunlin showed a preference of roosting in the restored habitat. Curlew showed a tendency to change from the reference to the restored site at mid tide for feeding.

Table 5.2: A) Percentage of birds (SE) observed exhibiting each behaviour across all monitoring sites between June 2004 and June 2005. B) *F*- values from a two-factor ANOVA of tidal height and monitoring site on bird activity from data collected between June 2004 and June 2005.

A)

Taxa	Feeding	Roosting
Curlew	60.5(14.0)	39.5 (14.0)
Dunlin	85.8 (13.5)	14.3 (13.5)
Golden Plover	30.5 (23.6)	69.5 (23.5)
Mallard	43.5 (19.5)	56.5 (19.5)
Redshank	65.2 (22.5)	34.8 (22.5)
Shelduck	69.8 (10.4)	22.2 (10.4)
Teal	87.2 (10.1)	12.8 (10.1)

b)

Taxa	Tide	Site	Tide x site
Shelduck	0.756	5.54**	0.842
Teal	0.829	6.38*	1.63
Mallard	0.149	1.38	0.721
Golden Plover	0.405	2.81***	0.316
Dunlin	0.616	3.11***	1.43
Curlew	2.31	0.104	2.31***
Redshank	0.589	0.137	0.726

* $p < 0.0001$; ** $p < 0.001$, *** $p < 0.05$

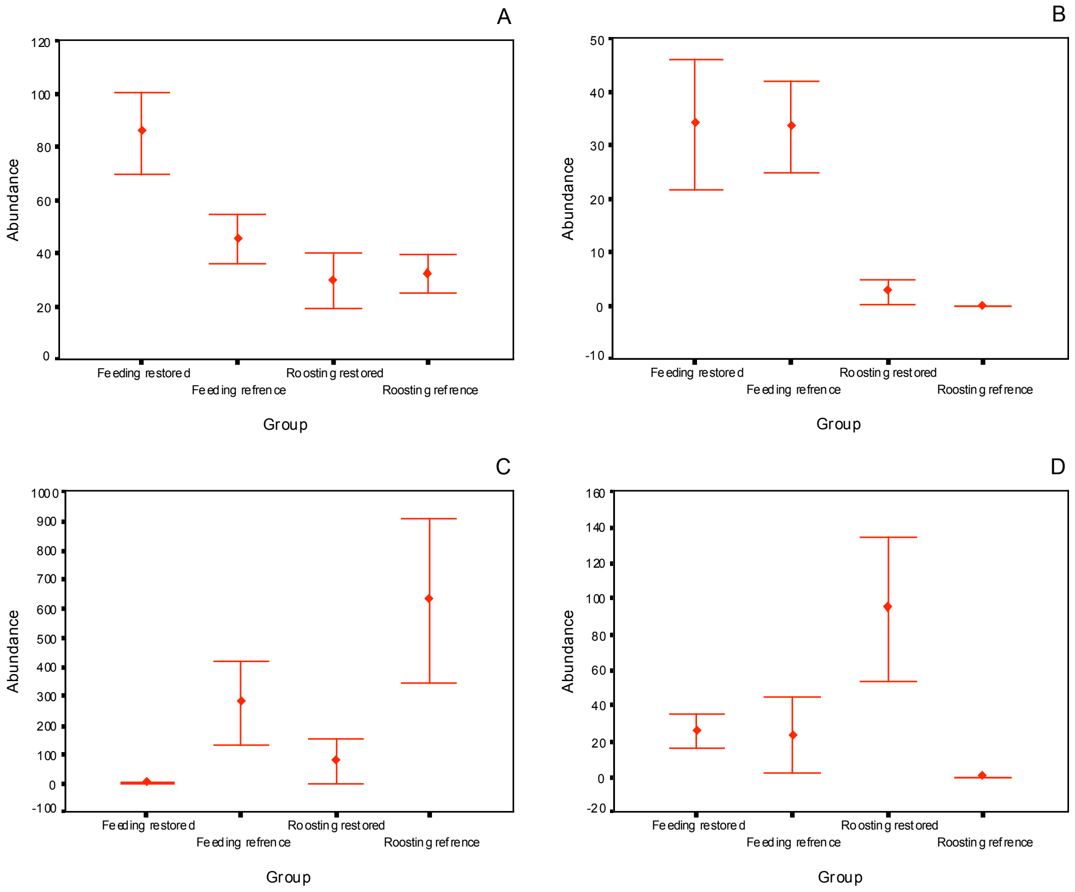


Fig. 5.7: Mean number (± SD) of observed A) Shelduck, B) Teal, C) Golden Plover and D) Dunlin feeding and roosting in reference and restored sites at Paull Holme Strays between June 2004 and June 2005.

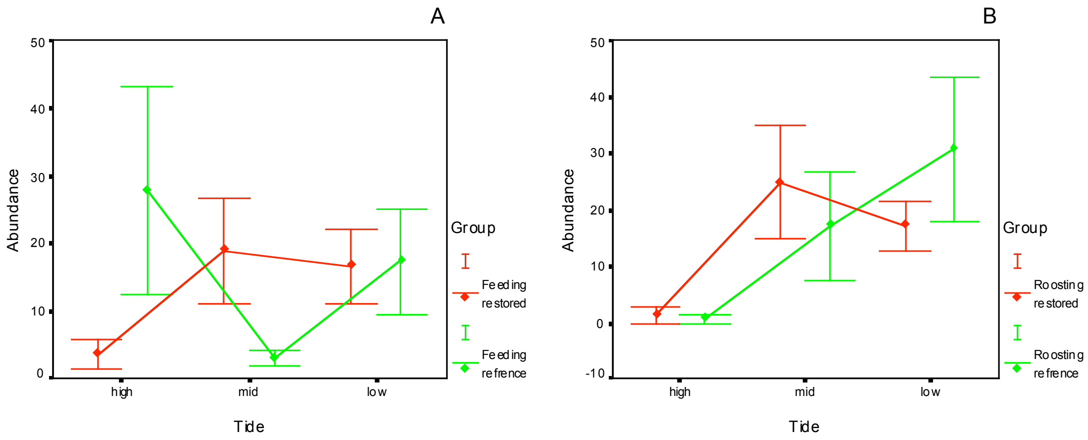


Fig. 5.8: Mean number (± SD) of Curlew observed A) feeding and B) roosting in reference and restored sites based on tidal height, at both reference and restored sites at Paull Holme Strays between June 2004 and June 2005.

5.4. Discussion

Bird densities were lower in the restored habitat of Paull Home Strays for every target species but Shelduck, suggesting that the expectation of waders and waterfowl rapidly colonizing (within the first year post breach) the restored site is not occurring. Both Hemesath and Dinsmore, 1993 and Passell, 2000 suggested that both waders and waterfowl would rapidly colonize (within the two years of restoration) restored habitats. Evaluation of bird densities within the restored habitat a Paull Holme Strays suggests that this habitat is suitable for wetland birds, but that colonization will take longer than expected. Lower than expected density levels in the restoration site could be due to a number of factors that influence the suitability of tidal wetlands for particular bird species. Intrinsic factors, such as amounts of food and space at a site, for example and extrinsic factors such as success on summer breeding sites and at other locations used in migratory paths. Brawley *et al.* (1998), found that some intrinsic factors may in fact help increase the number of wetland birds, they saw as the amount of water at a site increased so did the amount of *Spartina anglica* in the high marsh which is a favourable habitat for wetland birds to breed in. However as there was no significant difference in overall bird density using the habitat adjacent to the restored habitat at Paull Holme Strays before or after breaching there has been an overall increase in the number of birds that can be supported by the middle Humber estuary.

As with most estuarine saltmarsh systems, the Humber estuary is highly productive. The large amount of organic matter exported from the saltmarshes towards the mudflats leads to high macrozoobenthos productivity. The developing restoration study site Paull Holme Strays within the Humber estuary was able to produce 12.3 g/m^2/year of ash free dry weight in 2005 (see section 2.3.3 for a detailed analysis on organic content). Which is enough to support the 2.1 g/m^2/year estimated annual food intake by waders (Eybert *et al.*, 2003). Therefore food supply should not be a limiting factor for wetland birds within the restoration site at Paull Holme Strays.

Dunlin, curlew, redshank, shelduck, teal and mallard ducks consume a wide range of prey items (Del Hoyo and Elliot, 1996), Dunlin forage for polychaete worms, small gastropods, crustaceans and bivalves preferentially extracted from soft sediments covered by a film of water. Curlews, preferentially located in the upper tidal zone consume crabs, small bivalves, large polycheates and surface dwelling insects. Redshanks consume amphipods, gastropods and polycheates. While the three duck species have a generalist diet, they will consume anything from seeds to bivalves to polycheates. But the ducks do have dietary preferences based upon the time of year for example Teal, eat primarily gnats and snails during their nesting season but prefer plant seeds when migrating. Golden plover focus on polychaetes (*Nereis diversicolor*, *Arenicola marina*) (Del Hoyo and Elliot, 1996),

Avian density and species richness tends to be greatest in wetlands with an adequate supply of water (Capen and Low, 1980), an interspersion of open water and vegetation (Weller and Spatcher, 1965) and diversity of vegetation types (Craig and Beal, 1992). Reinert *et al.* (1981) found a direct positive relationship between the density and diversity of waders, shore birds, waterfowl, gulls and the availability of open water (permanent pools). They also determined that short *Spartina anglica* communities, (which started to appear within the restoration site in 2005) (section 3.3.1) are an essential habitat feature for breeding shorebirds. As one of the key borrow pits within the restoration site at Paull Home Strays has filled in since breaching there has been a net loss in the area of the restoration site that makes up permanent pools. The loss of this borrow pit could in part then explain why bird densities were lower in the restored site.

Evaluation of behaviour is an infrequently used but potentially informative tool to assess restored site function (Brusati *et al*., 2001). Monitoring wader and waterfowl behaviour may provide important information about whether wader and waterfowl actually exhibit a natural range of behaviours in restored sites, not simply whether or not they occur. This study suggests that most waders and waterfowl exhibit the natural range of behaviours in the restoration site as both measured behaviours (feeding and roosting) were observed to happen the restoration site. The number of birds observed roosting per hectare was higher or equal that of the reference site for all species examined except for golden plover. The number of birds observed feeding per hectare was higher or equal that of the reference site for all species examined except for golden plover. Since golden plover use the restoration site less than the natural habitat for both measured behaviours it can be said that while they return to restored habitat it takes longer for them to do so, thus indicating that complete ecosystem functioning takes time to establish.

Additionally, Wader and waterfowl behaviour is governed by tidal fluctuations (Recher, 1966; Stenzel *et al*., 1976) and, in the case of the Curlew, tidal cycle had an effect on behaviour. It is possible that tidal cycle would have played more of a role in the behaviour of the other bird species if the habitat type in both the reference and restored sites were more varied. Both sites consisted of tidal flats, which are generally used for feeding while areas of higher elevations, and vegetated areas are used as roosting sites (Warnock and Takekawa, 1995). The findings of the present study agree with Brusati *et al*. (2001) who detected few differences in the behaviour between natural and created sites.

It should be noted that Slavin and Shisler (1983) found the abundance and diversity of waders, shore birds, waterfowl and gulls increased following the tidal restoration of a New Jersey marsh that had been diked and drained for the crop production and over time the total bird biomass was far greater on the restored marsh than on nearby tidally restricted land. Therefore, the restoration site at Paull Holme Strays could be considered successful as waders, shore birds, waterfowl and gulls are using the site and they were not prior to breaching.

Chapter 6

Use of restored mudflats by fish at Paull Holme Strays

6.1 Introduction

Mud and sand flats are a dominant feature of many estuaries, and may comprise a significant component of the total estuarine habitat available to fish (Morrison *et al*., 2001). Estuaries are regarded as important sites for fish, as nursery grounds, as over wintering sites, migration routes and as areas that support large numbers of fish (McHugh, 1967; Elliott and Hemingway, 2002). Many studies have assessed the distribution and abundance of juvenile fish of estuarine systems, ranging from truly estuarine species, to those that have a more broad-based range extending beyond estuarine areas (Lenaton and Potter, 1987; Kneib, 1997; Elliott *et al*., 2007) Differences in abundance and population size structures have been quantified across a range of spatial and temporal scales including latitude, harbours (by way of infrastructure) (Edgar *et al*., 1999), season (Claridge *et al*., 1986 and Potter *et al*., 1997), and habitats (Nagelkerken *et al*., 2000). The nursery function of individual estuarine habitats has only recently received attention in Europe (Frid and James, 1989; Drake and Arias, 1991; Jager, 1999). Additionally, connectivity between ocean habitats and estuarine habitats is receiving attention. A study done by Able (2005) showed that the degree of estuarine use by individual fish species is variable; and that use by an individual species could by influenced by a number of factors including but not limited to temperature, salinity, turbidity and food availability.

The assumption that species exhibit movements/ migrations during their life cycles, at various spatial scales, ranging from daily habitat movements to larger, permanent movements between systems is central to such studies. These movements are implied by documenting large seasonal changes in abundance and size structure of various species of fish (Gillanders, 1997). There has been little work done on the role of smaller scale movements over short temporal and spatial scales for estuarine fishes. This is the scale at which habitat and resource usage are likely to be most tightly coupled. Small-scale movement patterns may be directly related to feeding dynamics, shelter usage and avoidance of potential predators. Knowledge of short- term (hours to days) movement of small fish into and out of estuarine systems is confined mainly to situations where movements into and out of distinctive vegetated habitats have been assessed indirectly, e.g. mangroves (Lin and Shao, 1999), sea grass beds (Rooker and Dennis, 1991) and salt- marsh (Thomas and Connolly, 2001). Such movements are most likely to be related to feeding and predator avoidance, and may be strongly structured by tidal and day-night cycles. Quantification of such movements is central to understanding how different fish species, and size ranges within species, utilize different components of estuarine systems (Kneib, 2000).

The aim of this chapter is to examine fish abundance and diversity; to see how both abundance and diversity change over time within the constructed wetland at Paull Holme Strays; to see how abundance and diversity from within constructed wetland at Paull Holme Strays compares with natural habitats adjacent to Paull Holme Strays; to examine the diet of *Platichthys flesus* (flounder).

6.2. Materials and methods

6.2.1 Field Methods

As each species of fish has a distinct behaviour, different preferences of habitat and different method of feeding, a multiple- method, fish sampling strategy was needed to accurately sample intertidal mudflats (Southward, 1970). Therefore, a range of qualitative techniques in a standard seasonal format at fixed sites was developed to provide the best chance of developing a full assessment of the status of the local fish communities (Colclough, *et al*., 2005). The following methods were used in this study: sampling occurred using a V winged fyke net, a seine net and minnow traps (Hemingway & Elliott 2002). The fyke net was set monthly from April to September in 2004, 2005 and 2006 and bimonthly from September to April in 2004, 2005, and 2006. The seine net was deployed in June, July and August of 2006 and the minnow traps were set annually in July of 2004, 2005 and 2006.The decision was made to deploy both the seine net and the minnow traps only in the summer as numerous papers (Madon, Williams, West and Zedler, 2001; Colclough *et al*. 2002; Laffaille *et al*. 2000) have suggested seasonal variation in the proportion of species utilizing the intertidal area is variable and abundance peaks in summer as species prepare to return to deep water for winter and because of limitations to site accesses during other parts of the year.

The fyke nets used were 2- metres long with an initial hoop diameter of 56 cm, wings measured 3 meters long and 56 cm high, and were made of 14 mm knotless mesh. A net was staked out just inside of the breach site at Paull Holme Strays (Fig.6.1). Nets were also staked out at Cherry Cobb and Welwick in areas of the mudflat that represent habitat similar to that of which Paull Holme Strays is trying to recreate in terms of tidal inundation and sediment characteristics (Fig. 6.2).

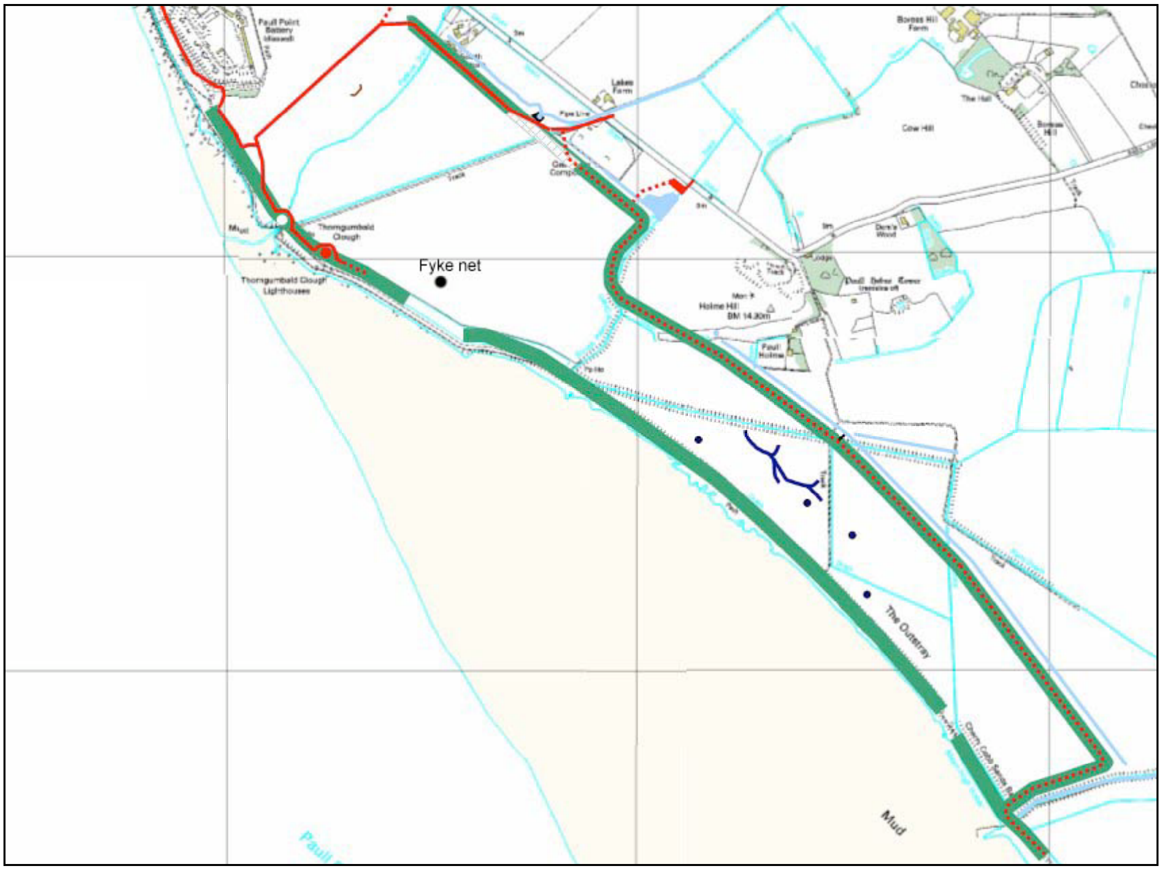


Fig. 6.1: Location of fyke net deployment within the restoration site at Paull Holme Strays. ** Map modified from IECS map of PHS

Nets were deployed on a spring tide of each month. The nets were set at low water and were allowed to stay in position for the completion of one tidal cycle. The nets were positioned to allow them to sample fish draining off of the upper marsh areas through the creek systems. The nets were also deployed on a spring tide for a period of 12 hours to determine differences between nocturnal and diurnal fish assemblages using the sites. Problems with water velocities during the time of sampling were experienced, similar difficulties have been reported by Bozeman & Dean (1980), as well as Colclough *et al*. (2005). Depending on the location, the nets were considered to be sampling most of the fish populations leaving the marsh, but results cannot be considered fully quantitative.

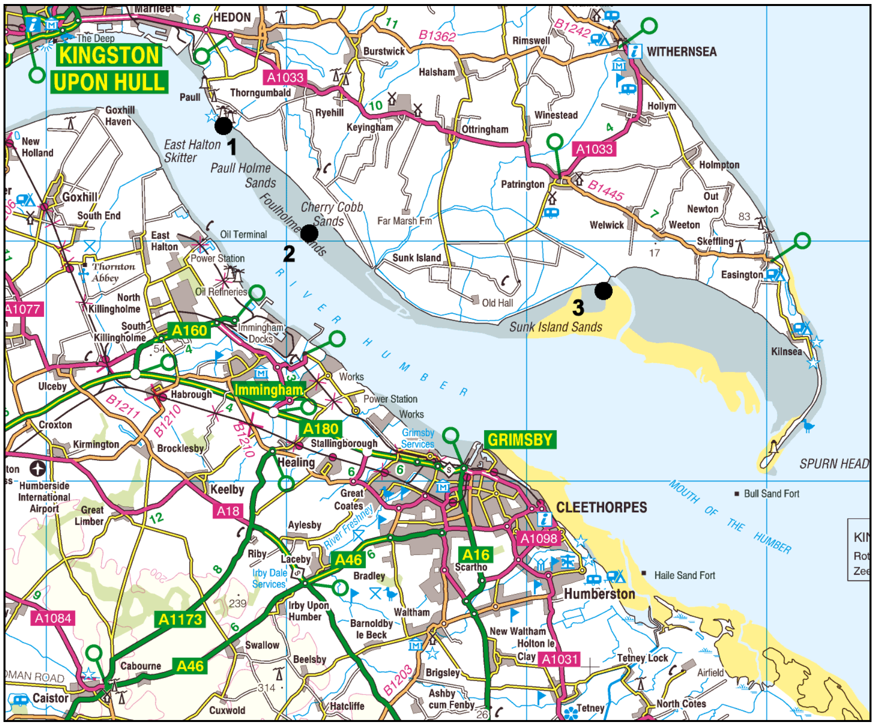


Fig. 6.2: Study sites along the Humber estuary. 1) Paull Holme Strays, 2) Cherry Cobb sands and 3) Welwick. **map modified from googlemaps.com

During the upper part of the tidal cycle, two other methods were employed to study the fish use of the upper creeks and salt marsh:

- One 27 metres X 4 meters knotless mesh seine net with 5mm centre and 14mm wings was deployed repeatedly at appropriate locations at or near high water slack tide on both spring and neap high tides. The area enclosed by the net during sampling was approximately 10 m^2
- Bottle traps were staked at low tide at 20 randomly selected pools and creeks within the Paull Holme Strays (Fig. 6.3). They were left for a tidal cycle.

Upon the collection of the nets and traps, they were opened and everything that was inside was stored in an ice chest and kept at approximately

0° C, until returned to the laboratory. Once in the lab samples were stored at 0° C in a freezer until processing.

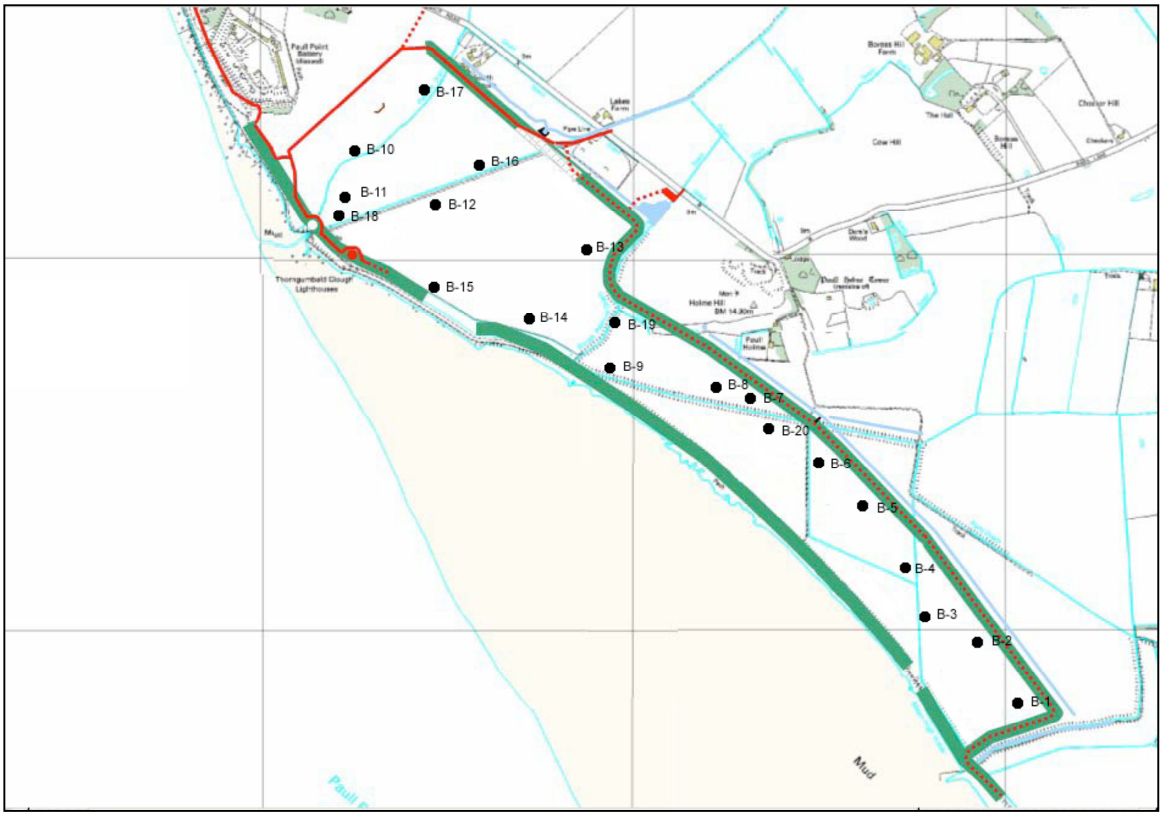


Fig. 6.3: Location of bottle trap deployment inside of the restoration site Paull Holme Strays. ** Map modified from IECS map of PHS

6.2.2. Laboratory methods

The nekton was thawed in the laboratory. All invertebrates caught as bi catch were identified, weighed (wet weight) and measured. In the case of shrimp body length was taken from the rostrum to the tail end and for crabs carapace width was recorded. Fish had their total length and standard length (Bagenal, 1968) to the nearest 1mm recorded, as well as total body mass to the nearest 0.01 g for each specimen. Cutting the esophagus of each sample, near the throat, and the intestines at the vent gutted each fish. The stomachs, gonads and livers were removed, and separated from one and other. The gonads were then used to determine the sex and maturity of the fish. The stomachs were then blotted dry weighed to the nearest 0.01 g. The stomachs were then dissected, and the contents of each stomach sorted. The contents were then classed as still identifiable or debris. Contents that were still identifiable were then separated further into the highest taxonomic separation possible. Once the contents of the stomachs were completely separated into groups, mostly at the phylum level, each content group was weighed.

6.2.3. Data Analysis

The relative composition of the species, in terms of abundance was expressed as a percentage of the total and calculated using mean data for each site per month for all sampling techniques. Abundance ratio (A/S) values were also calculated along with the Shannon Weiner diversity index (H’) and Pielous index of evenness (J), for all data collected using the fyke net.

Species richness and abundance (number of individuals captured per tidal cycle) data collected using bottle traps was analyzed by comparing year using the 20 traps as replicates using a Kruskal- Wallis test, followed by the Games-Howell test, which assumes unequal variance as homogeneity of variance (tested for by a Levene’s test) could not be achieved through transformation of the data.

Seasonality was compared using a one-way ANOVA looking at abundance per month. A Kruskal- Wallis test was used to look at differences in abundance based upon time of day; homogeneity of variance (tested for by a Levene's test) could not be achieved through transformation of the data.

Species richness per 10 m^2 was examined using a one-way ANOVA where the factor was location and abundance per 10 m^2 was examined using a Kruskal- Wallis test where the factor was location as homogeneity of variance (tested for by a Levene's test) could not be achieved through transformation of the data. Additionally analysis of the abundance of individual species per 10 m^2 based upon location was preformed. All of this data was tested for homogeneity of variance (Levene's test) and statistical testing of abundance was done using a one-way ANOVA. In cases where homogeneity of variance could not be achieved through transformation of the data, a Kruskal- Wallis test was used.

The importance of each prey item in the diet was expressed as a percent of the numerical abundance (I_N), occurrence of food items in stomachs (I_O), and weight (I_W) (Hylsop, 1980). To study diet variation with fish size, fish were separated into 20 mm length classes, which were then compared to one and other. A cluster analysis was performed based upon I_N, for each length class. Diet differences between length classes were checked for homogeneity of variance using a (Levene's test then tested using a one-way ANOVA, if homogeneity of variance could not be achieved through transformation of the data, a Kruskal- Wallis test was used.

Food selectivity was evaluated by comparing prey availability and diet composition in numerical terms using a Spearman rank correlation. A Sorenson's similarity index was used to produce clusters via the group average sorting of the similarity coefficient matrix of niche overlap. While diversity and evenness were determined with a Shannon- Weiner and ‘evenness’ indices, these are generally accepted as an indication of dietary niche breadth (Hamerlynck and Cattrijsse, 1994).

6.3. Results

A total of 11 fish species were encountered using Paull Holme Strays after 3 years of development since breaching (Table 6.1) using a fyke net fish sampling strategy. Overall the fish community was dominated by flounder, (*Platichthys flesus*). Species evenness was higher 2005 and 2006 than it was in 2004 just after breaching (Table 6.2). Species richness was highest in 2005 (Table 6.2). Fish species diversity (H’) was highest in 2005 (Table 6.2)

Table 6.1: Abundance of fish at Paull Holme Strays based upon fyke net catch from 2004 to 2006.

					Year				
	2004			2005			2006		
	Total Abundance	% Dom	Cum %	Total Abundance	% Dom	Cum %	Total Abundance	% Dom	Cum %
3- Spine stickle back	0	0	0	1	0.8	0.8	0	0.0	0.0
Gasterosteus aculeatus trachurus var.									
Bass	0	0	0	2	1.7	2.5	0	0.0	0.0
Dicentrarchus labrax									
Cod	0	0	0	3	2.5	5.0	1	1.0	1.0
Gadus morhua									
Eel	0	0	0	2	1.7	6.6	4	3.8	4.8
Anguilla anguilla									
Flounder	1	100	100	105	86.8	93.4	76	73.1	77.9
Platichthys flesus									
Goby	0	0	100	0	0.0	93.4	1	1.0	78.8
Pomatoschistus minutus									
Herring	0	0	100	0	0.0	93.4	1	1.0	79.8
Clupea harengus									
Place	0	0	100	2	1.7	95.0	16	15.4	95.2
Pleuronectes platessa									
Sole	0	0	100	4	3.3	98.3	0	0.0	95.2
Microstomus kitt									
Sprat	0	0	100	1	0.8	99.2	1	1.0	96.2
Sprattus sprattus									
Whiting	0	0	100	1	0.8	100.0	4	3.8	100.0
Merlangius merlangus									

Table 6.2: Biological parameters of fish captured at Paull Holme Strays (means ± SD) for each year of sampling.

Year	H'	J'	S	A	A/S
2004	0(0)	0(0)	0.2(0.44)	0.2(0.44)	0.2(0.44)
2005	0.37(0.29)	0.32(0.25)	2.5(1.22)	20.5(10.74)	8.58(5.53)
2006	0.34(0.42)	0.29(0.32)	2.14(1.57)	19.14(17.74)	7.02(4.87)

* S= number of species; A= number of fish caught; H'= Shannon Weiner diversity; J'= Pielou's evenness; A/S= abundance ratio.

The results of three successive years of bottle trapping inside the restored site (Paull Holme Strays) show that in 2004 the catch per unit effort (CPUE), was five species and 112 individuals. Compared to the CPUE in 2005, which was one species at one station, and six species and 260 individuals in 2006. This indicates that not all species were found every year (Table 6.3). For example four 3-Spined Sticklebacks (*Gasterosteus acuteatus*) were caught in 2004, while none were caught in 2005 and 2006. Ditch shrimp (*Palaemonetes varians*) and Mysid Shrimps (*Neomysis integer*) were only caught in 2006.

Table 6.3: Abundance of nekton at Paull Holme Strays based upon bottle trapping from 2004 to 2006.

			year						
		2004			2005			2006	
	Total Abundance	% Dom	Cum %	Total Abundance	% Dom	Cum %	Total Abundance	% Dom	Cum %
Common littoral crab *Carcinus maenas*	42	38	38	0	0		11	4	4
Brown shrimp *Crangon crangon*	42	38	76	1	100		10	4	8
3-Spine stickle back *Gasterosteus aculeatus*	4	3	79	0	100		0	0	8
Pink shrimp *Pandalus montagui*	1	0.9	80	0	100		0	0	8
Common goby *Pomatoschistus minutus*	23	20	100	0	100		37	14	22
Opossum shrimp *Neomysis integer*	0		100	0	100		51	20	42
Ditch shrimp *Palaemonetes varians*	0		100	0	100		151	58	100

There was a significant difference in the number of individuals found per year (abundance) in the bottle traps (χ^2= 39.56; d.f.= 2; p< 0.01). A Post Hoc Games-Howell test for number of individuals per year showed that there was a significant difference (p<0.05) between 2005 and 2006, and that there was a significant difference (p<0.05) between 2004 and 2006 (Fig. 6.4).

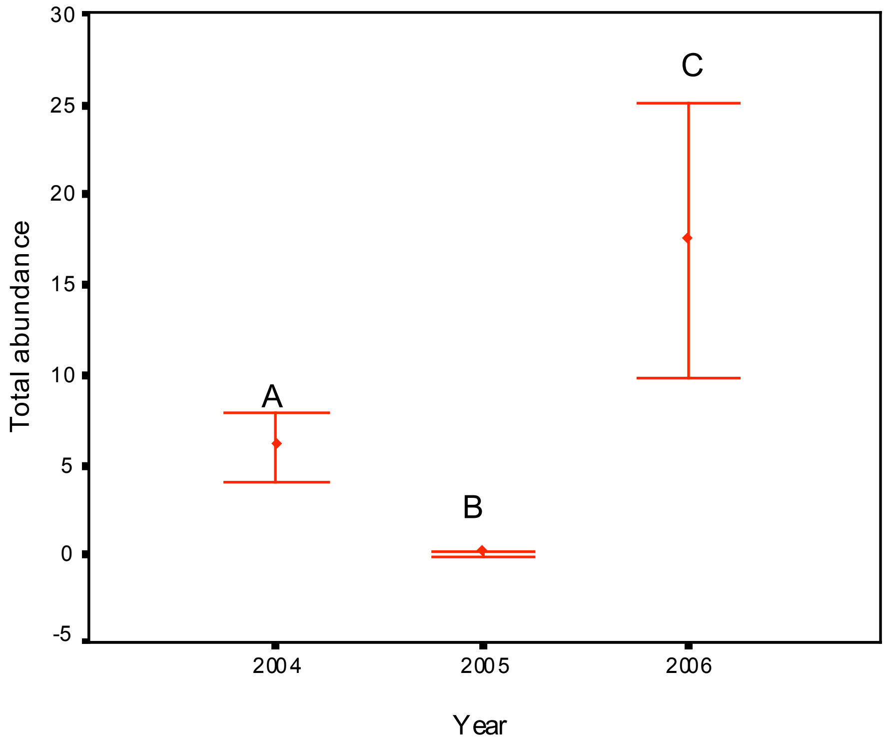


Fig. 6.4: Temporal changes in mean abundance (± SD) of nekton captured in bottle traps between 2004 and 2006. Letters indicate a significant difference in abundance between years based upon a Games– Howell test ($p< 0.05$).

There was a significant difference in the total number of species found per year (species richness) ($\chi^2= 41.50$; d.f.= 2; $p< 0.01$). A Games-Howell test revealed that the difference in the number of species found between both 2004 and 2005 ($p<0.05$) and 2006 and 2005 ($p<0.05$) but not between 2004 and 2006 (Fig. 6.5).

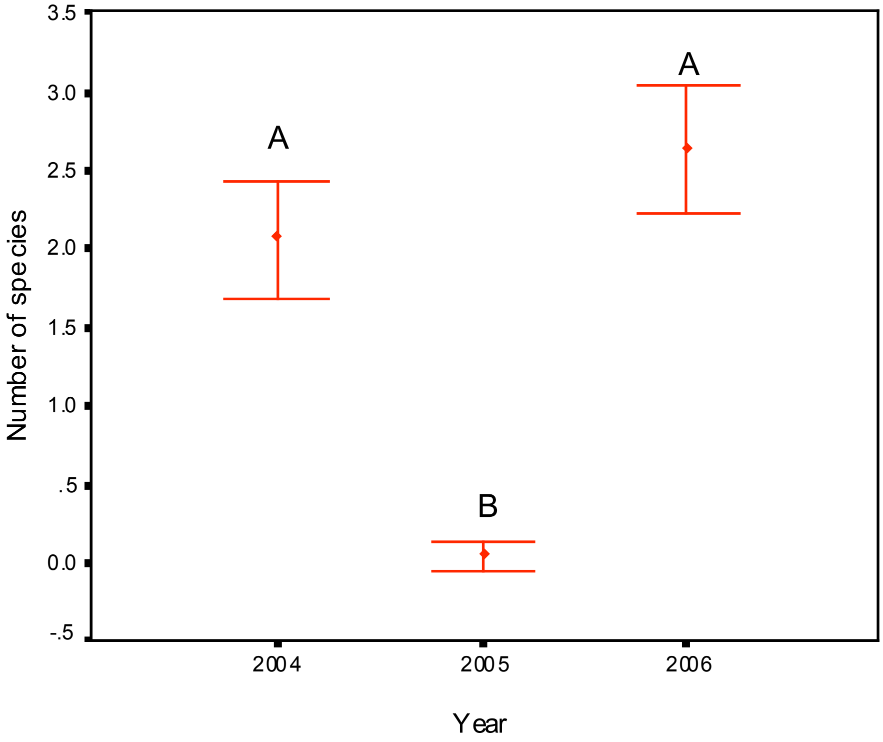


Fig. 6.5: Temporal changes in the number of species (± SD) captured in bottle traps between 2004 and 2006. Letters indicate a significant difference in the number of species between years based upon a Games– Howell test ($p < 0.05$).

In 2006 Paull Holme Strays was compared to two reference marshes and a total of 7 species were found (Table 6.4). Species evenness and species richness were highest at Paull Holme Strays (Table 6.4) Fish species diversity (H') was 0.14 at Cherry Cobb Sands which half as much as what was found at both Paull Holme Strays and the reference site Welwick, both sites had a diversity index (H') of 0.36 (Table 6.4).

Table 6.4: Biological parameters (means ± SD) of fish captured in 2006 for Paull Holme Strays (the restoration site), Welwick and Cherry Cobb sands (reference sites).

Site	S	A	H'	J'	A/S
Paull Holme Strays	7(1.48)	19.12(16.42)	0.36(0.40)	0.38(0.34)	6.93(4.52)
Welwick	6(2.06)	5.37(12.81)	0.36(0.55)	0.35(0.49)	1.27(2.10)
Cherry cobb sands	3(1.12)	7.62(10.51)	0.14(0.27)	0.16(0.26)	4.10(5.16)

* S= number of species; A= number of fish caught; H'= Shannon Weiner diversity; J'= Pielou's evenness; A/S= abundance ratio.

By comparing the fish catch from diurnal and nocturnal samplings it was shown that there was no significant differences between day and night in the number of fish (abundance) ($\chi^2 = 1.33$; d.f. = 2; $p > 0.05$) (Fig. 6.6) or the number of species ($\chi^2 = 0.103$; d.f. = 2; $p > 0.05$) present at Paull Holme Strays (restoration site), or the natural habitats at Welwick and Cherry Cobb (Fig.6.7).

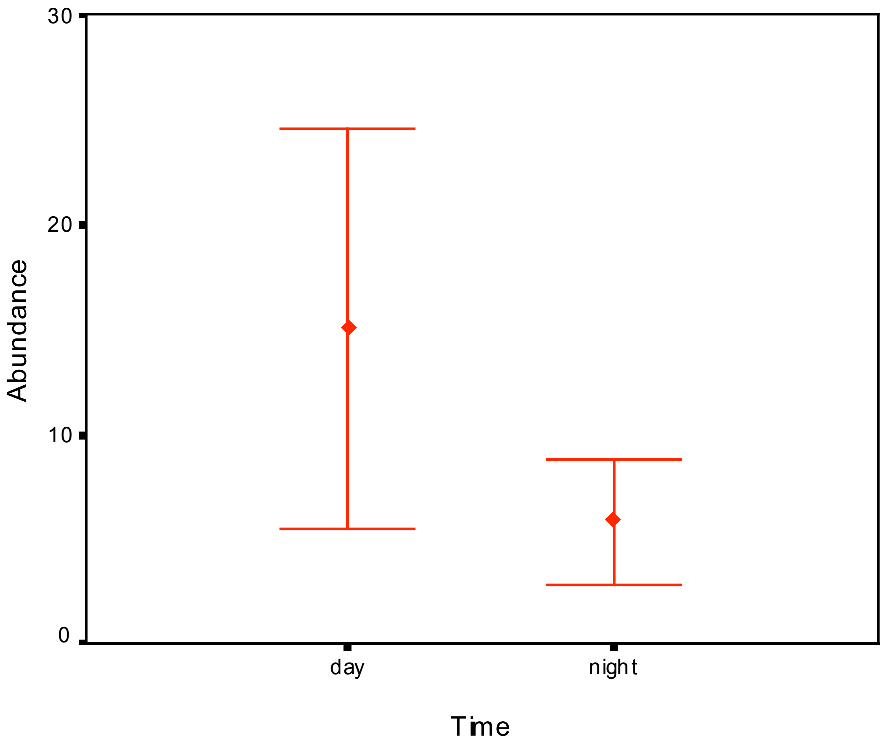


Fig. 6.6: Mean abundance ±(SD) of fish caught in nocturnal and diurnal samples.

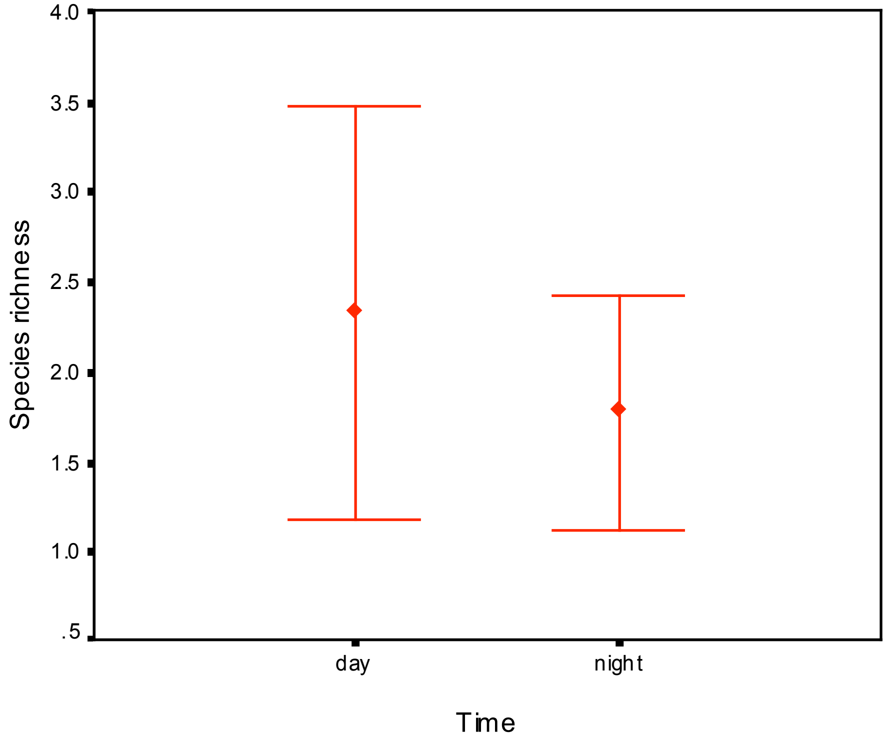


Fig. 6.7: Mean number of species ±(SD) of fish species caught in nocturnal and diurnal samples.

While there was a trend of less fish being present in the winter/ early spring there was no significant difference in the number of fish observed based upon the time of year (f =6.23; d.f. = 7,37; p> 0.05), (Fig. 6.8).

A total of 11 species were captured using the seine net, *Atherina presbyter*, *Gasterosteus aculeatus*, *Gadus morhua*, *Anguilla Anguilla*, *Platichthys flesus*, *Pomatoschistus minutus*, *Clupea harengus*, *Pleuronectes platessa*, *Microstomus kitt*, *Sprattus sprattus*, *Nerophis ophidion* and *Chelon labrosus*. Of these 11 species 3 were only captured using the seine net; the smelt *Atherina presbyter*, the pipefish *Nerophis ophidion* and the mullet *Chelon labrosus*. The use of a seine net showed that there was a greater diversity of fish using the restored habitat (Paull Holme Strays) per 10 m^2 than at the natural habitat (Cherry Cobb). (f = 4.87; d.f. = 1,39; $p < 0.05$) (Fig. 6.9). However the abundance of fish per 10 m^2 did not differ between the two sites ($\chi^2 = 0.961$; d.f.= 1; $p > 0.05$). Additionally nether the abundance of fish ($\chi^2 = 0.00$; d.f.= 1; $p > 0.05$) or the species richness ($\chi^2 = 2.24$; d.f.= 1; $p > 0.05$) per 10 m^2 differed based upon tidal height. Looking a individual species of fish showed that there was no significant difference found in the abundance of *Platichthys flesus* ($\chi^2 = 2.05$; d.f.= 2; $p > 0.05$), *Gasterosteus aculeatus* ($\chi^2 = 1.50$; d.f.= 2; $p > 0.05$), *Gadus morhua* ($\chi^2 = 1.00$; d.f.= 2; $p > 0.05$), *Anguilla Anguilla*, ($\chi^2 = 0.45$; d.f.= 2; $p > 0.05$), *Clupea harengus* ($\chi^2 = 0.70$; d.f.= 2; $p > 0.05$), *Pleuronectes platessa* ($\chi^2 = 1.40$; d.f.= 2; $p > 0.05$), *Microstomus kitt* ($\chi^2 = 1.00$; d.f.= 2; $p > 0.05$), *Pomatoschistus minutus* ($\chi^2 = 1.00$; d.f.= 2; $p > 0.05$), or *Sprattus sprattus* ($\chi^2 = 2.36$; d.f.= 2; $p > 0.05$) based on time of year. However, *Atherina presbyter* was only found in June of 2006 but it was present at both locations; while *Nerophis ophidion* and *Chelon labrosus* were only found in June of 2006 at the restoration site Paull Holme Strays.

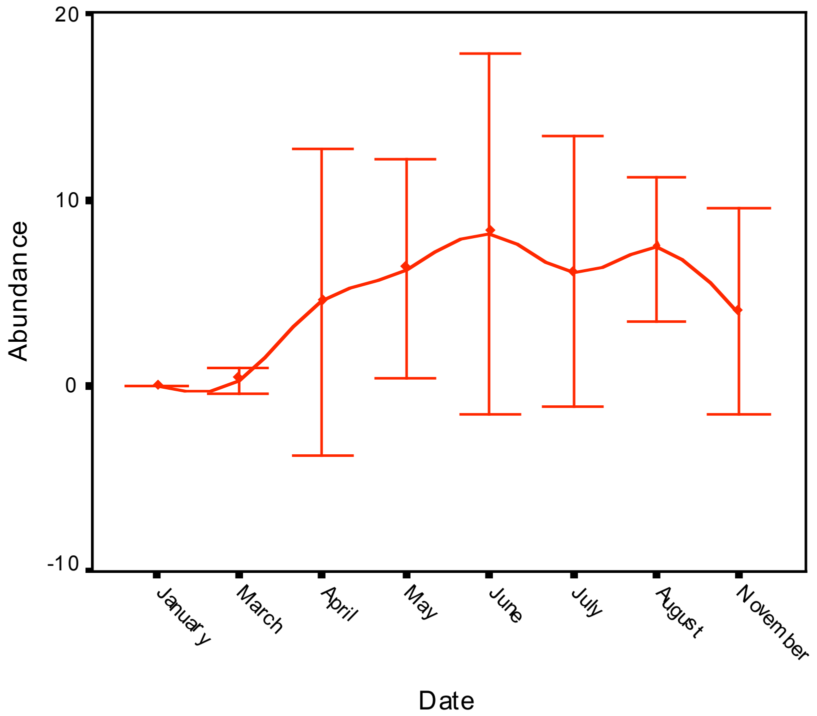


Fig. 6.8: Fish Abundance at Paull Holme Strays based upon the time of year based upon fyke net samples taken from 2004- 2006.

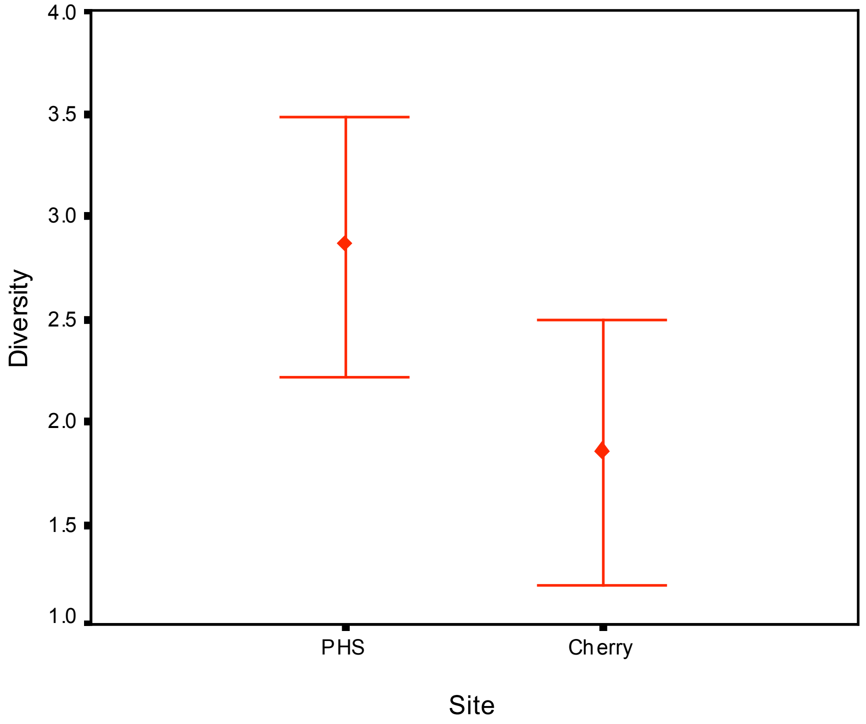


Fig. 6.9: Mean (± SD) of the number of species of fish caught per 10 m^2 at both the Paull Holme Strays (restoration site) and the reference site Cherry Cobb.

Both the plaice *Pleuronectes platessa* ($\chi^2 = 7.50$; d.f. = 1; $p < 0.05$) and the herring *Clupea harengus* (f= 5.61; d.f.= 1; 14; $p < 0.05$) had significantly more individuals per 10 m^2 present in the natural habitat (Cherry Cobb) than at the restoration site (Paull Holme Strays) (Fig. 6.10). No other species caught were found to vary in abundance per 10 m^2 based upon the location. However, *Clupea harengus* was captured using the fyke net only at the restored site (Paull Holme Strays) in July 2006, as was *Gadus morhua*. The sprat, *Sprattus sprattus*, as with *Clupea harengus* was only caught using the fyke net at Paull Holme Strays in April of 2006. No significant differences were found in the numbers of the common eel *Anguilla anguilla* present based upon site ($\chi^2 = 2.15$; d.f.= 2; $p > 0.05$). As with *Anguilla anguilla*, the number of plaice *Pleuronectes platessa* found did not vary by site (f= 0.78; d.f.= 2,3; $p > 0.05$). The number of whiting *Merlangius merlangus* did not differ between the restored site and the natural habitat at Welwick ($\chi^2 = 2.07$; d.f.= 1; $p > 0.05$) although no whiting were found in the natural habitat at Cherry Cobb. *Platchthys flesus* numbers were not significantly different between the restoration site and the natural habitat Cherry Cobb (f= 3.98; d.f.= 2, 18; $p > .05$), but both of these sites had significantly more individuals present than at Welwick (mean diff. 13.42; SE = 4.76; $p < .05$).

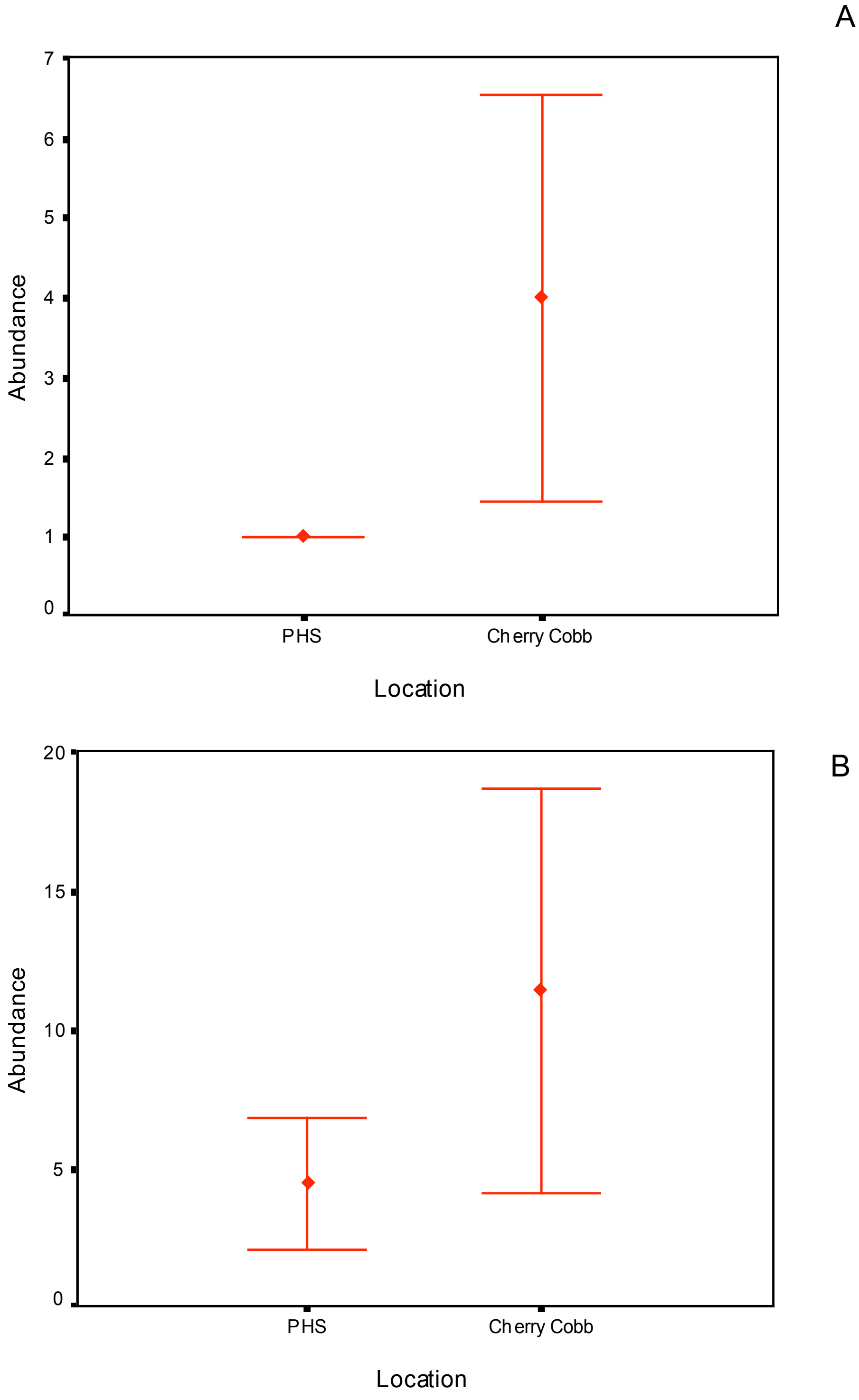


Fig. 6.10: Mean number of individuals (abundance) (± SD) of (A) *Pleuronectes platessa* and (B) *Clupea harengus* per 10 m^2 based upon location in 2006.

As flounder were the most abundant fish caught within the restoration site (Table 6.5) they were used to assess feeding behaviour in the restoration site. The flounder ranged in length from 27 to 305 mm (Fig. 6.11). Males had a mean length of 137 mm and females had a mean length of 145 mm and fish ranged in age from 0 + to 6 years of age, with fish between 135 and 145 mm representing age 2 based on samples taken during this study. Males and females were equally abundant up to about 140 mm, with larger fish being mostly female. A majority of the individuals sampled were found to be mature (72 %). The overall percentage of empty stomachs was 26%. A log-linear model showed that a change in the percentage of empty stomachs was not dependent on length-group (Fig. 6.12) (χ^2= 6.374; d.f. = 6; $p > 0.05$). The percentage of empty stomachs was also shown not to be dependent upon the season of collection (χ^2= 2.52; d.f. = 2; $p > 0.05$).

Table 6.5: Monthly actual abundance of flounder in samples from Paull Holme Strays from April 2004 to April 2006

Month	N	% all fish species
June 04	1	100
July 04	0	0
August 04	0	0
September 04	0	0
October 04	0	0
April 05	23	82.14
May 05	1	100
June 05	18	100
July 05	26	86.67
August 05	23	85.19
November 05	15	78.95
January 06	0	0
March 06	1	100

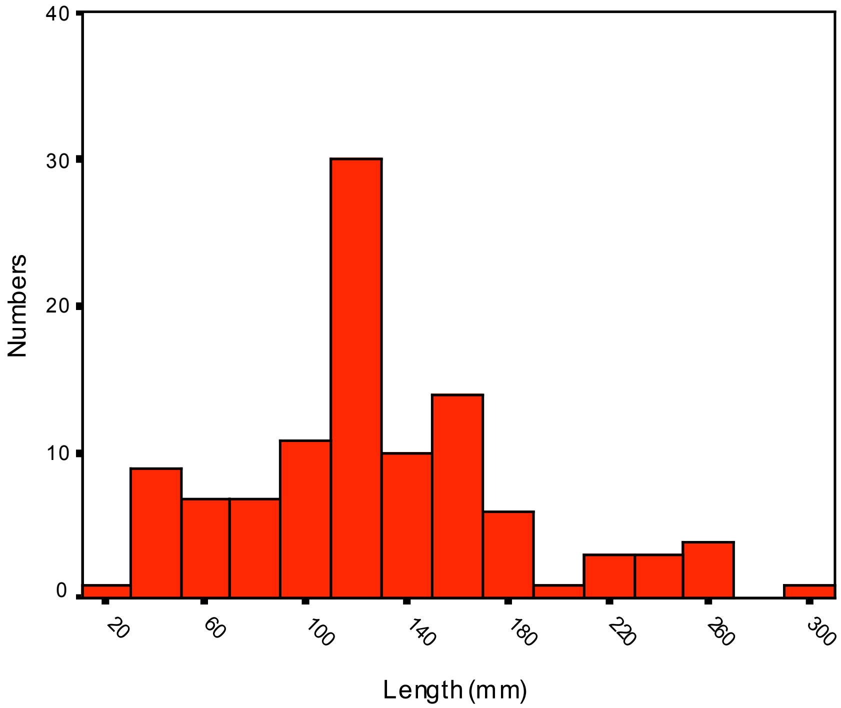


Fig. 6.11: Length frequency histogram for flounder samples from Paull Holme Strays. Data is for the whole sampling period(2004-2006) so growth over time would merge the cohorts.

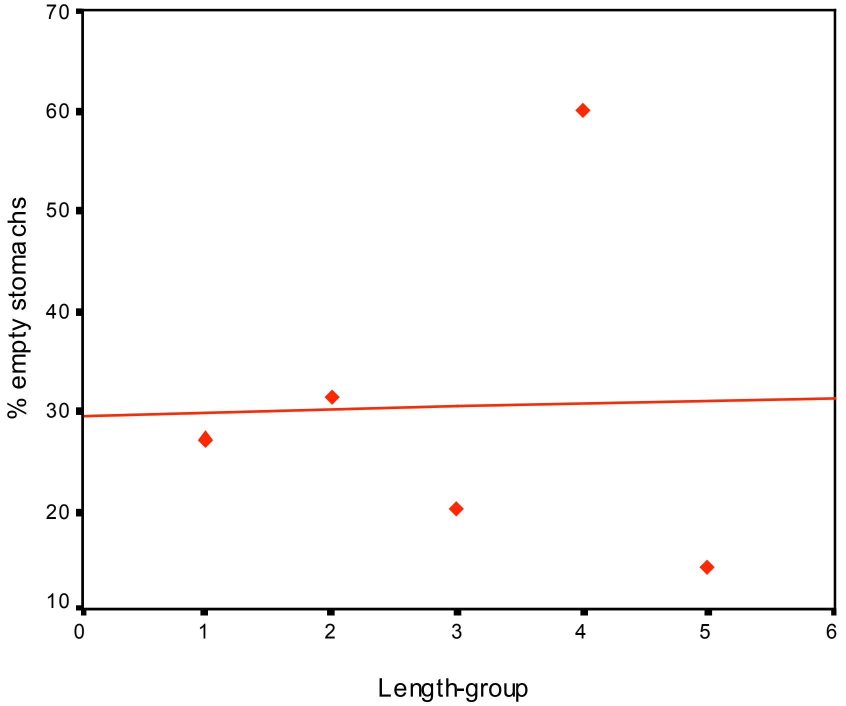


Fig 6.12 Percentage of empty stomachs based upon size class. 1= >100mm, 2= 100-119, 3= 120-139, 4= 140-179, 5= <180.

Of the stomach contents analyzed, 85% of the items were identified to one of thirteen prey categories (Table 6.6). Annelids were the most dominant group, occurring in 45% of the stomachs, contributing to 60% of the prey in terms of abundance and 34% of the total prey weight or biomass. *Nereis (Hediste) diversicolor* was the most abundant species (40% of total prey), contributing 34% to the biomass. Similarly, 16% of all prey was the amphipod *Corophium volutator*, and this species contributed 6% to the total biomass, while the mysid *Schistomysis* sp. contributed 12% in terms of abundance and 5% to the biomass. Other prey found in the stomachs included molluscs, insects, bivalves and various other annelid species, in total contributing 18% to the prey biomass.

Table 6.6: Prey composition of Flounder diet from samples taken from Paull Holme Strays

Prey taxa	% FO	% N	% W
Nereidae			
Nereis diversicolor	39.60	31.08	34.08
Tubificidae			
Tubificoides benedii	3.96	16.02	+
Enchytraeidae	0.99	12.55	+
Total annelids	44.55	59.65	34.18
Tellinidae			
Tellinacea sp.	1.98	0.58	0.65
Macoma balthica	4.95	4.05	0.61
Tellinacea sp. Juv.	7.92	2.90	0.76
Hydrobiidae			
Hydrobia ulvae	0.99	+	+
Total molluscs	15.84	7.72	2.03
Portunidae			
Carcinus maenas	6.93	1.54	38.71
Crangonidae			
Crangon crangon	1.98	0.58	13.01
Corophiidae			
Corophium volutator	13.86	16.22	6.11
Mysidacea			
Schistomysis sp.	11.88	12.16	5.42
Total crustaceans	34.65	30.50	63.25
Tipulidae	0.99	+	+
Total insecta	0.99	+	+
bi-valve siphons	2.97	1.74	0.53
Total other	2.97	1.74	0.53

% FO = percent frequency of occurrence, %N = percent number, %W = percent by weight, + indicates presence but percentage less than < 0.5.

The size of the predator had no effect on the abundance of prey items in their stomach (Fig. 6.13).

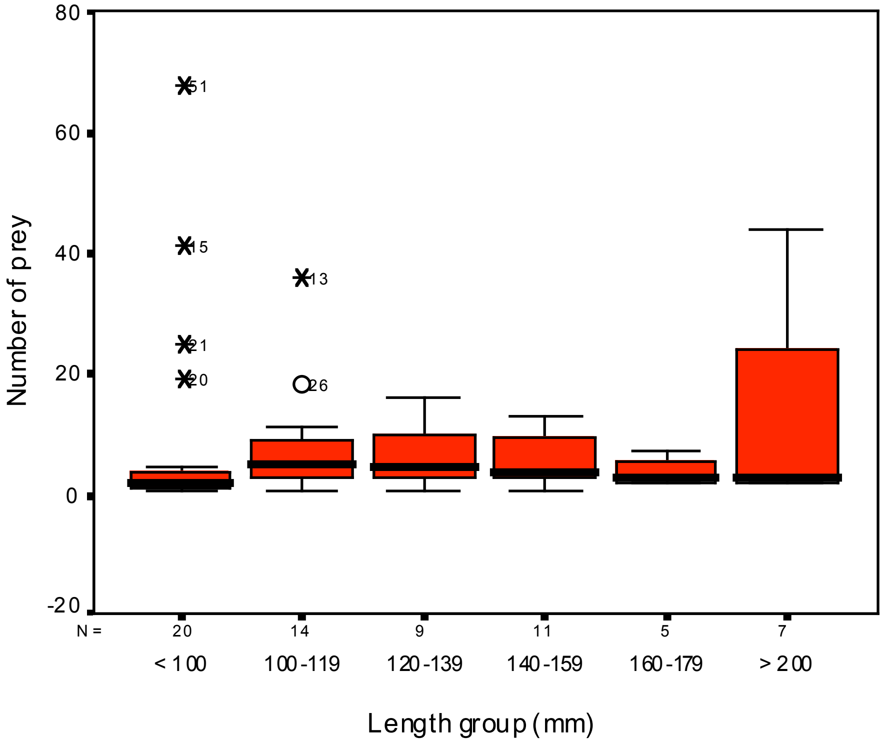


Fig. 6.13: Number of prey items for stomachs having some contents. Median, box limits 25% and 75%. Whiskers indicate the inner fence, rings are values within the outer fence, and asterisks are far outside values. The number of non-empty stomachs with an identifiable number of prey is given at the bottom of the figure.

In stomachs with 3 or more prey, 53% contained both relatively fresh and severely digested prey simultaneously. Neither the percentage of empty stomachs ($\chi^2 = 1.01$; d.f.= 2; p> 0.05) or diet ($\chi^2 = 2.34$; d.f.= 2; p> 0.05) differed between sexes. There were no statistically significant seasonal differences in the diet based upon size class ($\chi^2 = 7.03$; d.f.= 6; p> 0.05). The influence of predator size on diet composition shows that the main pattern of diet was similar, however as predator length increased the importance of *Nereis diversicolor* to the overall prey weight biomass decreased and the importance of other prey like *Carcinus maenas* became more important to the prey weight biomass (Fig. 6.14, Table 6.6). This reflects a change of feeding behaviour - moving from taking relatively sedentary prey to prey that is more mobile. Conversely, the importance of both *Corophium volutator* and *Schistomysis* sp. seems to be relatively stable. Neither niche breadth or niche overlap of the predator's diet seem to be affected by size (Fig. 6.15,Table 6.7).

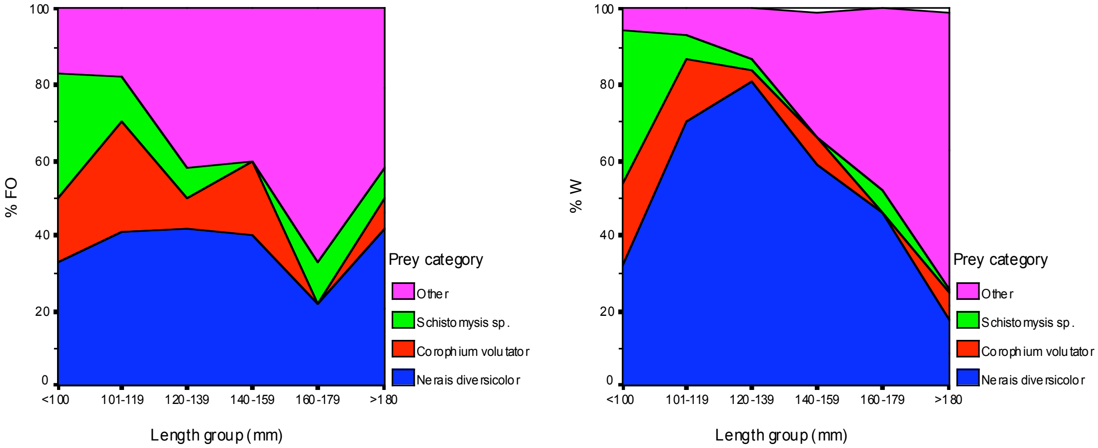


Fig. 6.14: Main prey categories of flounder (*Platichthys flesus*) by 20 mm length groups, presented as frequency of occurrence (%FO) and percentage by weight (%W).

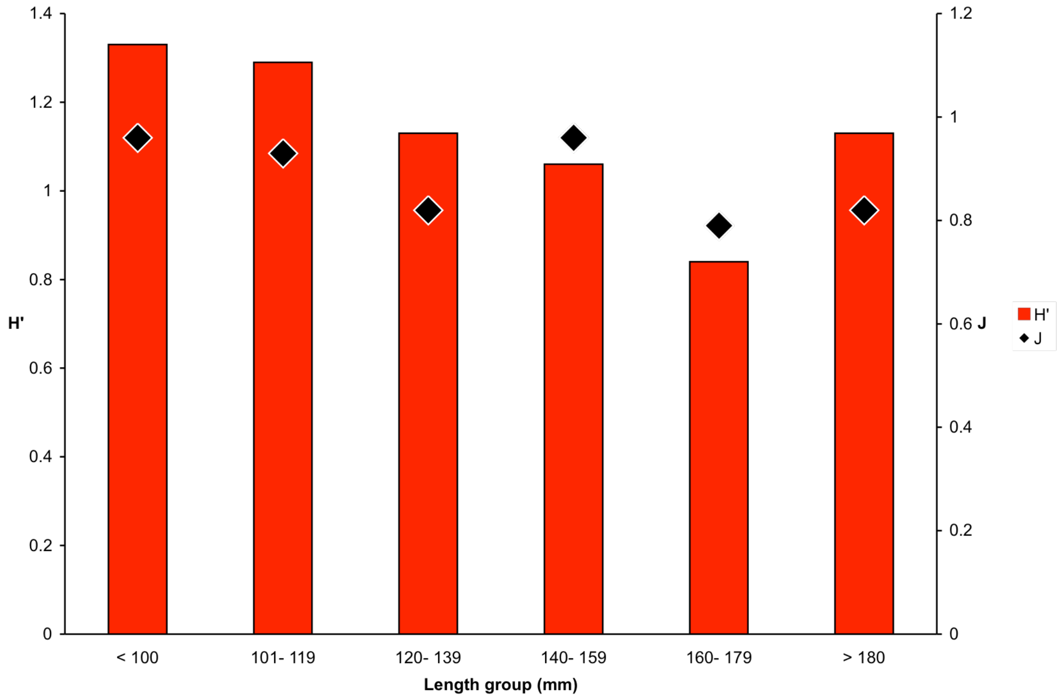


Fig. 6.15: The Shannon- Wiener diversity index (H') and Pielous index of evenness (J) for the diets of the flounder based upon size class.

Table 6.7: Sorenson's similarity index showing the similarity of diet between the different length groups of flounder in this study.

	<100	101-119	120-139	140-159	160-179
<100					
101-119	1.00				
120-139	1.00	1.00			
140-159	0.86	0.86	0.86		
160-179	0.86	0.86	0.86	0.67	
>180	1.00	1.00	1.00	0.86	0.86

Core samples indicated that a total of 15 species made up the benthic community table (Table 6.8). *Nereis diversicolor* was the most abundant species found, and 90% of the community consisted of *N. diversicolor*, Nematoda, Acarina, *Hydrobia ulvae*, *Paranais littoralis* and Enchytraeidae. Spearman rank correlation revealed that diet was not reflective of prey availability (p> 0.5) although *N. diversicolor* was the most abundant species both in the predators' stomachs and in the cores.

Table 6.8: Species abundance per m^{-2} and percent make up of the benthic community based upon core samples.

Species	Mean abundance	% of population
Nereis diversicolor	333.64	49.33
Nematoda	121.32	17.94
Acarina	48.53	7.18
Hydrobia ulvae	42.46	6.28
Paranais litoralis	30.33	4.48
Enchytraeidae	27.3	4.04
Crophium volutator	27.3	4.04
Tubificoides benedii	15.17	2.24
Macoma balthica	12.13	1.79
Pygospio elegans	3.03	0.45
Streblospio shrubsolii	3.03	0.45
Aphelochaeta marioni	3.03	0.45
Capitella sp.	3.03	0.45
Chone/ Jasmineira spp.	3.03	0.45
Manayukia aestuarina	3.03	0.45

6.4 Discussion

Creeks and mudflats contain many spatial and temporal microhabitats, highlighting the necessity both to sample at a range of scales (Zedler, 2000) and to assess each site over an entire tidal cycle to make assessments of the suitability of different sampling methods. The continuous redistribution and clustering of fry in high intertidal habitats can only be shown by combining static nets with a variety of mobile sampling gear in a highly adaptive manner (Cololough, *ed al.,* 2005).

The data collected in this study shows that there is a seasonal trend in fish utilization of the created marsh-flat at Paull Holme Strays with a peak in abundance in June (Fig. 6.8). Additionally, flounder were found to be the most commonly captured user of the newly created marsh-flat habitat. Marsh-flat researchers have aimed to determine what nektonic animals use saltmarshes and when they use them. A generalization from studies of nekton is that fish and crustaceans using the marsh flat are mainly resident on or near the saltmarsh for their entire life cycle, while juvenile fish congregating around the edge of the salt-marsh are juveniles of species that spawn elsewhere in the estuary or oceanic waters (Peterson and Turner, 1994).

A more detailed classification by Peterson and Turner (1994) places nekton into four distributional categories:

1. On the marsh flat at all times (remains in pools at low tide, dominated by Gasterosteidae such as sticklebacks).
2. On the marsh flat at high tide, but retreating to subtidal fringing vegetation at low tide, (killifish, minnows (*Cyprinodon*), grass shrimp (*Palaemonetes*), and portunid crabs (*Callinectes*)).
3. Onto the marsh edge at high tide, but penetrating only a few meters onto the flat, and into subtidal creeks at low tide (this type comprises mainly juveniles of commercially important species such as mullet (*Mugilcephalus*) and penaeid shrimp).
4. Remaining subtidal, not really entering salt marsh at all, but being in close proximity in creeks (e.g. juvenile Atlantic menhaden (*Brevoortia tyranmus*), a major commercial fish in the USA).

A total of 85 species of fish are known to inhabit the Humber estuary and of those 43 are known to inhabit the middle estuary; of those 43, 11 were found to be using the mudflats and salt-marsh of restored and references sites in the middle estuary. This indicates that only a small number of species occurring in the Humber are using the upper mudflats and marshes of the mid Humber Estuary.

The majority of the fish taken in the present study were juveniles, which provides evidence that despite extensive anthropogenic disturbance in the system, this portion of the estuary is a nursery area. Shenker and Dean (1979) highlighted a degree of utilization of intertidal creek habitat by larval and juvenile fishes, and stated that differences in diurnal and nocturnal activity, coupled with variation in catch sizes, serve to reduce space and energy competition. While no significant differences in diurnal and nocturnal catch sizes were seen, the trend was towards a greater abundance of fish present during the day. Copp and Jurajba (1993) found significantly greater number of potentially piscivorous fishes at night in marsh habitats and concluded that saltmarsh habitats are too shallow for larger fish to risk entering during daylight hours for feeding, for risk of stranding or predation by birds. Paterson and Whitfield (2000) found a high positive correlation between water depth and number of piscivorous fish in a temperate southern African estuary. Water depth at the present study sites was relatively deep, (Section 2.3.1) thus partly explaining why no significant differences were found in diurnal/ nocturnal samples.

Many fish species complete their life cycles in tropical and subtropical estuaries (Blaber *et al*., 1989; Roberson and Duke, 1990), but there is less evidence they do so in temporal regions (Claridge *et al*., 1986; Potter *et al*., 1990). However, Elliott and Dewailly (1995) listed only 26 out of 186 species occurring in 16 European estuaries as being estuarine dependent meaning that they live there whole lives in the estuary. Therefore most European temperate estuaries are not critical to the survival of their visitors, as they can complete their entire life cycle without entering an estuary with the exception of diadromous species such as *A. anguilla*.

Abiotic water conditions (salinity and temperature) are often suggested as controls for seasonal patterns of species occurrence (Thiel *et al*., 1995; Marshall and Elliott, 1998). Temperature can and will effect the distribution of fish through different thermal tolerances based upon species (Marshall and Elliott, 1998). Although it has not been recorded for estuarine species, an upper lethal limit of 25 °C the species with the lowest thermal tolerance (Poxton and Allouse, 1992). 25 °C was greater than the highest temperature recorded during this study (Section 2.3.6); therefore it is unlikely that thermal tolerances alone are the main factors influencing the distribution of the species examined. Salinity has been shown to have a greater influence on the species composition within the Humber than temperature (Marshall and Elliott, 1998). As each species of fish has a distinct salinity tolerance. As salinity during this study remained relatively constant only dipping in August of 2004 after a severe rain storm (Fig. 2.19) it is thought that salinity effects on fish distribution would be constant only allowing individuals capable of surviving within the specific salinity range of the created habitat (Paull Holme Strays) to take advantage of it. While both salinity and temperature were measured during this study, correlations with seasonal fish distributions could not be made, as the data was not taken simultaneously. Additionally the proximity to the mouth of the estuary can and does influence the number of species taken.

Of the species captured, flounder was the most common followed by plaice, both are flatfish. Like other flatfish they inhabit areas based upon prey availability seasonal temperature changes and substrate type (Rackowski and Pikitch, 1989). In the mid Humber estuary where this study took place the sediment is predominantly fine silt and clay (Section 2.3.3) with little benthic macrovegetation. These are ideal conditions for flatfish which if food is available and temperature is with in range choose their habitats based upon the size of the sediment grains (Riley *et al*., 1981).

Flounder (Platichthys *flesus*) feed based upon the perception of water currents coming from their prey by the lateral line on their blind side and site (Bergnballe, 1961; Muus 1967). They assume an attack posture with their heads lifted off of the substratum, braced on the rays of their dorsal and ventral fins, then suddenly bury their heads into the substrate, while hovering in the water with their dorsal fin above the substratum (Summers, 1979). They feed by both day and night moving inshore approximately two hours before low tide and return to the sublitoral zone two hours before the next low tide (Buckley, 1989; Summers 1979). Data collected during this study shows that a flounders (*P.*

flesus) diet consists mainly of polychaetes (60% of their diet) and crustaceans (16% of their diet). This agrees with the findings of de Groot, (1971) who characterized *Platichthys flesus* as being in the Pleuronectidae of type II, which feed mainly on crustaceans but also consume polychaetes and molluscs.

15 invertebrate species were identified from the core samples (Section 4.3). Of these, only 6 were found in their stomachs. The cores may be biased in that they sampled the top 10 cm of sediment, whereas flounder may only be able to capture items within the top 1-2 cm, thus over estimating potential prey items. Of the species found in both the cores and stomachs annelids made up 44 percent of the diet. This may reflect the fact that most annelids live within the top 2 cm. Molluscs only formed 15% of the diet indicating that there is little dietary selection for them by the flounder; this may reflect the fact that most molluscs burrow within the sediment, reducing the ability of flounder to take the prey (More and More, 1976), together with the relative indigestibility of the shell. However, the corer captures infauna that may emerge at some point and thus is available to the flounder. Hardisty (1986) suggests and Marshall (1995) found that some prey items might not be accounted for in benthic cores due to *Platichthys flesus* adaptability in capturing prey thus allowing it to feed on free-swimming organisms in the water column as well as prey buried in the sediment. The results of this study agree with these findings as both *Carcinus maenas* and *Schistomysis* sp. were found as components of the fishes' diet. Several studies (Ramos, 1981; Lagardere, 1987;Molinenro and Flos, 1991; Cabral, 2000) on other species of flat

fish have suggested that as the fish become larger so do the prey, which was seen in this study.

As indicated by the stomach content analysis and the occurrence of prey items, many of the prey taken are more likely to occur in the intertidal areas than in the subtidal areas, including: *Nereis, Corophuim* and *Macoma* all of which occurred in relatively high densities within the core samples taken (Section 4.3). Hence, *Platichthys flesus* appears to be dependent on intertidal areas for feeding. There were no significant differences in the type of prey that was taken by *Platichthys flesus* based upon size class, therefore there is a niche overlap based upon size class. This could lead to strong intra-specific competition between the various size classes of *Platichthys flesus*, which are competing with one and other for a limited amount of food that is available at high tide on the intertidal mud flats. This competition could be a limiting factor to the overall population of *Platichthys flesus* as not only are they competing with themselves for food but also there is also inter-specific competition with other species of fish such as sole that are dependent on the intertidal mud flats at high tide for food (Marshall, 1996).

Chapter 7

Final Discussion and Conclusions

Wetlands frequently are constructed to mitigate and/or compensate for loss of ecosystem habitats and services caused when human activities, such as: surface mining, highway and pipeline construction, and urban development, lead to the destruction or degradation of natural wetlands (Zedler *et al*., 2001, Thayer, 1992). Wetland creation typically involves many aspects, including structural components as well as ecosystem functioning (www.ser.org; Ruiz-Jaen and Aide, 2005). The emphasis in this thesis has been on the initial biological response to habitat restoration. However, after this, many wetland creation projects fail to monitor the development of all wetland dependent ecosystem services following construction (National Research Council, 1992).

There is much debate on how to define restoration success. One approach is to use clearly specified targets, set in agreements, contracts or permits (Kentula, 2000). Other possibilities for assessing success are to compare the ecological structure or functioning of a restored site with one or more reference sites (Edwards and Proffitt, 2003). Historical reference can also be used for assessing success; in this case success criteria are based on what the wetland was like before the industrial revolution and before the application of artificial fertilizer (de Jonge and de Jong, 2002). Regardless of the method used to measure restoration success the bigger question still remains, should the aim of restoration be focused on replacing ecosystem structure or on ecosystem functioning. Zedler and Lindig-Cisneros (2000) defined structure as a condition at one point in time (e.g. species diversity) and function as a process that occurs over time (e.g. primary production), and concluded that structural measures are often used as substitutes for functioning. Zedler and Callaway (1999) furtherer point out that the restoration of function often takes much longer than the restoration of structure its self. While I agree with the fact that proper structure and function are key to a successful restoration, this thesis focuses on the restoration of the structural component, as it is the first and most important stage in wetland restoration.

The present project utilized several methods to assess the success of development of Paull Holme Strays. Ecological structure within the restored wetland of Paull Holme Strays was compared with one or more reference sites. This method of monitoring was carried out as the primary goal of Paull Holme Strays was and is to provide flood protection, with habitat creation regarded only in terms of an additional benefit (Richardson, 2004). Problems with this type of monitoring arise from reference site choice as this choice can strongly influence the outcome of comparisons (Kentula, 2000; Morgan and Short, 2002). Besides, comparing conditions with a natural reference system may not be realistic because restoration may start on a different substrate or at a different elevation (Thom, 2000), or because the reference site its self may be degraded. Doing so showed several contrasts in the ecological structure between the natural and restored areas.

Saltmarsh a key component to a successful wetland restoration is the vegetated part between the land and sea, which receives frequent tidal inundation (Adam, 1990). Once vegetation has established it can serve many different functions, for example, sediment trapping (accretion Chapter 2), nutrient cycling redox potential (Chapter 2), dissipation of wave energy, spawning areas for fish, feeding areas for both fish (Chapter 6) and birds (Chapter 5), breeding and resting areas for birds (Chapter 5) ect.

Saltmarsh colonization rates and species diversity were highest at the highest elevation of a restoration site. Superimposed on the importance of surface elevation is the effect of bioturbation and herbivory by the ragworm *Nereis diversicolor*. Ragworms generally occur at the level of low intertidal mudflats up to the transition to pioneer salt marsh where vegetation is sparse. In this transition area they may have a negative effect on the survival of *Salicornia* seedlings, but the effect should be viewed in relation to physical parameters of tidal currents and soil properties. In general it can be concluded that the higher the surface elevation relative to sea level the better the colonization potential for saltmarsh species (Cornu and Sadro, 2002; Williams and Orr, 2002). However, competition from less salt-tolerant species will become increasingly important with increasing elevation. In this respect, land use before and after de-embankment may play a role in the establishment of salt-marsh species.

Benthic invertebrate abundance and diversity remained lower than those in the natural sites after two years. The rate of establishment of invertebrates can be related to their mobility both as adults and as juveniles where many have planktonic stages (Atkinson *et al*., 2001). Colonisation of restoration sites also seems to be related to the availability of suitable sediments (Garbutt *et al*., 2006).

Evans *et al*. (1998) found that colonisation of a newly created area of intertidal land by invertebrates was slower than expected due to compaction caused by earth moving equipment during the restoration process. Similar conclusions were derived from Chapter 4, which showed that the most successful areas for invertebrate colonization were those with high levels of newly accreted sediments (Chapter 2). Overall the restoration site showed an increase in abundance and diversity over the two years was seen which suggests that the restoration site is on a recovery trajectory towards the natural system. Shorebirds and fish, though substantially more mobile than benthic invertebrates, also frequently differed in ecological structure between natural and restored sites. In this case of shorebirds, however, the abundance and diversity was often as high or higher in the natural areas, suggesting that the restored habitat was able to quickly develop both intertidal mudflats (Chapter 2) and saltmarsh (Chapter 3) that were suitable to shorebirds. Most bird species were only found in a subset of the study site (which were pooled to compare to natural habitats). However, each subset contributed to the integrity of a larger landscape level landscape wetland matrix, in which a diverse shorebird assemblage comparable to other UK estuarine wetlands (Colwell and Landrum, 1993) was encountered indicating that the restoration of Paull Holme Strays increased the carrying capacity of the Humber Estuary for shorebirds (Chapter 5). Furthermore, subsets of the restored site that were not heavily used by shorebirds may have achieved additional restoration goals, including provision of biogeochemical processes or habitat for marsh vegetation, invertebrates, and fish. Therefore, wetland restoration should be considered on a large, landscape scale that considers the types of habitats that already exist within the wetland and those that are lacking. Defining restoration goals to encompass the range of processes and target organisms, from annelids to shorebirds, will aid in the creation of restoration sites that contribute to the integrity of the regional wetland landscape and supplement pre-existing habitats.

The present study utilized several approaches to evaluate habitat restoration success. Assessments of faunal assemblages in restored coastal wetlands demonstrated some of the structural similarities and differences between natural and restored areas. Experimental manipulations revealed finer scale contrasts in species assemblages. These studies suggest that even if faunal colonization of restored areas occurs, biotic and aboiotic interactions among community components will not necessarily be similar as well. A combination of both monitoring and experimental studies will provide a more thorough understanding of the degree of restoration success on both structural and functional levels.

Long-term post-construction monitoring of constructed wetlands suggests that there is an initial lag before these ecosystems provide the same level of ecosystem service as the natural wetlands they were designed to replace (Craft et al., 1988, Levin et al., 1996 and Zedler, 2000). In some cases selected ecosystem services lost to natural wetland degradation or destruction require considerable time to develop (Craft et al., 2002) or are never adequately replaced (Zedler, 1993). Therefore the challenge comes from how to determine if a wetland restoration has been successful, and in how do we measure that success (Elliott *et al*., 2007). If wetland creation and restoration is to be successful at replacing habitat loss, it is important to know how much time is needed for these ecosystems to achieve equivalence to natural wetlands and once equivalence is achieved, whether they provide long-term ecological benefits.

The initial monitoring of Paull Holme Strays has shown that the landward realignment of coastal defences quickly produced intertidal mudflats on low-lying agricultural land (Chapter 2) with ecological structure as seen by the colonization of saltmarsh plants (chapter 3), invertebrates (Chapter 4), fish (Chapter 6) and birds (Chapter 5). This was possible with fairly minimal pre-treatment, and simple management, which allows tidal ingress through two uncomplicated and small breaches. What has yet to be seen is if the restoration site at Paull Holme Strays will gain ecosystem functioning.

In general, it can be concluded that managed realignment is a suitable means for the restoration of wetlands on formerly reclaimed land and offers an excellent opportunity to study the initial stages of wetland ecosystem development and primary succession.

References

Environment Agency 2003. Questions and answers. Planning for the rising tides: the Humber Estuary, update on managed realignment. Environment Agency July 2003: 1- 9. Leeds.

ABP Research.1998. Review of coastal habitat creation, restoration and recharge schemes. R.909.

Able, K.W. 2005. A re-examination of fish estuarine dependence: Evidence of connectivity between estuarine and ocean habitats.

Adam, P. 1990. *Salt-marsh Ecology*. Cambridge University Press, Cambridge.

Adam, P. 2002. Salt marshes in a time of change. Environmental Conservation. 29: 39-61.

Alhorn, F. and Kunz, H. 2002. The future of historically developed summerdikes and polders: a saltmarsh use conflict. *Littoral 2002, The Changing Coast*, 365-374.

Allen, J. R. L. 2000. Morphodynamics of holocene salt marshes: a review sketch from the Atlantic and Southern North Sea coasts of Europe. Quartenary Science Reviews. 19: 1155-1231.

Allen, J.H. 2000. Saltend benthic invertebrate and sediment survey 2000 and assessment of variability in baseline benthic community from 1998 to 2000. Institute of Estuarine and Coastal Studies, University of Hull, Hull, HU6 7RX. Unpublished Report No. Z091-M-2000-Invert.

Allen, J.H. and Elliott, M. 1999. Baseline invertebrate and sediment survey: Saltend, spring 1998. Institute of Estuarine and Coastal Studies, University of Hull, Hull, HU6 7RX. Unpublished Report No. Z091-F-99-Invert.

Allen, J.R.L. and Pye, K. 1992. *Salt marshes: Morphodynamics, conservation and engineering*. Cambridge University Press, Cambridge.

Anderson, T.J. 2001. Seasonal variability in erodibility of two temperate, microtidal mudflats. Estuarine, Coastal and Shelf Science. 53: 1- 12.

Andrews, J.E., Samways, G., Dennis, P.F. and Maher, B.A. 2000. Origin, abundance and storage of organic carbon and sulfur in the Holocene Humber

Anon. 1992. Coastal zone conservation. English Nature's rationale, objectives and practical recommendations. English Nature, Peterborough, UK.

Armitage, A.R. 2003. Community structure and trophic interactions in restored and natural estuarine mudflats: Complex trophic cascades and positive and negative effects of nutrients. Unpublished P.hD. University of California Los Angeles.

Armstrong, W., Wright, E. J., Lythe, S. and Gaynard, T. J. 1985. Pant zonation and the effects of thespring-neap tidal cycle on soil aeration in a Humber salt marsh. Journal of Ecology. 73: 323-339.

Atkinson, P.W., Crooks, S., Drewitt, A., Grant, A., Rehfisch, M.M., Sharpe, J. and Tas, C.J. 2004. Managed realignment in the UK- the first 5 years of colonization by birds. The International Journal of Avian Science. 146: 101-110.

Aubry, A. and Elliott, M. 2006. The use of environmental integrative indicators to assess seabed disturbance in estuaries and coasts: application to the Humber Estuary, UK. Marine Pollution Bulletin. 53: 175–185.

Aust, W.M., Lea, R. 1991. Soil temperature and organic matter disturbance in a forested wetland. Soil Science Society of America Journal. 55:1741-1746.

Austen, I., Anderson, T.J. and Edelvang, K. 1999. The influence of benthic diatoms and invertebrates on the erodablity of an intertidal mudflat, the Danish Wadden Sea. Estuarine, Coastal and Shelf Science. 49: 99- 111.

Bagenal, T.B. 1968. Methods for assessment of fish production in fresh waters. Ricker, W.E. (Ed.) I.B.P. Hand Book No. 3, Blackwell, Oxford.

Bakker, J. P., Esselink, P., Dijkema, K. S., van Duin, W. E. & de Jong, D. J. 2002. Restoration of saltmarshes in the Netherlands. Hydrobiologia. 478: 29-51.

Baltz, D.M., Rakocinski, C. and Fleeger, J.W. 1993. Microhabitat use by marsh-edge fishes in a Louisiana estuary. Environmental Biology of fishes, 36: 109-126.

Bannister, A. and Raymond, S. 1975. Surveying. Pitman Publishing, London.

Bannister, A., Raymond, S. and Baker, R. 1998. Surveying. Addison Wesley Longman, Harlow, Essex, UK.

Barnett, B.E. 1992. National standard methodology for benthic sampling. I. Intertidal soft sediments. National Rivers Authority. Anglian region.

Barlocher, F., Mackay, R.J. and Wiggins, G.B. 1978. Detritus processing in a temporary vernal pool in southern Ontario. Archives of Hydrobiology. 81: 269–295.

Bartels-Hardege, H.D. & Zeeck, E., 1990. Reproductive behaviour of *Nereis diversicolor* (Annelida: Polychaeta). Marine Biology. 106: 409-412.

Beatty, S.W. 1984. Influence of microtopography and canopy species on spatial patterns of forest understory plants. Ecology. 65: 1406-1419.

Beeby, A. 1993. Applied Ecology. Chapman and Hall. London.

Beeftink, W.G. 1979. The coastal salt marshes of western and northern Europe: an ecological and phytosociological approach. *In*: Chapman, V.J. (Ed), Wet Coastal Ecosystems. Elsevier, Amsterdam.

Beeftink, W.G. 1979. The structure of salt-marsh communities in relation to environmental disturbances. *In*: Jefferies, R.L. and Davy, A.J. (eds). Ecological Processes in Coastal Environments. 77-93. Blackwell, Oxford, UK.

Begon, M., Harper, J.L. and Townsend, C.K. 1990. Ecology. Individual populations and communities. Blackwell, Oxford..

Benedetti-Cecchi, L. and Cinelli, F. 1996. Patterns of disturbance and recovery in littoral rock pools: nonhierarchical competition and spatial variability in secondary succession. Marine Ecology Progress Series. 135: 145- 161.

Bertness, M.D. and Ellison, A.M. 1987. Determinants of pattern in a New England salt marsh plant community. Ecological Monographs. 57:129-147.

Bettinetti, A., Pypaert, P. and Sweerts, J. 1996. Application of an integrated management approach to the restoration project of the lagoon of Venice. Journal of Environmental Management 46: 207- 227.

Bishel- Machung, I., Brooks, R.P., Yates, S.S. and Hoover, K.L. 1996. Soil properties of reference wetlands and wetland creation projects in Pennsylvania. Wetlands. 16: 532-541.

Bischoff, A. 2002. Dispersal and establishment of floodplain grassland species as limiting factors in restoration. *Biological Conservation*, 104: 25-33.

Bishop, T., 2000. A study of the intertidal invertebrate communities at three sites on the mud flats of the Humber Estuary. Unpublished B.Sc. (Hons.) Thesis.

Bissels, S., Hözel, N., Donath, T. W., and Otte, A. 2004. Evaluation of restoration success in alluvial grasslands under contrasting flooding regimes. *Biological Conservation*, 118: 641-650.

Blaber, S.J.M., Brewer, D.T. and Salini, J.P. 1989. Species composition and biomasses of fishes in different habitats of northern Australian estuary: their occurrence in the adjoining sea and estuarine dependence. Estuarine, Coastal and Shelf Science, 29: 247- 255.

Blaber. S.J.M. 1997. Fish and fisheries of Tropical Estuaries. Chapman & Hall, London. 367pp.

Black, K.S., Tolhurst. T.J., Hagerthey, S.E. and Patterson, D.M. 2002. Working with natural cohesive sediments. J. Hydraul. Eng. 128: 1- 7.

Blamey, M. and Grey-Wilson, C. 1989. *The Illustrated Flora of Britain and Northern Europe*. Hodder & Stoughton, London.

Boesch, D.F. and Turner, R.E. 1984. Dependence of fishery species on salt marshes: the role of food and refuge. Estuaries. 7(4A): 460- 468.

Bolam, S.G. Fernandes, T.F. and Huxham, M. 2002. Diversiy, Biomass, and Ecosystem Processes in the Marine Benthos. Ecological Monographs, 72: 599-615.

Bolam, S.G. and Whomersley, P. 2005. Development of macrofaunal communities on dredged material used for mudflat enhancement: a comparison of three beneficial use schemes after one year. Marine Pollution Bulletin. 50: 40-47.

Bouma, J. 1983. Hydrology and genesis of soils with aquatic moisture regimes. p. 235-281. *In* L.P. Wilding and N.E. Smeck (eds.) Pedogensis and Soil Taxonomy. I. Concepts and Interactions, Elsevier, Amsterdam, The Netherlands.

Boydes, S. and Mazik, K. 2004. Paull Holme Strays- Accretion/ Erosion Monitoring. Report to Halcrow Group Ltd. Institute of Estuarine and Coastal Studies (IECS), University of Hull, Hull, UK

Branch, G.M. and Branch, M.L. 1980. Competition in *Bembicium auratum* (Gastropoda) and its effects on microalgal standing stocks in mangrove mud's. Oecologia. 46: 106- 114.

Brawley, H.A., Warren, S.R. and Askins, R.A. 1998. Bird use of restoration and reference marshes within the Barn Island Wildlife Management area, Stonington, Connecticut, USA. Environmental Management. 22:625- 633.

Brettar ,I. and Hofle, M.G. 2002. Close correlation between the nitrate elimination rate by denitrification and the organic matter content in hardwood forest soils of the upper Rhine floodplain (France). Wetlands. 22: 214–224.

Brinker, R. C. and Wolf, P. R. 1984. *Elementary surveying* (7th ed.). New York: Harper and Row

Brinson, M.M. 1993. A hydrogeographic classification for wetlands. U.S. Army Corps of Engineers, Washinton, DC, USA.

Brinson, M.M. and Rheinhardt, R.D. 1996. The role of reference wetlands in the functional assessment and mitigation. Ecological Applications. 6: 69-76.

Brooke, J.S. 1992. Coastal defence: the retreat option. Journal of the Institute for Water and Environmental Management 6: 151-157.

Broome, S.W., Seneca, E.D. and Woodhouse, W.W. 1988. Tidal salt marsh restoration. Aquatic Botany 32: 1- 22.

Broom, S.W., Craft, C.B., and Toomey, W.A. 2001. Soil organic matter effects on infaunal community structure in restored and created tidal marshes. *In*: Weinstein, M.P., Kreeger, D.A. (eds.) Concepts and controversies in tidal marsh ecology. Kluwer Academic Publishers, Dordrecht.

Brown, S.C., and C.R. Smith. 1998. Breeding season bird use of recently restored versus natural wetlands in New York. Journal of Wildlife Management 62:1480-1491.

Brusati, E.D., DuBowy, P.J. and Lacher, T.E. 2001. Comparing ecological functions of natural and created wetlands for shorebirds in Texas. Waterbirds 24: 371-380.

Buchanan, J.B. 1984. Sediment analysis. *In:* Holme, N.A. and McIntyre, A.D. Methods for the study of marine benthos. 2nd edition. Blackwell Scientific Publications, Oxford. Pp 41- 65.

Budd, G.C., 2007. *Hediste diversicolor*. Ragworm. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme*. Plymouth: Marine Biological Association of the United Kingdom. Available from: http://www.marlin.ac.uk/species/Hedistediversicolor.htm

Burchett, M.D., Pulkownik, A., Grant, C. and Macfarlane, G. 1998. Rehabilitation of Saline wetlands, Olympics 2000 site, Sydney (Australia) –I: management strategies based on ecological needs assessment. Marine Pollution Bulletin 37: 515- 525.

Burd, F. 1989. *The salt marsh survey of Great Britain. An inventory of British salt marshes*. Research and Survey in Nature Conservation No. 17, Nature Conservancy Council, Peterborough.

Burd, F. 1992. *Erosion and vegetation change on the salt marshes of Essex and North Kent between 1973 and 1988.* Research and Survey in Nature Conservation No.42, Nature Conservancy Council, Peterborough.

Burd, F. 1994. Guide to the Identification of British salt-marsh plants. National Rivers Authority (NRA), UK.

Burd, F. 1995. *Managed retreat: A practical guide.* English Nature, Peterborough.

Burger, J., Shisler, J. and Lesser, F.H. 1982. Avian utilization on six salt marshes in New Jersey. Biological Conservation 23: 187-212.

Caddy, J.F. 1967. Maturation of gametes and spawning in *Macoma balthica* (L.). Canadian Journal of Zoology. 45: 955-965.

Campbell, D.A., Cole, C.A. and Brooks, R.P. 2002. A comparison of created and natural wetlands in Pennsylvania, USA. Wetlands Ecolology and Management. 10: 41–49.

Castillo, J. M., Rubio-Casal, A. E., Luque, C. J., Nieva, F. J., and Figueroa, M. E. 2000. Wetlands lossby erosion in Odiel marshes (SW Spain). Journal of Coastal Research. SI 36: 134-138.

Cearreta, A., Irabien, M.J., Ulibarri, I., Yusta, I., Croudace, I.W. and Cundy, A.B. 2002. Recent salt marsh development and natural regeneration of reclaimed areas in the Plentzia estuary, N. Spain. Estuarine, Coastal and Shelf Science 54: 863- 886.

Center for Ecology and Hydrology (CEH). 2000. Modular Analysis of Vegetation Information System (MAVIS) Plot Analyzer 1.00. Produced by DART Computing for CEH at Merlewood, Cumbria. Available from the CEH website http://www.ceh.ac.uk/products_services/software/mavis.htm

Chapman, M.G., Underwood, A.J. and Skilleter, G.A. 1995. Variability at different spatial scales between subtital assemblages exposed to the discharge of sewage and two control assemblages. Journal of Experimental Marine Biology and Ecology. 189: 103- 122.

Chapman, V. J. 1974. Salt Marshes and Salt Deserts of the World. Cramer, Lehre.

Clancy, J. 1991. *Site surveying and leveling*. Butterworth Heinemann, Oxford.

Clapham, A. R., Tutin, T.G. and Moore, D.M.1987. Excursion Flora of the British Isles. Cambridge University Press, Cambridge.

Claridge, P.N., Potter, I.C. and Hardisty, M.W. 1986. Seasonal changes in movements, abundance, size composition and diversity of the fish fauna of the Severn estuary. Journal of the Marine Biology Association of the United Kingdom. 66: 229-258.

Clay, J. 2003. Flooded Britain. 60 minutes. Produced and Directed by Jonathan Clay for the Open University. Broadcast on BBC 4, October 2003.

Coats, R., Williams, P.B., Cuffe, C.K., Zedler, J.B., Reed, D., Watry, S.M. and Noller, J.S. 1995. Design guidelines for tidal channels in coastal wetlands. Report prepared for the U.S. Army Corps of Engineers. Waterways Experiment Station. 45pp.

Colclough, S., Fonseca, L., Astley, T., Thomas, K. and Watts, W. 2005. Fish utilization of managed realignments. Fisheries Management and Ecology. 12:351-360.

Cole, C.A., Brooks, R.P. and Wardrop, D.H. 1997. Wetland hydrology as a function of hydrogeomorphic (HGM) subclass. Wetlands 17: 456–467.

Colwell, M.A. and Landrum, S.L. 1993. Nonrandom shorebird distribution and finescale variation in prey abundance. Condor. 95: 94- 103.

Confer, S.R. and Niering, W.A. 1992. Comparison of created and natural freshwater emergent wetland in Connecticut. Wetlands Ecology and Management. 2: 143–156.

Cooper, N. J., Cooper, T. and Burd, F. 2001. 25 years of salt marsh erosion in Essex: Implications forcoastal defence and nature conservation. *Journal of Coastal Conservation*,9: 31-40.

Copp, G.H. and Jurajda, J. 1993. Do Small riverine fish move inshore at night? Journal of Fish Biology, 43: 229- 241.

Cornu, C. E. and Sadro, S. 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's south Slough. Restoration Ecology. 10: 474- 486.

Costa, M., Catarino, F. and Bettencourt, A. 2001. The role of salt marshes in the Mira estuary (Portugal). Wetland Ecology and Management. 9: 121- 134.

Costa, M., Catarino, F. and Bettencourt, A. 2001. The role of salt marshes in the Mira estuary (Portugal). Wetland Ecology and Management. 9: 121- 134.

Cox, R., Wadsworth, R. A. & Thomson, A. G. 2003. Long-term changes in salt marsh extent affectedby channel deepening in a modified estuary. Continental Shelf Research. 23: 1833-1846.

Craft, C.B. and Sacco, J. 2003. Long-term succession of benthic infauna communities on constructed *Spartina alterniflora* marshes. Marine Ecology Progress Series. 257: 45-58.

Craft, C.B. 2000. Co-development of wetland soils and benthic invertewbrate communities following saltmarsh creation. Wetlands Ecology and Management. 8: 197-207.

Craft, C.B., Reader, J., Sacco, J.N. and Broome, W. 1999. Twenty-five years of ecosystem development of constructed *Spartina alterniflora* (Loisel) marshes. Ecological Applications. 9: 1405- 1419.

Craft, C.B., Broome, S.W. and Campbell, C.L. 2002. Fifteen years of vegetation and soil development following brackish water marsh creation. Restoration Ecology. 10: 248-258.

Craft, C.B., Broome, S.W. and Seneca, E.D. 1988. Nitrogen, phosphorus and organic carbon pools in natural and transplanted marsh soils. Estuaries. 11: 272-280.

Craig, R.J. and Beal, K.G. 1992. The influence of habitat variables on marsh bird communities. Wilson Bulletin 104: 295-311.

Croker, R.A. and Hatfield, E.B. 1980. Space partitioning and interactions in an intertidal sand burrowing amphipod. Marine Biology. 61: 79- 88.

Crooks, S., Schutten, S., Sheern, G.D., Pye, K. and Davy, A.J. 2002. Drainage and Elevation as Factors in the restoration of salt marsh in Britain. Restoration Ecology. 10 (3): 591-602.

Cutts, D.B. 1979. Birds of the Humber estuary. The Humber estuary. The Natural Environment Research Council. Publication series.

Davidson, N.C. 1991. Estuaries, wildlife and man. A summary of nature conservation and estuaries in Great Britain. Peterborough, Nature Conservancy Council.

Davidson, N.C., D’A Laffoley, D., Doody, J.P., Way, L.S., Key, R., Drake, C.M., Pienkowski, M.W., Mitchell, R. and Duff, K.L. 1991. Nature Conservation and Estuaries in Great Britain. Nature Conservancy Council.

de Brouwer, J.F.C., Bjelic, S., de Deckere, E.M.G.T. and Stal, L.J. 2000. Interplay between biology and sedimentology in a mudflat, Biezelingse Ham, Westerschelde, The Netherlands. Cont. Shelf Res. 20: 1159- 1177.

de Deckere, E.M.G.T., de Brouwer, J.F.C. and Heip, C.H.R. 2000. Variation in macro- and meiobenthos densities over intertidal mudflats. Pages 11- 20 *in* Faunal Influence on Sediment Stability in Intertidal Mudflats. Netherlands Institute of Ecology, Yerseke, Netherlands.

de Deckere, E.M.G.T., Tolhurst, T.J. and de Brouwer, J.F.C. 2001. Destabilization of cohesive intertidal sediments by infauna. Estuarine, Coastal and Shelf Science. 53: 665- 669.

de Jonge, V.N. and de Jong, D.J. 2002. Ecological restoration in coastal areas in the Netherlands: concepts, dilemmas and some examples. Hydrobiologia 478: 7- 28.

Decho, A.W. and Fleeger, A.W. 1988. Microscale disruption of meiobenthic copepods in response to food- resource patchiness. Journal of Exparimental Marine Biology and Ecology. 118: 229- 243.

Defew, E.C., Tolhurst, T.J. and Patterson, D.M. 2002. Site-specific features influence sediment stability of intertidal flats. Hydrology and Earth System Science. 6: 971- 981.

DeLaune, R.D., Jugsujinda, A., Peterson, G.W. and Patrick, W.H. 2003. Impact of Mississippi river freshwater reintroduction on enhancing marsh accretionary processes in a Louisiana estuary. Estuarine, Coastal and Shelf Science 58: 653- 662.

Del Hoyo, J. and Elliot, J. 1996. Handbook to birds of the world, Vol. 3 Sargatal (Eds.), Lynx editions, Barcelona.

Department of the Environment. 1992. Planning Policy Guidance for the coast (PPG 20). HMSO, London.

Department of the Environment. 1995. Policy guidelines for the coast. HMSO, London.

Department of the Environment. 1996. Coastal zone management-towards best practice. HMSO, London.

Dhondt, A.A. 1988. Carying Capacity: a Confusing Concept. Oecologia generalis. 9: 337- 346.

Dierschke, J. and Bairlein, F. 2004. Habitat selection of wintering passerines in salt marshes of the German Wadden Sea. Journal of Ornithology, 145, 48-58.

Dickie, I. and Pilcher, R. 2001. Coastal flood defence: recent economic evidence. RSPB Discussion Paper.

Dijkema, K.S., Reineck, H-E. and Wolff, W.J. (eds) 1980. *Geomorphology of the Wadden Sea area.*Report 1 of the Wadden Sea Working Group. Balkema, Rotterdam, the Netherlands.

Dijkema, K. S. 1987. Changes in saltmarsh area in the Netherlands Wadden Sea after 1600.*Vegetation between land and sea* (eds A. H. L. Huiskes, C. W. P. M. Blom & J. Rozema), pp. 42-49. Dr. Junk Publishers, Dordrecht, the Netherlands.

Dixion, A.M., Leggett, D. and Weight, R.C. 1997. Habitat creation opportunities for landward coastal realignment- Essex case studies. Paper presented at the Chartered Institution for Water and Environmental Management winter meeting, 1997. Regents Park Zoo, London. Journal of the Chartered Institution for Water and Environment Management. 12: 107- 112.

Dixon, A.M., Leggett, D.J and Weight, R.C. 1998. Habitat creation opportunities for landward coastal realignment: Essex case studies. Journal of the Chartered Institution of Water and Environmental Management 12: 107-112.

Dixon, M. 2000. Sustainable Flood Defences Project Scoping Document. Environment Agency.

Doody, J. P. 1992. The conservation of British saltmarshes. Saltmarshes. Morphodynamics, conservationand engineering significance (eds J. R. L. Allen and K. Pye), pp. 80-114. Cambridge University Press, Cambridge.

Doody, J. P. 2004. Coastal squeeze - an historical perspective. Journal of Coastal Conservation, 10: 138.

Drake, P. and Arias, A.M. Ichthyoplankton of a shallow coastal inlet in southwest Spain: factors contributing to colonization and retention. Estuarine, Coastal and Shelf Science. 32: 347-364.

Duncan, C.P. and Groffman, P.M. 1994. Comparing microbial parameters in natural and constructed wetlands. Journal of Environmental Quality. 23: 298–305.

Dryer, K.R. 1998. The typology of intertidal mudflats. *In:* Black, K., Paterson, D.M. and Cramp, A. (eds), Sedimentary processes in the intertidal zone. Geological Society, London, Special Publications, 139: 115- 124.

Easytide. 2004. On-line tidal predictions available from the East Tide website www.Easytide.ukho.gov.uk.

Edgar, G.J. and Barrett, N.S. 1999. The distribution of macro invertebrates and fish in Tasmanian estuaries. Journal of Biogeography. 26: 1169-1191.

Edwards, A.M.C. 2004. ‘Planning for the Rising Tides’ seminar, presented by Tony Edwards (Humber Strategies Manager, Environment Agency, North East Regional Office, Leeds), at the University of Hull, March 2004.

Edwards, A.M.C. and Winn, P.S.J. 2006. The Humber Estuary, Eastern England: Strategic planning of flood defences and habitats. Marine Pollution Bulletin, 53: 165- 174.

Eertman, R.H.M., Korman, B.A., Stikvoort, E. and Verbeek, H. 2002. Restoration of the Sieperda Tidal Marsh in the Scheldt Estuary, The Netherlands. Restoration Ecology. 10 (3): 438- 449.

El- Sayed, M.K. 1996. Rising sea-level and subsidence of the Northern Nile delta: a case study. *In*: Milliman, J.D. and Haq, B.U. (Eds.) 1996. *Sea- level rise and coastal subsidence: causes, consequences and strategies*, Ch. 11. Kluwer Acadamic Publishers, Dordrecht.

Elliott, M. 2002. Introduction. *In*: Elliott, M. and Hemingway, K. (eds.) Fishes in Estuaries. Blackwell Science, Oxford. pp 1-9.

Elliott, M. 2002. The role of the DPSIR approach and conceptual models in marine environmental management: an example for offshore wind power. Marine Pollution Bulletin 44: iii- vii.

Elliott, M. and Dewailly, F. 1995. The structure and components of European estuary fish communities. Netherlands Journal of Aquatic Biology, 29: 397- 419.

Elliott, M. and Ducrotoy, J.P. 1991. Estuaries and Coasts: Spatial and temporal intercomparisons. Olsen and Olsen: Fredensborg, Denmark.

Elliott, M. and Hemmingway, K. 2002. *Fish in estuaries*. Blackwell Science, Oxford.

Elliott, M. and Quintino, V. 2007. The Estuarine Quality Paradox, Environmental Homesostasis and the difficulty of detecting anthropogenic stress in naturally stressed areas. Marine Pollution Bulletin, 54: 640- 645.

Elliott, M. and Taylor, C.J.L. 1989. The production ecology of the subtitle benthos of the Fourth Estuary, Scotland. In: Ros, J. (Ed.) 1989. Topics in Marine Biology: proceedings of the 22nd European Marine Biology Symposium, Barcelona, Spain, August 1987. Scientia Marina, 53 (2-3) European Marine Symposium, 22: 531-541.

Elliott, M., Nedwell, S., Jones, N.V., Read, S.J., Cutts, N.D. and Hemingway, K.L. 1998. Intertidal Sand and Mudflats & Subtidal Mobile Sandbanks: An overview of dynamic and sensitivity characteristics for conservation management of marine SACs. Scottish Association for Marine Science (UK Marine SACs Project).

Elliott, M. and Marshall, S. 2000. The Biology of Fishes of the Humber Estuary, UK. Coastal Zone Topics: Process, Ecology & Management, 4: 85-96.

Elliott, M., Whitefield, A.K., Potter, I.C., Blaber, S.J.M., Cyrus, D.P., Nordie, F.G., and Harrison, T.D. 2007. The Guild Approach to Categorizing Estuarine Fish Assemblages: A Global Review. Fish and Fisheries. 8: 241- 268.

Elliott, M., Burdon, D., Hemmingway, K.L. and Apitz, S.E. 2007. Estuarine, costal and marine ecosystem restoration: Confusing management and science- A revision of concepts. Estuarine Coastal and Shelf Science. 74: 349- 366.

Emmerson, R.H.C., Manatunge, J.M.A., Macleod, C.L. and Lester, J.N. 1997. Tidal exchanges between Orplands Managed retreat site and Blackwater Estuary, Essex. Journal of the Chartered Institution for water and environmental management. 11: 363- 372.

Environment Agency. 2005. Humber estuary flood defence strategy: Paull Holme Strays. Environmental monitoring report 2005.

Environment Agency. 2000. Planning for the Rising Tides- the Humber Estuary Management Plan. Environment Agency, North East Regional Office, Leeds.

Environment Agency. 1999. Planning for the Rising Tides- Options Consultation Document. Environment Agency, North East Regional Office, Leeds.

Environment Agency.1995. *A guide to the understanding and management of salt marshes*. R & D note 324, Environment Agency, Bristol.

Environmental Action Plan, February 2002 (Issue 5, Revision 0). Environment Agency, North East Regional Office, Leeds.

Esselink, P., Zijlstra, W., Dijkema, K.S. and van Diggelen, R. 2000. The effects of decreased management on plant-species distribution patterns in a saltmarsh nature reserve in the Waden Sea. Biological Conservation. 93: 61- 76.

Estuary: emphasizing human impact on storage changes. *In:* Shennan, I. and Andrews, J. (Eds). Holocene Land-Ocean Interaction and Environmental Change around the North Sea. Geological Society, London. Special publications 166: 145- 170.

Evans, P.R., Ward, R.M., Bone, M. and Leakey, M. 1998. Creation of Temperate-Climate Intertidal Mudflats: Factors Affecting Colonization and Use by Benthic Invertebrate and their Bird Predators. Marine Pollution Bulletin. 37: 535-545.

Eybert, M.C., Geslin,T., Questiau, S. Feuteun, E. 2003. Shorebird community variations indicative of a general perturbation in the Mont-Saint-Michel bay (France). Comptes Rendus Biologies. 326:140- 147

Fairbairn, S.E. and Dinsmore, J.J. 2001. Local and landscape- level influences on wetland bird communities of the Prairie Pothole Region of Iowa. Journal of the Iowa Academy of Science 108: 8-14.

Featherstone, N. and Lee- Elliott, E. 2003. Reeds Natural Almanac 2004. Nautical Data, Westborne, Hampshire.

Federal Interagency Committee for Wetland Delineation (FICWD). 1989. Federal manual for identifying and delineating jurisdictional wetlands. USACE, USEPA, USDI-FWS and USDA-SCS. Cooperative technical publication, United States Government Printing Office, Washington, DC, USA.

Finlayson, C.M. and Eliot, I. 2001. Ecological assessment and monitoring of coastal wetlands in Australia’s wet-dry tropics: a paradigm for elsewhere? Costal Management 29: 105- 115.

Ford, R.B., Thrush, S.F. and Probert, P.K. 1999. Macrobenthic colonization of disturbances on an intertidal sandflat: the influence of the season and buried algae. Marine Ecology Progress Series. 191: 163- 174.

Frankhauser, S. 1995. Protection vs. retreat: the economic costs of sea-level rise. Environment and Planning A 27: 299-319.

French, P.W. 2006. Managed Realignment- The developing story of a comparatively new approach to soft engineering. Estuarine Coastal and Shelf Science 67: 409- 423.

French, C.E., French, J.R., Clifford, N.J. and Watson, C.J. 2000. Sedimentation-erosion dynamics of abandoned reclamations: the role of waves and tides. Continental Shelf Research 20: 1711- 1733.

Frenkel, R.E. and Morlan, J.C. 1991. Can we restore our salt-marshes? Lessons from the Salmon River, Oregon. Northwest Environmental Journal. 7:119-135.

Frid, C. and James, R. 1989. The marine invertebrate fauna of British coastal salt-marsh. Hilarctic Ecology. 12: 9-15.

Galatowitsch, S.M., and Vander Valk, A.G., 1994. Restoring prairie wetlands: An Ecological Approach. Iowa State University Press.

Galatowitsch, S.M., and Vaner Valk, A.G. 1996. Characteristics of recently restored wetlands in the prairie pothole region. Wetlands 16:75-83.

Galatowitsch S.M. and Vander Valk A.G. 1996. Vegetation and environmental conditions in recently restored wetlands in the prairie pothole region of the USA. Vegetation 126: 89–99.

Garbutt, R.A., Reading, C.J., Wolters, M., Gray, A.J. and Rothery, P. 2006. Monitoring the development of intertidal habitats on former agricultural land after managed realignment of coastal defences at Tollesbury, Essex, UK. Marine Pollution Bulletin. 53: 155- 164.

Garbutt, A . 2004. Correspondence with Angus Garbutt, Centre for Ecology and Hydrology, 08/04.

George, T.L. and Zack, S. 2001. Spatial and temporal considerations in restoring habitat for wildlife. Restoration Ecology 9: 272-279.

Gill, J., O'Riordan, T. and Watkinson, A. 2002. Redesigning the coast - science workshop. Proceedings of a workshop organised by the Tyndall Centre and the Centre for Socio-Economic Research on the Global Environment.

Gillanders, B.M. 1997. Patterns of abundance and size structure in the blue groper: evidence of links between estuaries and coastal reefs. Environmental Biology of Fisheries. 49: 153- 173.

Glowka, L., Burhenne-Guilmin, F., Synge, H., McNeely, J.A. and Gundling, L. 1994. A Guide to the Convention on Biological Diversity- Environmental Policy and Law Paper No. 30. World Conservation Union, also Know as the International Union for the Conservation of Nature. Gland, Switzerland.

Golet, F.C. 1986. Critical issues in wetland mitigation: A scientific perspective. National Wetlands Newsletter 8:3-6.

Golet, F.C., Calhoun, A.J.K., DeRagon, W.R., Lowry, D.J. and Gold, A.J. 1993. Ecology of red maple swamps in the glaciated northeast: a community profile. US Fish and Wildlife Service, Washington, DC, USA.

Goodwin, P., Mehta, A. J. & Zedler, J. B. 2001. Tidal wetland restoration: An introduction. Journal of Coastal Research. SI27: 1-3.

Gray, J.S. and Elliott, M. 2009. Ecology of Marine Sediments. Oxford University Press

Gray, A.J., & Benham, P.E.M. 1990. *Spartina anglica - a research review*. Institute of Terrestrial Ecology.

Gray, A.J., Marshall, D., and Raybould, A.F. 1991. A century of evolution *in*
Gray, J.S. 1974. Animal-sediment relationships. Oceanography and Marine Biology: An Annual Review. 12: 223- 261.

de Groot, S.J. 1971. On the interrelationships between morphology of the alimentary tract, food and feeding behavior in flatfishes (Pisces: Pleuronectiformes). Netherlands Journal of Sea Research. 5:121-196.

Goss-Custard, J.D., 1985. Foraging behaviour of wading birds and the carrying capacity of estuaries. *In*: Sibly, R.M., Smith, R.H. (Eds.), Behavioural Ecology: Ecological Consequences of Adaptive Behaviour. Blackwells, Oxford, pp. 169–188.

Goss-Custard, J.D., Charman, K., 1976. Predicting how many wintering waterfowl an area can support. Wildfowl. 27: 157–158.

Goss- Custard, J.D., Stillman, R.A., West, A.D., Caldow, R.W.G., McGrorty, S. 2002. Carrying Capacity in Overwintering Migratory Birds. Biological Conservation. 105: 27- 41.:

Headrich, R.L. 1983. Estuarine Fishes. Chapter 7. *In:* B. Ketchum (Ed.) Estuaries and enclosed seas. Elsevier, Amsterdam. Pp. 183- 207.

Halcrow Group LTD. 2002. Thrngumbald Flood Alleviation Scheme.
Halcrow, with CSERGE and CRU 2002. Managed Realignment Review. Report to DEFRA and the Environment Agency. Policy Research project FD 2008.

Haltiner, J., Zedler, J.B., Boyer, K.E., Williams, G.D. and Callaway, J.C. 1997. Influence of physical processes on the design, functioning and evolution of restored tidal wetlands in California (USA). Wetlands Ecology and Management. 4 (2) SI: Hydrologic Restoration of Coastal Wetlands: 73- 91.

Haltiner, J.P. and Williams, P.B. 1987. Hydraulic design in salt marsh restoration. Proceedings National Symposium of Wetland Hydrology. Association of Wetland Managers. Chicago, IL. Sept. 16-18.

Hanslip, V.L. 2002. The Application of a Conceptual Model, a Decision Tree, and a Logical Framework Approach to Managed Realignment Schemes: a case studying the Humber Estuary, UK. Msc thesis, University of Hull, Hull, UK.

Hemesath, L.M. and Dinsmore, J.J. 1993. Factors affecting bird colonization of restored wetlands. Prairie Naturalist 25: 1-11.

Hemingway, K.L. and Elliott, M. 2002. Field Methods. *In*: Fishes in Esutuaries (Eds.) Elliott, M. and Hemingway, K.L. Blackwell science, Oxford.

Hiscock, K. and Kimmance, S. 2002 Review of Current and Historical Seabed Biological time-series studies in the UK and near Europe. Joint Nature Conservation Committee. Report #336.

Hölzel, N. and Otte, A. 2003. Restoration of a species-rich flood meadow by topsoil removal and diaspore transfer with plant material. Applied Vegetation Science. 6: 131-140.

Howard, D.M. and Howard, P.J.A. 1993. Relationships between CO2 evolution, moisture-content and temperature for a range of soil types. Soil Biology and. Biochemistry. 25: 1537–1546.

Howes, B.L., Howarth, R.W., Teal, J.M. and Valiela, I. 1981. Oxidation- reduction potentials in a salt marsh: spatial patterns and interactions with primary production. Limnology and Oceanography. 26:350-360.

HR Wallingford 2002. National Flood Risk Assessment 2002. Executive Summary. HR Wallingford Report n0 EX 4722 to the Environment Agency.
HR Wallingford, Halcrow Maritime and Flood Hazard Research Center. 2000.

Hubbard, C. E. 1984. *Grasses*. Penguin Books Ltd, Harmondsworth, Middlesex, UK.

Hughes, R.G. and Paramor, O.A.L. 1999. Salt marsh erosion and management of salt marsh restoration; the effects of infaunal invertebrates. Aquatic Conservation- Marine and Freshwater Ecosystems 9: 83- 95.

Hughes, R.G. and Paramor, O.A.L. 2004. On the loss of saltmarshes in south east England and methods for their restoration. Journal of Applied Ecology. 41: 440-448.

Huiskes, A.H.L., Koutstaal, B.P., Herman, P.M.J., Beeftink, W.G., Markusse, M.M. and De Munck, W. 1995. Seed dispersal of halophytes in tidal salt marshes. Journal of Ecology. 83: 559-567.

Hulme, M., Jenkins, G.J., Lu, X., Turnpenny, J.R., Mitchell, T.D., Jones, R.G., Lowe, J., Murphy, J.M., Hassell, D., Boorman, P., McDonald, R. and Hill, S. 2002. Climate Change Scenarios for the United Kingdom: The UKCIP02 Scientific Report. Tyndall Centre for Climate Change Research, School of Environmental Sciences, University of East Anglia, Norwich, UK. 120pp.

Humber Bibliography. 2007. http://www.humber-bib.hull.ac.uk/index.php

Humber Estuary and Coast. 1994. Institute of Estuarine and Coastal Studies, University of Hull. Hull, UK.

Hunter, R.G. and Faulkner, S.P. 2001. Denitrification potential in restored and natural bottomland hardwood wetlands. American Society of Soil Science Journal. 65: 1865–1872.

Hutchings, M.J. and Russell, P.J. 1989. The seed regeneration dynamics of an emergent salt-marsh. Journal of Ecology. 77: 615-637.

Institute of Terrestrial Ecology. 1998. *Managed realignment at Tollesbury and Saltram*. Report to MAFF, project CSA 2313.

Intergovernmental Panel on Climate Change 2001. Third Assessment Report, Science Report, Cambridge, University Press.

Jackson, A., 2000. *Hydrobia ulvae*. Laver spire shell. *Marine Life Information Network: Biology and Sensitivity Key Information Sub-programme*. Plymouth: Marine Biological Association of the United Kingdom. Available from: http://www.marlin.ac.uk/species/Hydrobiaulvae.htm

Jager, Z. 1999. Floundering: Processes of tidal transport and accumulation of larval flounder (*Platichthys flesus*) in the Ems- Dollard nursery. PhD Thesis, University of Amsterdam.

Jarvie, H.P., Neal, C. and Robson, A.J. 1997. The geography of the Humber catchments. Science of the Total Environment 194: 87- 99.

Jenny, H. 1961. Derivation of the state factor equations of soil and ecosystems. Soil Science Society of America Proceedings. 25:385-388.

Joint Nature Conservation Committee. 1997. *Coasts and seas of the United Kingdom*. Coastal Directories series, Joint Nature Conservation committee, Peterborough.

Jones, L.H. 1978. Studies of heavy metals in the sediments, selected animals and algae from the Humber estuary with particular reference to *Hediste diversicolor*. Ph.D. Thesis. Hull University.

Jones, N.V. 1988. The Humber ecosystem- an assessment. *In*: The Humber Ecosystem. Proceedings of a conference in support of the European Year of the Environment. Humber Estuary Committee. University of Hull. Pp. 81- 85.

Jones, N.V. and Key, R.S. 1989. The Biological value of mudflats in the Humber Estuary (England): Areas proposed for land reclamation. Proceedings of the International Symposium on Coastal Ecosystems: Planning, Pollution and Productivity. 2: 19-32.

Kentula, M.E., Brooks, R.P., Gwin, S.E., Holland, A.D., Sherman, A.D. and Sinfneos, J.C. 1992. An approach to improved decision making in wetland restoration and creation. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, OR, USA.

Kentula, M.E. 2000. Perspectives on setting success criteria for wetland restoration. Ecological Enginerring. 15: 199-209.

Kirkman, L.K., Goebel, P.C., West, L., Drew, M.B., and Palik, B.J. 2000. Depressional wetland vegetation types: a question of plant community development. Wetlands 20: 373–385.

Kneib, R. 1997. The role of tidal marshes in the ecology of estuarine nekton. Oceanography Marine Biology: An Annual Review. 35: 163-220.

Kneib, R. 2000. Salt- marsh ecoscapes and production transfers by estuarine nekton in the southeastern United States. *In*: Concepts and Controversies in tidal marsh ecology (Weinstien, M.P. and Kreeger, D.A. eds.) Kluwer Academic Publishers, The Netherlands. pp. 267- 291.

Koutstaal,B.P., Markusse, M.M. and De Munck, W. 1987. *In*: Aspects of seed dispersal by tidal movements. Vegetation between land and sea. Structures and processes (eds Huiskes, A.H.L., Blom, C.W.P.M. and Rozema, J.), pp. 226-233. Dr. W. Junk Publishers, Dordrecht, the Netherlands.

Krebs, C.J. 1994 *Ecology*: *the experimental analysis of distribution and abundance*, 4th edition. Addison- Wesley Publishers, Inc., Menlo Park, CA.

Kusler, J.A. 1990. Views on scientific issues relating to restoration and creation of wetlands. p. 217-230. *In* G. Bingham, E.H. Clark, L.V. Haywood and M. Leslie (Eds.) Issues in wetland protection. The Conservation Foundation, Washington, DC, USA.

Kusler, J.A. and Kentula, M.A. 1989. Wetland creation and restoration: the status of the science. EPA/600/3-89/038. U.S. Environmental Protection Agency, Environmental Research Laboratory, Corvallis, Organ, USA.

Kusler, J.A., and M.E. Kentula. 1990. Executive summary. Pages xvii-xxv. In J.A. Kusler and M.E. Kentula (eds.). Wetland creation and restoration: The status of the science. Island Press, Washington, DC.

LaGrange, T.G., and Dinsmore J. J. 1989. Plant and animal community responses to restored Iowa wetlands (USA). Prairie Naturalist 21: 39-48.

Langlois, E., Bonis, A., and Bouzille, J.B. 2003. Sediment and plant dynamics in saltmarshes pioneer zone: *Puccinellia maritime* as a key species? Estuarine, Coastal and Shelf Scinece 56: 239-249.

Lapointe, L. and Bourget, E. 1999. Influence of substratum heterogeneity scales and complexity on a temperate epibenthic marine community. Marine Ecology Progress Series. 189: 159- 179.

Le Preste, P.G. (Ed). 2002. Governing Global Diversity. Ashgate, Aldershot, UK.

Ledoux, L., Crooks, S., Jordan, A. and Turner, R.K. 2000. Implementing EU biodiversity policies: UK experiences. Land Use Policy. 17: 257- 268.

Ledoux, L., Turner, K. and Cave, R. 2002. The use of scenarios in catchments and coastal zone management: the case of the Humber Estuary, UK. Littoral 2002, The Changing Coast 211-219.

Lee, E.M. 1998. *The implication of future shoreline management on protected habitats in England and Wales.* R&D Technical Report W150, Environment Agency, Bristol.

Lee, M. 2001. Coastal Defence and the Habitats Directive: predictions of Habitat change in England and Wales. The Geographical Journal 167: 39-56.

Lenanton, R.C.J. and Potter, I.C. 1987. Contribution of estuaries to commercial fisheries in temperate Western Australia and the concept of estuarine dependence. Estuaries. 10: 28-35.

Levin, L.A., Talley, D. and Thayer, G. 1996. Succession of macrobenthos in a created salt-marsh. Marine Ecology Progress Series. 141: 67-82.

Levine, J. M., Brewer, J. S. and Bertness, M. D. 1998. Nutrients, competition and plant zonation in a New England salt marsh. Journal of Ecology. 86: 285-292.

Levinton, J S. 2001. Marine Biology — Function, Biodiversity and Ecology. Oxford University Press, Oxford/New York.

Lin, H.J. and Shao, K.T. 1999. Seasonal and diurnal changes in subtropical mangrove fish assemblage. Bulletin of Marine Science. 65; 775- 794.

Little, C. 2000. The Biology of Soft Shores and Estuaries. Oxford University Press, England.

Long, L.L. and Ralph, C.J. 2001. Dynamics of habitat use by shorebirds in estuarine and agricultural habitats in northwest California. Wilson Bulletin 113: 41-52.

Long, S.P. and Mason, C.F. 1983. *Saltmarsh Ecology*. Blackie, Glasgow.

Lowry, D.J. 1990. Restoration and creation of palustrine wetlands associated with riverine systems of the glaciated northeast. Pages 267-280. In J.A. Kusler and M.E. Kentula (eds.). Wetland creation and restoration: The status of the science. Island Press, Washington, DC.

Malakoff, D. 1998. Restored Wetlands Flunk real world test. Science. 280: 371-372.

Maltby, E. 1987. Soils science as a base for freshwater wetland mitigation in the northeastern United States. p. 17-52. *In:* J.S. Larson and C. Neill (eds.) Mitigating Freshwater Wetland Alterations in the Glaciated Northeastern United States: and Assessment of the Science base. Environmental Institute, University of Massachusetts, Amherst, MA, USA.

Mander, L., Cutts, N.D., Mazik, K. 2007. The impact on newly created habiat on estuarine bird communities. Estuarine Coastal Shelf Science. 175: 163-174.

Marshall,S. 1995. The structure and functioning of the fish assemblage of the Humber Estuary, UK. Unpublished P.hD. University of Hull.

Marshall, S. and Elliott, M. 1998. Environmental influences on the fish assemblage of the Humber estuary, UK. Estuary, Coastal and Shelf Science., 46: 175-184.

Marshall, S. and Elliott, M. 1997. A comparison of univariate and multivariate numerical and graphical techniques for determining inter and intraspecifc feeding relationships in estuarine fish. Journal of Fish Biology. 53: 526-545

Marsluff, J.M. Ewing, K. 2001. Restoration of fragmented landscapes for the conservation of birds: a general framework and specific recommendations for urbanizing landscapes. Restoration Ecology 9: 280-292.

Mazik, K., Simth, J.E., Leighton, A. and Elliott, M. 2007. Physical and biological devlopment of a newly breached managed realignment site, Humber estuary, UK. Marine Pollution Bulletin. 55: 564-578.

Mazik, K. 2004. The influence of a petrochemical discharge on the bioturbation and erosion potential of an intertidal estuarine mudflat, Humber estuary (U.K.). Unpublished Ph.D. Dissertation, University of Hull.

Mazik, K. 2006. Paull Holme Strays Monitoring Program 2006: Benthic Invertebrate Monitoring. . Institute of Estuarine and Coastal Studies, University of Hull, Hull, HU6 7RX. Unpublished Report No. ZBB634-F-2006.

Mazik, K. and Elliott, M. 2000. The effects of chemical pollution on the bioturbation potential of estuarine intertidal mudflats. Helgoland Marine Research. 54: 99-109.

McDermontt, J.J. 1983. Food web in the surf zone of an exposed sandy beach along the mid- Atlantic coast of the United States. *In*: McLachlan, A. and Erasmus, T. (eds.) 1983. Sandy beaches as ecosystems. Junk., The Netherlands.

McHugh, J.L. 1967. Estuarine nekton. *In:* Lauff, G.H. (ed.), Estuaries. Vol. 83. American Association of Advance Science special publication, Washington, DC, pp.581-619.

McLachlan, A. 1983. Sandy beach ecology: a review. *In*: McLachlan, A. and Erasmus, T. (eds.) 1983. Sandy beaches as ecosystems. Junk., The Netherlands.

McLusky, D.S. 1989. The Estuarine Ecosystem. Blackie, Glasgow.

McLusky, D.S., Bryant, D.M. and Elliott, M. 1992. The impact of land- claim on macro benthos, fish and shore birds on the Forth Estuary, eastern Scotland. Aquatic Conservation: Marine and Freshwater Ecosystems. 2: 211- 222.

McLusky, D.S. and Elliott, M. 2004. The Estuarine Ecosystem: Ecology, Threats and Management. Wiley, New York, 2004.

McMahon, R.F. and Britton, J.C. 1985. The relationship between the vertical distribution, thermal tolerance, evaporative water loss rate, and behavior on emergence. *In*: six species of mangrove gastropods, from Hong Kong. In: Morton, B. and Dudgeon, D. (Eds.), The Malacofauna of Hong Kong and Southern China, vol. 2. Hong Kong Univ. Press, Hong Kong, pp. 563- 582.

Meadows, P.S. and Tait, J. 1989. Modification of sediment permeability and shear strength by two burrowing invertebrates. Mar. Biol. 101: 75- 82.
Meets, J., Dewicke, A. and Hamerlynck, O. 1993. Seasonal composition and spatial distribution of hyperbenthic communities along estuarine gradients in the Westerschelde. Netherlands Journal of Aquatic Ecology. 27: 359- 376.

Melvin, S.L. and Webb, J.W. 1998. Differences in avian communities of natural and created *Spartina alterniflora* slat marshes. Wetlands 18: 59-69.

Mendelssohn, L.A., McKee, K.L. and Patrick, W.H. 1981. Oxygen deficiency in *Spartina alterniflora* roots: metabolic adaptation to anoxia. Science. 214:439-441.

Meng, L., Powell, J.C. 1999. Linking juvenile fish and their habitats: An example form Narrgansett Bay, Road Island. Estuaries, 22: 905- 916.

Mermillod-Blondin, F., Marie, S., Desrosiers, G., Long, B., de Montety, L., Michaud, E. and Stora, G. 2003. Assessment of the spatial variability of intertidal benthic communities by axial tomodensitomertry: importance of fine-scale heterogeneity. Journal of Exparimental Marine Biology and Ecology. 287: 193-208.

Millennium Ecosystem Assessment. 2005. ECOSYSTEMS AND HUMAN WELL-BEING: WETLANDS AND WATER Synthesis. World Resources Institute, Washington, DC.

Minello, T.J. and Webb, J.W. 1997. Use of natural and created *Spartina alterniflora* salt- marsh by fishery species and other aquatic fauna in Galveston Bay, Texas, USA. Marine Ecology Progress Series. 154: 165- 179.

Ministry of Agriculture, Fisheries and Food. 1993. *A strategy for flood and coastal defense in England and Wales.* MAFF PB 1471.

Ministry of Agriculture, Fisheries and Food. 1993. *Coastal defense and the environment: A guide to good practice.* MAFF PB 1191.

Ministry of Agriculture, Fisheries and Food. 1995. *Shoreline Management Plans: A guide for coastal defense authorities.* MAFF PB 2197.

Mitsch, W.J. and Gooselink, J.G. 1993. *Wetlands*. Van Nostrand Reinhold, New York, NY, USA.

Morgan, P.A. and Short, F.T. 2002. Using functional trajectories to track constructed saltmarsh development in the Great Bay Esturary, Mane/ New Hampshire, USA. Restoration Ecology. 10: 461- 473.

Morrison, M., Hartill, B., Shankar, U. and Drury, J. 2001. Pahurehure Inlet, Manukau Harbor habitat map. NIWA Information Series No. 12

Mortimer, R.J.G., Davey, J.T., Krom, M.D., Wadson, P.G., Frickers, P.E. and Clifton, R.J. 1999. The effect of macrofauna on porewater profile and nutrient fluxes in the intertidal zone of the Humber estuary. Estuarine and Costal Shelf Science. 48: 683-699.

Murby, P. 2001. The causes, extent and implications of intertidal change- a regional view from eastern England. Ph.D. University of Hull.

Myers, R.A., MacKenzie, B.R., Bowen, K.G. and Barrowman, N.J. 2001. What is the Carrying Capacity for Fish in the Ocean? A Metaanalysis of Population Dynamics of North Atlantic Cod. Canadian Journal of Fish Aquatic Science. 58: 1464- 1476.

Myrick, R.M. and Leopold, L.B. 1963. Hydraulic Geometry of a small tidal estuary. United States Geological Survey Professional Paper. 422 B.

Nagelkerken, I., Dorenbosh, M., Verberk, W.C., de la Moriniere, C.E. and van der Velde, G. 2000. Day- night shifts of fishes between shallow- water biotopes of a Caribbean bay, with empisis on the nocturnal feeding of Haemulidae and Lutjanidae. Marine Ecology Progress Series. 194: 55- 64.

National Appraisal of Assets at risk from flooding and coastal erosion, Report for MAFF. Reports No SR 573 and TR 107

National Research Council. 1992. *Restoration of aquatic ecosystems*. National Academy Press, Washington D.C., USA.

Nicholls, R.J. 2004. Coastal Flooding and Wetland Loss in the 21st Century: Changes Under the SRES Climate and Socio- Economic Scenarios. Global Environment Change. 14: 69- 86.

Odum, E.P. 1969. A research challenge: evaluating the productivity of coastal and estuarine water. *In*: Hamilton, P. and MacDonald, K.B. (Eds.) 1980. Estuarine and Wetland Processes with emphasis on modeling. Plenum Press, London.

Odum, E.P. 1980. The Status of three ecosystem-level hypotheses regarding salt marsh estuaries: tidal subsidy, outwelling, and detritus- based food chains. Pp. 485- 495. *In*: Kennedy, V. Estuarine Perspectives, Academic, New York.

OFFWELL WETLAND SURVEY. 2004. Ecological Survey Methods. Available from the Offwell Woodland & Wildlife Trust website http://www.offwell.free-online.co.uk

Paratley, R.D. and Fahey, T.J. 1986. Vegetation- environment relations in a confer swamp in central New York. Bulletin of the Torrey Botanical Club. 113: 357-371.

Passell, H.D. 2000. Recovery of bird species in minimally restored Indonesian tin strip mines. Restoration Ecology 8: 112-118.

Paterson, A.W. and Whitfield, A.K. 2000. Do shallow- water habitats function as refuge for juvenile fishes? Estuarine Coastal and Shelf Science, 51: 359- 364.

Pearson, T.H. and Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. Oceanorgaphy and Marine Biology: An Annual Review. 16: 229- 311.

Pestrong, R. 1965. The development of drainage patterns on tidal marshes. Stanford University Publications Geol. Sci. Vol. X. No. 2. 87pp.

Peterson, C.H. 1991. Intertidal zonation of marine invertebrates in sand and mud. American Scientist. 79: 236- 249.

Peterson, G.W. and Turner, R.E. 1994. The value of salt marsh edge vs. interior as habitat for fish and decapod crustations in a Louisiana tidal marsh. Estuaries. 17: 235-262.

Pethick, J.S. 2002. Estuarine and tidal wetland restoration in the United Kingdom: policy versus practice. Restoration Ecology. 10 (3): 431- 437.

Phillips, J. D. 1986. Coastal submergence and marsh fringe erosion. Journal of Coastal Research. 2: 427-436.

Pilcher, R., Burston, P. and Davis, R. 2002. Seas of change- the potential for intertidal habitat creation around the coast of mainland Britain. Royal Society for the Protection of Birds, Tring.

Pinckney, J. and Sandulli, R. 1990. Spatial autocorrelation analysis of meiofaunal and microalgal populations on an intertidal sandflat: scale linkage between consumers and resources. Estuarine, Coastal and Shelf Science. 30: 341- 353.

Ponnamperuna, F.N. 1972. The chemistry of submerged soils. Advances in Agronomy. 24:29-96.

Porter, E. 1793. Pollution in four industrialized estuaries. HMSO publications.

Potter, I.C., Beckley, L.E., Whitefield, A.K. and Lenanton, R.C.J. 1990. Comparisons between the roles played by estuaries in the life cycles of fishes in temperate Western Australia and Southern Africa. Environmental Biology of Fishes, 28: 143- 178.

Potter, I.C., Claridge, P.N., Hyndes, G.A. and Clarke, K.R. 1997. Seasonal, annual and regional variations in ichthyofaunal composition in the inner Severn Estuary and inner Bristol Channel. Journal of the Marine Biological Association of the United Kingdom. 77: 507-525.

Poxton, M.G and Allouse, S.B. 1982. Water quality criteria for marine fisheries. Aquacultural Engineering. 1:153-191.

Pye, K. & French, P.W. 1992. *Targets for coastal habitat creation*. Unpublished report to English Nature, Peterborough (F72-04-22/ES22).

Pywell, R. F., Bullock, J. M., Hopkins, A., Walker, K. J., Sparks, T. H., Burke, M. J. W. and Peel, S. 2002. Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. Journal of Applied Ecology. 39: 294-309.

Race, M.S. and Christie, D.R. 1982. Coastal zone development: mitigation, marsh creation and decision- making. Environmental Management. 6: 317- 328.

Race, M.S. and Christie, D.R. 1982. Coastal zone development: mitigation, marsh creation and decision- making. Environmental Management. 6: 317- 328.

Rackowski, J.P. and Pikitch, E.K. 1989. Species Profiles: life histories and environmental requiremnts of coastal fishes and invertebrates (Pacific Southwest)- Pacific and Speckled Sanddabs. 11(107) (Ed.).

Rand, T. A. 2002. Variation in insect herbivory across a salt marsh tidal gradient influences plant survival and distribution. Oecologia, 132: 549-558.

Ranwell, D S. 1964. Spartina salt marshes in Southern England. II. Rate and seasonal pattern of sediment accretion. Journal of Ecology, 52: 79-94.

Ratccliffe, P.J. 1979. The proceedings of a joint symposium on the Humber estuary, The University of Hull/ Humber advisory group.

Ratti, J.T., Rocklage, A.M., Giudice, J.H., Garton, E.O. and Golner, D.P. 2001. Comparison of avian communities on restored and natural wetlands in North and South Dakota. Journal of Wildlife Management 65: 676-684.

Recher, H.F. 1966. Some aspects of the ecology of migrant shorebirds. Ecology 47: 393- 407.

Rees, I.E.S., Nicholaidou, A. and Laskaridou, P. 1977. The effects of storms on the dynamics of shallow benthic organisms. 11th EMBS. Pergamon Press, Oxford. Pp. 465- 474.

Richardson, H. 2004. Correspondence with Helen Richardson, Humber Ecologist, Environment Agency, Willerby, Hull. Between June 2004 and September 2004.

Riley, J.D. 1979. The Biology of young fish in the Humber estuary. The Natural Environment Research Council. Publication series. 20: 20- 24.

Riley, J.D., Symonds, D.J. and Woolner, L. 1981. On the factors influencing the distribution of 0-group demersal fish in coastal waters. Rapporte et Proces-verbaux des Reunions de Conseil Permanent Internatinal pour l'Exploration de la Mer. 178:223- 228.

Ringrose, S., Vanderpost, C. and Matheson, W. 2003. Mapping ecological conditions in the Okavango delta, using fine and coarse, resolution systems including simulated SPOT vegetation imagery. International Journal of Remote Sensing 24: 1029- 1052.

Roberson, A.I. and Duke, N.C. 1990. Recruitment, growth and residence time of fishes in a tropical Australian mangrove system. Estuarine, Coastal and Shelf Science, 31: 723-743.

Robertson, P. 2004. Habitat mapping and development at the Paull Holme Strays Managed Realignment site on the Humber estuary. MSc University of Hull.

Rodwell, J.S. (Ed) 2000. *British Plant Communities*. Volume 5 — Maritime communities and vegetation of open habitat. Cambridge University Press, Cambridge.

Rooker, J.R. and Dennis, G.D. 1991. Diurnal, lunar and seasonal changes in a mangrove fish assemblage off southwestern Puerto Rico. Bulletin of Marine Science. 49: 648- 698.

Rose, F. 1981. *The Wild Flower Key*. Frederick Warne, London.

Ruiz-Jaen, M.C., Aide, T.M. 2005. Restoration success: how is it being measured? Restoration Ecology 13: 569-577.

Sanders, H.L. 1968. Marine Benthic Diversity: A Comparative Study. The American Naturalist. 102: 243- 282.

Scatolini, S.R. and Zedler, J.B. 1996. Epibenthic invertebrates of natural and constructed marshes of San Diego Bay. Wetlands. 16: 24-37.

Schwimmer, R. A. 2001. Rates and processes of marsh shoreline erosion in Rehoboth Bay, Delaware, USA. Journal of Coastal Research.17: 672-683.

Shaffer, P.W. and Ernst, T.L. 1999. Distribution of SOM in freshwater wetlands in the Portland, Oregon area. Wetlands. 19: 505–516.

Shenker, J.M. and Dean, J.M. 1979. Larval and juvenile fishes in saltmarshes. Estuaries, 2: 154- 163.

Simenstad, C., Reed, D. and Ford, M. 2006. When is Restoration Not? Incorporating Landcape- Scale Processes to Restore Self- Sustaining Ecosystems in Coastal Wetland Restoration. Ecological Engineering. 26: 27- 39.

Smith, J.E. In prep. Variation in sediment properties in relation tidal cycles and benthic community structure at the Paull Holme Strays managed realignment site, Humber estuary, U.K. Unpublished MSc Dissertation, University of Hull.

Snelgrove, P.V.R. 1999. Getting to the Bottom of Marine Biodiversity: Sedimentary Habitats. Bioscience. 49: 129- 138.

Snow, A. A. & Vince, S. W. (1984) Plant zonation in an Alaskan salt marsh. II. An experimental study of the role of edaphic conditions. *Journal of Ecology*, 72: 669-684.

Soil Survey Staff. 1975. Soil Taxonomy: a basic system of soil classification for making and interpreting soil surveys. United States Government Printing Office. Washington, DC, USA.

Solt, M.H., Genthner, M.H., Daniels, W.L., Groover, V.A., Nagle, S. and Haering, K.C. 2000. Comparison of soil and other environmental conditions in constructed and adjacent palustrine reference wetlands. Wetlands. 20:671-683.

Southward, A.J. 1970. Improved methods of sampling post larval young fish and macroplankton. Journal of the Marine Biology Association UK. 50:689-712.
Spartina anglica. *Advances in Ecological Research*. 21: 1-62.
Stapleton, C.M. 1994. The Humber Estuary and coast. The University of Hull.

Spencer, D.R., Perry, J.E. and Silberhorn, G.M. 2001. Early secondary succession in bottomland hardwood forests of southeastern Virginia. Environmental Management. 27: 559–570.

Stenzel, L.E., Huber, H.R. and Page, G.W. 1976. Feeding behavior and diet of the long billed curlew and willet. Wilson Bulletin 88: 314- 332.

Stewart, A., Pearman, D.A. & Preston, C.D. 1994. *Scarce plants in Britain*. Joint Nature Conservation Committee, Peterborough.

Tabor, R. 2003. Turning the tides for salt marshes. British Wildlife, October 2003.

Thayer, G.W., editor. 1992. Restoring the nations marine environment. Maryland Sea Grant, College Park, Maryland, USA.

Thiel, R., Sepulveda, A., Kafemann, R. and Nellen, W. 1995. Environmental factors as forces structuring the fish community of the Elbe Estuary. Journal of Fish Biology, 46: 47-69.

Thom, R.M. 2000. Adaptive management of coastal ecosystem restoration projects. Ecological Enginerring. 15: 365- 372.

Thomas, B.E. and Connolly, R.M. 2001. Fish use of subtropical salt- marshes in Queensland, Australia: relationships with vegetation, water depth and distance onto the marsh. Marine Ecology Progress Series. 209: 275-288.

Thompson, R.C., Wilson, B.J., Tobin, M.L., Hill, A.S. and Hawkins, S.J. 1996. Biologically generated habitat provision and diversity of rocky shore organisms at a hierarchy of spatial scales. Journal of Experimental Marine Biology and Ecology. 202: 73- 84.

Thrush, S.F. Pridmore, R.D. and Hewitt, J.E. 1994. Impacts of soft-sediment macrofauna: the effects of spatial variations on temporal trends. Ecological. Applications. 4: 31- 41.

Tiner, R.W. 1995. Wetland restoration and creation. Pp 517-534. *In:* W.A. Nierenberg (Ed.). *Encyclopedia of Environmental Biology*. Volume 3. Academic Press, New York, N.Y.

Toft A.R. and Maddrell, R.J. (Eds) 1995. A Guide to Understanding and Management of Salt marsh. R & D note 324. National Rivers Authority, Bristol.

Tolhurst, T.J., Jesus, B., Brotas, V. and Patterson, D.M. 2003. Diatom migration and sediment armoring- an example from the Tagus Estuary, Portugal. Hydrobilogia. 503: 183- 193.

Turner, R.K., and Adger, N. 1995. Assessing the economic costs of sea level rise. Environment and Planning A 27: 1777-1796.

Underwood, A.J. and Barret, G. 1990. Experiments on the influence of oysters on the distribution, abundance and size of the gastropod *Bembicium auratum* in the mangrove swamp in New South Wales, Australia. Joural of Experimental Marine Biology and Ecology. 137: 25- 45.

Underwood, A.J. and Chapman, M.G. 1996. Scales of spatial patterns of distribution of intertidal invertebrates. Oecologica. 107: 212- 224.

Underwood, A.J. and Chapman, M.G. 1998. A method for analyzing spatial scales in variation in composition of assemblages. Oecologica. 117: 570- 578.

Ungar, I. A. 1998. Are biotic factors significant in influencing the distribution of halophytes in saline habitats? *The Botanical Review*. 64: 176-199.

Van Cleve, F.B., Leschine, T., Klinger, T. and Simenstad, C. 2006. An evaluation of the influence of natural science in regional-scale restoration projects. Environmental Management. 37: 367-379.

van der Wal, R., van Lieshout, S., Bos, D. and Drent, R. H. 2000. Are spring staging brent geese evicted by vegetation succession? Ecography, 23: 60-69.

Verhoeven, J.T.A., Whigham, D.F., van Logtestijn, R. and O'Neill, J. 2001. A comparative study of nitrogen and phosphorus cycling in tidal and non-tidal riverine wetlands. Wetlands 21: 210–222.

Vepraskas, M.J. 1992. Redoximorphic features for identifying aquatic conditions. North Carolina Agricultural Research Service Technical Bulletin 301.

Warnock, S.E. and Takekawa. J.Y. 1995. Habitat preferences of wintering shorebirds in a temporally changing environment. Journal of American Ornithologists. 112: 920-930.

Warren, R.S. and Niering, W.A. 1993. Vegetation change on a northeast tidal marsh: Interaction of sea-level rise and marsh accretion. Ecology. 74: 96-103.

Warwick, R.M. 1990. Measuring ecological stress: Variations on a theme. Marine Biology. 106: 305- 308.

Watts, C.W., Tolhurst, T.J., Black, K.S. and Whitmore, A.P. 2003. In situ measurements of erosion shear stress and geotechnical shear strength of the intertidal sediments of the experimental managed realignment scheme at Tollesbury, Essex, UK. Estuarine Coastal and Shelf Science. 58: 611- 620..

Weinstein, E.W. 1999. Right triangle- Pythagorean Theorem. Available form the Math world website. www.mathworld.wolfram.com/RightTriangle.html

Weinstein, M.P., Litvin, S.y., Guida, V.G. 2005. Considerations of habitat linkages, estuarine landscapes, and the trophic spectrum in wetland restoration design. Journal of Coastal Reserch. 40: 51-63.

Wells, F.E. 1986. Distribution of mollusks across a pneumatophore boundary in a small bay in northwestern Australia. Journal of Molluscan Studies. 52: 83- 90.

Weltzin, J.F., Bridgham, S.D., Pastor, J., Chen, J. and Harth, C. 2003. Potential effects of warming and drying on peatland plant community composition. Global Change Biology. 9: 141- 151.

West T.L., Clough, L.M. and Ambrose, W.G. 2000. Assessment of function in an oligohaline environment: lessons learned by comparing created and natural habitats. Ecological Engineering. 15: 303- 321.

Whittecar G.R. and Daniels W.L. 1999. Use of hydrogeomorphic concepts to design created wetlands in southeastern Virginia. Geomorphology 31: 355–371.

Widdows, J., Brinsley, M.D., Salkeld, P.N. & Lucas, C.H., 2000. Influence of biota on spatial and temporal variation in sediment erodability and material flux on a tidal flat (Westerschelde, The Netherlands). Marine Ecology Progress Series. 194: 23-37.

Widdows, J., Brown, S., Brinsley, M.D., Salkeld, P.N. and Elliott, M. 2000. Temporal changes in intertidal sediment erodibility: influence of biological and climatic factors. Continent and Shelf Science Research. 20: 1275-1289.

Wiegert, R.G., Pomeroy, L.R. and Wiebe, W.J. 1981. *The Ecology of a Salt Marsh.* Springer-Verlag, New York.

Wiehe, P. O. 1935. A quantitative study of the influence of tide upon the population of *Salicornia europaea. Journal of Ecology*. 23: 323-333.

Williams, G. and Hall, M. 1987. The loss of coastal grazing marshes in south and east England, with special reference to east Essex, England. Biological Conservation. 41: 243-253.

Williams, P. B. and Orr, M. K. 2002. Physical evolution of restored breached levee salt marshes in the San Fransisco Bay estuary. Restoration Ecology. 10: 527-542.

Winn, P.J.S., Young, R.M. and Edwards, A.M.C. 2003. Planning for the rising tides: the Humber Estuary shoreline management plan. The Science of the Total Environment 314- 316: 13- 30.

Winter, T.C. 1999. Relation of streams, lakes, and wetlands to groundwater flow systems. Journal of Hydrogeology. 7: 28–45.

Wolff, W. J. 1992. The end of a tradition: 1000 years of embankment and reclamation of wetlands in the Netherlands. Ambio. 21: 287-291.

Wolters, M., Bakker, J.P., Bertness, M.D., Jefferies, R.L. and Möller, I. 2005. Saltmarsh erosion and restoration in south-east England: squeezing the evidence requires realignment. *Journal ofApplied Ecology*. 42: 884-851.

Wolters, M., Garbutt, A. and Bakker, J.P. 2005. Plant colonization after managed realignment: the relative importance of diaspore dispersal. Journal of Applied Ecology. 42: 770-778.

Wolters, M., Geertsema, J., Chang, E.R., Veeneklaas, R.M., Carey, P.D. and Bakker, J.P. 2004. Astroturf seed traps for studying hydrochory. Functional Ecology. 18: 141-147.

Wolters, M. and Bakker, J. P. 2002. Soil seed bank and driftline composition along a successional gradient on a temperate salt marsh. Applied Vegetation Science. 5: 55-62.

Zedler, J.B. 1993. Canopy architecture of natural and planted cordgrass marshes: selecting habitat evaluation criteria. Ecological Applications. 3: 123-138.

Zedler, J.B. 1996. Tidal wetland restoration: a scientific perspective and southern California focus. Report No. T-038, California Sea Grant College System, University of California, La Jolla, CA.

Zedler, J. B. 1996. Coastal mitigation in southern California: The need for a regional restoration strategy. Ecological Applications. 6: 84-90.

Zedler, J. B. 1996. Ecological issues in wetland mitigation: An introduction to the forum.Ecological Applications,. 6: 33-37.

Zedler, J.B. 2000. Progress in wetland restoration ecology. Trends in Ecology and Evolution. 15: 402-407.

Zedler, J.B. and Callaway, J.C. 1999. Tracking wetland restoration: do mitigation sites follow desired trajectories? Restoration ecology. 7 (1): 69- 73.

Zedler, J.B. and Callaway, J.C. 2000. Evaluating the progress of engineered tidal wetlands. Ecological Engineering. 15: 211-225.

Zedler, J.B. and Lindig- Cinseros, R. 2000. Functional equivalency of restored and natural salt marshes. Pp 596- 582 *In:* M.P. Weinstein and D.A. Kreeeger, editors. Concepts and Controversies in Tidal Marsh Ecology. Kluwer Academic Publishers, Dordrecht, The Netherlands.

Zedler, J.B., and J.C. Callaway. 1999. Tracking wetland restoration: Do mitigation sites follow desired trajectories. Restoration Ecology. 7: 69-73.

Zedler, J.B., Callaway, J.C. and Sullivan, G. 2001. Declining biodiversity: why species matter and how their functions might be restored in Californian tidal marshes. BioScience. 51:1 005-1017.

Zedler, J.B. 2001. *Handbook for restoring tidal wetlands*. CRC Press, New York, US.

Zedler, J.B., Josselyn, M. and Onuf, C. 1982. Restoration techniques, research, and monitoring: Vegetation. *In*: Josselyn, M. (ed) Wetland restoration and enhancement in California. 63-72. California Sea Grant. Report No. T-CSGCP-007, La Jolla, CA.

Zedler, J.B., Nordby, C.S. and Kus, B.E. 1992. The ecology of Tijuana Estuary: a National Estuarine Research Reserve. NOAA Office of Coastal Resource Management, Sanctuaries and Reserves Division, Washington DC, US.

Zedler, J.B. and Lindig-Cisneros, R. 2000. Functional equivalency of restored and natural saltmarshes. *In*: Weisnstein, M.P., Kreeger, D.A. (Eds.) Concepts and Controversies in Tidal Marsh Ecology. Kluwer, Dordrecht. 565- 582.

Zobel, M., van der Maarel, E. and Dupré, C. 1998. Species pool: the concept, its determination and significance for community restoration. Applied Vegetation Science. 1: 55-66.