Influence of vegetation on sediment accumulation in tidal saltmarshes: An integrated field and modelling study

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A thesis submitted for the degree of
Doctor of Philosophy

October 2016
Declaration

I, Darryl Christopher Price, confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

Name: ...........................................

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Abstract

Tidal saltmarshes in the UK, and especially in the estuaries of southeast England, have been subject to degradation and erosion over the last few decades, primarily caused by sea-level rise and coastal squeeze. This is of great concern as saltmarshes play a key coastal defence role and function as important habitats for a variety of plant and animal species. Sediment accumulation is critical for the maintenance of marsh elevation within the tidal frame and for delivering the aforementioned functions and services. Key questions still remain, however, regarding the processes that govern deposition and the role of vegetation in enhancing sedimentation. The research presented in this thesis focuses on a case study at the managed realignment site and adjacent natural marshes at Tollesbury, in the Blackwater Estuary, Essex, UK. An innovative combination of secondary data analysis, intensive field campaigns and numerical modelling is used to advance our understanding of the processes controlling sedimentation.

Results from a series of hierarchical deployments of sediment traps indicate the role of vegetation in marsh development is less clear-cut than previously thought. Gross sedimentation rates were statistically higher in non-vegetated areas, and vegetation has no overall influence over trapping efficiency. However, sediment retention was higher at the vegetated sampling points. This implies that vegetation acts primarily to inhibit resuspension by waves rather than by facilitating deposition from tidal flows.

The performance of the realignment site in terms of vegetation development has been poor compared to other schemes, with only half of the site having been colonised after 18 years of tidal inundation. No evidence was found for erosion of the natural marshes surrounding the site, and it is shown that previous estimates of marsh erosion are erroneously high. It is thus speculated whether the realignment of large stretches of coastal defences are actually necessary or worthwhile.
Acknowledgements

First and foremost, I would like to thank my parents, Lianne and Bob Price for their unwavering help, support and encouragement throughout the entirety of my PhD. Without their assistance, I would have been unable to undertake this research, as they helped to plan and accompanied me on the majority of my fieldwork and carried out half of the seasonal data sampling. A particular thanks to my brother, Justin, who was also made to endure getting up in the early hours of many Sunday mornings in order to trudge around a very muddy saltmarsh in Essex with us.

Thanks must go to my primary supervisor, Prof Jon French, who inspired me to get this research off the ground, and provided expert guidance throughout the lifetime of this project. As well as assisting with the planning of fieldwork, he also helped considerably with the hydrodynamic modelling that was undertaken.

A big thank you to Dr Helene Burningham, my second supervisor, for the many hours spent poring over my fieldwork results trying to tease out relationships between the multitude of variables in my data. Her suggestions for which analyses to carry out were particularly gratefully received. Helene also assisted with data collection during the two intensive fieldwork monitoring periods.

A particular mention must be given to Ian Patmore, the fieldwork technician in the Geography Department for his considerable support and ideas to ensure the smooth running of my entire fieldwork programme. From coming up with a simple tool to make around 1,000 pins for my sediment traps to meticulously planning the installation and retrieval of the ADCP, Ian was always enthusiastic and willing to help. The data collection aspect my research would certainly have suffered without him. Janet Hope and Tula Maxted also aided in sourcing fieldwork equipment and provided advice on laboratory methods.

Many thanks to Miriam Fernandez Nunez, Mandy Green and Lizzie Gardner for helping out during the intensive fieldwork periods. My fieldwork programme was partly funded by grants from the British Society of Geomorphology, and I was awarded a one-year studentship by the Geography Department.

I’d also like to thank David Welsh, from the Environment Agency for supplying the aerial photography of the study area from 1997, 2008 and 2013; and Timothy Poate, from Plymouth University for providing detailed tidal predictions for fieldwork planning.
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<td>ADCP</td>
<td>Acoustic Doppler Current Profiler</td>
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<tr>
<td>BADC</td>
<td>British Atmospheric Data Centre</td>
</tr>
<tr>
<td>BODC</td>
<td>British Oceanographic Data Centre</td>
</tr>
<tr>
<td>CCA</td>
<td>Canonical Correspondence Analysis</td>
</tr>
<tr>
<td>CEH</td>
<td>Centre for Ecology and Hydrology</td>
</tr>
<tr>
<td>CSTM</td>
<td>Community Sediment Transport Model</td>
</tr>
<tr>
<td>DAFOR</td>
<td>Dominant, Abundant, Frequent, Occasional, Rare (a scale to estimate vegetation cover)</td>
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<tr>
<td>Defra</td>
<td>Department for Environment, Food and Rural Affairs</td>
</tr>
<tr>
<td>DEM</td>
<td>Digital Elevation Model</td>
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<tr>
<td>dGPS</td>
<td>Differential Global Positioning System</td>
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<td>DHI</td>
<td>Danish Hydraulic Institute</td>
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<tr>
<td>DSM</td>
<td>Digital Surface Model</td>
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<tr>
<td>DTM</td>
<td>Digital Terrain Model</td>
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<tr>
<td>EA</td>
<td>Environment Agency</td>
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<tr>
<td>ECOMSED</td>
<td>Estuarine Coastal Ocean Model with Sediment Transport</td>
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<tr>
<td>GIS</td>
<td>Geographic Information System</td>
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<td>Global Positioning System</td>
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<tr>
<td>HAT</td>
<td>Highest Astronomical Tide</td>
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<tr>
<td>iCOASST</td>
<td>Integrating Coastal Sediment Systems</td>
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<tr>
<td>INS</td>
<td>Inertial Navigation System</td>
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<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
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<tr>
<td>ITE</td>
<td>Institute of Terrestrial Ecology (now CEH)</td>
</tr>
<tr>
<td>LiDAR</td>
<td>Light Detection And Ranging</td>
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<tr>
<td>MAFF</td>
<td>Ministry of Agriculture, Fisheries and Food (now Defra)</td>
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<td>Mean High Water Neaps</td>
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<td>Met Office Integrated Data Archive System</td>
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<td>Nature Conservancy Council</td>
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<td>NERC</td>
<td>Natural Environment Research Council</td>
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<td>NSE</td>
<td>Nash-Sutcliffe Efficiency</td>
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<td>Ordnance Survey</td>
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<td>PCA</td>
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<td>Root Mean Square Error</td>
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<td>Regional Ocean Model System</td>
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<td>SSSI</td>
<td>Site of Special Scientific Interest</td>
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<tr>
<td>USGS</td>
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<td>Window of Opportunity</td>
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1. Introduction

1.1 Importance of Saltmarshes

Saltmarshes are wetlands of the upper intertidal zone characterised by normally dense strands of halophytic vegetation (Chapman, 1960; Frey and Basan, 1985; Allen and Pye, 1992; Mitsch and Gosselink, 2007). They are typically located in low wave energy environments, fronted by mudflats or sandflats, and may naturally transition into freshwater marshes or terrestrial habitats (Steers, 1977; Adam, 2002). Saltmarshes are regularly flooded by astronomical tides and these inundations supply the marsh surface with sediment, increasing its elevation in relation to the tidal frame, and allowing vegetation to colonise and grow (Stoddart et al., 1989; Craft et al., 1993; Allen, 2000).

Historically, saltmarshes and other wetlands worldwide were often viewed as barren wastelands with little or no economic value, and even considered as unattractive and unwholesome (Patrick, 1994; Boorman, 1999). As such, many were drained, impounded or filled and converted for human use (Stevenson and Kearney, 2009). Coastal saltmarshes in particular were seen as having little benefit and large areas have been reclaimed from the sea for agriculture, industry, housing or recreational areas such as caravan parks and marinas (King and Lester, 1995). According to Davidson et al. (1991), at least 85% of British estuaries have been subject to land claim, with over 25% of intertidal areas being lost from many estuaries.

It is now widely recognised that wetlands are amongst one of the most valuable habitats in the world, and are vitally important ecologically and to environmental management (Keddy, 2010; Barbier et al., 2011). Intertidal saltmarshes and mudflats in particular, although only occupying a small fraction of the Earth’s surface (4%; Spencer and Harvey, 2012), provide a wide variety of functions due to their location at the interface between freshwater and marine environments (Frey and Basan, 1985; Lefevre and Dame, 1994).

1.1.1 Human Uses

Despite the long-held view of saltmarshes as wastelands, humans have made use of them for centuries wherever they are found. The oldest and most common use of marshes is as pastureland for sheep, cattle and other livestock, which graze on the vegetation present (Doody, 1992b). Grazing on marshes may have begun as early as 4000 BC on the Danish Baltic coast, 600 BC in The Netherlands, and from colonial
times in the Americas (Queen, 1977; Adam, 2002). In the UK, people in the medieval period first realised the potential of using marshes for pasture, and it became such a popular practice that the Essex saltmarshes supported 18,000 sheep by 1086 (Leggett and Dixon, 1994). Very few mature marshes in Europe will not have been grazed at some point, as the tradition spread throughout the continent (Davy et al., 2009). Although marshes are not widely used for grazing in North America today, they are in China, Chile and parts of Europe, with marsh-fed lamb regarded as a delicacy in France and Britain (Gedan et al., 2009). As well as being eaten by livestock, some marsh vegetation is also harvested for human consumption. Marsh samphire (Salicornia europaea), the most commonly collected plant, has been commercially gathered along the UK east coast for generations, and is eaten as a vegetable in salads, pickled or boiled (Adnitt et al., 2007). Although not as regularly eaten as in the past, with only 100 harvesters left in Norfolk (EFTEC et al., 2006), samphire has been enjoying a surge of popularity as a luxury good in London’s markets again.

Marsh plants are also harvested for animal fodder, and used to feed cattle and sheep as hay (Oliver, 1925; Queen, 1977). In the United States and Canada, Spartina patens was cut annually and became known as ‘saltmarsh hay’ as it was so highly regarded for its nutrition. In the late 1800s, products made from animals from marsh farms in North America were highly sought after, especially butter made from the milk of cows that grazed on marsh hay (Gedan et al., 2009). During the 20th century, however, haymaking declined considerably on both sides of the Atlantic (Adam, 1990). The other main traditional use of marsh plants was as thatching for farmhouse roofs, with the shoots from the coarse Spartina alterniflora or the reed Phragmites communis mainly chosen for their strength (Bandaranayake, 1998). Reed cutting was carried out in 12 of the 155 UK estuaries covered by the Nature Conservancy Council’s Estuaries Review in 1991 (Davidson et al., 1991), with the highest yields coming from the Tay Estuary in Scotland, where commercial mechanical harvesting began in 1975 (Adam, 1990; Adnitt et al., 2007). Dried marsh vegetation has also been used as a packing material (Queen, 1977).

Saltmarshes have a long history of being home to a number of small-scale industries. Salt extraction from European marshes was introduced in Roman times, becoming more commercialised in the seventh and eighth centuries in The Netherlands and France (Gedan et al., 2009). Along the Wadden Sea coast in northern Europe, peat was dug from marshes and tidal bogs and burned, with salt collected from boiling the ashes (Knottnerus, 2005). By the end of the Middle Ages, most tidal bogs had
disappeared due to the destructive nature of this process, and the last saltern closed in the Netherlands in the 18th century. In warmer climates, such as southern Europe and Western Australia, salt is produced from evaporating sea water, with evaporation ponds often constructed on mudflats, saltmarshes and mangroves (Adam, 2002). In the United States, the San Francisco Bay area was heavily modified for industry following the start of the Gold Rush in 1849; over 90% of the 76,000 ha of marshes were subsequently drained and diked and converted to farmland and salt production ponds (Nichols et al., 1986; Mitsch and Gosselink, 2007).

As is the case for coastal environments, saltmarshes provide a range of recreational experiences and amenities, such as rambling, dog walking, drawing and painting, swimming, fishing and boating (Dixon et al., 1998). Much of the appeal of using marshes in this way stems from the distinctive aesthetic qualities of the marshland landscape, as well as the proximity to the wider marine environment and its associated leisure activities (De Groot et al., 2002). The wildlife found in marshes also attracts birdwatchers and wildfowlers (Doody, 1992b; Myatt et al., 2003). Wildfowling, the shooting of ducks, geese and other birds for food and sport, has been popular in saltmarshes in the UK for centuries, and was very important to some coastal economies of Essex in the 19th century (Gramolt, 1961).

1.1.2 Habitat

Saltmarshes have been known to be ecologically important in Europe and the United States since the 1950s (Steers, 1959; Chapman, 1960; Frey and Basan, 1985). This is in part due to their high rates of primary productivity, making them some of the most biologically productive environments on the planet (Adams et al., 2012). Saltmarsh plants are able to produce high levels of biomass during the growing season (Lillebø et al., 1999), leading to the creation of large amounts of organic matter from its decomposition and production of particulate and dissolved organic carbon (Odum, 1968; Nixon, 1980; Odum, 1980). Grazing is usually limited in saltmarsh environments, such that the food chain is dominated by decomposers. This results in the marsh sediments and channel systems exhibiting high nutrient loads (Wolff et al., 1979; Dame, 1982; Dankers et al., 1984), which sustains a variety of wildlife including birds, fish and invertebrates (Dixon et al., 1998; Cattrijsse and Hampel, 2006).

Although the saltmarsh ecosystem may appear simple at first glance, with only a small number of plant types present, they actually display considerable vegetational
diversity (Adam, 1981). This diversity arises from the position of marshes on the boundary between the terrestrial and marine environments, and there are approximately 40 species of halophytes around the UK (Oliver, 1913; Adam, 1990). Halophytes are specially adapted plants that colonise, germinate and grow in highly saline environments, and are generally restricted to saltmarshes by poor competitive ability under non-saline conditions (Chabreck, 1988). The number of species at a particular point in an individual marsh at a particular time is influenced by a number of environmental and biotic factors, such as water depth, and water or soil salinity, climate, land use history, as well as the size of the marsh itself (Adam, 1981, 1990). Saltmarshes in the UK, and especially in East Anglia, are home to a number of rare plant species, such as *Suaeda vera*, *Sarcocornia perennis* and *Inula crithmoides* (Davidson, 1990; Garbutt et al., 2008). Most of these species are found in upper marsh areas, and are becoming less common as a result of human activities (Ranwell, 1981). Boorman and Ranwell (1977) documented that Maplin sands in Essex contains the only extensive population in Britain, and possibly Europe, of the rare marsh grass *Spartina maritima*, a pioneer species.

Intertidal estuarine and coastal environments, such as saltmarshes and mudflats provide important habitats for a range of bird species worldwide (Boorman, 1999). Shorebirds, waders, wildfowl and passerines utilise saltmarshes as feeding, roosting and nesting areas that are relatively free of disturbance by humans and other mammals (Brusati et al., 2001; Hughes, 2004). Although the marsh vegetation itself is eaten by relatively few species such as some geese and ducks, many more eat the seeds of the plants, as well as the invertebrates found in abundant numbers on mudflats, in salt pans on the marsh surface, and in the creek network (Brown and Atkinson, 1996). Saltmarshes are particularly important in areas where meadowland has been converted to agriculture and no other suitable habitat is available; Oystercatchers (*Haematopus*), Lapwing (*Vanellus*) and Redshank (*Tringa*) are frequently cited examples of birds in the UK associated with former meadows now found extensively in saltmarshes (Boorman, 1999). According to Brindley et al. (1998) saltmarshes support approximately 45% of the population of Common Redshank (*Tringa totanus*) breeding in the UK.

As well as supporting a number of resident bird species, saltmarshes are critical sites for migrating birds (Mander et al., 2007). The majority of shorebirds are equatorial migrants, travelling from the high Arctic to southern Africa or South America for winter, stopping at estuaries or bays for feeding. In North America, migrating birds may reach 100,000 – 200,000 on a single count in individual estuaries, with four large bay
systems on the east and west coasts of Canada and the United States supporting more than one million shorebirds during the migration season (Burger et al., 1997). Intertidal habitats in the UK are an important stopover site for birds migrating along the East Atlantic Highway, which runs from the Arctic down along the west coast of Africa. Each winter, UK estuaries support roughly 1.4 million waders and another 1.4 million wildfowl migrating south (Holt et al., 2012). As well as these migrating birds, British coastal wetlands provide wintering grounds to many waterbirds, which benefit from the fairly mild climate and large tidal ranges, meaning that intertidal areas rarely freeze (Crowther, 2007).

Juvenile fish have long been known to make extensive use of saltmarshes as nursery habitats during high tide when they move in from nearby mudflats and estuary channels (Mathieson et al., 2000; Tinch, 2003; Green et al., 2009). The intricate creek network found in most marshes adds complex topography to the system, providing a variety of depths and tidal regimes for different species (Desmond et al., 2000). The shallow water and dense vegetation provide protection and shelter from predators to small fish and other marine creatures such as crustaceans (Dixon et al., 1998; Cattrijsse and Hampel, 2006); with thick swaths of Spartina being especially important as refuge for amphipods from larger fish in New England saltmarshes (Vince et al., 1976). The higher summer temperatures and lower salinity found in the marshes and creeks is preferred by younger fish, but not older ones, thus reducing the number of predators in these habitats (Wiegert and Pomeroy, 1981).

Fluxes of nutrients and organic matter produced by decomposition of the marsh plants are especially important in sustaining fisheries. Detritus is an important food resource for young fish, whether eaten directly or indirectly via the consumption of insects and crustaceans (Boesch and Turner, 1984). The vegetation also provides an increased surface area for food sources, while some species, such as grey mullet feed on algae growing on the creek banks (King and Lester, 1995; Colclough et al., 2005). High levels of saltmarsh primary productivity result in some estuaries having very high biodiversity. For example, Mont Saint-Michel Bay in north west France supports at least 100 fish species despite the relatively short inundation period within its marshes and mudflats (Laffaille et al., 2000). In the UK, many of the juvenile species supported in estuarine systems later become part of important fish stocks and have much higher yields due to the range of benefits provided by marsh systems (Boesch and Turner, 1984). In Essex, these commercially fished species include herring, sea bass, and eels as well as lobster, oyster and crab (Green et al., 2009).
Although birds and fish are generally the most valued animals present in saltmarshes, intertidal environments support a wide range of other species. In the UK there are at least 293 resident saltmarsh invertebrate species, of which 148 are found exclusively in saltmarshes, ranging from small molluscs to large burrowing crabs (Doody, 1992b). They are typically found in high densities both above and below the sediment surface, with Dixon et al. (1998) reporting 12,000 individuals of the marine gastropod, *Hydrobia ulvae*, per square metre above ground, and 2,000 below-ground in Essex saltmarshes. Other small fauna include beetles, shore bugs, worms and spiders (Boorman, 2003; Petillon and Garbutt, 2008). Amphibians may also be associated with saltmarsh environments; in the UK, 16 of the 49 colonies of the threatened natterjack toad (*Bufo calamita*) are found on marshes in the north west of England and south west Scotland (McGrath and Lorenzen, 2010). The occurrence of low salinity pools that develop in the early summer corresponds with their breeding season and provides protection from predators (Adnitt et al., 2007). A number of mammals are also found in saltmarshes throughout North America and Europe, including voles, rats, mice, rabbits, hares, foxes and badgers, all attracted by the food sources available (Dixon et al., 1998; Van der Wal et al., 1998, 2000; Hotaling et al., 2010; Eaton et al., 2011; DeSa et al., 2012).

The importance of saltmarsh as a habitat has been recognised in law both nationally and internationally (Doody, 1992b; Crooks et al., 2002). On a global scale, the Ramsar Convention on Wetlands of International Importance is an intergovernmental treaty adopted in 1971 whose mission is “the conservation and wise use of all wetlands through local, regional and national actions and international cooperation” (Ramsar Convention Secretariat, 2013: 7). As of January 2013, 163 countries have signed the treaty, and over 2,060 wetlands, covering more than 197 million hectares have been designated as Ramsar Sites. Of this total, 453 sites are categorised as intertidal marshes, and cover 40 million hectares. In the UK, 80% of saltmarshes have been designated as Sites of Special Scientific Interest (SSSI), a legal designation for land of special nature conversation interest introduced in the late 1940s to protect the habitat from development (Davidson, 1990; Brampton, 1992). Several estuaries are covered by one large SSSI designation, for example The Wash, while other areas have multiple SSSI units, such as the Firth of Forth with eight, and the Humber estuary with four (Davidson, 1990). Other initiatives within the UK include National Nature Reserves and Biodiversity Action Plan habitats to prevent the loss of these environments. At an EU level, saltmarshes are protected under the 1992 Habitats
Directive; throughout Europe, over 3,800 km$^2$ of saltmarshes and salt meadows have been designated as a Special Area of Conservation (Evans, 2006). As well as aiming to achieve a ‘favourable conservation status’ for the different habitats, the Directive seeks to maintain the same area of each habitat, such that new saltmarsh must be created each time natural saltmarsh is lost (Ledoux et al., 2000; Crooks et al., 2002).

1.1.3 Ecosystem Functions and Services

Intertidal environments, such as saltmarshes and mudflats, deliver a wide range of ecosystem functions and services, defined as ‘the benefits that people obtain from ecosystems’ by the Millennium Ecosystem Assessment (Luisetti et al., 2011). These benefits can be classified into four categories: provisioning services, the goods obtained from ecosystems such as food and fuel; regulating services, including water purification, nutrient retention and erosion control; cultural services, non-material benefits such as recreation, spiritual and aesthetic experiences; and supporting services, which are necessary for all the other ecosystem services, including primary production, nutrient cycling and providing habitat (World Resources Institute, 2003; Sousa et al., 2010a; Luisetti et al., 2011). These functions and services can have significant value in monetary terms to national economies; in the UK this has been estimated at £48 billion, or 3.46% of the national income, despite the fact that coastal margin habitats only make up 0.6% of the UK’s land area (Jones et al., 2011).

Much of the work into ecosystem services in saltmarshes has stemmed from the outwelling theory proposed by Eugene Odum in the 1960s, in which he argued that marshes supply nutrients and organic matter to the wider coastal system, thereby supporting marine productivity (Odum, 1968). The general absence of large herbivores is a key factor favouring the preferential role of microbial populations in transforming organic carbon into bacteria-rich detritus, much of which is exported to the estuarine environment (De la Cruz, 1978; Nixon, 1980; Wiegert et al., 1981; Lefeuvre and Dame, 1994). Teal (1962) reported that 45% of the organic material produced in a US east coast marsh was transferred to estuarine and coastal waters. In their classic work at Sapelo Island, Georgia, Odum and de la Cruz (1967) found that the organic detritus produced in the marsh was the key link between primary and secondary productivity, with the protein content of *Spartina alterniflora* detritus over double that of the living grass (24% compared with 10%). Although the majority of studies support the
outwelling theory, researchers have found evidence along the New England coast that some estuaries and marshes import nutrients instead (Wolaver and Spurrier, 1988).

This decomposition and export of nutrients is just one part of the key role that saltmarshes play in nutrient cycles, as they transform and recycle nutrients through a variety of processes. Marshes predominately receive nutrients from tidal inflows and suspended sediment, but also from terrestrial runoff, rivers as well as the atmosphere and rainfall (EFTEC et al., 2006; Barbier et al., 2011). Nitrogen is the main nutrient involved, and different marshes may be either a sink or source of nitrogen depending on, for example, tidal levels and energy, salinity, species composition, oxygen release, as well as the age of the marsh (Pomeroy and Imberger, 1981; Asjes and Dankers, 1994; Sousa et al., 2008). In general, young marshes, dominated by pioneer vegetation are net importers of nutrients, whereas older mature marshes export nutrients into estuaries and coastal systems (Hughes, 2004). Nitrogen is removed and stored in the marsh sediment by nitrogen-fixing bacteria and algae, taken up into plant roots and tissues for growth or recycled by denitrification (Dankers et al., 1984; Craft et al., 1988; Reddy and Graetz, 1988).

Another supporting service provided by saltmarshes is oxygen production. Every plant produces oxygen via photosynthesis in their leaves and shoots, and some, including Spartina alterniflora, export oxygen into the marsh soil through their roots and alter its pH (Lee et al., 1999). As Koretsky et al. (2000) found, this process increases respiration and microbial activity in the surrounding sediment, thus enhancing the rate of nutrient cycling and increasing productivity.

The main regulating services that these functions support is the filtering of water entering the marsh, and the retention of nutrients and heavy metals, thereby reducing or preventing eutrophication and pollution of the habitat (Gedan et al., 2009). Wetlands in general are considered so effective in removing contaminants from water, that they are constructed in some areas instead of traditional wastewater treatment works to purify water before it enters adjacent ecosystems (Bastian and Benforado, 1988; Zedler and Kercher, 2005). As water passes through a marsh system, it slows due to the baffling effect of the vegetation, and sediment settles from the water column. Nutrients, especially phosphorus, bind to suspended sediment particles, so are trapped by the marsh plants (Zedler and Kercher, 2005) and either retained and buried in the marsh soil, or assimilated into the plant biomass. Different plant species are more effective at sequestering different nutrients than others; Sousa et al. (2010a) found that Sarcocornia fructicosa, Sarcocornia perennis and Atriplex portulacoides were most efficient for
nitrogen (N) sequestration, and *Spartina maritima* best for phosphorus (P) retention in two Portuguese estuaries. The chemical composition of the soil also influences nutrient fixation, with acidic, clayey sediment with high aluminium and iron concentrations increasing the P-binding capacity and thus its removal from the water column (Smil, 2000).

Saltmarsh soils are also widely understood to be sinks for a range of heavy metals including zinc, iron, manganese, nickel and copper (e.g. Cundy *et al.*, 2005; EFTEC *et al.*, 2006). The predominant source of these metals into marshes is industrial waste entering estuarine waters, but some contaminants enter the soil directly via atmospheric deposition, e.g. lead from vehicular transport and mercury from manufacturing processes (Emmerson, 1997; Kim *et al.*, 2004; Hung and Chmura, 2006). Heavy metals are taken up by plants in the same way as nutrients, with the majority of halophytes retaining them in their roots or the soil via phytostabilisation, ensuring the bioavailability of the metals is low (Weis and Weis, 2004; Anjum *et al.*, 2013). As with nutrients, different species are more effective at storing metals than others. Weis and Weis (2004) found *Phragmites australis*, an invasive species in the northeastern US marshes they studied, much more effective at sequestering heavy metals into the soil than the native *Spartina alterniflora* which actually leached metals back into the ecosystem through its leaves. Retention also varies between seasons, tidal inundation and the level of metal input (Williams *et al.*, 1994).

The main provisioning service of saltmarshes to people is the food products obtained directly for animal or human consumption or indirectly through the detritus-based food chain, which supports fish in the wider estuary system. In addition to the commercially fished species mentioned in Section 1.1.2, the flux of nutrients from marshes, and the sheltered conditions in their creeks provide ideal conditions for farming oysters, mussels and other shellfish (EFTEC *et al.*, 2006). In the United States, oyster habitats are a very valuable commodity; commercial fishing of the eastern oyster *Crassostrea virginica* began as early as the 1600s in the Hudson River estuary (Kirby, 2004). In the UK, both native species of oysters and hardier introduced varieties are produced commercially (King and Lester, 1995; zu Ermgassen *et al.*, 2013). Other products and materials gathered by people from marshes include seaweed for food, cosmetics and medicines; reeds for fencing, mats and craft goods; and worms for use as bait in fishing (Bandaranayake, 1998; EFTEC *et al.*, 2006; Doody, 2008). In addition to cultural functions and values discussed in Section 1.1.1, other perceived benefits of saltmarshes include the preservation of archaeological artefacts in sediment, their role as
an educational resource, and their historical and cultural importance, which is especially appreciated in the UK given its rich maritime history (Emmerson, 1997; EFTEC et al., 2006; Barbier et al., 2011).

1.1.4 Carbon Sequestration

The saltmarsh ecosystem service most recently recognised by the scientific community is their major role as a sink in the global carbon cycle (Kathilankal et al., 2008; Chmura, 2013). Carbon sequestered within saltmarshes and other vegetated coastal environments such as mangroves and seagrass beds is known as ‘blue carbon’ and, despite the small land area of these ecosystems, is disproportionately important to the global carbon pool when compared with terrestrial forests (Connor et al., 2001; Chmura et al., 2003; Canuel et al., 2012). As uncertainties exist regarding the total areal extent of saltmarshes (especially in China and South America), their contribution to the global carbon budget has not yet been quantified accurately (Chmura, 2013), but Mitra et al. (2005) have estimated that wetlands in general contain 350-535 Gt of carbon, equating to 20-25% of the world’s soil carbon, even though they only occupy 4-6% of the land area.

Wetlands, and saltmarshes in particular, are able to sequester carbon over long time scales in their living tissues both above- and below-ground, and within the sediment itself, potentially for millennia (Barbier et al., 2011; Mcleod et al., 2011). Carbon dioxide is removed from the atmosphere by plants through photosynthesis, and fixed into their leaves, stems, branches and roots (Sousa et al., 2010a; Chmura, 2013); in saltmarsh halophytes such as Spartina spp. This carbon storage can range from 100-1000 gC m\(^{-2}\) yr\(^{-1}\), compared with only 1.4-7.6 gC m\(^{-2}\) yr\(^{-1}\) in tropical forests (Sousa et al., 2010b; Mcleod et al., 2011). These larger values are linked to the high rates of primary production in marshes (Callaway et al., 2012). Carbon is also stored within the ever-increasing volume of soil substrate through the continual deposition of sediment on the marsh surface, which traps and buries organic matter (Zedler and Kercher, 2005; Keller et al., 2012). The saturated soil conditions preserve this matter, as microbial decomposition is constrained due to the lack of oxygen. This shifts the carbon into the long-term carbon cycle where it remains buried for thousands of years (Mayor and Hicks, 2009; Chmura, 2013).

The rate at which carbon can be sequestered varies significantly between saltmarshes, and over time within the same ecosystem. Different halophyte species have
different photosynthetic pathways and thus will store varying levels of carbon in their roots and leaves. In their Portuguese study, Sousa et al. (2010a) found that Spartina maritima sequestered the most carbon into its roots and the surrounding sediment, while Olsen et al. (1996) found Salicornia bigelovii to be a better species for carbon storage in a laboratory study due to its lower decomposition rate. The maturity of the vegetation is also significant, with the carbon storage potential of a marsh generally increasing with the level of vegetation development (Sousa et al., 2010b; Adams et al., 2012). This does not automatically relate to the age of the marsh or vegetation, however, as demonstrated by the results from the study by Adams et al. (2012) of restored marshes in the Blackwater Estuary, SE England. The 4 year old Abbott’s Hall restored site had higher surface soil carbon densities than the 11 year old Tollesbury site (3.3 wt.% versus 2.4 wt.% respectively), as a more substantial area of mid-marsh vegetation had developed over the shorter time period. A more mature plant structure with more complexity and larger canopy allows larger pieces of organic matter to be trapped and retained within the marsh. Many researchers agree that the differences exhibited in carbon cycling and sequestration in saltmarshes are mainly controlled by their own individual physicochemical characteristics and the hydrological properties of the area (Connor et al., 2001; Li et al., 2010; Callaway et al., 2012). These include soil salinity, hydroperiod, water velocity, nutrient availability, oxygen levels, drainage, pH and sediment type, as well as biological factors such as herbivory and bioturbation, as well as vegetation species (Sousa et al., 2010b; Mcleod et al., 2011). The other major control of carbon sequestration rates is the rate of sediment accumulation. This can override all other factors if the rate of sedimentation is very high, as is often the case in newly created managed realignment sites (Connor et al., 2001; Chmura et al., 2003; Adams et al., 2012).

The processes relating to carbon sequestration in wetlands are expected to change under the elevated CO$_2$ levels and higher temperatures expected to result from anthropogenically-induced climate change (Erickson et al., 2007; Mayor and Hicks, 2009). These conditions should result in a rise in photosynthesis and primary productivity, but also increase decomposition rates in wetland soils (Sousa et al., 2010a). In saltmarshes, however, the overall influence of these factors is expected to be minor, as there is only a weak relationship between mean annual temperature and carbon storage in these ecosystems (Chmura et al., 2003). The greatest impact on these coastal habitats will be from sea-level rise, and the changes in carbon storage resulting from this are likely to be complex due to the close links between the geomorphological
and biological processes (Mcleod et al., 2011). For example, vegetation growth depends on physical conditions, such as the hydroperiod, but it can also alter these conditions as plants affect the elevation of the marsh surface by trapping sediment. This makes it hard to predict longer-term carbon accumulation (Kirwan and Mudd, 2012). Mudd et al. (2009) modelled sequestration in a South Carolina marsh and their results suggest that rates of carbon accumulation will increase with sea-level rise until a critical threshold is reached that drowns the vegetation and stops all accumulation.

1.1.5 Coastal Defence Value

It is widely acknowledged that one of the most important benefits provided by saltmarshes is their ability to dissipate wave energy and provide protection against flooding and erosion (Brampton, 1992; Shepherd et al., 2007; Wang et al., 2008; Balke et al., 2012). Although quantitative measurements from field studies have only fairly recently been provided (Möller et al., 1996, 1999, 2001, 2014), qualitative observations of the ability of saltmarsh vegetation to attenuate wave action dates back to the latter half of the 19th century. In his study of the New England coast in the United States, Shaler (1886: 364) documents that “the deadening of the current when the lowered tide brings the top of the plants near the surface is very noticeable” resulting in an assortment of floating matter and detritus to collect in the dense vegetation and sink to the bottom. Yapp et al. (1917) also noted that the very thick swards of vegetation in the saltmarshes of the Dovey Estuary in west Wales were very effective at reducing wave action and promoting sediment deposition.

One of the first attempts at quantifying the extent to which saltmarsh vegetation mitigates shoreline erosion was undertaken by Dean (1978) who suggested that marshes are analogous with an array of vertical cylinders in a water column, which instigate drag forces on the waves and tidal currents that pass through them, resulting in a reduction in wave height and energy (Knutson, 1988; Möller et al., 1996; Bouma et al., 2010). Dean developed a wave dissipation model based on laboratory tests of various arrangements of cylinders, which was modified by Knutson et al. (1982) and used to calculate wave energy reductions in Spartina alterniflora marshes in Chesapeake Bay, USA. They found that, on average, 64% of the incoming wave energy was lost within the first 2.5 m of the marsh, and all energy was dissipated after 30 m of marsh vegetation, due to the very dense stands and tall canopies of the Spartina stems. Other early physical
modelling studies suggested a wave height reduction of approximately 40% over an 80 m wide saltmarsh (Brampton, 1992).

Saltmarsh vegetation moderates erosion of the coast by a number of mechanisms. As mentioned, the primary means is through the increased surface roughness and frictional resistance generated by the physical structure of the plants; stems and leaves hinder the flow of currents and waves, causing the velocity and power of the water to be reduced (Shi et al., 2000; Neumeier, 2007; Coulombier et al., 2012). Wave energy decreases exponentially with distance across a salt marsh, as found by Knutson et al. (1982) and Möller et al. (1999), with plants at the edge of the marsh experiencing the largest hydrodynamic forces (Ysebaert et al., 2011). This reduction in vertical shear and turbulence in the plant canopy promotes the accumulation of sediment, raising the elevation of the marsh surface in comparison to water level, further reducing the erosion potential (Frey and Basan, 1985; Leonard and Reed, 2002).

Vegetation also increases soil stability, as the roots bind the sediment together and increase its shear strength (Pestrong, 1969; Dean, 1978). In terrestrial soils, Gray (1974) reported that vegetated soils had a shear strength 2-3 times higher than unvegetated areas due to the increased cohesion of the sediment particles. Few empirical results are available from saltmarsh soils, but comparable findings between tidal flats and marsh areas have been reported in California (Knutson, 1988). These areas of dense root systems increase the erosion threshold of the soil, reducing the rate at which blocks of sediment at the marsh edge or along channel banks collapse (Chen et al., 2011).

The history of actively using saltmarshes as a form of coastal defence dates back to the 1920s and 1930s in the UK, when Spartina townsendii was planted throughout many estuaries to stabilise the shoreline and promote the deposition of sediment (Lambert and Davies, 1940; Ranwell, 1967). Marshes were also established in the US from the 1950s onwards to prevent erosion along tidal stretches of rivers in Virginia, a practice that spread throughout the country in the decades that followed (Knutson et al., 1981). In China, Spartina anglica and S. alterniflora were first planted in 1963 and 1980 respectively to strengthen estuarine defences, enabling them to better resist severe typhoons (Qin et al., 1997; Chung, 2006). Following research from the UK Environment Agency (formerly the National Rivers Authority) in the early 1990s, based on the work undertaken by Brampton (1992), estimates of the cost of building new sea walls fronted by various widths of saltmarsh were produced (Toft et al., 1995). As shown in Table 1.1, a 6 m wide saltmarsh can halve the required height of a sea wall relative to that needed if no marsh is present and reduce the construction costs by over
Table 1.1: Sea wall construction costs associated with levels of saltmarsh protection
Source: King and Lester (1995) and based on data for Essex, southeast England.

<table>
<thead>
<tr>
<th>Width of saltmarsh (m)</th>
<th>Wall height (m)</th>
<th>Cost of new wall (£ m$^{-1}$)*</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>12</td>
<td>5000</td>
</tr>
<tr>
<td>6</td>
<td>6</td>
<td>1500</td>
</tr>
<tr>
<td>30</td>
<td>5</td>
<td>800</td>
</tr>
<tr>
<td>60</td>
<td>4</td>
<td>500</td>
</tr>
<tr>
<td>80</td>
<td>3</td>
<td>400</td>
</tr>
</tbody>
</table>

*Prices based on 1992 costings

two-thirds. Savings are also made through the lower maintenance costs associated with the lower sea walls (King and Lester, 1995; Dixon et al., 1998).

Although these savings have been frequently quoted in the coastal management literature (e.g. Leggett and Dixon, 1994; Dixon et al., 1998; Cooper, 2005), Möller (2003) describes them as only a ‘best guess’ estimate. This is due to the fact that they were based on physical modelling studies with a limited set of hydrodynamic conditions, did not use real nor simulated vegetation in the laboratory, and viewed the width of the marsh as the key factor governing the attenuation of wave energy. It was only since the 1990s that more thorough laboratory and field work was carried out to fully quantify the role played by marshes and their vegetation in attenuating wave energy, and it was realised that the traditional view of halophytes as cylinders in the water column was overly simplistic (Möller et al., 1999).

Table 1.2 lists the percentage of wave height and energy lost over different types of saltmarshes based on a synthesis of the relevant literature. Attenuation distances studied in the more recent work are greater than those found in the early American studies, where both Wayne (1976, cited in Cooper, 2005) and Knutson et al. (1982) reported that virtually all wave energy was dissipated over just 20 to 30 m of marsh. This is likely due to dense cover of tall plant species in these systems as well as very low incident wave energy levels encountered in most of the US studies (Möller et al., 1996, 1999; Yang et al., 2012). A comparison of the results from the vegetated and non-vegetated stretches of the marshes clearly demonstrates the crucial role played by halophytes. Although wave height and energy are reduced over the tidal flats by bottom friction and variations in micro-topography, the increased frictional resistance and drag created by the vegetation results in a much higher rate of energy dissipation: an average
Table 1.2: Wave height and wave energy reduction in different saltmarshes

<table>
<thead>
<tr>
<th>Location</th>
<th>Marsh Species</th>
<th>Marsh Width</th>
<th>Wave Height Reduction</th>
<th>Wave Energy Reduction</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Adams Beach, Florida, USA</td>
<td><em>Spartina alterniflora</em></td>
<td>20 m</td>
<td>71%</td>
<td>92%</td>
<td>Wayne (1976)*</td>
</tr>
<tr>
<td></td>
<td><em>Thalassia testudinum</em></td>
<td>20 m</td>
<td>42%</td>
<td>67%</td>
<td></td>
</tr>
<tr>
<td>Chesapeake Bay, Virginia, USA</td>
<td><em>Spartina alterniflora</em></td>
<td>30 m</td>
<td>94%</td>
<td>100%</td>
<td>Knutson <em>et al.</em> (1982)</td>
</tr>
<tr>
<td>Stiffkey, Norfolk, UK</td>
<td>Sand flat</td>
<td>197 m</td>
<td>15%</td>
<td>29%</td>
<td>Möller <em>et al.</em> (1999)</td>
</tr>
<tr>
<td></td>
<td>Diverse saltmarsh</td>
<td>180 m</td>
<td>61%</td>
<td>82%</td>
<td></td>
</tr>
<tr>
<td>Tillingham, Essex, UK</td>
<td>Mudflat</td>
<td>147 m</td>
<td>21%</td>
<td>35%</td>
<td>Möller and Spencer (2002)</td>
</tr>
<tr>
<td></td>
<td>Diverse saltmarsh</td>
<td>163 m</td>
<td>87%</td>
<td>99%</td>
<td></td>
</tr>
<tr>
<td>Yangtze Delta, China</td>
<td>Mudflat</td>
<td>185 m</td>
<td>11%</td>
<td>21%</td>
<td>Yang <em>et al.</em> (2008)</td>
</tr>
<tr>
<td></td>
<td><em>Spartina alterniflora</em></td>
<td>13.5 m</td>
<td>37%</td>
<td>60%</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Scirpus maritiner</em></td>
<td>16.5 m</td>
<td>16%</td>
<td>29%</td>
<td></td>
</tr>
</tbody>
</table>

* data from Cooper (2005)

of 1.82% m$^{-1}$, compared to 0.17% m$^{-1}$ for the recent studies in Table 1.2 (Putnam and Johnson, 1949; Möller and Spencer, 2002; Strong and Ayres, 2009).

Vegetation type will also affect the amount of energy dissipated, as the biophysical parameters between plant types such as height; buoyancy; stem diameter, flexibility and density; and leaf characteristics will vary (Gleason *et al.*, 1979; Bouma *et al.*, 2005b; Neumeier and Amos, 2006; Feagin *et al.*, 2011). Several researchers have examined the effect of different species on wave attenuation, including comparison of *Spartina alterniflora* with other plants, such as *Scirpus maritiner*. In their study in the Yangtze Delta, China, Yang *et al.* (2008), found *Spartina* attenuated wave energy by 4.44% m$^{-1}$, considerably higher than the 1.76% m$^{-1}$ for *S. maritiner*. In a similar study, Ysebaert *et al.* (2011) recorded results akin to these values when comparing the two species, noting that *S. alterniflora* dissipated 2.5 times as much hydrodynamic energy, primarily due to the greater height and biomass of these plants in the water column. Shorter plants are fully submerged more quickly than taller ones, so the friction factor
reduces significantly when the percentage of water occupied by vegetation falls (Koch et al., 2009). Möller et al. (2001) suggest that there is probably a threshold water depth above which the saltmarsh will have no extra dampening effect on tidal currents and waves. Thus marshes will likely prevent damage to flood defences during average water depths, but during extreme events, such as severe weather or storm surges that increase the power of the waves and the water depth, the “value of saltmarshes as sea defences is more questionable” (Möller et al., 2001: 115). Other factors affecting wave attenuation include weather conditions, tidal currents and velocity, sediment type, and the season, which influences plant height and density (Möller et al., 1996, 1999; Shi et al., 2000; Möller, 2003; Neumeier and Ciavola, 2004).

1.2 Saltmarsh Biogeomorphology

Saltmarshes are complex morphosedimentary systems and their formation and maintenance is dependent upon a multitude of natural physical and biological factors interacting together in a number of different ways (Allen, 2000, 2009). This diversity of controls gives rise to a wide range of saltmarsh environments, both in the UK and around the world (Adams, 2001). No two marshes are the same; they all exhibit different geomorphic characteristics influenced by their geographical location, and more recently, the level of human utilisation (Allen, 2000; French and Reed, 2001).

The global distribution of coastal saltmarshes is primarily controlled by temperature, and as such they are essentially confined to the mid- and high-latitude temperate regions (Chapman, 1977; Steers, 1977). They are also located in some low-latitude areas, generally alongside their tropical equivalent, mangroves, such as the USA Gulf Coast and central Florida, southern Australia and New Zealand (Chapman, 1960). Saltmarshes occur along the coastlines of most continents, primarily on open and barrier coasts and in estuaries (Frey and Basan, 1985). They range in size from a narrow fringe occupying an area of raised land, to large swaths measuring hundreds of square kilometres (Allen, 2000). Data on the exact areal extent of marshes is hard to come by; in the UK estimates range from 45 km² (Burd, 1989) to over 350 km² (Dijkema, 1987).

1.2.1 Controls on Saltmarsh Occurrence and Type

The specific location, size, character and behaviour of saltmarshes is controlled by four main factors: the accommodation space within the coastal environment, relative sea level, tidal conditions, and the sediment regime (Frey and Basan, 1985; French and
Reed, 2001). Although these elements combine in different ways in different environments, many authors (Steers, 1977; Reed, 1990; Allen and Pye, 1992), note that the occurrence of marshes is primarily determined by coastal physiography and geology, or the accommodation space. At a continental and regional setting, large-scale variations in geology and topography influence the physical space available for marsh development. Tectonically active areas, such as the US Pacific coast, result in disintegrated marshes forming narrow fringes, whereas much larger continuous marshes have developed on the broader coastal plains of the Atlantic, where the wide, gently sloping continental shelf creates a larger intertidal area (Frey and Basan, 1985; Townend et al., 2011). On a local scale, the establishment of halophytic vegetation can only occur in areas with low energy wave conditions to allow for the germination of seedlings before they are washed away by the tide (Chapman, 1977). Therefore marshes are generally located on protected coastlines, in embayments and estuaries, behind spits and offshore barrier islands, where the physical conditions afford areas of shallow and sheltered water (Wiegert et al., 1981; Friedrichs and Perry, 2001).

Changes in relative sea level are closely related to the accommodation space available for coastal saltmarshes (Townend et al., 2011). The rise in sea level since the end of the last glacial maximum has created accommodation space in which halophytes can colonise and establish and for sediment to build marsh surfaces (Allen and Duffy, 1998a; Allen, 2000; Madsen et al., 2007). Small variations in sea level are linked to the formation and development of marshes around coastal and estuarine areas, with minor fluctuations (both positive and negative) initiating an increase in marsh area and elevation (French and Reed, 2001). Where accommodation space is limited, erosion occurs and vegetated saltmarshes are replaced by bare mudflats, but when the geomorphology and sea level conditions in an area combine to increase the accommodation space over time, intertidal areas will be more conducive to healthy halophyte communities (Townend et al., 2011).

In addition to the larger-scale, longer term changes associated with sea-level rise, the local tidal regime plays a major role in determining the vertical and horizontal extent of saltmarsh area, as well as the geomorphological and biological processes occurring in the marsh itself (Allen and Pye, 1992; Allen, 2009). Tidal range determines the zone within which sediment accretion and vegetation growth can occur, generally understood to be roughly between Mean High Water Neap Tide (MHWNT) and the Highest Astronomical Tide (HAT), as shown in Figure 1.1 (Allen, 2000; French and Reed, 2001). At MHWNT, the marsh surface is exposed for just long enough during
low water for the seedlings of pioneer species to take root, whilst the upper marsh vegetation cannot exceed the level of the highest tides due to the limited frequency and duration of tidal flooding and sediment input at this elevation (Chapman, 1977; Allen and Pye, 1992). The greater the tidal range, therefore, the greater the vertical range available for marsh growth along the coast. Localities with large tidal ranges also have a greater potential for a larger horizontal extent in marsh area, due to the flattening of the surface gradient that typically occurs over time (French, 1993, 2006a). Tidal range varies considerably throughout the world, and especially around the UK coastline, leading to considerable spatial variability in the vertical range available for marsh formation (Adams, 2001). Many studies have shown a link between tidal range and net sediment accretion, with macro- and meso-tidal coasts being more favourable to continual marsh development due to the larger spatial extent, the longer hydroperiod, higher suspended sediment concentrations and the potential for sediment transport (Harrison and Bloom, 1977; Stevenson et al., 1986; French, 1994, 2006a; D’Alpaos et al., 2012). In contrast, micro-tidal regions are more reliant on storm surges or river floods to supply sediment and so accrete rapidly during these short-term events when the inundation period is greater and the sediment concentration significantly higher than normal (Leonard et al., 1995; Friedrichs and Perry, 2001; Townend et al., 2011).

As a depositional environment, the final main factor controlling the occurrence and behaviour of saltmarshes is the availability of fine sediment (Allen, 2009). The abundance of sediment in suspension in the coastal zone influences the rate at which marshes can develop and respond to changes in the environment, sea-level fluctuations especially (Allen and Pye, 1992). The most extensive and healthy marshes are generally located in those areas with high sediment loads that are readily available and easily deposited on the marsh surface (Chapman, 1977; Allen and Duffy, 1998b). The import
of sediment is critical for the maintenance of the marsh surface and for providing nutrients to the vegetation (Delaune and Patrick, 1980). The sources of external fine grained sediment to the intertidal zone include silt from rivers flowing into an estuary or nearby open coast, material from eroding cliffs, and offshore mud deposits or other sedimentary formations on the sea floor (Allen, 2000). The exact source of sediment for a particular marsh is often not easy to determine, however, as mud is easily suspended and transported by tidal currents, can travel a significant distance from its source, and will mix with other materials before being eventually deposited (Allen and Pye, 1992). In the UK, the majority of inorganic sediment available to marshes derives from cliff erosion, with relatively little supplied by river catchments (Eisma and Kalf, 1987; McCave, 1987). Factors affecting the availability of sediment are generally related to the marine energy level within the local environment, and include tidal and wind velocity, changes in tidal asymmetry, erosion, as well as biological activity and human activity in the coastal zone (Townend et al., 2011). Due to the construction of coastal defences to control cliff erosion, the supply of sediment to marshes along the UK’s North Sea coast has been reduced over the last 60 years (Allen, 2000). This is a reversal of the increase in sediment supply due to deforestation and agricultural practices that occurred from the mid Holocene (Macklin et al., 2000).

1.2.2 Initial Formation and Sediment Dynamics

Where conditions are right, mudflats will form through the continual deposition and accumulation of thick deposits of sediment (Steers, 1959). Once the surface has been raised to a sufficiently high enough level in the tidal frame, algae and halophytic vegetation will start to colonise the mud, a process that can only occur once the mudflat is only covered by the tide for approximately 6 hours per day (Allen and Pye, 1992; Boorman, 1999). This allows marsh seeds, carried to the surface by the tide or birds, to germinate; inundation must not be too frequent or long, otherwise they will not be able to take root (Friedrichs and Perry, 2001). This often occurs during neap tides, where much of the surface will not be inundated, allowing plant growth to begin (Boorman, 1999). Balke et al. (2014) has termed these disturbance-free periods as ‘Windows of Opportunity’ (WoO), which must last for a critical minimum duration directly following diaspore dispersal to allow for seedling anchoring. Although wind and water movements are required to disperse seeds, they can also inhibit their establishment if no WoO occurs (Hu et al., 2015). The minimum duration of a WoO required for seeds of each species to germinate varies between 2-3 days for the saltmarsh pioneer Salicornia
sp. to 5 days for Aster spp. (Wiehe, 1935; Chapman, 1960). Coles (1979, cited in Adam, 1990) also suggested that diatoms and other algae are required to stabilise sediment in order for vascular plants to be established.

The exact distribution of individual plants will depend on seed dispersal across the mud surface (Oliver, 1925). As plant cover grows, the rate of sedimentation increases, as particles are trapped as the surface roughness decreases the velocity of water flow (Allen and Pye, 1992; Boorman et al., 1998). The nature of the plant cover depends partly on the type of sediment deposited, whether primarily sand or mud (Steers, 1959). This pioneer zone is generally dominated by species of Salicornia, Suaeda maritima or Spartina anglica (Boorman, 1999). This interaction between physical and ecological factors is present in all stages of marsh development.

Tidal creeks are an important part of all saltmarsh systems, acting as channels for exchanges of biogenic material and nutrients, as well as water and sediment throughout the marsh area (Pye and French, 1993a; French, 1994). The creek network found in marshes originates and is inherited from the colonisation of the mudflat, and is influenced by geological, topographical and physical factors (Allen, 2009). Vegetation also controls the development and lateral migration of creeks, with those channels in vegetated areas being more sinuous with marked meandering than in unvegetated zones (Minkoff et al., 2006). The formation and enlargement of creeks is a complicated erosional process associated with interactions between physical, chemical and biological processes, and transfers the material from tidal flats and marshes back to the wider coastal system (Chen et al., 2011). After the initial development of the creek network, it is generally stable, and the subsequent development of marsh vegetation reinforces and preserves it (Townend et al., 2011).

Sediment dynamics of saltmarshes are a complex result of the interaction of a number of factors, such as tides, wind and wave regimes, coastal morphology and vegetation, leading to spatial and temporal variability in sedimentation (Silva et al., 2009; Coulombier et al., 2012). Inorganic material is supplied to the marsh surface in suspension via tidal flows, and in the early stages of marsh development, sediment accumulation is rapid, measuring centimetres annually, but this decreases in a non-linear fashion as mean marsh height increases towards HAT (French, 1994; Allen, 2000; 2009; Spencer and Harvey, 2012). Provided there is an adequate supply of sediment, the marsh surface will continue to increase in elevation, reducing tidal inundation time which feeds back again on sedimentation, driving marshes towards an
equilibrium elevation relative to mean sea level (FitzGerald et al., 2008; Townend et al., 2011).

It has long been thought that on short-term timescales, tidal currents control sedimentation and during slack water the suspended sediment load is deposited onto the marsh surface (FitzGerald et al., 2008). Over longer timescales, storm events provide larger sediment volumes in a single event that are orders of magnitude greater than sedimentation rates on a single tide (Leonard and Reed, 2002). French (2006), however, supports the argument that sedimentation occurs progressively throughout each tidal cycle, and not just during the brief slack water periods. Vegetation reduces turbulence and slows tidal currents, allowing the sediment particles to settle out (Leonard et al., 1995). As a result, sedimentation rate decreases with distance from the seaward edge of the marsh and from the internal creek network, a finding supported by field evidence from northwest European saltmarshes (Temmerman et al., 2003a), where much of the incoming sediment is lost from suspension prior to high water. Over long timescales, marsh accretion is positive, but during specific tidal events and over a particular season, material may be resuspended from the marsh surface by rainfall, wind waves, storms and tidal currents (FitzGerald et al., 2008).

It has been demonstrated that saltmarshes are uniquely complex systems and are dominated by non-linear interactions between biologic and geomorphic processes (Fagherazzi et al., 2004; Viles et al., 2008; Balke et al., 2012). These strong feedbacks play a decisive role in both the initial formation from mudflats and the longer-term evolution and maturation of marshes (Murray et al., 2008; Nolte et al., 2013a). Many researchers are increasingly calling for saltmarsh processes to be studied using a biogeomorphological approach in order to gain a much fuller understanding of the way these habitats function (Viles and Naylor, 2002; Reinhardt et al., 2010). The theory of biogeomorphology explicitly considers both the influence of landforms and geomorphology on the distribution and development of plants, animals and microorganisms as well as the influence of plants and other biota on earth surface processes and landform development (Reed, 2000; Viles et al., 2008). The theory was initially pioneered by Viles (1988) and overturned the previously long-standing paradigm that only physical processes shaped landscapes and constrained biological agents (D’Alpaos et al., 2012). Over the past 25 years, biogeomorphology has developed into a well-established field with a range of studies being published examining the two-way interplay between organisms and geomorphology (Naylor et al., 2002; Naylor, 2005; Kim, 2014). These include papers exploring the role of lichens as
weathering agents on limestone buildings (Carter and Viles, 2004), beaver dams on sedimentation rates (Butler and Malanson, 2005), topography on the spatial patterning of plant communities (Alpert, 1985), as well as the baffling effect of vegetation in saltmarshes (Bouma et al., 2005b; Temmerman et al., 2005b; Möller, 2006).

In saltmarshes, biological processes depend on and are affected by physical processes in several ways and in turn, flora and fauna exert controls on geomorphology and hydrodynamics (Viles, 1988; Möller et al., 1999; Corenblit et al., 2011; Balke et al., 2012). Figure 1.2 shows some of the many positive and negative biogeomorphological interactions occurring within saltmarsh systems. Geomorphology influences plant biomass and zonation through relationships between rising sea level and tidal range, surface elevation change and hydroperiod, whilst vegetation contributes to marsh growth through trapping and binding inorganic sediment brought in by tidal flows as well as direct accretion of organic matter from plant detritus (Fagherazzi et al., 2004; Murray et al., 2008; Viles et al., 2008; Nolte et al., 2013a). A further factor, livestock grazing, is largely unstudied, but may have a major effect on vegetation and sediment dynamics in saltmarshes (Elshot et al., 2013; Nolte et al., 2013b). Livestock increases sediment compaction through trampling, and reduces vegetation density through grazing, which in turn may modify flow patterns and reduce sedimentation rates (Berg et al., 1997). Livestock has also been shown to increase the accumulation of biomass in the soil, as plants which experience grazing produce more roots, which may help to further bind the sediment surface and thereby reduce erosion (Esselink et al., 1998). The biogeomorphological approach to studying marsh processes thus reveals

Figure 1.2 Interactions between biogeomorphological components in saltmarsh systems
Adapted from Nolte et al. (2013a).
additional complexity in the system as a whole, but provides further understanding to inform management and restoration of these habitats (Viles and Naylor, 2002; Naylor, 2005; Spencer and Harvey, 2012).

1.3 Resilience of Saltmarshes to Change

In their position at the interface between land and sea, coastal saltmarshes are potentially sensitive to changing environmental conditions, with any alteration in sediment accretion and erosive processes resulting in potentially major changes to the marsh system (Haslett, 2009). Global sea-level rise, increasing wave energy and storminess, and subsidence are still being compounded by anthropogenic activities such as land reclamation, pollution, port developments and coastal defence measures (Cox et al., 2003; Osgood and Silliman, 2009; Junk et al., 2013). Up to 90% of the coastal marshes worldwide are predicted to suffer degradation or be lost by 2100 (Nicholls et al., 1999; Stevenson and Kearney, 2009). The main threat to saltmarshes is a combination of sea-level rise from climate change (eroding and drowning the ocean edge), and human development of the coastal zone (through reclamation or flood defences), squeezing the available intertidal area and reducing the capacity of marshes to migrate inland (Titus, 1991; Cooper et al., 2001; Hughes, 2004; Gedan et al., 2009).

Given the severity of these predictions, efforts are underway to conserve existing marshes, reconstruct those previously subject to reclamation, as well as creating new habitats to mitigate for those lost (Doody, 2004). Restoration approaches include planting halophyte species, increasing suspended sediment concentrations, placement of dredged material and decreasing wave activity via construction of breakwaters (Cooper et al., 2001; Mossman et al., 2012b). There is now growing support to increase the space available for saltmarshes to develop or migrate landward to counteract the problem of coastal squeeze (Nicholls and Mimura, 1998; Davy et al., 2009; Mulder et al., 2011). This is undertaken by removing or breaching sea defences and embankments through a process of managed realignment with the expectation that marsh vegetation will colonise the previously protected area and provide equivalent habitat and coastal defence value as naturally-occurring marsh (Williams and Faber, 2001; Williams and Orr, 2002; Blackwell et al., 2004). Managed realignment schemes most often occur at sites that historically were saltmarshes before being reclaimed (Leggett et al., 2004).
1.3.1 Historic Degradation and Loss

Although it is acknowledged that saltmarshes experience cyclical periods of erosion as well as accretion in their growth and evolution (Allen, 2000; Singh Chauhan, 2009), there has been a continued trend towards the global deterioration and erosion of marshes over the past 100-200 years (Kennish, 2001; Gedan and Silliman, 2009). On average, approximately 50% of the original marsh ecosystems around the world are in decline or have been lost completely (Barbier et al., 2011). There is no one cause that has led to this large-scale loss, but rather a multitude of factors, potentially combining synergistically, to create a complex picture of erosion in most saltmarsh systems (Hartig et al., 2002). Different marshes can experience varying levels of pressure from subsidence, sea-level rise, direct and indirect human interference including land claim and pollution, as well as biological forcings such as herbivory and bioturbation. It is often difficult, if not impossible, to separate out the relative role of each factor in the deterioration of the ecosystem (Phillips, 1986; Lotze et al., 2006; Holdredge et al., 2009; Coverdale et al., 2012).

Historically, the majority of saltmarsh in NW Europe has been lost through reclamation for agriculture and industry (Doody, 1992b). Even major cities such as Amsterdam and Venice now stand on former marshes (Gedan et al., 2009). In the Dutch and German Wadden Sea area, conversion of marshes to farmland and freshwater lakes began around the 11th century. Reclamation continued until the 1960s, when there were growing concerns over land claim and the focus switched to conservation (Wolff, 1997; Airoldi and Beck, 2007). By that time about 1,500,000 hectares of coastal wetlands had been lost and the Wadden Sea is now nearly half of its original size (Reise, 2005). Embankment and land claim have taken place since Roman times in Britain, with most activity carried out in the shallow waters around large estuaries such as the Wash, and the Severn, Humber, Ribble and greater Thames estuaries (Doody, 1992a; Davy et al., 2009). It has been estimated that 53,400 hectares of saltmarsh has been lost from these major estuaries, approximately 50% of all marsh area since Roman times (Doody, 1992b; Crowther, 2007). Most land claim was undertaken on a piecemeal basis, slowly acquiring land from the sea, with earth banks and sea walls built to keep the water out. Despite a significant decline in reclamation for agriculture since the 1800s, it continued in the UK up until 1979, when it became uneconomical to create new land for arable farming (Doody, 2004).

Over the same time period, global and locally higher relative sea-level rise has placed pressure on remaining saltmarsh (Morris et al., 2002; Temmerman et al., 2004).
The natural response of saltmarshes to a gradual rise in sea level is to migrate inland, or ‘roll-over’ to maintain their position within the tidal frame, gradually shifting landward to a lower energy environment (Reed, 1995; Crooks, 2004). In pre-industrial times, the global rate of sea-level rise was 0.1-0.2 mm yr\(^{-1}\), but over the last 200 years, it rose to approximately 1-2 mm yr\(^{-1}\), putting a greater pressure on marshes to migrate more quickly in order to survive (Tooley, 1992). Since 1993, global sea-level rise has accelerated to 3mm yr\(^{-1}\) (Church and White, 2006). In some autochthonous systems, this rate of increase, combined with subsidence from regional downwarping and sediment compaction, has been too rapid and marshes become waterlogged and plants start to die (Clarke and Hannon, 1970; Cooper, 1982). This decrease in biomass further reduces the marsh’s ability to accumulate sediment and maintain its relative elevation (Reed, 1990), and the marsh becomes totally submerged and converts to bare mudflat or open water (Deegan \textit{et al.}, 1984; Phillips, 1986).

Also of concern, especially within estuaries, is the interaction between a rising sea level and the presence of fixed coastal defences. The natural landward migration of saltmarshes during sea-level rise is prevented by sea walls and other structures such that the intertidal area becomes progressively narrower, limiting the amount of space available for marsh vegetation (Dixon \textit{et al.}, 1998; Cooper \textit{et al.}, 2001). This has been termed ‘coastal squeeze’, as the marsh is effectively squeezed out of existence by rising sea levels and will eventually drown and disappear (Titus, 1991; French, 1997; Hughes, 2004). Sea walls may also reflect wave energy back onto the vegetation, causing erosion at the toe of the wall and in the marsh interior (Morris \textit{et al.}, 2004). According to Cooper \textit{et al.} (2001), the reduction in the amount of intertidal area to receive eroded sediment leads to a decrease in estuary depth, as the sediment builds up in the remaining sub-tidal channel. This loss of depth, combined with the loss of width, results in an increase in the tidal velocity within the estuary, further increasing the potential for erosion at the marsh edge.

Coastal squeeze is widely believed to be one of the main factors leading to large losses of saltmarsh in SE England, where over 1,200 ha were lost between 1973 and 1988 in the 11 Essex and Kent estuaries studied by Burd (1992). In Essex, 40 ha of marsh, or about 2% of the current resource, is currently lost every year (Hughes, 2004). Part of the explanation for these losses lies in the variations in relative sea-level rise around the UK (Jenkins \textit{et al.}, 2008; Wahl \textit{et al.}, 2013). In Scotland, relative land uplift is occurring at a rate of approximately 1.6 mm yr\(^{-1}\), whilst the south of England is subsiding, at 1.2 mm yr\(^{-1}\), on top of which sediment compaction and consolidation, at a
rate of 0.5-1.1 mm yr\(^{-1}\) in southeast England must be added (Shennan and Horton, 2002). Rates of relative sea-level rise are considerably higher here, for example 2.23 mm yr\(^{-1}\) and 2.57 mm yr\(^{-1}\) at Sheerness and Lowestoft, compared to 1.60 mm yr\(^{-1}\) in Liverpool and just 0.87 mm yr\(^{-1}\) in Aberdeen over the last 50 to 100 years (Woodworth \textit{et al.}, 2009).

Table 1.3 lists the area of saltmarsh in each of Essex’s main estuaries in 1973, 1988 and 1998, the comparison of which shows that almost 1,000 ha was lost over this period, primarily caused by erosion from the marsh edges due to sea-level rise and coastal squeeze as well as direct wave action (Doody, 1992b; Cooper \textit{et al.}, 2001). Erosion has been most serious in the outer estuaries, such as the Blackwater and the Stour, while in more sheltered areas such as the Colne, less severe rates have occurred. At all sites, vertical sediment accumulation was able to keep pace with sea-level rise (Van der Wal and Pye, 2004). There is also significant temporal variation in erosion within many of the estuaries studied. Some of this is explained by changes in the wind and wave climates, with records indicating an increase in the number of strong gales occurring in the East Anglia area in the 1960s and 1970s, before falling after their peak in the early 1980s (Lamb, 1982). Wave data from the North Sea follows a similar pattern, with a peak in mean wave heights around 1980 before reducing again; in 1984, wave heights were 13 per cent lower than the peak (Bacon and Carter, 1991). These trends are not clearly evidenced by the data shown in Table 1.3, but other studies focusing on specific estuaries show a clear link between increased periods of wind and

<table>
<thead>
<tr>
<th>River / Estuary</th>
<th>Total marsh area (ha)</th>
<th>Net loss (ha and (%))</th>
</tr>
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<tbody>
<tr>
<td>Orwell</td>
<td>100</td>
<td>70</td>
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<tr>
<td>Stour</td>
<td>264</td>
<td>148</td>
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<tr>
<td>Hamford Water</td>
<td>876</td>
<td>765</td>
</tr>
<tr>
<td>Colne</td>
<td>792</td>
<td>744</td>
</tr>
<tr>
<td>Blackwater</td>
<td>880</td>
<td>739</td>
</tr>
<tr>
<td>Dengie Peninsula</td>
<td>474</td>
<td>437</td>
</tr>
<tr>
<td>Crouch</td>
<td>467</td>
<td>347</td>
</tr>
</tbody>
</table>
wave activity and erosion. In the Dengie Peninsula, Harmsworth and Long (1986) found times of rapid erosion correlated with these periods, such as a rate of loss of 16 ha yr\(^{-1}\) between 1970 and 1973 and 9 ha yr\(^{-1}\) between 1978 and 1981, before falling to 3 ha yr\(^{-1}\) in the mid 1980s. Protection works carried out within the area in the 1980s also reduced wave attack and slowed marsh loss at this time (Reed and Pethick, 1987). The reduction in the rate of erosion in the Blackwater estuary from 9.5 ha yr\(^{-1}\) in 1973-1988 to 5.5 ha yr\(^{-1}\) in 1988-1998 is also partly explained by conservation efforts, in particular the creation of managed realignment schemes in a number of locations throughout the estuary (Cooper et al., 2001; Van der Wal and Pye, 2004).

As well as erosion of the marsh edge, enlargement of tidal creek systems within saltmarshes is a significant contributor to the recent losses seen in SE England and elsewhere (Kirwan et al., 2008). Caused by a range of mechanisms including hydraulic erosion on the bank face and mass movement due to bank slumping, the dissection of creeks is generally a long term process, converting vegetated marsh habitat into bare mudflats and channel beds (Minkoff et al., 2006). In the Beaulieu estuary, in southern England, creek erosion rates ranged between 1.1-1.5 cm yr\(^{-1}\), depending on vegetation type (Chen et al., 2011). The nature of enlargement is generally episodic; when a block fails and slumps into the channel, it offers protection to the remaining bank until the material is eroded away by the tidal currents (Gabet, 1998).

Herbivory and bioturbation also lead to marsh loss, through the grazing of above- and below-ground vegetation and roots and the destabilisation of the sediment (De Deckere et al., 2001; Talley et al., 2001; Coverdale et al., 2012). Various species of fauna, including crabs, amphipods, snails and polychaetes burrow through the soil and change the sediment particle size, which increases water content, reducing its shear strength and increasing its vulnerability to erosion (Hughes and Gerdl, 1997; Hughes and Paramor, 2004; Wilson et al., 2012). In Essex marshes, the infaunal polychaete *Nereis diversicolor* has been found by some researchers to be a major cause in the reduction of halophyte biomass (Emmerson, 2000; Paramor and Hughes, 2004), but this research has been called into question by others (Morris et al., 2004; Wolters et al., 2005a). Rapid degradation and loss of vegetation is also one of the effects of marsh dieback (Silliman et al., 2009). Although more prevalent in the United States, plant dieback has occurred in UK sites. For example, dieback affected approximately 200 ha of marsh in the Lymington Estuary in southern England in the 1950s (Goodman et al., 1959; Johnson, 2000; Alber et al., 2008). The characteristics of the sediment itself partly control how vulnerable it is to erosion. Crooks and Pye (2000) found that those
marshes deficient in calcium carbonate in the clay sediment allowed sodium ions to dominate, which leads to the development of thick water films around the clay particles, slowing sediment compaction. Essex soils tend to be lacking in these minerals, and are thus less resistant to erosion and creek dissection (Hodgkinson and Thorburn, 1995).

Anthropogenic activities have always impacted upon coastal saltmarshes, but their effects have dramatically accelerated over the past 150-300 years (Lotze et al., 2006; Wolanski et al., 2009). On a local scale, humans have destroyed marshes directly through dredging, spoil dumping, canal cutting, leveeing, as well as exacerbating subsidence by extracting groundwater, oil and gas (Kennish, 2001). Indirect impacts include nutrient enrichment, the introduction of invasive species and pollution from agricultural and industrial effluents (Davy et al., 2009). High nutrient inputs from herbicides and pesticides can reduce the photosynthetic efficiency of saltmarsh plants and epipelic diatoms, which normally form a biofilm that helps bind the surface sediment together (Mason et al., 2003). Although eutrophication of marshes increases above-ground biomass, the dense below-ground biomass of roots is reduced, leading to reduced sediment stability, increased bank collapse and the conversion of marshes into unvegetated mud (Deegan et al., 2012). Increasing global population requires an increase in food production, and thus more pressure is placed upon coastal systems through herbicides and fertilisers; herbicide use has more than doubled over the last 30 years, and this trend is set to continue into the future, as are all major threats to coastal wetlands (Mason et al., 2003; Junk et al., 2013).

1.3.2 Future Marsh Loss and Methods of Restoration

The major threat facing saltmarshes going into the future is anthropogenically-caused climate change and the many potential impacts associated with it (FitzGerald et al., 2008). The most serious of these with regards to coastal wetlands is a rise in relative sea level, caused mainly by thermal expansion of the world’s oceans (estimated to account for 70% of the rise) and the melting of glaciers, ice caps and ice sheets (Hulme et al., 2002; Lowe et al., 2009; Cazenave and Llovel, 2010). As stated in the IPCC 5th Assessment Report, sea level is predicted to rise between 0.42 m and 0.80 m, based on the SRES A1B scenario, but potentially by 0.98 m relative to 1990 levels by the end of the 21st century (Church et al., 2013). The worst case scenario predicts that the rate of sea-level rise could be as high as 15.7 mm yr⁻¹ in 2100. The increase in the rate of sea-level rise is not expected to be linear, instead models predict that there will be a ‘tipping
point’ around 2040 when the rate of increase will accelerate (Kearney and Rogers, 2010). The variation between the predictions are mainly due to the differences in human attitudes in the SRES scenarios, and how important the environment is viewed in the future (Nicholls, 2004). Many of the impacts of a continually rising sea level can be considered as an exacerbation of the current situation (FitzGerald et al., 2008); up to 22% of global coastal wetlands could be drowned by 2080 due to a rise in sea level alone, but 70% may be lost when the other impacts associated with human activities are taken into account (Nicholls et al., 1999).

A rising sea level will increase the levels of inundation and erosion of saltmarshes around the world that has occurred over the last 50-100 years. Marshes will be flooded for a greater period of time in the tidal cycle, and the vegetation may be submerged for longer than can be tolerated, leading to a loss of biomass which will increase the overall marsh vulnerability to further erosion (Hartig et al., 2002). The transformation of vegetated marsh to bare mud and open water is not predicted to be linearly correlated to the rate of sea-level rise, but instead there will be a critical threshold after which sediment accumulation is unable to keep up with the rising water levels (Reed, 2002; FitzGerald et al., 2008). Once passed, an irrevocable process of submergence and drowning will begin; the rate of marsh deterioration will increase as an ever-increasing area of vegetation is lost, leading to locally-generated wind waves which erode creek banks (Nicholls et al., 2007). The threshold will vary between different coastal systems, depending on hydrodynamic and sedimentary characteristics.

The impacts of a global rise in sea level will be spatially non-uniform, due to local vertical crust movements, either subsidence or uplift, and the variations between local coastlines, such as lithology, landforms, wave climates and longshore currents (Gornitz, 1991). In the UK, sea level may be between 2 cm below and 58 cm above current levels in western Scotland, and between 26 cm and 86 cm above present in the London area (Hulme et al., 2002). Alongside this increase in mean relative sea level, extreme sea levels and surges, formed through a combination of high tides, higher energy waves and strong winds, will be experienced more frequently in all coastal areas (Bird, 1993). Again, this will vary spatially, with Hulme et al. (2002) predicting that an extreme water level with a two per cent probability of occurring in any year at present, will have an occurrence probability of 33 per cent by the 2080s. The UK Met Office has estimated that the combined effect of rising sea levels and storm surges in the future will lead to water levels 2.7 m higher than normal tidal levels, putting many homes and
businesses at risk, and possibly overtopping sea defences in many areas (FitzGerald et al., 2008; Doody, 2013).

An increase in the magnitude of severe storms is also predicted, but there is considerable uncertainty over the likely projection of their frequency, and the locations at which they will occur (Collins et al., 2013). Regardless of their location, however, saltmarsh erosion will escalate with an increase in storm and associated wave activity, as larger waves reach the coast and penetrate further inland than at present (Airoldi and Beck, 2007). However, stormier conditions also lead to further cliff and foreshore erosion, increasing the amount of sediment carried by tides and waves, which may lead to sediment deposition, possibly offsetting any marsh loss from erosion and submergence (Bird, 1993). In areas affected by tropical storms, warmer ocean surface temperatures may further increase the severity and frequency of storms (Webster et al., 2005).

The predicted increase in temperature and changes to rainfall due to climate change may also impact the composition and distribution of plant species within saltmarshes. Changing rainfall amounts and patterns could affect soil salinity, either negatively or positively, whilst increasing temperatures may increase evaporation and raise salt concentrations, affecting vegetation growth (Hughes, 2004; Adam, 2009). Growth could also be inhibited by higher temperatures, as climate affects photosynthesis, transpiration and nutrient cycling; an increase in UV-B radiation may also reduce photosynthesis in some species (Bird, 1993; Davy et al., 2009). However, the associated rise in atmospheric carbon dioxide concentrations will increase plant productivity and biomass production (Stevenson and Kearney, 2009).

Given the loss of saltmarsh that has already occurred, and the concerns over its vulnerability in the future, management action is increasingly being considered to protect and restore these vital habitats. There is also a legal requirement in both Europe and the US to create new areas of marsh when existing areas are lost (Atkinson et al., 2004; Mossman et al., 2012b). As discussed previously, marshes can play a key role in coastal defence, so there is a pressing need for marsh restoration and creation to accommodate the increased water levels associated with sea-level rise, and reduce the high construction and maintenance costs of traditional hard defence options (Denny, 1995; Spencer and Harvey, 2012). There are a range of options available to protect existing marshes, including reducing wave activity before it reaches the marsh (through constructing offshore breakwaters), increasing the suspended sediment concentration (by supplying sediment through dredging), increasing the trapping efficiency of marshes
(by constructing polders or groynes), or spraying sediment directly onto the marsh surface to increase its elevation in the tidal frame (Cooper et al., 2001; Stevenson and Kearney, 2009). Managed coastal realignment, however, is becoming the most widely used strategy in creating new accommodation space for marshes to colonise (Alexander et al., 2012; Mossman et al., 2012b). Realignment involves breaching or removing sea walls to create intertidal habitats on the formerly protected land, allowing the tide to inundate the surface, as depicted in Figure 1.3 (Cooper, 2003; Atkinson et al., 2004). Lower and cheaper defences are then constructed behind the realignment area. The first site in the UK to be realigned was on Northey Island in the Blackwater estuary, Essex, where 0.8 ha of land was flooded in 1991 (Pye and French, 1993e). Realignment is the favoured technique in overcoming marsh loss in the UK, and is the first choice option by the EA and Defra (Morris et al., 2004). It is less expensive than other options over the long term, but implicit in these schemes is the development of saltmarsh vegetation on the restored land, and the attainment of similar ecological conditions as those previously lost, but this is not always the case (Emmerson, 1997; Mossman et al., 2012b). Further research is required to understand the development of these sites, and whether they fully solve the problems of habitat loss and flood management.

![Schematic representation of a managed realignment site](image)

**Figure 1.3: Schematic representation of a managed realignment site**
Adapted from French (2001)
1.4 Some Unresolved Questions in Saltmarsh Science

Saltmarshes have been of interest to researchers for over 100 years (Shaler, 1886; Yapp et al., 1917). Early studies in the 1930s and 1940s concentrated on describing the marsh environment and comparing marshes in different regions, as well as the distribution of plant and animal communities (Nicol, 1935; Chapman, 1938, 1939, 1940, 1941). In the following period in the 1950s to 1980s, a large amount of research was conducted on the physical and biological interactions within marshes, the flow of energy and materials and their overall productivity (Teal, 1962; Odum and de la Cruz, 1967; Lefeuvre and Dame, 1994). Although more recent work has examined the relationships between vegetation and sedimentation (e.g. Möller et al., 1996, 2001; Möller and Spencer, 2002), comparatively little is known about the geomorphology of saltmarshes in contrast to our botanical and ecological knowledge (Allen, 2009). To address this, further research is required into the basic processes operating within saltmarshes, to improve our understanding of their functioning, as well as how they will respond to future disturbance (Pye and Allen, 2000; Spencer and Harvey, 2012).

Restored saltmarshes present us with an opportunity to study these fundamental processes as they are forming and evolving in the present time. Allen (2009) has argued that more focus should be given to how contemporary marshes develop over time and mature; those marshes recently restored are ideal case studies for this purpose. In the context of coastal erosion and worldwide wetland degradation and efforts undertaken towards their restoration, it is critical that we fully understand the geomorphological, ecological and biogeochemical processes and their interrelationships in order to determine the success of such schemes (Scott et al., 2012; Spencer and Harvey, 2012). As with the trends in wider saltmarsh research mentioned above, most of the literature pays attention to ecological studies on vegetation, birds, invertebrates and fish (e.g. Havens et al., 2002; Atkinson et al., 2004; Colclough et al., 2005; Mander et al., 2007; Balke et al., 2012). Biogeochemical studies have also been undertaken at a number of realignment sites in the UK (e.g. Chang et al., 2001; Blackwell et al., 2004; Keller et al., 2012; Kirwan and Mudd, 2012) to examine the impact of reclamation and subsequent restoration on marsh sediments, as well as the potential role of these schemes in carbon sequestration now, and in the future (Spencer and Harvey, 2012). Comparatively little is known, however, about the geomorphological processes within restored marshes. Some recent biogeomorphological research has been undertaken at the Freiston Shore realignment site in Lincolnshire, including quantifying sediment provenance and the impacts of hydrodynamic reconfiguration on saltmarsh dynamics.
(Rotman et al., 2008; Friess et al., 2012); but there are still a number of questions yet to be fully answered regarding the present and future relationships between the physical and biological processes.

1.4.1 Role of Vegetation

It has been recognised for many years that halophytic vegetation reduces erosion and enhances sediment deposition through wave dissipation in coastal saltmarshes (French et al., 1995; Allen and Duffy, 1998a; Reed et al., 1999; Cahoon et al., 2000; Möller, 2003). However, recent research has called these assumptions into doubt, leading to a re-thinking of the role vegetation plays, or at least the specific mechanisms by which plants enhance sedimentation (Silva et al., 2009; Mudd et al., 2010; Gedan et al., 2011). Some authors have even found that halophytes increase erosion within saltmarshes. Feagin et al. (2009) conducted laboratory flume tests and field experiments at a saltmarsh in Galveston Island, Texas, to determine whether marsh plants prevented wave-induced erosion at the marsh edge. In both their simulated and real-world studies, there was no statistically significant difference between the levels of erosion between vegetated and unvegetated sediments; in the flume, mean erosion was actually higher in the vegetated tests. In the Westerschelde estuary in The Netherlands, Temmerman et al. (2007) discovered that although vegetation patches reduced erosion on a local scale, erosion was steadily increasing between these areas. The plants obstructed the flow around their immediate location, causing an increase in flow velocity and shear stress between the patches, leading to channel erosion that steadily worsened as the vegetation patches expanded in size and the flow around them concentrated further.

Other researchers have concluded that vegetation is not as important in the sedimentation process as previously thought, with other controls, such as elevation and the distance from sediment source the key determining factors (Silva et al., 2009; Coulombier et al., 2012; Moskalski and Sommerfield, 2012). Contrary to the long-standing assumption, short-term sedimentation rates in some saltmarshes are higher in unvegetated areas; Neumeier and Ciavola (2004) actually report a statistically significant negative correlation between sediment deposition and vegetation density. To explain this, they suggest that the generally low suspended sediment concentration during their study period meant that little sediment was available once the currents had passed over the unvegetated areas closer to the sediment source. Important sediment transport only occurs during larger storm events when there are greater quantities of
sediment transported in suspension, and the vegetation is probably more efficient at trapping it (Neumeier and Ciavola, 2004). In the Humber estuary in NE England, physical conditions are more important than vegetation cover, with no evidence found that *Spartina* enhances deposition in the pioneer zone, and in some cases actually increases turbulence (Brown, 1998). However, the plants did help to stabilise sediment during periods of erosion; bare sites were steadily eroded whilst the vegetated areas remained fairly consistent.

This stabilising effect is one of the mechanisms by which halophytes are known to influence longer-term surface elevation change in saltmarshes. Although many studies have shown that the vegetation may act as a barrier to sediment deposition (e.g. Leonard and Luther, 1995; Yang, 1999), flow resistance by the canopy and sediment binding by the roots strongly reduces erosion in saltmarshes by consolidating and strengthening the sediment surface (Ward *et al*., 1984; Shi *et al*., 2000). These processes reduce the resuspension of sediment from vegetated areas, protecting the bed from waves. A number of studies present results of substantially reduced or even no erosion in vegetated marsh areas (e.g. Brown, 1998; Yang *et al*., 2008), whilst significant erosion has occurred in the unvegetated flats. Suspended sediment concentrations reported by Coulombier *et al*., (2012) for a *Spartina alterniflora* dominated marsh in the lower St. Lawrence Estuary, Canada were 40% higher in unvegetated zones when compared with vegetated marshes, caused by local resuspension of sediment from the bare surface. These findings support the hypothesis by some authors that saltmarsh vegetation is more effective at retaining sediment during periods of erosion, rather than promoting its deposition (Scoffin, 1970; Harper, 1979; Brown, 1998).

1.4.2 Sediment Availability and Trapping Efficiency

A sufficient supply of sediment is crucial to the existence of saltmarshes and maintaining a positive sediment budget is necessary for marshes to keep pace with the predicted rise in sea level (French, 1993; Morris *et al*., 2004; Temmerman *et al*., 2004; French, 2006a). The majority of models and predictions of marsh response to environmental change assume that sediment will be available for future deposition and elevation growth, and make comparisons between sedimentation rates and sea-level rise (Orford and Pethick, 2006). However, as French (2006) and Mudd *et al*., (2010) argue, this is too simplistic and neglects complex feedbacks between sedimentation and hydrodynamics; instead estimates of sediment supply and trapping efficiency should be
used to assess marsh response. Even when sedimentation rates are greater than sea-level rise, some authors have predicted that marshes may drown, such as in the Venice Lagoon (Marani et al., 2007; Kirwan and Temmerman, 2009).

Most studies, however, show that current sources of sediment are adequate to support the vertical rate of growth of marsh surfaces (French, 1994); in the Greater Thames Estuary, the lateral loss of saltmarsh alone (40ha, average height of 1-1.5 m) sustains the sedimentation rate of 2-5 mm yr⁻¹ over the remaining 4000 ha (Van der Wal and Pye, 2004). Other sediment sources, such as fluvial sediment, eroding cliffs and resuspension from the seabed further increase the sediment available (McCave, 1987; Reed, 1988; Dyer and Moffat, 1998; Nicholls et al., 2000). However, in future, this is likely to change, due to coastal protection measures. For example, during the 20th century, material from the cliffs on the Isle of Sheppey has decreased by a third, whilst the supply from fluvial sources in the Thames area has been declining over the last 200 years (Bowen, 1972; Nicholls et al., 2000). Other anthropogenic impacts, such as dredging, harbour expansion and river diversions have also reduced the sediment available to the coastal environment (Day et al., 2000; Streever, 2001).

Regardless of the level of suspended sediment in the water column, it is the volume trapped by a saltmarsh that is important, i.e. its trapping efficiency, the ratio of measured deposition to the sediment available (Boorman et al., 1998; Moskalski and Sommerfield, 2012). Despite this fact, very few studies have examined the relationship between sediment supply and sediment deposition, partially due to the difficulties in estimating the volume of sediment transported between coastal and estuarine waters and into the saltmarsh ecosystem (French, 2006a). Those that have considered the trapping efficiency of a marsh have tended to compare the suspended sediment concentration on the flood tide entering the marsh with that on the ebb tide leaving it (e.g. Stumpf, 1983; French, 1993) or calculated the sediment budget for the marsh as a whole, incorporating all sources and sinks (e.g. Stevenson et al., 1988; Van Proosdij et al., 2006b). Such approaches have provided varying results between different systems, for example in the large estuary of Chesapeake Bay, marshes trap 5-11% of the annual sediment influx, half that of that previously estimated (Stevenson et al., 1988). French (1989) calculated a trapping efficiency of 54% (temporally varying between 30% and 79%) for a macro-tidal marsh in Norfolk, UK, whilst in a small micro-tidal marsh in Delaware, US approximately 80% of suspended sediment was lost from the water column within 12 m of a creek (Stumpf, 1983). Efficiency thus appears to be correlated with tidal range, with micro-tidal systems trapping virtually all of the sediment available to maintain
their elevation in the tidal frame, and macro-tidal marshes utilising a much smaller proportion of the sediment supply (French, 2006a). As Moskalski and Sommerfield (2012) note, however, these studies do not provide significant understanding of the controls on trapping efficiency, or of the small spatial variations that can occur within marshes. Their work seeks to provide some answers to those questions, by calculating efficiency across a number of transects in the St John’s River Estuary, Delaware, US. Trapping efficiency decreased with distance from the source of sediment (a tidal channel), likely related to vegetation effects, hydroperiod and flocculation processes. Further work is required to fully understand these influences, especially that of vegetation, and to produce a reliable measure of trapping efficiency.

1.5 Aims and Objectives

There are three main aims to this study, each with a subset of objectives and research questions. The research is conducted via an intensive case study of a restored saltmarsh and adjacent natural marshes in the Blackwater Estuary, Essex, in eastern England.

The first aim is to investigate the recent biogeomorphological evolution of both the restored and natural saltmarshes and how the nature of these marshes has changed over the last 20 years. To achieve this the following questions will be considered:

- Can the colonisation of the realignment site by halophytic vegetation be quantified using a series of high-resolution aerial photographs?
- Is there evidence of a reduction in the areal extent of the natural saltmarsh system over the last 20 years?
- Can the evolving topography of both the realignment site and the natural marshes be detected using LiDAR data and ground-based surveys?

The second aim is to examine and ascertain the relative controls on sediment accumulation processes and rates in natural and restored saltmarshes. The following research questions have been identified:

- Does seasonal climate variation influence sediment accumulation in both natural and restored saltmarshes?
- How do short-term sediment accumulation rates vary spatially across both saltmarsh systems?
Do geomorphological and hydrodynamic factors such as elevation, tidal conditions and suspended sediment concentration have a comparable influence on sediment deposition in natural and restored saltmarshes?

The third aim is to investigate the role of vegetation in sediment accumulation processes. Does vegetation increase sedimentation by reducing shear stress, or does it act to inhibit resuspension by reducing bottom stresses by wind waves? This aspect of the research will focus primarily on the restored saltmarsh, and the following questions will be addressed:

- Does the presence of vegetation affect the mass flux of sediment deposited?
- Does the presence of vegetation affect the amount of sediment retained at the marsh surface?
- Are variations in sediment deposition and retention related to plant species abundance and community structure?
- Does vegetation affect the efficiency with which the restored marsh sequesters sediment?
- Can hydrodynamic modelling increase our understanding of the processes driving sedimentation, especially regarding the role of vegetation?

Sediment accumulation is critical for the maintenance of marsh elevation within the tidal frame and for delivering the ecosystem functions and services marshes provide. This study will help to shed further light on the sedimentation process in saltmarshes through a combination of historic data analysis, intensive field research and numerical modelling.
2. Methodology

This study is conducted through a combination of historic data analysis, an intensive programme of field measurements and hydrodynamic modelling, with reference to a case study site in the Blackwater Estuary, Essex, southeast England. Saltmarshes in the Blackwater have a history of degradation and loss through both natural and human influences, and have been the focus of restoration efforts over the last couple of decades. The Tollesbury Fleet marshes were the site of the earliest full-scale managed realignment experiment in the UK. This provides an opportunity to investigate processes in a recently restored marsh surrounded by natural marshes that are currently experiencing erosional degradation and loss.

2.1 Research Design

The research design for the project has been split into three core areas of study in order to answer the three key strands of the objectives set out in the previous section. The first focuses on the recent evolution of both the restored and natural marshes at Tollesbury using aerial photographs to detect changes in vegetation, and topographical development using LiDAR data, and local-scale field measurements. The second comprises a comprehensive fieldwork campaign, divided into two halves. The first set of measurements was designed to answer the research questions for the second aim presented in Section 1.5, and was collected across both marsh systems over a year-long period. The detailed sampling strategy is discussed in Section 2.4.1. Following a preliminary analysis of the data collected from these field investigations, the findings showed a more exciting strand of research to be pursued, and the third aim regarding the role of vegetation was conceived. This avenue of research required a different strategy centred solely on the realignment site, to take advantage of the proximity of vegetated and unvegetated areas, and utilised a much more intensive programme of data collection and monitoring over shorter timescales. Some of the data collected as part of this scheme feeds into the third strand of research: hydrodynamic modelling. This will extend the conclusions drawn from the fieldwork and allows for the scaling-up of short term point measurements to gain insights into the longer term evolution of the realignment site.

Following the background information regarding the study site in the next section, the rest of the methodology chapter and the following results chapters are
organised into the three strands outlined above, before the broader implications and conclusions are drawn together in the final two chapters. Figure 2.1 shows how the research aims and objectives map onto the structure of the results chapters (Chapters 3-6).

<table>
<thead>
<tr>
<th>Aims &amp; Objectives</th>
<th>Chapters &amp; Sections</th>
</tr>
</thead>
<tbody>
<tr>
<td>Biogeomorphological evolution</td>
<td>3: Evolution of Tollesbury Marshes</td>
</tr>
<tr>
<td>Colonisation of realignment site</td>
<td>3.1: Vegetation Changes (aerial photographs)</td>
</tr>
<tr>
<td>Natural marsh extent</td>
<td>3.2 Marsh Fragmentation</td>
</tr>
<tr>
<td>Topographical development</td>
<td>3.3 Topographical Development (LiDAR and ground surveys)</td>
</tr>
<tr>
<td>Sedimentation processes</td>
<td>4: Sediment Accumulation Processes</td>
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<td>4.2 Temporal variation</td>
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<tr>
<td>Spatial variation</td>
<td>4.3 Spatial variability</td>
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<td>Forcing factors</td>
<td>4.4 Influence of vegetation</td>
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<td>5: Role of Vegetation</td>
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<td>5.2 Controls on Sedimentation</td>
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<td>Species abundance and structure</td>
<td>5.3 Influence of Vegetation</td>
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<tr>
<td>Trapping efficiency</td>
<td></td>
</tr>
<tr>
<td>Hydrodynamic modelling</td>
<td>6. Hydrodynamic Modelling</td>
</tr>
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</table>

**Figure 2.1: Linkages between thesis aims and objectives and the structure of the results chapters**
2.2 Case Study Location

The Blackwater Estuary (Figure 2.2) connects the catchments of the Rivers Chelmer and Blackwater (CEH, 2013), with the outer Thames Estuary and the southern North Sea. The catchment is approximately 1,200 km$^2$ and has a surface geology dominated by London clay and glacial deposits of chalky clay, sand and gravel (Hazelden and Boorman, 2001). The estuary has a tidal length of approximately 23 km from Beeleigh (just upstream of Maldon) in the west to Mersea Island in the east. The Blackwater is a vertically and longitudinally well-mixed estuary; salinity variation is thus not a factor controlling any processes (Emmerson et al., 1997b). The spring tidal range averages 4.5 m, so the estuary can be classified as macro-tidal. The tides are the dominant energy input into the estuary, since mean discharge from the local rivers is minimal in comparison to the tidal prism due to abstraction for public water supplies, 1.9 m$^3$s$^{-1}$ and 1.4 m$^3$s$^{-1}$ for the Chelmer and Blackwater respectively (Emmerson et al., 1997b). In the period between 1993 and 2011, sea level rose by 4.4 ± 1.1 mm yr$^{-1}$ at Lowestoft (101.4 km northeast of Tollesbury) and 7.4 ± 4.5 mm yr$^{-1}$ at Sheerness (35.7 km south of Tollesbury), leading to great concern over the future of marshes in the area (Wahl et al., 2013). In the estuary, the intertidal sediments comprise medium to fine grained silts.

Figure 2.2: Location of the Blackwater Estuary in the UK (inset) and areas of natural saltmarsh within the estuary
The study site near Tollesbury is marked. Saltmarsh areas derived from Burd (1992).
which have formed large areas of mudflats as well as narrow fringes of tidal marshes on the main estuary shores (as shown in Figure 2.2) (Emmerson et al., 1997b). More extensive marshes have developed in the sheltered embayments around the estuary, for example near Maylandsea and Abbots Hall, as well as on the large islands in the centre of the main channel (especially Northey Island) (Pye and French, 1993c). The saltmarsh area within the Blackwater Estuary was estimated at 684 ha in 1998 (Cooper et al., 2001).

The area of saltmarsh within the Blackwater Estuary and Essex as a whole was significantly higher in the medieval period than at present; over 40,000 ha were reclaimed for agriculture and protected by clay walls in the intervening period (Dixon et al., 1998). In the Blackwater, reclamation commenced around the late 1500s, and continued until approximately 1744 (Gramolt, 1961). Approximately 95% of the estuary is now defended by earth embankments; any areas left unprotected from the medieval period were enwalled following the devastating storm surge in 1953 (Pye and French, 1993b; Adams et al., 2012). The remaining saltmarsh is being eroded at a rate of 2% per year (Dixon et al., 1998), predominately by wave action, and is prevented by naturally migrating inland due to coastal squeeze. Approximately 90% of the saltmarsh area in Essex has been lost via human actions and natural processes over the last five centuries, but the proportions related to each is unknown (Burd, 1992; Dixon et al., 1998). In the Blackwater, almost 200 ha of marsh was eroded between 1973 and 1998 (Cooper et al., 2001). To help protect and preserve the marsh that is left, the entire estuary has been designated as a Site of Special Scientific Interest (SSSI), with some areas further protected as National Nature Reserves or Ramsar sites (Crossley, 1999).

The Blackwater has been the primary research site chosen by the Environment Agency and the Department for Environment, Food and Rural Affairs (Defra, formerly known as MAFF) for studies into saltmarsh restoration, and especially managed realignment (Emmerson et al., 1997a). The estuary was selected partly due to its tidal and sediment properties, but it is also still a relatively natural system when compared with other estuaries in the region; for example 21% of its intertidal area was lost to reclamation, in comparison with 88% in the nearby Crouch Estuary (Townend and Pethick, 2002). There are four major managed realignment trials in the Blackwater: at Tollesbury, Orplands, Abbots Hall and Northey Island (Figure 2.2). The last of these was the first such scheme to be carried out in the UK. The works were carried out between 1991 and 2002, and the size of the restored marshes ranges from 0.8 to 38 ha, totalling 92 ha throughout the estuary (Adams et al., 2012). Extensive monitoring
programmes collected data to examine the response of each site to realignment, as well as the wider impact on the estuary as a whole. As well as these four managed restoration sites, a natural breaching on Northey Island, occurring over 100 years ago during a storm, provides a mature analogue to the younger realignment sites.

2.2.1 Tollesbury Marshes

The field measurements for the present study were undertaken on the saltmarshes at the end of the Tollesbury Fleet on the north side of the Blackwater Estuary (Figure 2.3). An extensive area (130 ha) of mature natural marsh is dissected by a complex anastomosing creek system (Figure 2.4) and abundant saltpans. The marsh habitat is split into two named areas: the Tollesbury marshes and Old Hall marshes. The latter was purchased by the RSPB and has been run as a nature reserve since 1981 (Williams and Hall, 1987). The marsh edges are cliffed, and there are many unvegetated patches with mud mounds, many of which have been colonised by the algae *Enteromorpha* (Figure 2.5). Clay embankments, some reinforced with concrete blocks to repair erosion damage, back the

![Figure 2.3: Tollesbury Fleet, showing the realignment site (red bordered area) surrounded by natural saltmarsh.](image)
Species visible include *Salicornia spp.*, *Suaeda maritima* and *Atriplex Portulacoides*.

Algae on mounds are *Enteromorpha spp.*
natural marshes to protect the surrounding farmland. In some areas, wooden groynes have been constructed to minimise marsh losses through erosion (Garbutt et al., 2006b). The marsh vegetation is divided into two distinct communities: mid marsh flora and pioneer species; upper marsh and transition species are rare due to the unnatural barriers imposed by the sea walls (Garbutt et al., 2008). The mid marsh flora are found on the upper reaches of the marsh surfaces, and comprise primarily *Atriplex portulacoides* and *Puccinellia maritima*. Other species in this zone include *Limonium vulgare*, *Armeria maritima*, *Salicornia spp* and *Suaeda maritima*. The pioneer plants are mainly found on the creek margins, with *Aster tripolium* dominating this species-poor community; some *Salicornia spp* and *Suaeda maritima* are also present.

The 21 ha managed realignment site adjacent to these natural marshes was formerly arable land. 150 years ago, the site was reclaimed and drained from marshland for grassland and protected by a sea wall, before being converted to agricultural use and divided into three fields, separated by an open ditch and hedgerows (Watts et al., 2003; Burden et al., 2013). Planning on the realignment began in 1993, with work commencing on a new earth-bank sea wall, to form a secondary defence and boundary to the west and south of the site. This was completed in July 1995, and a 50 m breach was made in the old sea wall on 4 August 1995 (Meadowcroft et al., 1996). At the time, this was the largest such scheme carried out in the UK, and was primarily an experiment to test the concept of managed realignment on a large scale and its longer-term development. Prior to breaching, the only engineering work undertaken within the site was to connect the main ditch to the tidal entrance (Garbutt et al., 2006b).

Following realignment, a short-term research programme was initiated to monitor the development of the site, with results published in reports for the EA and Defra (e.g. Reading et al., 2002, 2006, 2008b), as well as a number of scientific papers (Wolters et al., 2005b; Garbutt et al., 2006b; Garbutt and Wolters, 2008). Research focused initially on the vegetation and hydrology, but later expanded to include invertebrates and other fauna, changes in the soil structure as well as sediment accretion studies. The main monitoring ended in 2008, but a number of other studies have investigated the biogeochemistry, bird colonisation, vegetation productivity as well as the general evolution of the realignment site since this date (e.g. Boorman and Ashton, 1997; Emmerson et al., 1997b; Chang et al., 1998; Cahoon et al., 2000; Atkinson et al.,
A variety of research continues to be undertaken at Tollesbury to this day.

The realignment site has evolved considerably over the 20 years since restoration of tidal exchange. In common with most formerly reclaimed land in the region, the elevation was 1.0 to 1.5 m lower than the surrounding marshes (Pethick and Burd, 1996), with those areas nearest the breach site lower than areas fronting the new sea wall to the south and west. Sediment accreted rapidly across the site after the breaching, with rates of 25 mm yr\(^{-1}\) averaged over the first five years (Reading et al., 2008b). Accretion slowed after 2003 (13 mm yr\(^{-1}\) averaged over 2003-2007), due to the infilling and associated reduction in hydroperiod and sediment supply. By 2007, 12 years after the exposure to tidal inundation, approximately 13 ha of the site was covered by saltmarsh vegetation made up of 21 species, all typical of the region (Garbutt et al., 2008). Around the fronting edge of the new sea wall, _Puccinellia maritima_ dominates, with some _Salicornia europaea_, _Limonium vulgare_ and _Atriplex portulacoides_ (Figure 2.6). The rest of the vegetated area is characterised mainly by _Spartina anglica_ and _Salicornia europaea_, the latter dominating the central vegetated zone near the breach (Figure 2.7). The nationally scarce _Inula crithmoides_, _Spartina maritima_, _Sarcocornia perennis_ and _Suaeda vera_ are also found in the realignment site. Many areas of the site remain devoid of vegetation, however, and are characterised by thick, poorly consolidated mud deposits. Invertebrates have also colonised the realignment site; in fact there are more species there (21) than in the natural marshes (16) (Reading et al., 2008a). They are also found in higher numbers within the restored site. However, the rate of increase in invertebrate density has slowed after its peak in 1998. The gastropod _Hydrobia ulvae_ is the most abundant species at 14,362 individuals per square metre on average.

The Tollesbury managed realignment site and surrounding marshes are an excellent case study for studying the biogeomorphological processes within restored and natural saltmarsh systems. As the first large-scale realignment project in the UK, there is considerable background information and monitoring data freely available; this is often not the case for other sites. The partial development of vegetation allows for the mudflats and vegetated boundary to be studied. Mudflats are often not present in other sites, as these have colonised more rapidly; plants would need to be removed from these locations to recreate the same conditions found naturally at Tollesbury.
Figure 2.6: High marsh species near the northern sea wall in the realignment site
Species visible include *Spartina anglica* and *Atriplex Portulacoides*.

Figure 2.7: Mud mounds and pioneer species near the breach in the realignment site
Vegetation in this part of the realignment site is comprised almost entirely of *Salicornia perennis*, with some patches of *Enteromorpha* algae.
2.3 Historic Data Analysis

2.3.1 Aerial Photography

High-resolution aerial photography was obtained directly from the Environment Agency (EA) and Bing Maps in order to investigate the longer-term colonisation of vegetation within the managed realignement site, and changes in the vegetation cover in the natural saltmarshes at Tollesbury. Many researchers have used aerial photographs to study coastal developments, and they are regarded as a source of reliable, accurate data with few limitations (Moore, 2000; Cooper et al., 2001; Cox et al., 2003; Smith, 2009; Baily and Inkpen, 2013). The main weakness is that a photograph is a snapshot in time, and may not represent the average condition of the environment (Smith and Zarillo, 1990), but this is unlikely to be a problem in a saltmarsh, due to the timescales over which change occurs. Table 2.1 lists the details of the imagery studied. All photographs were taken at low water on a single day during the summer months of the years listed. The 2001 image was only available as a hard copy photograph, so a high-resolution (1200 dpi) scan was made. The other images were sourced in a digital format and analysed in the GIS program ArcMap 10.2.2. The 2001 scan was georeferenced against the 2008 image using the automatic registration tool to ensure each dataset could be compared. All images capture the entire region around Tollesbury, except 1997 which excludes one third of the realignment site and all the natural marshes to the west of this area.

The spread of vegetation in the realignment site was investigated separately to any changes in the natural marshes, so a mask was used to exclude the sea wall and surrounding areas, to simplify the change detection process. A supervised classification (maximum likelihood method) (Tso and Mather, 2009; Richards, 2013) was conducted on the five images, (each comprising three bands – red, blue and green) to categorise each pixel into a particular class. A low level classification of each image was the final

<table>
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<th>Image</th>
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<th>Source</th>
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<tr>
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<td>EA</td>
</tr>
<tr>
<td>2001</td>
<td>14/08/2001</td>
<td>1:5,000</td>
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</tr>
<tr>
<td>2013</td>
<td>Summer 2013</td>
<td>1:5,000</td>
<td>EA</td>
</tr>
</tbody>
</table>
result desired, where just two classes were present – vegetated and unvegetated. For the purposes of this study, any areas of algae and water are considered as unvegetated (Garbutt et al., 2002). A test classification was undertaken using training areas for these two classes, but many pixels were classified incorrectly due to the wide variation and overlapping nature of vegetation colour, mud and algae. Extra classes were thus used in the classification process which would then be re-classed into the relevant vegetated / unvegetated group. These groups included algae, water, shadow and scrubland (shrubs and grasses at the foot of the sea wall). The result of the initial classification of the 2008 aerial photograph is shown in Figure 2.8. During this process, 154 training areas were used which were well distributed and representative of each class; these areas equated to 20.3% of the total image. A similar proportion of training areas were used for each photograph.

Although it captures the broad areas of mud, vegetation and most of the algae, there are significant errors in the classification, including the areas of scrub in the centre of the site, and the patches of bare mud that are shown to be near the southern sea wall. If all areas of algae were reclassified as bare ground, and scrub as vegetation, the result would not be correct. A similar first result was obtained for the four other images. In

![Figure 2.8: Initial classification of 2008 aerial photograph](image-url)
order to achieve a better classification, each dataset was then manually edited using the ARIS GRID Editor for ArcMap (ARIS B.V., 2014). Groups of incorrectly classified pixels (using the aerial photograph to verify) were re-classed to either vegetated or unvegetated ground. The change in area of each class between each dataset was calculated by subtracting one map from another.

In the natural marshes, no changes were detectable from a simple visual comparison of each photograph. As manual edits of every classified map for the entire marsh area (240 ha) would be too time-consuming, two small test areas were chosen to determine if there was any change in vegetation between the 1997 and 2011 images. The 2013 image had not been obtained when the analysis was undertaken. Each test area was classified in the same way, using training areas that represented bare mud, vegetation, and shadows. No conclusive changes were detected in either area, so the process was not expanded to the rest of the natural marsh area.

2.3.2 Elevation Data

Historic LiDAR datasets were also obtained from the Environment Agency as part of the NERC iCOASST project (Nicholls et al., 2012). LiDAR (Light Detection and Ranging), also known as scanning laser altimetry, is an airborne terrain mapping system, analogous to Radar and Sonar in its principle of operation (Woolard and Colby, 2002; Blott and Pye, 2004). Fitted to a helicopter or aeroplane flying at a height of 300-800 m, LiDAR systems emit near-infrared laser pulses, and measure the distance between the scanner and the ground using the time it takes for the pulse to hit the ground and return to the detector (French, 2003; Hladik and Alber, 2012; Klemas, 2013). Combined with the precise position and altitude of the scanner unit each time a pulse is emitted, collected from the aircraft flight systems, the coordinate of the reflection point on the ground is calculated (Reutebuch et al., 2005). A Digital Elevation Model (DEM) is generated by interpolating the point data over the surveyed area (Cobby et al., 2001). The LiDAR products supplied by the EA are Digital Surface Models (DSMs) in that they represent the ground surface and all reflective surfaces on it, including buildings and vegetation. This is in contrast to a Digital Terrain Model (DTM), which depicts the elevation of the bare ground without these objects (Maune et al., 2001). Further details on the processes of how LiDAR data is captured and processed is given in Wehr and Lohr (1999).
LiDAR has now become the leading technology for high-resolution topographical coastal surveys, surpassing the use of traditional ground-based techniques (Carter et al., 2001; Reutebuch et al., 2005; Tarolli, 2014). Cross-sections across transect lines collected at intervals along the coast using a theodolite or total station, taken with reference to a fixed benchmark yield an accuracy of ±5 cm, whilst more modern dGPS techniques providing grid-based spot heights can achieve vertical accuracy of ±1 cm (Morton et al., 1993; Saye et al., 2005). These ground-based methods have several disadvantages: they are too labour-intensive to create a high-resolution dataset over a large area, and are impractical in inaccessible or unsafe areas, such as cliffs or saltmarsh channels and mudflats (Mason et al., 2000; Lohani and Mason, 2001). In contrast, LiDAR data commonly achieves horizontal and vertical accuracies of ±10 cm and better, and can be collected easily and quickly over an extensive study area, regardless of whether the location is suitable for ground surveying (Rayburg et al., 2009; Roering et al., 2013). Geomorphological applications that have used LiDAR surveys include investigations of landslides (McKean and Roering, 2004; Glenn et al., 2006; Tarolli and Tarboton, 2006), volcanoes (Davila et al., 2007; Csatho et al., 2008; Bisson et al., 2009), drainage channel extraction (Tarolli and Dalla Fontana, 2009; Pirotti and Tarolli, 2010; Pelletier, 2013), river morphology (Magirl et al., 2005; Thoma et al., 2005; Jones et al., 2007; Hilldale and Raff, 2008; De Rose and Basher, 2011), as well as saltmarshes (Blott and Pye, 2004; Morris et al., 2005; Rosso et al., 2006; Millette et al., 2010; Bertels et al., 2011; Hladik and Alber, 2012; Hladik et al., 2013).

The available LiDAR data covering the Tollesbury region are listed in Table 2.2. The exact date of each flight is unknown, but data were collected at low tide using an Optech ALTM Gemini sensor (Environment Agency, 2015). After pre-processing, the LiDAR data were provided with the Easting, Northing, and elevation of each point in the British National Grid coordinate system and gridded into 2 km × 2 km tiles. Elevation measurements are provided at 2 m horizontal intervals between 1999 and 2002, and at 1 m intervals from 2009. The 2008 flight recorded data in 0.25 m grid cells, but unfortunately the survey captured only a small portion of the natural marshes (42%) and none of the realignment site (Table 2.2). It was thus excluded from all analysis undertaken. The 2009 dataset was also excluded from the analysis of the realignment site, due to the low coverage (66.5%), but was included in that of the natural marsh area. The small areas of no data in the 2000, 2001, 2002, 2011 and 2013 surveys are due to areas of standing water being impenetrable to the laser pulses and
Table 2.2: LiDAR datasets covering the Tollesbury marshes

<table>
<thead>
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<th>Year</th>
<th>Date flown</th>
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<th>Natural marshes</th>
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<td>Sept 00</td>
<td>2</td>
<td>99.7</td>
<td>98.3</td>
<td></td>
</tr>
<tr>
<td>2001</td>
<td>Jun 01</td>
<td>2</td>
<td>99.7</td>
<td>98.4</td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>Oct 02</td>
<td>2</td>
<td>100.0</td>
<td>98.8</td>
<td></td>
</tr>
<tr>
<td>2008</td>
<td>Jun – Oct 08</td>
<td>0.25</td>
<td>0.0</td>
<td>42.4</td>
<td></td>
</tr>
<tr>
<td>2009</td>
<td>Mar 09</td>
<td>1</td>
<td>66.5</td>
<td>90.3</td>
<td></td>
</tr>
<tr>
<td>2011</td>
<td>Nov 11 – Feb 12</td>
<td>1</td>
<td>99.9</td>
<td>98.7</td>
<td></td>
</tr>
<tr>
<td>2013</td>
<td>May 13</td>
<td>1</td>
<td>99.9</td>
<td>99.5</td>
<td></td>
</tr>
</tbody>
</table>

should not cause any problems in the analysis. The 1999 dataset, however, had more serious issues with the data coverage. Shown in Figure 2.9, the areas of no data are located only within a ~700 m wide diagonal stripe covering a portion of both the realignment site and the surrounding marshes. These areas do seem to be mainly confined to the creek network within the natural marshes, indicating they represent regions of surface water. The reason they only appear in this stripe and not in the rest of the dataset is unclear; whether the issues were caused during the data capture from that specific fly past or in subsequent processing is unknown. In addition to these missing areas, the elevation of the marsh surface and channel beds within this stripe are 0.5 – 2.0 m higher than those points recorded outside this region. This survey was thus also excluded from all subsequent analysis.

The horizontal accuracy of the remaining surveys, quoted as ±5 cm (Marks and Bates, 2000), was tested by comparing the position of a number of control points in the village of Tollesbury (primarily the edges of buildings) taken from the OS 1:1,000 MasterMap with the same points visible in each of the LiDAR datasets. Due to the nature of the topography, no definitive control points could be found closer to the marsh area. The summary of the results is listed in Table 2.3. The mean residuals improve with the more recent surveys, but they are all above the quoted accuracy of ±5 cm. This is considered to be acceptable for the present study, however, especially as the residuals listed are likely to overestimate the errors in the LiDAR positioning due to the difficulty in defining the exact location of the edge of a building in the elevation data.
Figure 2.9: 1999 LiDAR elevation of the Tollesbury marshes

Table 2.3: Horizontal difference between map control points and LiDAR surveys

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Horizontal residuals (m)</td>
<td>Mean</td>
<td>1.84</td>
<td>1.55</td>
<td>1.29</td>
<td>0.79</td>
<td>0.78</td>
</tr>
<tr>
<td></td>
<td>Minimum</td>
<td>0.60</td>
<td>0.75</td>
<td>0.57</td>
<td>0.16</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>Maximum</td>
<td>3.41</td>
<td>3.43</td>
<td>2.40</td>
<td>1.83</td>
<td>1.82</td>
</tr>
<tr>
<td></td>
<td>Count</td>
<td>18</td>
<td>22</td>
<td>22</td>
<td>14</td>
<td>25</td>
</tr>
</tbody>
</table>

Most importantly, there is no evidence for any systematic horizontal offset in any of the surveys.

The vertical accuracy of the LiDAR is claimed to be less than ±15cm, with recent surveys close to ±5cm and random height errors expected to be no more than ±5cm (Environment Agency, 2015). Vertical inaccuracies occur in LiDAR data due to errors in the GPS and altitude systems in the aircraft, local air turbulence, limitations
with the laser scanner angle and range, and during processing and coordinate conversion (Lohani and Mason, 2001; French, 2003; McKeen et al., 2014). These errors can lead to greater differences between the actual elevation and that measured than that stated by the data provider (Blott and Pye, 2004; Rosso et al., 2006; Rayburg et al., 2009). The characteristics of the area surveyed are also important in determining the accuracy with low-altitude flat terrain likely to yield a much more accurate dataset than areas with more complex topography (Aguilar et al., 2010). Hodgson and Bresnahan (2004) found observable elevation errors on steeper slopes (e.g. 25°) were twice as large as those on shallow slopes (less than 4°).

To determine the vertical accuracy of the 2013 LiDAR survey, the recorded elevation was compared with that measured at 36 control points within the realignment site. These spot heights were measured using a dGPS system (Leica 500 and 1200 base/rover combination) in March 2014 whilst undertaking fieldwork. The mean difference between the measurements was 7 cm, with the maximum difference being 26 cm. In all but three cases, the LiDAR data was higher than that measured with the dGPS, suggesting a positive bias likely due to interference from vegetation (Morris et al., 2005). The LiDAR products are all DSMs, which include vegetation, whereas the dGPS was used to measure the ground surface. Regardless of this issue, the average difference falls well within the quoted accuracy of the LiDAR system used.

In order to measure the accuracy of the earlier surveys, a set of control points were defined in Tollesbury village, with the elevation at each point compared with that of the 2013 dataset. The points were all located on hard level surfaces (chosen from the 2011 aerial photograph), primarily in the centre of roads, which should not have changed significantly in elevation over time. The summary of the differences at these points between the earlier surveys and that from 2013 is shown in Table 2.4. Although the roads used for the control points are not totally smooth surfaces and may well have

<table>
<thead>
<tr>
<th>Elevation difference (mm)</th>
<th>2000</th>
<th>2001</th>
<th>2002</th>
<th>2009</th>
<th>2011</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>180</td>
<td>87</td>
<td>166</td>
<td>100</td>
<td>6</td>
</tr>
<tr>
<td>Minimum</td>
<td>-199</td>
<td>-123</td>
<td>-12</td>
<td>0</td>
<td>-70</td>
</tr>
<tr>
<td>Maximum</td>
<td>488</td>
<td>379</td>
<td>299</td>
<td>180</td>
<td>90</td>
</tr>
<tr>
<td>Count</td>
<td>147</td>
<td>147</td>
<td>147</td>
<td>102</td>
<td>147</td>
</tr>
</tbody>
</table>

Table 2.4: Difference between 2013 LiDAR control point elevations and prior surveys
changed slightly (due to resurfacing for example), they provide a measure of the baseline elevation variability in the earlier surveys (Rosso et al., 2006). In general, the difference in elevation increased alongside the age of the survey, with the majority of the points within 20 cm of the 2013 elevations. To bring the older surveys in line with the most recent dataset, the mean elevation difference listed in Table 2.4 was added to the relevant survey. No further processing of the LiDAR was undertaken before the analysis with the exception of the removal of some anomalously high pixels (20-40 m high) in the 2000 survey.

Although the LiDAR products provide the ability to calculate elevation change at a high horizontal resolution throughout the Tollesbury marshes, their vertical accuracy are typically an order of magnitude below that needed to resolve saltmarsh elevation change over annual timescales (Cahoon et al., 2000; Thomas and Ridd, 2004). Data from six Sediment-Erosion Tables (SETs) installed by Cahoon et al. (2000) in 1995 at Tollesbury and measured repeatedly by Prof J French and Dr H Burningham have been obtained to compliment the LiDAR products and provide a more accurate assessment of the longer-term changes in marsh elevation. Originally designed by Boumans and Day (1993), an SET is a portable, mechanical levelling device that integrates both surface (deposition and erosion) and sub-surface processes (e.g. compaction, shrinking and swelling) (French et al., 2000). At each sampling location, a seat pipe is driven into the ground to refusal (3-6 m) and provides a stable datum for measurement. A removable arm is slotted into the top of the pipe, secured and levelled and nine pins are lowered from the end of the arm to the sediment surface. A diagram showing the SET is given in Figure 2.10. The length of each pin above the SET is measured using a ruler; over time these data are used to calculate elevation change. The arm can be locked into four different orientations around the seat pipe, giving 36 measurements for each sampling location. On this basis, the SET provides a measure of elevation change with a vertical accuracy of ±1.5 – ±2.0 mm (Boumans and Day, 1993; French et al., 2000). At Tollesbury, four SETs were installed in the realignment site, and two in the natural marshes just outside the site’s northern sea wall and were monitored between 1995 and 2014. A location map is provided when the results are discussed in Chapter 3.
Figure 2.10: Diagram of Sediment-Erosion Table installed and monitored at Tollesbury
Adapted from (Cahoon et al., 1995, 2002).
2.4 Field Data Acquisition

2.4.1 Sampling Strategy

The fieldwork programme for this study has been conducted through a series of campaigns designed to fulfil the different objectives of the research. As described in Section 2.1, these campaigns fall into two categories: those used to study the seasonal and spatial variation in sediment accumulation across both marsh systems, and those designed to investigate the role of vegetation in the sedimentation process in the realignment site. The following paragraphs describe the timings of and sampling schemes used for each fieldwork campaign.

The seasonal fieldwork campaign was carried out in the natural and restored marshes at Tollesbury between May 2011 and February 2012. A two-week deployment of sediment traps (see below) was undertaken every three months (Table 2.5). The seasonal sampling strategy allows for the effect of the vegetation growth cycle to be captured, as well as varied weather conditions. Individual dates were primarily selected to ensure suitable tide times and heights to allow safe access at low water.

The aerial photographs and LiDAR obtained from the Environment Agency were used to choose the locations of each sediment trap. A regular sampling strategy was precluded by the difficulties of safely accessing specific locations over a long enough portion of the tidal cycle. Accordingly, the positions of existing paths and bridges over the main creeks were discerned from the aerial imagery and used to locate a set of sampling points that covered a range of elevations, proximity to the channel system and vegetation cover. A number of these correspond with locations used in the original monitoring studies by Reading et al. (2002, 2006, 2008b), as well as the SETs installed in 1995 (Cahoon et al., 2000). The final sampling scheme comprised 33 points (Figure 2.11).

Table 2.5: Seasonal fieldwork campaign dates

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Day 1</th>
<th>Day 8</th>
<th>Day 15</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2011</td>
<td>08/05/11</td>
<td>15/05/11</td>
<td>22/05/11</td>
</tr>
<tr>
<td>August 2011</td>
<td>07/08/11</td>
<td>14/08/11</td>
<td>21/08/11</td>
</tr>
<tr>
<td>November 2011</td>
<td>06/11/11</td>
<td>13/11/11</td>
<td>20/11/11</td>
</tr>
<tr>
<td>February 2012</td>
<td>12/02/12</td>
<td>19/02/12</td>
<td>26/02/12</td>
</tr>
</tbody>
</table>
In the seasonal campaign, three pairs of points were purposely located across the saltmarsh – mudflat transition within the realignment site in order to capture the local effect of the presence or absence of vegetation. In each pair (points 19 and 20, 23 and 24, 30 and 31 in Figure 2.11), one sample point was positioned approximately 0.5-1.0 m in front of the marsh margin in the unvegetated area, and the other 0.5-1.0 m into the vegetation. As the points were so close, all other influencing factors such as elevation and the distance from sediment source were essentially equal. Painted marker poles were installed at every point to facilitate easy relocation of their position during the main fieldwork period. Each point was numbered prior to the post installation, and that numbering system has been retained to avoid confusion, even though points 25 and 26 were never used. The locations of these two points was originally planned to be within the large expanse of bare mudflat between points 24 and 27 (Figure 2.10), but on the first field campaign it was decided it would be too dangerous to repeatedly traverse such a large stretch of unconsolidated mudflat.

Following the analysis of the results obtained from these transition sites, the second research aim was conceived and the role of vegetation in the sedimentation process was investigated more thoroughly. This was achieved through two further fieldwork campaigns carried out in May 2012 and March 2014. These trips were conducted over five or six consecutive days, during which sediment traps were...
deployed and monitoring equipment installed (detailed in sections 2.4.2 and 2.4.5 respectively). Some of this equipment was left in position for 4-6 weeks before being retrieved. Table 2.6 lists the dates over which these two campaigns were carried out.

The May 2012 fieldwork was designed to gather more data to add support to the preliminary findings gathered from the seasonal campaign. A subset of 23 points was selected from the original 33 to ensure that every site could be visited each day for the four day deployment. These are shown in Figure 2.12. Five points in the natural marshes were included to investigate any difference in sedimentation between these and the realignment area over the shorter timescales being studied during this period.

The results from this campaign did not yield enough data for a thorough investigation of the influence of vegetation, so a further campaign was conducted in March 2014, the sampling strategy for which was totally overhauled and focused solely on the realignment site. All data collected during this fieldwork was specifically designed to study the role of vegetation, no other factors were considered when designing the trap layout. The previous campaigns negated the need to include points in the natural marshes. The sampling strategy was based around nine transects, each containing four points – two located in the transition zone around the vegetation margin, 0.5-3 m in front of and within the vegetation; one between 20 and 50 m within the vegetation from this margin, and one 5-25 m into the unvegetated zone. In this scheme, therefore, half of all points are located in vegetated areas, and half in unvegetated areas. This was the primary factor in choosing the locations of the transects; the north east and south west corners were excluded because they comprise solely of unvegetated and vegetated areas respectively and would not have provided a comparison across the vegetation transition. Figure 2.13 shows the layout of the transects; although 13 of the original sites were used, a further 23 were added and a new numbering sequence was created.

Table 2.6: Intensive monitoring field campaigns

<table>
<thead>
<tr>
<th>Campaign</th>
<th>First Day</th>
<th>Final Day</th>
<th>Retrieval</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2012</td>
<td>29/04/12</td>
<td>03/05/12</td>
<td>29/05/12</td>
</tr>
<tr>
<td>March 2014</td>
<td>22/03/14</td>
<td>27/03/14</td>
<td>07/05/14</td>
</tr>
</tbody>
</table>
Figure 2.12: Sampling strategy for May 2012 sediment traps

Figure 2.13: Sampling strategy for March 2014 sediment traps
2.4.2 Sediment Deposition

There are a variety of techniques available to measure the rate of sediment deposition and/or surface elevation change within saltmarshes. These include marker horizons, erosion pins, Sedimentation-Erosion Tables (SETs) and radionuclide activity profiles (e.g. Boumans and Day, 1993; Childers et al., 1993; Cundy and Croudace, 1995; Darke and Megonigal, 2003; Swift et al., 2006; Cundy et al., 2007; Goodman et al., 2007; Takekawa et al., 2010). The method chosen generally reflects the resources available, the accuracy and spatial and temporal resolution required and the timescale of interest (Thomas and Ridd, 2004; Marion et al., 2005).

In this study, short-term accretion rates were determined from the mass of sediment deposited on filter paper sediment traps placed directly on the marsh surface. First used by Reed (1989), the collection of deposited sediment rather than the measurement of its vertical accretion allows for more frequent sampling and the investigation of short-term sediment dynamics and specific events, such as storms. Another key advantage in measuring mass flux is that it excludes sediment compaction, which varies between vegetation types and affects the longer-term elevation change (Leonard et al., 1995). Many researchers (e.g. Reed, 1992; French et al., 1995; Leonard, 1997; Brown, 1998; Neumeier and Ciavola, 2004; Van Proosdij et al., 2006a) have used filter papers as an inexpensive method with a high vertical resolution well suited for intertidal environments. Large numbers of traps can be used over multiple deployments and provide a good spatial resolution of sedimentation (Marion et al., 2009). Extensive sampling arrangements can be labour-intensive, however, given the need to install and revisit for each measurement (Callaway et al., 2001).

The present study utilised 55 mm diameter Whatman 542 hardened ashless filter papers placed on the lid of a plastic Petri dish and secured to the marsh surface with four galvanised wire pins (Figure 2.14). The Petri dish lid protects the filter paper from the underlying sediment and facilitates easy retrieval. A small strip of wire mesh was placed on the surface beneath the Petri dish lid to ensure the trap did not sink and become flush with or lower than the surrounding sediment. Care was taken to minimise disturbance during site visits. During the seasonal campaign, at each of the 33 sampling points, a total of nine traps were laid during each deployment (Table 2.5). On day 1, six traps were laid at each point, in a semi-circular arrangement around the marker post, for easy relocation. On day 8 (a week later), three of these filter papers were collected and replaced with a fresh set, and on day 15 (two weeks after the initial placement), all six remaining traps were retrieved. Three papers were used for each period in case some
were lost to the tide or disturbed by birds. This overlapping hierarchy resulted in a set of papers collecting sediment for the entire two-week deployment, one set for the first week, and the final set covering the second week. This technique, as used by Hutchinson et al. (1995) allows for a measure of sediment retention to be calculated based on the sum of the one week traps compared with those for the whole two week period. The difference between the two values will primarily be due to any reworking or resuspension of the already deposited sediment, a process that is crucial in determining the influence of vegetation in the sedimentation process (Christiansen, 1998).

Prior to deployment, each filter paper was dried at 50°C for three hours in a drying cabinet to remove any moisture, placed in a desiccator to cool, and then weighed to four decimal places. Each paper was handled with tweezers at all times to minimise contamination and stored within a labelled Petri dish before and after being deployed. Once retrieved and back in the laboratory, each filter paper was photographed, any foreign objects such as algae and Hydrobia ulvae shells removed, and then placed in a Buchner funnel connected to a vacuum pump. 250 ml of fresh water was added to the funnel and allowed to rest for five minutes to dissolve any salt particles. The water was then passed slowly through the filter using the pump, before the paper was placed back inside the Petri dish, dried at 50°C for at least 24 hours, cooled and re-weighed as before.
To quantify the error associated with the processing of the filter papers in the laboratory, 15 extra papers were used as blanks for each fieldwork period. These papers were weighed alongside those used for deployment, but then stored until after the others were collected from the marsh. They were then subjected to the same rinsing and drying before being reweighed. Any difference in their weight was thus associated with the laboratory handling and processing. This method for identifying errors follows the procedure used by Dyer and Moffat (1998) and French et al. (2000). All traps and blanks were processed both before and after deployment in a random order so that any systematic errors, such as changes in temperature or humidity in the laboratory, which may have led to batches of filters with slightly different moisture content, were eliminated. Deposition rates have been normalised with respect to time, and expressed in units of g m\(^{-2}\) tide\(^{-1}\) to standardise the data throughout the sampling periods and for easy comparison. It does not imply that the same amount of sediment was actually deposited on each tidal cycle for the deployment concerned.

During the May 2012 intensive monitoring field campaign, sediment traps were deployed across 23 of the original 33 sampling sites over a three day period. A total of eight filter papers were used per point, this time in batches of two. One set was deployed for the whole three day period, whilst the other three sets were retrieved after being used over the intervening individual days, in the same hierarchical manner as the main fieldwork programme. The March 2014 campaign was conducted in the same manner, deploying traps over a daily timescale and focusing solely on the 36 sites outlined above in the realignment site. Again, eight traps were used per site over the study period. Each paper was processed as described above.

One of the main influencing factors controlling the spatial variation in sediment deposition is elevation, so in October 2011, a differential-GPS survey (Leica 500 and 1200 base/rover combination) was carried out. Three measurements were taken at each site, at 30 second intervals, to obtain the best possible precision and accuracy of the horizontal and vertical position. The location of the nearest Ordnance Survey benchmark, approximately 15 km away from Tollesbury, was also measured using the rover unit so that the coordinates obtained within the marshes were related to the national system as accurately as possible. The survey was repeated in February 2014 to record the heights and locations of the new sites included in the March 2014 campaign.
2.4.3 Suspended Sediment Concentration

By comparing the amount of sediment deposited on the marsh surface with the total available suspended in the water column, the efficiency at which the marsh sequesters sediment can be determined (French, 2006a; Van Proosdij et al., 2006b). Sampling the suspended sediment concentration (SSC) across saltmarshes and mudflats is made difficult by the practicalities of accessing such environments when the measurements must be obtained – during the flood tide. Some studies collect bottle samples directly from the water surface from small boats, but this is expensive, and logistically impossible in the majority of locations (Collins, 1976). SSC is thus typically measured using equipment installed on the marsh during low tide, which either monitors the sediment concentration continuously, e.g. optical backscatter sensors (Christiansen et al., 2000; Dyer et al., 2000), or takes a water sample during the flood tide, which is then analysed in the laboratory (Leonard et al., 1995; Darke and Megonigal, 2003). For ease of application and cost, this final method was used at Tollesbury.

In this study, suspended matter was collected using a siphon sampler (Bouma et al., 2005a; Chen et al., 2005), a schematic of which is shown in Figure 2.15. Comprised of a one litre plastic bottle, two tubes are driven into a cork bung inserted into the bottle opening. Water enters the bottle via the lower inlet tube and air is allowed to escape via the vent. During the seasonal field campaign, siphon samplers were used in pairs at six points (numbers 2, 3, 5, 16, 19, 27 in Figure 2.11). On day 1 of each seasonal period, a hole was dug in the sediment using a bulb planter, and the bottle inserted so that the tip of the inlet tube was approximately 10 cm above the sediment surface. A wire pin was then used to secure the bottle in place. The samplers were located near to the sediment traps, but not so close as to disturb the flow patterns around them. On day 8, the bottles were removed from the surface, sealed, and a new batch was re-inserted into the hole. These bottles were then recovered on day 15, so that two one-litre water samples were collected for each week of the study period. Once retrieved from the field, the water samples were stored in a cold room to minimise biological activity.

In May 2012, the number of SSC sample sites was expanded to 12 (points 7, 10, 11, 16, 17, 19, 27, 28, 30, 32, 34, 35; Figure 2.12). As before, a pair of siphon samplers was installed at each point, with the bottles being retrieved after two tidal cycles. A total of six samples were thus collected from each site over the three day period. In March 2014, the 12 sites chosen for SSC sampling were sites 5-8, 9-12 and 21-24 (see Figure 2.13). These were selected to investigate any change in SSC levels, and thus trapping efficiency, across the transition from unvegetated mudflat to vegetated saltmarsh along
Figure 2.15: Diagram of the siphon sampler setup used

three of the transects in the realignment site. The three transects investigated incorporate a range of other key variables in the restored area, including distance from the breach, elevation and vegetation type. Six samples were collected in total from each point over the three day period in the same manner as in 2012.

The process for obtaining the SSC from each water sample is similar to that of the sediment deposition described above. Instead of the Buchner funnel, a vacuum manifold was used to pass the water sample through the filter paper (Buller, 1975; Wolanski et al., 2000; Day et al., 2009). Its volume was determined using a measuring cylinder, and the sample poured into the funnel over the filter paper, and drawn through it by the vacuum. Both the measuring cylinder and the sample bottle were thoroughly rinsed and emptied into the funnel to be captured by the filter. Once the water had drained, the paper was placed in a Petri dish and dried for at least 24 hours at 50 °C and re-weighed. Extra papers were subjected to the same filtering procedure using one litre
of tap water to test the analytical error of the technique: five for the seasonal periods, and seven for the later campaigns (as performed by Dyer and Moffat, 1998).

2.4.4 Sediment and Vegetation Characteristics

The properties of marsh sediment have a considerable influence on its susceptibility to resuspension and erosion (Jepsen et al., 1997; Aberle et al., 2004; Grabowski et al., 2011; Seco et al., 2016), so investigating their spatial variation will be useful in explaining the longer term development of the natural marshes and realignment site, as well as the shorter term rates of sediment deposition. Accordingly samples of the surface sediment were acquired at all trap sampling points. Sediment bulk density was determined using density rings (Watts et al., 2003; Burden et al., 2013), with samples then sealed in labelled pre-weighed bags and stored in a cold room to retain moisture. Three samples were taken from each point. In the laboratory, the samples were weighed in the bags, before being dried. Due to the size of the samples, they were left in the drying cabinet at 50°C for a week, and then dried to a constant weight, still within the original bag (Callaway et al., 2001). Samples for the original 33 points were collected in March 2011, and all final 36 points in May 2014. The wet and dry bulk density and the water content for each sample were calculated using the formulas given by the British Standards Institution (1998):

\[
\text{Wet density} = \frac{m_w}{v_s} \quad (2.1)
\]

\[
\text{Dry density} = \frac{m_d}{v_s} \quad (2.2)
\]

\[
\text{Moisture content} = \frac{m_w - m_d}{m_w} \times 100 \quad (2.3)
\]

where \(m_w\) is the mass of the wet sediment, \(m_d\) is the mass of the dry sediment, and \(v_s\) is the volume of the sediment.

The shear strength of the sediment at each sampling point was measured using a hand-operated 19 mm Pilcon Shear Vane (Couperthwaite et al., 1998; Whitehouse et al., 2000; Jones et al., 2006). Shear strength was measured at 10 cm intervals up to 1 m in depth or until obstruction to obtain a depth profile. Measurements were collected in March 2011 during the seasonal fieldwork campaign and in March 2014.
As one of the key factors potentially influencing sediment deposition, the abundance and species composition of the vegetation around each sampling point was recorded. During the seasonal field campaign, a 1 m$^2$ quadrat was placed over each point and the total vegetation abundance (as a percentage) was estimated (see Bullock, 2006). The Dafor scale (Table 2.7) was used to measure vegetation composition to enable more rapid data collection than determining the exact percentage cover for each species. It allows for a quick and simple estimate of cover using a broad rating scheme (Joint Nature Conservation Committee, 2004). Although the scale is fairly coarse, the wide range of cover for each rating reduces subjectivity and recording errors (Kent, 2012). Both the total vegetation abundance and a Dafor rating for each species present were recorded at every sampling point for each of the four seasonal deployment periods. During the March 2014 field campaign the Dafor scale was not used, instead abundance of each species present was recorded to the nearest five per cent and summed to calculate the total vegetation abundance for each point. Vegetation data were not recorded during May 2012.

### 2.4.5 Tidal and Meteorological Monitoring

Tidal water level data were acquired throughout the study period. Water levels were monitored using self-recording pressure sensors (Schlumberger Mini-Diver and In-Situ RuggedTroll 100). One was installed in the main creek entering the realignment site from the breach in March 2011, prior to the seasonal fieldwork campaign (for location see Figure 2.11). Pressure data were corrected for atmospheric pressure variation with reference to a separate barometric pressure recorder (Schlumberger Baro-Diver and In-

---

**Table 2.7: Dafor scale for vegetation cover**

Source: (Wheater et al., 2011).

<table>
<thead>
<tr>
<th>Dafor Scale Rating</th>
<th>Vegetation Cover (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dominant</td>
<td>&gt; 75</td>
</tr>
<tr>
<td>Abundant</td>
<td>51 – 75</td>
</tr>
<tr>
<td>Frequent</td>
<td>26 – 50</td>
</tr>
<tr>
<td>Occasional</td>
<td>11 – 25</td>
</tr>
<tr>
<td>Rare</td>
<td>&lt; 11</td>
</tr>
<tr>
<td>Absent</td>
<td>0</td>
</tr>
</tbody>
</table>
Situ Rugged BaroTroll 100). The sensor in the main creek recorded data every 15 minutes until it was removed in May 2014.

As part of the intensive monitoring period during March 2014, additional instruments were deployed to collect data required to support subsequent numerical modelling of the tidal hydrodynamics within the realignment site. An automatic weather station (Davis Weather Monitor II) was set up in the natural marshes north of the realignment site to record wind speed and direction every five minutes. Two flow meters (Valeport model 105) were used to monitor the velocity and direction of water flows within the main channels in the realignment site. Suspended from support beams at the creek near seasonal sampling point 17 (referred to as the west creek), and in the main creek just inside the breach, they recorded data every five minutes when the instrument was underwater. Six high frequency pressure sensors (Druck PTX1830 interfaced to Campbell Scientific CR10X data loggers) placed around the realignment site were set up to capture pressure data in short bursts to measure wave activity. All these sensors were deployed for the four day sediment trap deployment.

Other instruments recorded data for a longer period in order to run and validate the hydrodynamic model. They were installed during the trap deployment in March and retrieved in May 2014, providing six weeks of data. Two additional pressure sensors were installed, one alongside the flow meter in the west creek, and one in the main Tollesbury Fleet channel (approximately 2.5 km from the realignment site). The latter provided the surface water height required to drive the model simulations. All three pressure sensors (including the one in the main creek) recorded data at five minute intervals during this period. Finally, an Acoustic Doppler Current Profiler (ADCP; RDI 1200KHz Workhorse Sentinel) was placed outside the breach in the main Tollesbury Creek. This recorded surface water levels, velocity and suspended sediment concentration every five minutes. Manual bottle samples were collected and filtered for calibration purposes. The locations of all the instruments deployed will be shown in Chapter 6. A similar arrangement of sensors was installed in May 2012, but due to various issues with a number of instruments, the data have not been included in this study.

2.5 Modelling Study

In order to extend the insights obtained from the relatively short-term field research, numerical hydrodynamic and sediment transport modelling was undertaken to simulate the hydrodynamics and sedimentation within the realignment site. Modelling provides
further understanding into the processes driving sedimentation and also allows the scaling up of short term point measurements to gain insights into the longer term evolution of the topography. Very few studies of this nature have been conducted before. Temmerman et al. (2005b), however, used a 3D model to investigate the impact of vegetation, micro-topography and water level changes on the flow and sedimentation patterns within a tidal marsh system.

The two-dimensional, finite-element Telemac modelling system was chosen for this study (Hervouet, 1999). It is a proven flow and sediment transport model that has been widely used for a variety of coastal and estuarine studies, and a number of authors have used it to simulate saltmarsh processes (e.g. Robins, 2009; Robins and Davies, 2011; Spearman, 2011; Luo et al., 2013). The system is also freely available, with the open source code having been released. Telemac comprises a number of modules, the flow (Telemac-2D) and sediment transport modules (Sisyphe) being the key ones used here. After the hydrodynamic model domain was constructed and validated, a series of simulations were performed to study the spatial variation in sediment deposition across the realignment site. These aim to fill in the gaps between the necessarily sparse field observations. The effect of vegetation was investigated through the adjustment of the bed roughness. Further details regarding the equations governing the Telemac model, how each module was implemented, the validation process, and the results of the various simulations performed are given in Chapter 6.
3. Longer-Term Evolution of Tollesbury Marshes

There is a well-documented history of change in tidal saltmarshes throughout the UK, and southeast England in particular (e.g. Dixon et al., 1998; Doody, 2004). In Essex, up to 40 hectares of marsh is lost every year (Hughes and Paramor, 2004), through sea-level rise, coastal squeeze, expansion of tidal creek systems and human activities (Cooper, 1982; Doody, 1992b; Kirwan et al., 2008; Wolanski et al., 2009). As discussed in Section 2.2, the Blackwater Estuary is no exception to this; it has been estimated that 2% of the marsh here is being eroded each year (Cooper et al., 2000). These findings date back to 1998, however, and are calculated for the estuary as a whole. To provide a more local-scale background and context to the primary data collection and modelling studies conducted in this research, more recent secondary data, in the form of aerial photographs, LiDAR and SET data covering the natural and restored marshes at Tollesbury have been analysed. These have been used to show the changes in vegetation and elevation in the realignment site since its formation, and to determine if the pattern of loss in the surrounding marshes has continued.

3.1 Vegetation Changes

3.1.1 Realignment Site

The five aerial photographs sourced from the Environment Agency and Bing Maps that feature the realignment site at Tollesbury are shown in Figure 3.1. The section under study is framed on each picture by a black border; all aerial photography and maps derived from the images in this chapter are north oriented. The ground area available for analysis is 12.5 ha for the 1997 photograph (which has partial site coverage due to a less extensive flight survey at this time), and 18.7 ha for those from 2001, 2008, 2011 and 2013. The following descriptions and changes detected in the aerial photography is supported by vegetation surveys conducted from the monitoring work undertaken at Tollesbury by staff from the Centre for Ecology and Hydrology (CEH) (Reading et al., 2002, 2006, 2008b).

In 1997, two years after the breaching of the sea wall, vegetation had started to colonise the realignment site (Figure 3.1(a)). Along the southern sea wall, a strip of saltmarsh is present, although it appears patchy in places, particularly in the eastern field at the furthest extent of the vegetation. Although the type of vegetation is not discernible from the photos, Garbutt et al. (2008) list the percentage frequency of
Figure 3.1: Aerial photographs of realignment site: 1997 (a) and 2001 (b)
All aerial images and maps are north oriented. Annotated areas in (b) are referred to in text.
Figure 3.1 (cont): Aerial photographs of realignment site: 2008 (c) and 2011 (d)
species recorded over transects within the realignment site. In 1997, the main species present was *Salicornia europaea* (15.6%), with a few scattered individuals of *Suaeda maritima* (0.5%) and *Spartina anglica* (0.01%). There was some algal growth present in the area surrounding the breach (likely *Enteromorpha* spp.), but the rest of the site was unvegetated mudflat. The horizontal lines visible in the central area to the south west of the breach site are possibly the remains of furrows and ridges formed during ploughing of the field in its prior form. The results of the supervised classification of the aerial photographs are shown in Figure 3.2. The area of vegetation that was classified in the 1997 photograph (Figure 3.2 (a)) is 0.88 ha, or 7.1% of the study area (Table 3.1), and also includes the hedgerow running perpendicular to the sea wall that separates the central and eastern fields in the southern half of the site. The seaward extent of the vegetation along the lower sea wall in these two fields is noticeably different: the average width of the marsh from the wall (measured in transects 30 m apart) is 33.2 m in the visible part of the central field, compared to 51.0 m in the eastern field. This is likely due to the higher elevation of the sediment surface in the eastern corner of the site (discussed further in Section 3.3).
Table 3.1: Area of pixels classified as vegetated and unvegetated in the realignment site

<table>
<thead>
<tr>
<th>Year</th>
<th>Vegetated</th>
<th>Bare</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997*</td>
<td>0.88 ha 7.1%</td>
<td>11.58 ha 92.9%</td>
</tr>
<tr>
<td>2001</td>
<td>2.34 ha 12.5%</td>
<td>16.32 ha 87.5%</td>
</tr>
<tr>
<td>2008</td>
<td>6.75 ha 36.2%</td>
<td>11.92 ha 63.8%</td>
</tr>
<tr>
<td>2011</td>
<td>8.40 ha 45.0%</td>
<td>10.26 ha 55.0%</td>
</tr>
<tr>
<td>2013</td>
<td>9.38 ha 50.2%</td>
<td>9.29 ha 49.8%</td>
</tr>
</tbody>
</table>

*Study area of 1997 image is 12.5 ha; 18.7 ha for other dates

The aerial photograph taken in 2001 (Figure 3.1(b)) shows that the area of algae near the breach has increased significantly; other patches have also developed to the north and in the westernmost of the three fields in the southern half of the site. None of these areas have been classed as vegetation, however, as Figure 3.2(b) shows. 2.3 ha (12.5%) of the realignment site has been classified as vegetated at this time, primarily comprising the marsh along the southern sea wall (on average now 46.8 m in width) and that fronting the western wall (20.1 m). The eastern wall is noticeable by the absence of any colonisation. The hedgerow separating the former fields appears to have diminished and thinned, this is clear from both the aerial photo and the vegetation map. This is likely due to the salt present in the tidal waters flooding the site continuing to affect the terrestrial vegetation. The other noticeable change since 1997 is the development of some minor creek networks on the mud surface adjacent to the main east-west channel.

After five years of tidal inundation the vegetation community of the site had not yet stabilised. New species continued to arrive, rising from four in 1997 to 17 in 2001, with all species increasing in frequency year on year (Garbutt et al., 2002, 2008). The most established species present at this time included Salicornia europea, Suaeda maritima, Spartina anglica and Puccinellia maritima. Although species richness and percentage cover increased over this period, the maximum seaward extent had been reached for most species after only one or two years of marsh establishment, and did not progress further downslope from the sea walls until 2001. They had already colonised the site at their lowest elevational range within the first one or two years and could not advance until further increases in the elevation of the mudflat (Garbutt and Boorman, 2009).

By 2008 (Figure 3.1(c)), there had been a substantial expansion in the marsh area; 36.2% of the realignment site was now colonised by vegetation. A major
Figure 3.2: Classified maps of realignment site: 1997 (a) and 2001 (b)
Figure 3.2 (cont): Classified maps of realignment site: 2008 (c) and 2011 (d)
Figure 3.2 (cont): Classified maps of realignment site: 2013 (e)

A proportion of this growth is the result of the conversion from bare mud and algae in the central zone around the breach to saltmarsh plants (dominated by *Salicornia europaea* and *Spartina anglica*). The ridge and furrow pattern first seen in the 1997 photograph in this area has remained; vegetation is growing on the ridges, whilst the furrows have developed into minor channels (Garbutt et al., 2008). This is visible in the 2008 photograph (Figure 3.1(c)) and classification map (Figure 3.2 (c)), and observed by the author on field visits (Figure 3.3). Evidence of scouring is also apparent either side of the main channel leading from the breach into the site, and the other minor channels in the mudflat areas noted in 2001 have also expanded, the banks of which have been colonised in places.

The marsh vegetation in the three former fields in the south of the site has expanded considerably, particularly in the westernmost. The plants here have progressed into the mudflat, and now reach the bank of the channel running through the centre of the site. This represents a shift in the marsh edge of 60.0 m on average in this area over 7 years. The seaward advancement in the vegetation in the central field was 22.6 m and 16.3 m in the eastern field. The marsh fronting the western sea wall expanded considerably too, up from 20.1 m in 2001 to 59.1 m in 2008. The percentage
frequency and abundance in most species increased too, especially with regards to *S. anglica*, which rose in frequency from 5.2% to 65.1% over the same period (Garbutt *et al.*, 2008). Communities were found at all elevations from the sea wall down to the mudflat edge, and as individual patches beyond.

*Spartina anglica* continued to be the dominant species in 2011 and beyond throughout the site, with the main vegetation community characterised by its tall clusters with a bed of *Salicornia europaea* and bare mud. This marsh covered 8.4 ha of the site (Table 3.1), with expansion evident in all areas since 2008. The progression in the lowest limit in the vegetation fronting the sea walls had slowed compared to the preceding period; the rate of increase fell from 4.8 m yr$^{-1}$ to 1.1 m yr$^{-1}$ along the southern sea wall, and from 5.6 m yr$^{-1}$ to 1.9 m yr$^{-1}$ in the west. The main area of growth was the expansion of the central zone near the breach, which increased from 0.8 ha in 2008 to 1.4 ha in 2011. Communities present on the banks of the minor channels also experienced expansion into the surrounding mud, as did the scattered tussocks spread throughout the southern and western mudflat areas.

Marsh vegetation continued to spread slowly in all regions, and had colonised half of the realignment site by 2013 (Figure 3.1 (e) and Figure 3.2 (e)). There was no
change in the former westernmost field in the southern section, as the vegetation had already reached the channel bank. The growth along the rest of the southern sea wall dropped to 0.94 m yr\(^{-1}\), with the average width of marsh vegetation extending to 75 m by this time. The marsh fronting the western and northern sea walls expanded by 0.54 m yr\(^{-1}\) and 1.08 m yr\(^{-1}\) respectively. Across the whole site, just one additional hectare of bare mud was vegetated between 2011 and 2013. This continues the slow down in marsh colonisation since 2008, as shown in Figure 3.4. At its peak between 2001 and 2008, 0.63 ha of marsh was colonised per year, dropping to 0.55 ha yr\(^{-1}\) by 2011, and 0.49 ha yr\(^{-1}\) by 2013. Once the site becomes predominately vegetated, it is inevitable that this increase is likely to plateau off in the coming years.

The spread of vegetation described above between 1997 and 2013 can be seen more clearly in the results of the change detection analysis undertaken on the five classified maps. The four maps showing the differences between the five time periods are shown in Figure 3.5. Each map shows four classes – those pixels that remained as vegetated and unvegetated between each time period, and those that changed from unvegetated to vegetated and vice versa. Table 3.2 summarises the quantitative analysis; annualised data is given in Table 3.3. The majority of each map shows no change; between 1997 and 2001 (Figure 3.5(a)), 95.7% of the study area did not change. However the change detection analysis reveals some changes that are not as obvious from simply comparing the classified maps. Although the lowest limit of vegetation along the southern sea wall extended 8.54 m over the first 4 years of analysis, the marsh plants became sparser, with 1,900 m\(^2\) changing from vegetation to bare ground within this region. This change could be partly due to slight positional errors in the

Figure 3.4: Percentage area of vegetation within the realignment site over time
Figure 3.5: Change detection between 1997 and 2001 (a) and 2001 and 2008 (b)
Figure 3.5 (cont): Change detection between 2008 and 2011 (c) and 2011 and 2013 (d)
Table 3.2: Change detection statistics for classified maps of the realignment area

<table>
<thead>
<tr>
<th>Time Period*</th>
<th>Bare (ha)</th>
<th>Vegetation (ha)</th>
<th>Bare to Vegetation (ha)</th>
<th>Vegetation to Bare (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997-2001</td>
<td>11.3</td>
<td>90.7%</td>
<td>0.6</td>
<td>5.0%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.3</td>
<td>2.3%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.3</td>
<td>2.0%</td>
</tr>
<tr>
<td>2001-2008</td>
<td>11.9</td>
<td>63.6%</td>
<td>2.3</td>
<td>12.2%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>4.5</td>
<td>23.9%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.1</td>
<td>0.3%</td>
</tr>
<tr>
<td>2008-2011</td>
<td>9.8</td>
<td>52.7%</td>
<td>6.3</td>
<td>33.9%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>2.1</td>
<td>11.1%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.4</td>
<td>2.3%</td>
</tr>
<tr>
<td>2011-2013</td>
<td>8.6</td>
<td>46.1%</td>
<td>7.6</td>
<td>40.6%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>1.8</td>
<td>9.6%</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>0.7</td>
<td>3.8%</td>
</tr>
</tbody>
</table>

*Area of 1997 image is 12.5 ha, compared to 18.7 ha for other dates

Table 3.3: Annualised change detection statistics for the realignment area

<table>
<thead>
<tr>
<th>Time Period*</th>
<th>Bare to Vegetation (ha yr^{-1})</th>
<th>Vegetation to Bare (ha yr^{-1})</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997-2001</td>
<td>0.08</td>
<td>0.08</td>
</tr>
<tr>
<td>2001-2008</td>
<td>0.64</td>
<td>0.01</td>
</tr>
<tr>
<td>2008-2011</td>
<td>0.70</td>
<td>0.13</td>
</tr>
<tr>
<td>2011-2013</td>
<td>0.90</td>
<td>0.35</td>
</tr>
</tbody>
</table>

georeferencing of the original aerial photography, as shown by the adjacent lines of vegetation growth and decline along the hedgerow or variations in the growth stages of the plants caused by the timing of the photograph in the summer season (the exact date of the 1997 image is not known). The vegetation survey conducted by Garbutt et al. (2002) does not indicate any overall thinning of the marsh during this time.

The main stage of change took place between 2001 and 2008 (Figure 3.5(b)). Almost a quarter of the realignment site evolved from bare mud and algae to saltmarsh vegetation during this time. An additional 4.4 ha of the site was vegetated at the end of 2008, averaging at a rate of increase of 0.63 ha per year. As noted above, the spread of vegetation occurred throughout the realignment site, but most visibly in the central breach area where 0.8 ha of mud was colonised. More significant, was the expansion of the vegetated area in the three fields in the south of the site. 1.64 ha of marsh developed in this seven year period, almost half of which (46.3%) developed in the westernmost field. The change detection map also shows the spread of halophytic vegetation in the northern and western edges of the site, and the initial colonisation on the banks around the minor drainage network developing to the west of the central vegetated area. As Table 3.2 shows, this period also exhibited the lowest conversion of vegetation to mud, just 0.3% of the study area.
A further 1.7 ha of mud was colonised between 2008 and 2011, with marsh plants covering 45% of the site at the end of this period (Figure 3.5 (c)). As discussed, a significant proportion of this growth occurred in the central area near the breach. This zone increased by 0.6 ha, representing 28.6% of the total increase in vegetation across the site over the 3 years. The expansion in the marsh communities around the sea walls slowed over time, from 0.45 ha yr\(^{-1}\) in 2001-2008 to 0.24 ha yr\(^{-1}\) in 2008-2011, likely due to a reduction in the rate of increase in the elevation of the sediment surface.

As the rate of vegetation colonisation continued to slow between 2011 and 2013, no one part of the realignment site saw any substantial growth in marsh area. Having already stabilised prior to 2008, the area of marsh fronting the southern sea wall only increased by 0.10 ha, whilst that in the north and west increased by just 0.18 ha overall. The banks surrounding the channels running through the site saw the greatest increase in marsh area, but compared to previous years was still not major, at 0.38 ha. Even the central vegetated area near the breach saw comparatively little change with 0.33 ha of bare mud colonised here. As Figure 3.5 (d) and Table 3.2 show, almost 4% of the site area changed from vegetation to bare mud. Although some of this is due to the expansion of the tidal channels into the marsh area, closer inspection of the original aerial photography reveals most of this change can be attributed to minor georeferencing errors with the 2013 image. This can especially be seen in the central vegetated area where there are many parallel lines of vegetation to bare mud conversion and vice versa.

Figure 3.6 shows the net change over the whole study period, with the corresponding statistics listed in Table 3.4. Due to the limited data coverage from the 1997 photograph, the missing portion has been filled with the change in vegetation cover between 2001 and 2013. Assuming there was very little overall development of vegetation in the missing 1997 section (as the findings described above suggest), the combined map and statistics can be considered a reasonable estimate of the change between 1997 and 2013 across the entire site. These results show over a third of the realignment area was colonised by vegetation over the whole period (6.9 ha). Although the majority of this occurred through the expansion of marsh plants around the sea walls (ranging from 30.4 m in the north to over 75 m in the southeast), 1.4 ha of vegetation was established from bare mud in the centre of the realignment site. A further 1.1 ha of plants grew along the banks of the minor channels running through the area.

A number of other changes to vegetation structure and distribution, and the geomorphology of the site are likely to have occurred since the site was restored, but
Figure 3.6: Net change in vegetation cover in the realignment site, 1997 – 2013

Table 3.4: Net change statistics for the realignment site, 1997 – 2013

<table>
<thead>
<tr>
<th>Section*</th>
<th>Bare</th>
<th>Vegetation</th>
<th>Bare to Vegetation</th>
<th>Vegetation to Bare</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997-2013</td>
<td>7.3 ha</td>
<td>58.9%</td>
<td>0.9 ha</td>
<td>6.9%</td>
</tr>
<tr>
<td>2001-2013</td>
<td>2.1 ha</td>
<td>34.3%</td>
<td>1.4 ha</td>
<td>22.6%</td>
</tr>
<tr>
<td>Combined</td>
<td>9.5 ha</td>
<td>50.7%</td>
<td>2.3 ha</td>
<td>12.1%</td>
</tr>
</tbody>
</table>

*Study area of 1997 image is 12.5 ha, section of 2001 image is 6.2 ha

have been at too fine a scale to be captured by the aerial photographs. A number of such changes have been seen at the realignment site whilst carrying out fieldwork. Examples of these include an increase in the extent and density of marsh vegetation along the minor creek channels over time. This is apparent by comparing two photographs taken of the eastern end of the channel that runs through the site from west to east, shown in Figure 3.7. The top photograph was taken on 07/08/11 and the lower on 17/08/14. Over the three year interval, the *Salicornia europaea* that grows on the creek banks has clearly expanded in area and colonised a larger area of the mud. New individuals have grown both lower down the bank as well as outwards on the marsh surface. The density
of the plants and the stem height also appears to have increased over time, with fewer mud patches visible amongst this community. The density of the marsh vegetation in the two main species types (*Spartina anglica* and *Salicornia europaea*) was noticed to have increased across the realignment site over the four years of fieldwork undertaken.

![Figure 3.7: Growth in vegetation on a creek edge between 2011 (a) and 2014 (b)](image)

Species present are primarily *Salicornia spp.* along the main creek edge visible, and *Spartina spp.* in the foreground. Photograph dates: 07/08/11 and 17/08/14.
Although the analysis of the aerial photography clearly shows a considerable growth in vegetation over time, there is evidence that localised erosion is taking place within the restored area. Figure 3.8 shows two examples of creek banks collapsing into the channels running through the marsh. Many similar collapses have been seen throughout the area whilst visiting the site. The specific causes of the collapses shown here (during August 2011) is not known, as these channels are located in a sheltered region away from the breach site, so flow velocities are generally low. The areas around the breach itself, and alongside the banks of the main channel running into the restored area, appear to have experienced substantial levels of erosion. Scouring has been a problem here since the sea wall was breached, inhibiting the development of any vegetation, and slowly excavating the hard, impermeable subsoil layer formed during the site’s former use for agriculture. This erosion is making the banks of the main drainage channel unstable; during February 2014, a significant section of the eastern bank collapsed into the channel (Figure 3.9). Although similar events have been noticed during the duration of this study, this was the largest, affecting approximately 5 m of the bank.

![Figure 3.8: Creek edge collapses in the realignment site](image)
Figure 3.8 (cont): Creek edge collapses in the realignment site

Figure 3.9: Bank collapse in the main channel leading from the breach
3.1.2 Natural Marshes

Prior analysis of aerial photography covering the Blackwater Estuary showed considerable saltmarsh erosion has occurred in the recent past. Between 1973 and 1988, over 200 ha of marsh was lost (Burd, 1992), with 54.9 ha being eroded between 1988 and 1998 (Cooper et al., 2001). This deterioration in marsh area took place at the seaward edge and within the marsh interior, both of which are apparent in the photographic analysis as an increase in the area of bare mud surrounding and within the marsh environment. To determine if this loss is continuing, the same classification and change detection process was undertaken using aerial photography of the natural marshes surrounding the Tollesbury managed realignment site. This analysis was carried out before the 2013 image was obtained, so only the photography from 1997, 2001, 2008 and 2011 was examined. The recent saltmarsh extents derived from the aerial photographs were not compared directly with the results from Burd (1989; 1992) and Cooper et al. (2001) due to the differences in the saltmarsh mapping methodologies used. In particular, the 1989 study by Burd considerably underestimated the area of saltmarsh present at the time due to the simple techniques used, which lead to overestimates in the rates of erosion. In their 2011 study (authored by Phelan et al., 2011), using aerial photography from 2006-2009, the Environment Agency mapped saltmarsh extent throughout England and Wales and produced a total area 3.1% higher than Burd (1989). The differences between the two studies ranged from -12.9% in the South West of England to +34.5% in the North East, with the extent in the Anglian region mapped at 10.3% larger than the 1989 report (Phelan et al., 2011). This has caused concern over the reliability of assessments into prior saltmarsh erosion, as the EA exercise suggests that the rate of marsh loss at both local and national levels has been slower than previously thought. This will be investigated further in the discussion (Chapter 7).

From initial inspection of the Tollesbury Fleet photographs from the four available years, it was not possible to visually detect any change of vegetation from shifts in the position of the marsh edge or expansion/contraction of the interior creek network. Due to the large area of marsh involved (130 ha), and the time required to produce a detailed classification (as documented in Section 2.3.1), two small test areas were chosen for analysis: one from the outer marsh closest to the main estuary (7.88 ha), and one from the interior, closer to the realignment site (1.88 ha). These two types were used to determine if there was any difference in erosion rates caused by wave action at the seaward edge and internal deterioration. The outer test area is located in the
far eastern section of the natural marsh, closest to the main estuary channel and therefore most likely to experience wave action; the inner test area chosen reflects the opposite conditions, but without being too close to the hinterland that it is rarely flooded. These are shown in Figure 3.10. Only the imagery from the 1997 and 2011 photographs were used, in order to detect the net change over the period available. Once the 2013 image was obtained, it was compared closely with the 2011 image in the same test areas. As there was no detectable change between them, it was decided not to repeat the following analysis with the new image.

Figure 3.11 shows the aerial photography for the outer test area from these two years. Apart from the difference in the colour of the marsh surface, there are no major changes visible in the vegetated area when comparing the two images. Minor changes between 1997 and 2011 include the construction (presumably by wildfowlers) of bridges to cross the marsh creeks (seen as narrow white lines across marsh segments), and the migration of the minor creeks within the larger channel in the south west corner of the images. The other noticeable difference is some small-scale changes and growth in algae at the outer marsh edge in the 2011 image; this is most visible at the eastern tip of the marsh.

Figure 3.10: Test areas for natural marsh change
Figure 3.11: Aerial photographs of outer marsh test area: 1997 (a) and 2011 (b)
The results of the corrected supervised classification of the photographs are shown in Figure 3.12, with the area of each class listed in Table 3.5. Due to the pronounced shadows in the 1997 photograph, an extra class was used in the classification process compared to the managed realignment site. In the 1997 photograph, 13.1% of the study area was classed as shadow, compared with 3.8% for 2011. Contrary to expectations, the area of vegetation increased by 0.38 ha over the 14 year period. The area of bare mud also increased, however, so it cannot be concluded whether the overall change is due to any loss or gain of saltmarsh area, or is simply caused by the reduction in the area of shadow in the 2011 scene.

For the change detection analysis, all areas of shadow have been regarded as containing no data, as it was not always possible to infer what the shadow was obscuring (vegetation or bare mud). The result of this calculation is presented in Figure 3.13. Areas with no change are shown in white, with all areas of change due to shadow movements in grey. Together, these classes make up 80% of the study region (Table 3.6). The remaining proportion of the data is split almost equally between the classes representing change between the vegetation and bare mud, with only 0.02 hectares of difference between them. Overall, therefore, there is essentially no net change between 1997 and 2011 in the outer marsh test area.

At a smaller level of scale, however, some changes are noticeable, but these are in part due to minor positional errors in the georeferencing of the aerial photographs, especially in the interior regions of the study area. There are also some differences in the classification of the areas of algae, particularly at the north-east facing marsh edge. Although less algae is visible in 1997, a higher proportion was classed as vegetation, thus leading to an apparent ‘loss’ of vegetation in the change detection process which is not supported by the aerial photography. This trend is reversed along the south-east facing edge of the marsh, where the growth in algae between 1997 and 2011 is shown as an increase in vegetation cover.

Table 3.5: Class extents for 1997 and 2011 outer marsh test area

<table>
<thead>
<tr>
<th>Class</th>
<th>Area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1997</td>
<td>2011</td>
</tr>
<tr>
<td>Vegetation</td>
<td>2.33</td>
</tr>
<tr>
<td>Bare</td>
<td>4.52</td>
</tr>
<tr>
<td>Shadow</td>
<td>1.03</td>
</tr>
</tbody>
</table>
Figure 3.12: Classified maps of outer marsh test area: 1997 (a) and 2011 (b)
Figure 3.13: Change detection between 1997 and 2011 for outer marsh test area

Table 3.6: Change detection statistics for the outer marsh test area

<table>
<thead>
<tr>
<th>Class</th>
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</tr>
</thead>
<tbody>
<tr>
<td>Vegetation to Bare</td>
<td>0.75</td>
</tr>
<tr>
<td>Bare to Vegetation</td>
<td>0.77</td>
</tr>
<tr>
<td>Shadow / No data</td>
<td>1.24</td>
</tr>
<tr>
<td>Unchanged*</td>
<td>5.11</td>
</tr>
</tbody>
</table>

*Includes areas that were shadow in both images

Aerial photography covering the second, inner marsh test area is displayed in Figure 3.14. Similarly to the outer test area, there are no obvious differences between the 1997 and 2011 images. The presence of the long shadows in the 1997 photograph is again made more apparent in the classification maps (Figure 3.15), with shadow comprising 22.9% of this image, compared to just 6.4% in 2011 (Table 3.7). The total area of both vegetation and bare mud increased in line with the reduction in shadow, but the percentage increase in vegetation was over three times greater at 28.4%, compared
Figure 3.14: Aerial photography of inner marsh test area: 1997 (a) and 2011 (b)
Figure 3.15: Classified maps of inner marsh test area: 1997 (a) and 2011 (b)
Table 3.7: Class extents for 1997 and 2011 inner marsh test area

<table>
<thead>
<tr>
<th>Class</th>
<th>Area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1997</td>
</tr>
<tr>
<td>Vegetation</td>
<td>0.95</td>
</tr>
<tr>
<td>Bare</td>
<td>0.50</td>
</tr>
<tr>
<td>Shadow</td>
<td>0.43</td>
</tr>
</tbody>
</table>

to an 8% increase in the area of bare ground. Whilst initially suggesting there may have been an increase in the area of vegetation over time, on closer inspection of the aerial photography, it simply reflects the fact that more of the shadows covered vegetated marsh in 1997 than bare mud. Although shadows were less pronounced in the 2001 image, they were still present and a visual comparison with the 2011 image revealed no detectable changes. The process of scanning and georeferencing the paper photographs would also have been likely to lead to further positional errors.

Figure 3.16 shows the output of the change detection analysis, with the corresponding statistics listed in Table 3.8. As before, the majority (80.3%) of the result is characterised as no change, or involving a change in shadow, with a very small difference between the conversion of vegetation to bare mud and vice versa (0.07 ha). Most of these changes appear to be due to the positional errors between the georeferencing of the two photographs. It can be concluded that the net change between 1997 and 2011 in the inner test area is, therefore, essentially zero.

These findings are consistent with a similar study carried out in the Deben Estuary in Suffolk (Environment Agency, 2013). The extent of saltmarsh was digitised manually using aerial photographs taken in 2000, 2006 and 2011. They found evidence of saltmarsh erosion in places, this was largely offset by accretion in other parts of the estuary. The area of marsh mapped in 2000 was 244.06 ha, and in 2011 was 240.78 ha, representing a net decrease of just 1.35%. This is likely within the error margin of the mapping techniques, and it was thus concluded that the overall net change measured is insignificant. It is clearly very difficult to detect reliable levels of change over these fairly short timescales. Although some degree of erosion was anticipated at Tollesbury over the 14-year study period, this assumption was based on the previously measured high estimates of erosion (10 ha per year across the whole of the Blackwater Estuary (Burd, 1992; Cooper et al., 2001)), which as mentioned at the start of this section, have now been called into question (Phelan et al., 2011).
While there are no detectable changes in the marsh vegetation from the aerial photography, vegetation monitoring conducted by the CEH between 1994 and 2005 revealed changes in the frequency of most species present in the natural marshes at Tollesbury. Several species declined significantly in the upper marsh. *Atriplex portulacoides* decreased by 25%, whilst *Suaeda maritima* fell by 28.1% before
recovering after 2000. Substantial increases were recorded in Aster tripolium, Limonium vulgare, Spartina anglica and Sarcocornia perennis. The picture was more mixed in the lower marsh areas, with Aster tripolium, Salicornia europaea and Suaeda maritima all showing increases and decreases in frequency over time, but with a net decrease between 1994 and 2007 (Garbutt et al., 2008). The cover of bare mud and algae was highly variable throughout the monitoring period. The research was conducted in order to determine whether any variations in the vegetation composition could be attributed to the creation of the realignment site (Garbutt et al., 2006a). Despite the changes detected in a variety of species at different parts of the marsh, there were no clear trends between them and any of the physical parameters tested (elevation, distance to breach, bathymetric change). Nor was there any evidence to indicate that the creation of the realignment site was the cause of the botanical changes. Instead, it is likely that processes operating at a larger scale are responsible for those changes which appear to be part of longer-term trends occurring on a regional scale (Garbutt et al., 2002, 2008).

As in the realignment site, other changes are detectable from a close inspection of the aerial photography. Most of these relate to changes in the large channels running throughout the natural marsh. Figure 3.17 shows a section of a meandering channel near

![Figure 3.17: Aerial photographs of outer marsh meander: 1997 (a) and 2011 (b)](image)
the breach site (position shown in Figure 3.10) from the 1997 and 2011 photographs. Although there is no visible change in the position of large meanders, the location of the thalweg within the centre of the main channel has changed over time. The meanders are more pronounced in places, and the channel appears to be deeper in places (based on the more pronounced shadows in the 2011 image).

Despite the lack of conclusive evidence of an overall change in the natural marshes from the aerial photography, many smaller signs of on-going erosion were observed during field campaigns undertaken between April 2011 and March 2014. These include a number of small holes and pans spread around the marsh interior. More significant longer-term erosional features are regularly found at the edges of marsh segments, and comprise large cracks, which over time appear to slowly slump into the creek system. Examples of both of these features are shown in Figure 3.18. These may be caused by longer-term erosional processes due to the continuous low energy tidal and wind wave currents as discussed by Chen et al. (2011). These small-scale changes, as well as the development of the managed realignment site will be further explored by analysing a series of LiDAR datasets available for the region in Section 3.3.
Figure 3.18: Natural marsh edge cracking and slumping
3.2 Marsh Fragmentation

Saltmarshes in the UK, especially in southeast England, have suffered extensive lateral erosion along their seaward boundaries that has variously been attributed to sea-level rise, coastal squeeze, direct wave action, and dwindling sediment supply (Cooper et al., 2001; Covey and Laffoley, 2002; Airoldi and Beck, 2007; Baily and Inkpen, 2013). Over the past century, 67% of the eastern coastline of the UK has displayed a landward shift of the low-water mark (Taylor et al., 2004). In addition to this retreat at the marsh edges, tidal creek enlargement within the marsh itself is causing internal dissection and fragmentation (Wolters et al., 2005a; Baily and Pearson, 2007; Chen et al., 2012).

Marsh fragmentation refers to the splitting up of large saltmarsh areas into smaller, more isolated blocks, with an increasing area of bare unvegetated mudflat between them, and is a major factor contributing to saltmarsh decline (Takekawa et al., 2006; Laegdsgaard et al., 2009).

Although this study has not found any definitive evidence for saltmarsh erosion either at the marsh edge, or through internal dissection in the Tollesbury marshes, the level of fragmentation has been investigated in order to judge the condition and quality of the natural marshes. The Environment Agency (EA) proposed a measure of fragmentation as the ratio between the area of saltmarsh and non-saltmarsh within a gridded framework, during a study of saltmarsh change carried out into the Deben Estuary in Suffolk (Environment Agency, 2013). Although no analysis was conducted on the results of the ratio, it can be used to identify which areas are potentially at the greatest risk of erosion. Here is presented a similar method to produce a fragmentation map of the Tollesbury marshes.

The marsh extent has been obtained from the EA’s saltmarsh mapping programme, carried out using aerial photography captured between 2006 and 2009 (Phelan et al., 2011). Analysts manually digitised the marsh area from standard orthorectified photographs and in places, near infrared imagery. As the EA’s main aim of the programme was to produce a saltmarsh extent baseline for the whole country, based on a consistent and repeatable methodology, areas of sparse low-density pioneer vegetation were excluded, and creeks narrower than 1.5 – 2 m were merged into the marsh extent. It was thus important to examine the marsh extent for the Tollesbury study area, compare it with the aerial photograph from 2008, and amend as necessary. In a very complex system such as that at Tollesbury, minor creeks are found extensively within the marshes, and a significant proportion are less than 2 m in width. These creeks are important in the delivery of tidal waters and sediment to the inner reaches of the
marsh (Reed et al., 1999; Allen, 2000) and it is therefore important that they are included. Upon close inspection of the mapped marsh and the 2008 aerial photograph, it was realised that there were many other inconsistencies and omissions to be corrected. Figure 3.19 (a) shows an example of the EA marsh extent with a lack of minor channels (i.e. the marsh area is greater than it should be), as well as many areas of marsh that were not mapped at all, or where only small portions of marsh segments were digitised. More puzzling is the fact the examples shown are adjacent to each other. Figure 3.19 (b) shows the amended marsh extent used in the present study.

Once all of the marsh extent had been checked and edited, the fragmentation map was created. The process is shown in Figure 3.20, with Figure 3.20 (a) showing the completed marsh extent for a section of the study area. Any major and minor creeks were identified and digitised separately (Figure 3.20 (b)). A grid was overlaid the whole area, and cropped to the edges of the major creeks (Figure 3.20 (c)). A 10 × 10 m grid was used initially, following the procedure used in the Environment Agency (2013) report. The marsh extent was intersected by the grid cells, and the area of marsh within each cell determined. A measure of fragmentation was calculated as the percentage of

![Figure 3.19: Saltmarsh extent mapped by the EA (a), and amended for this study (b)](image)

Blue regions show the areas of marsh mapped. The EA product (a) has sections with larger marsh extents than it should as minor channels have not been excluded (shown in top and left of map), as well as sections where too little has been mapped (bottom and right). These were corrected for this region (b) and as appropriate for the rest of the study area.
Figure 3.20: Process for creating marsh fragmentation maps
The marsh extent product from the EA is checked and amended as necessary (a), with major and minor creeks digitised separately (b). A grid was overlaid the full study area (10 × 10 m shown here), with the major creeks excluded (c). The marsh extent was intersected by the grid, and the area within each cell calculated. This area was divided by the area of the cell and a percentage of marsh area determined (d).
this area by the area of the cell, with those cells containing less marsh being more fragmented than those containing more marsh (Figure 3.20 (d)). Two further maps were created, using 20 × 20 m and 50 × 50 m grid cells, to explore the differences in the level of detail captured. The 20 m grid worked best in representing the level of fragmentation, and all further analysis was conducted using this map. Most cells within this grid will be 20 × 20 m squares, however, due to the cropping, many will be smaller in size and irregularly shaped.

The fragmentation map for the whole study area is shown in Figure 3.21; the red and orange cells indicate the regions of marsh comprising small fragmented patches with a larger proportion of unvegetated mudflat, whilst the blue cells show areas where the marsh is made up of larger segments with less bare mud. Although the level of fragmentation varies across the marshes, there appears to be a broad pattern that the least fragmented areas are closer to the marsh edges and internal creeks, whilst those areas furthest from creek edges are more likely to be fragmented. This was further explored by measuring the distance from the centre of each grid cell to the nearest major

![Fragmentation map for Tollesbury natural marshes](image)

**Figure 3.21:** Fragmentation map for Tollesbury natural marshes
creek (shown in Figure 3.21 by bold lines) as well as the nearest minor creek (shown as thinner lines). A boxplot of the distance from every cell (aggregated into the five percentage groups of marsh area) to the nearest major creek is shown in Figure 3.22. The range of values is very similar across all the groups, but there is a clear negative association between the percentage of marsh area and distance from the nearest creek. The coefficient of determination using the arithmetic mean of each marsh group (shown as solid squares in the boxplot) is 0.96; the trend line is also shown to highlight the pattern in the data. The trend is not as strong when the distance to the minor creeks are included, the $r^2$ value falls to 0.71.

The less fragmented areas of marsh nearest the major creeks and seaward edges receive more sediment from the flood tide than internal areas as the sediment settles out fairly quickly as the water enters the creek system. These areas are thus able to keep pace with increasing sea levels and wave action, and remain in a reasonably good condition. In contrast, much less sediment reaches the interior of the marsh, so these areas are not able to sustain the level of growth required to maintain their position in the tidal frame, and have thus started to degrade and split into smaller, disconnected patches.

**Figure 3.22: Distance to nearest major creek for each of the marsh area classes**
3.3 Topographical Development

3.3.1 LiDAR

Airborne LiDAR datasets covering the Tollesbury saltmarshes obtained from the surveys flown for the Environment Agency in 2000 and 2013 are shown in Figure 3.23. These, and all subsequent figures and analysis, have been subjected to the processing described in the Methods chapter (Section 2.3.2). These images show the elevation of the marsh area and channels and creeks at the earliest and most recently available time periods. The areas of highest elevation are displayed in white (generally showing the sea walls around the marsh edges, or buildings and boats in the marina), with the lowest in black (showing the deeper channels). The data have been thresholded at 4 m OD to better resolve the detail in the intertidal regions. Areas of no data are shown as blank pixels, and are mainly located within the creek network; these are caused by the inability of the red laser used in the EA LiDAR sensor to penetrate standing water left in these channels at low water. The morphology of the natural marshes has been captured well in each DSM, with all features ranging from the largest channels nearest the wider estuary, through to the smallest creeks and marsh segments clearly visible. There are few significant differences between the two surveys; the main one is the higher resolution and associated clarity in the 2013 DSM, most evident in the sharper edges of the creeks within the marsh area.

Focusing on the realignment site, the increased horizontal resolution (2 m in 2000, 1 m in 2013) is particularly apparent; a much greater level of detail was captured in the later survey, as seen between Figure 3.24 (a) and (b). At this higher zoom (compared to the full extent in Figure 3.23), the main channels running through the site are very ill-defined in 2000 such that the creek flowing from the breach is barely captured in places. In contrast, all the main channels are clearly defined in the 2013 DSM. The evolution of the site over the 13 year interval between the two figures is clear: the elevation has increased in front of the southern, western and northern sea walls along with the expansion of the marsh vegetation described in section 3.1. This has also occurred in the area near the breach, the elevation is slightly higher here (1.8 m in this region compared to 1.6 m in the unvegetated mudflat to the west). The minor drainage channels in this central vegetated area and elsewhere have developed from the featureless bare mud between 2000 and 2013.
Figure 3.23: LiDAR elevations of the Tollesbury marshes from 2000 (a) and 2013 (b)
Figure 3.24: LiDAR elevations of the realignment site from 2000 (a) and 2013 (b)
The main focus of this chapter is the evolution of the geomorphology of the realignment site and the adjacent natural marshes, and specifically in this section, the changes in the elevation of the sediment surface. To this end, more attention is given to the relative differences between the LiDAR DSMs rather than the absolute elevation values for each survey. The change in elevation between the surveys was calculated by subtracting one from the other in ArcMap. The difference in elevation between the first two surveys (2000 and 2001) for the full study area is shown in Figure 3.25. A positive change (i.e. accretion) is shown in shades of red, and negative changes (erosion) in blue. The most striking feature of this change detection map is the banding / striping visible running in a north-south direction throughout the entire image. The LiDAR data seem to show that in general, accretion occurred in alternate bands approximately 160 – 180 m in width, with erosion occurring in the remaining areas. On average, the elevation change in the red bands (signalling accretion) is ~85 mm higher than the blue bands across the whole study area. Clearly this pattern of change did not occur in reality, especially considering the time period between the surveys shown is just one year, and is being caused by vertical errors or artefacts present in the original LiDAR surveys. Similar errors are present in all the change detection maps produced between each

Figure 3.25: Tollesbury elevation change, 2000 – 2001
available survey. Even the 2011 to 2013 change map, shown in Figure 3.26, contains a similar type of error, despite these surveys being the most recent, and the most accurate (according to the Environment Agency). Here, there is a diagonal south-west to north-east line throughout the image, with the elevation difference noticeably greater in the north-west. In the 20 m either side of this line, the mean change in elevation is 27.9 mm higher on this side compared to the south-eastern section.

A number of studies have reported similar problems with DSMs produced from LiDAR data, and all suggest that the elevation discrepancies are linked to issues generated through the different flight lines taken by the aircraft during the survey (Huising and Gomes Pereira, 1998; Crombaghs et al., 2000; Lohani and Mason, 2001). This helps to explain the presence of the fairly straight lines seen in Figure 3.25 and Figure 3.26, associated with the multiple flight paths and where these overlap with each other. The exact causes of these errors and artefacts are often not known, but are likely attributable to imperfections in the GPS /INS (Inertial Navigation System) in the aircraft, misalignments between these systems and the laser scanner, and INS drift, tilt and initialisation errors (Crombaghs et al., 2000; Vosselman and Maas, 2001). Latypov (2002) reports that the main cause of height “steps” between the overlapping flight lines.

Figure 3.26: Tollesbury elevation change, 2011 – 2013
is caused by mirror misalignment with respect to the laser beam in the LiDAR sensor. Although these strip errors are not apparent in the elevation maps shown in Figure 3.23, some faint artefacts can be seen by examining the surveys at high zoom levels. They become significantly more apparent when two surveys are compared (as is the case in Figure 3.25 and Figure 3.26) as the range of change between surveys (-500 to +500 mm) is much smaller than the variability in elevation within an individual survey (-500 to +4000 mm) (Rosso et al., 2006).

Some researchers have attempted to eliminate these errors through a series of height adjustments, calibration with ground control points and modelling systematic and random errors (Albani and Klinkenberg, 2003; Sofia et al., 2013). After performing their strip adjustment procedure, Crombaghs et al. (2000) improved the quality of elevation data in their study area, and reduced the RMSE of the residuals between the LiDAR and ground control spot heights from 6-10 cm to 3-5 cm. Regarding the Tollesbury LiDAR data, however, these adjustments are not possible due to the lack of height data from control points in the natural marshes. The Geomatics group at the Environment Agency (who supplied the LiDAR products) were contacted regarding the errors, but they were unable to provide the raw data files, any details on how the data had been processed or any metadata, such as the location of flight lines. They pointed out that the errors in the 2000-2001 change detection map were likely within the ±25 cm accuracy of the sensor used at the time.

Although there are no control points located within the natural marshes at Tollesbury, a series of height measurements were recorded along the top of the sea wall surrounding the realignment site in 2009 by Dr H Burningham using dGPS survey equipment. The elevation at these points was extracted from each of the five LiDAR datasets, and the difference between these and the dGPS heights calculated. Points located in areas where the sea wall is known to be eroding were removed; at all other locations the height of the sea wall is assumed to have remained constant. No spot heights were collected around the breach area. The residuals at the 878 remaining control points were then interpolated using an inverse distance weighting (IDW) algorithm (Lu and Wong, 2008) The IDW method is a straightforward and non-computationally intensive model and is regarded as one of the standard spatial interpolation procedures (Longley et al., 2015) IDW was used over alternative methods, e.g. kriging as they are more complex to apply and require further initial analysis (Oliver and Webster, 1990). The results of the interpolation formed a gridded surface, which was added to the LiDAR datasets covering the realignment area only. This
should account for any offset errors within the original LiDAR and produce a corrected grid of elevation data. The results of this process were not particularly useful, however, as seen in Figure 3.27. This image shows the difference between the 2011 and 2013 elevations, after they had been adjusted using the LiDAR / dGPS residuals. Although the familiar features of the channels are visible, the map is dominated by artefacts generated from the residual interpolation. Similar results were found when the same process was repeated to generate adjusted change maps for each of the intervening time periods, with the artefacts preventing any further analysis of the changes in elevation.

These artefacts do, however, highlight the problems associated with using LiDAR in circumstances where the expected elevational changes are much smaller than the accuracy of the laser sensors. As Cahoon et al. (2000) note, the vertical accuracies achievable with remote sensing technologies are, at their most favourable, an order of magnitude lower than those required for detecting saltmarsh growth or erosion at anything less than a decadal timescale. To test this, further analysis was conducted on the differences between the LiDAR data and the control points from the sea wall obtained during the dGPS survey in 2009. Figure 3.28 shows a boxplot of the residuals.

![Image of elevation change map](image)

**Figure 3.27:** Realignment site elevation change, 2011-2013, after correction from dGPS control points
between the control points and each of the LiDAR surveys. The range is fairly similar across the five datasets, spanning from -900 mm to -1200 mm through to +800 mm to +1200 mm. At these extremes, the LiDAR data is therefore approximately 1 m higher or lower than the true elevation within the realignment site. There are no apparent differences between the 2011 residuals (the LiDAR survey carried out closest to the measurements of the control points in 2009) and any of the other residuals. The data with the arithmetic mean of the residuals closest to zero is the 2001 survey, at 2.23 mm; the 2013 survey has the highest difference at -194.95 mm. In every survey (except for 2011), the majority of the residuals were negative, indicating the LiDAR data was predominately higher than the control points. This finding has been shown in comparisons made by other researchers between LiDAR surveys and GPS control points, and tends to be caused by the presence of vegetation, as LiDAR cannot penetrate the canopy fully (French, 2003; Chassereau et al., 2011; Millard et al., 2013).

The spatial pattern of the residuals is highly variable and changes over time, as shown in Figure 3.29. These three maps show the difference between the LiDAR elevation and the dGPS heights at the location of the control points on the sea wall surrounding the realignment site in 2000, 2002 and 2013. Negative residuals (shades of blue) show where the LiDAR is higher in elevation than the control points, and positive residuals (shades of red) show where the LiDAR is lower. The majority of the residuals (58% on average) fall within the ±300 mm range, with just a few locations exceeding...
Figure 3.29: Elevation residuals between LiDAR and dGPS control points: 2000 (a) and 2002 (b)
±600 mm (6%). Along the southern and western sea walls (zones E and F in Figure 3.29), the residuals are primarily negative in all three survey years; in fact the average residuals become increasingly negative over time in both areas, from -68.5 mm to -252.8 mm between 2000 and 2013 in zone E, and from -283.7 mm to -429.3 mm in zone F. The same pattern occurs in zone A, however the residuals are positive in 2000 (+78.3 mm) before they fall to -51.2 mm in 2002 and then -288.2 mm in 2013. This trend seems to suggest that the sea wall in this region is increasing in elevation over time (by up to 37 cm on average along the northwest section), which is highly improbable. It is likely linked to a random error in the LiDAR, as the pattern in these zones does not seem to correspond with any of the structured artefacts present in the change detection analysis.

In the remaining three zones along the eastern sea wall (zones B, C and D in Figure 3.29) the spatial as well as the temporal variation in the residuals is more complex. In zones B and D, the change in residuals over time is the same, although those in zone D are consistently more negative. In 2000, both sets of residuals are positive (98.2 mm and 59.5 mm for B and D), falling to -97.9 mm and -229.3 mm in
2002, before rising again (but to a lower value than in 2000) to 18.1 mm and -106.9 mm. In zone C, the same change occurs between 2000 and 2002 (-51.9 mm falling to -499.9 mm), but in 2013 the average residual rises dramatically by 780 mm to 280.1 mm. As before, these errors do not coincide with any artefacts, and although they exhibit some structure, they are not systematic and therefore cannot be corrected for by the application of a uniform offset adjustment. This technique is often carried out by researchers, for example Montané and Torres (2006) and Millard et al. (2013) who simply subtracted 7.2 cm and 1.0 m respectively from the LiDAR surveys used in their studies after comparisons with dGPS data. This fails to take into account, however, any spatial variation in the elevation residuals that has been shown in this study to vary considerably even over small distances, and which could affect the validity of their results.

The large differences found between the two survey methods serve to confirm that LiDAR datasets cannot be used to assess such small-scale changes in elevation when investigating a study area such as the Tollesbury marshes as a whole. As Huising and Gomes Pereira (1998) note, despite theoretical research and some practical tests indicating the high accuracy achievable through LiDAR, in reality the results of elevational change studies frequently fail to meet these expectations. This is especially clear when the quoted accuracy of the LiDAR products is compared with similar statistics derived from the residual analysis performed above. Table 3.9 lists the vertical accuracy of the LiDAR surveys provided by the Geomatics team at the Environment Agency. The earliest products have an official accuracy of ±25 cm, and the most recent a much-improved accuracy of ±5 to ±15 cm. This is due to improvements in the laser

<table>
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<th>Year</th>
<th>Quoted Accuracy (cm)</th>
<th>Percentage of residuals within quoted accuracy</th>
<th>Quoted RMSE (m) *</th>
<th>RMSE of residuals (m)</th>
</tr>
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<td>0.126</td>
<td>0.312</td>
</tr>
<tr>
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<td>0.351</td>
</tr>
<tr>
<td>2002</td>
<td>±15 – ± 25</td>
<td>23.1 – 41.9</td>
<td>0.102</td>
<td>0.351</td>
</tr>
<tr>
<td>2011</td>
<td>±15</td>
<td>37.8</td>
<td>0.050</td>
<td>0.315</td>
</tr>
<tr>
<td>2013</td>
<td>±5 – ± 15</td>
<td>7.7 – 24.8</td>
<td>0.049</td>
<td>0.352</td>
</tr>
</tbody>
</table>

* average RMSE statistics provided by EA (pers. com)
systems used over time. It is not known how these figures were derived, whether they are from the manufacturer of the instrument or the Environment Agency themselves, but the analysis of the residuals derived above indicates that the accuracy actually achieved is significantly lower than that quoted. The percentage of the residuals falling within the listed accuracy is shown in Table 3.9. The earliest surveys fair the best, with just over half of the residuals within the ±25 cm value, but as the quoted accuracy increases, the percentage inside that range falls dramatically – just 8% of the residuals from 2013 are within the expected accuracy. The 2011 survey is the most accurate, with 58.3% of residuals falling within ±25 cm of the control points spot heights. 41.9% and 43.2% are within the same range from the 2002 and 2013 surveys, indicating these more recent, and supposedly more accurate datasets are actually less useful than the older products. All of these values are extremely low, considering the vast majority of the calculated residuals (except for a few probable anomalies) should be within this 25 cm error margin. The poorer performance of the more recent surveys is also reflected in the comparison between the quoted and observed root mean square error statistics (RMSE) (Table 3.9). Again, it is not known how the official error values were calculated, but they suggest a general improvement in the accuracy of the LiDAR products over time that is not supported by the residual analysis. The earliest survey has the lowest RMSE, whilst the 2013 dataset has the highest. The RMSE at the control points is considerably higher than expected – 2.5 times to 7.2 times higher than those values provided with the data. As the main purpose of the LiDAR analysis in this chapter is to provide a brief context to the recent changes at the Tollesbury marshes prior to the analysis of the primary data collected as part of this study, no further adjustments or corrections were attempted on the LiDAR elevations as it would be too time-consuming a process.

In one final attempt to glean any form of useful information from the LiDAR data, a series of transects were extracted from each survey in six locations across the realignment site. These are shown in Figure 3.30, on top of the LiDAR elevation change between 2000 and 2013, and are positioned in the key areas examined in the vegetation development section earlier. These include the three former fields in the southern region, the northwest corner and the central vegetated area near the breach. Two transects were also taken spanning creeks within the site: one across the main creek from the breach where some scouring has occurred, and the creek in the NE corner, the only significant area of the realignment site where no vegetation has ever developed.
Figure 3.31 shows the transects extending 220 m from the centre of the southern sea wall, beyond the vegetated area and out into bare mudflat, with those in the western and centre segments also including the creek running through the centre of the site. The elevations have been sampled at 2 m intervals and then filtered with a 10 m moving average to reduce the random errors and spikes from the LiDAR data. Three smoothing methods were tested: a 6 m, 10 m and 14 m moving average. The 6 m moving average transects still contained a significant level of noise, whilst the 14 m moving average removed too much detail. As Rayburg et al. (2009) found, point-to-point variability within LiDAR surveys can be of the order of 5-10 cm, partly caused by differences in how the micro-topography is captured by the laser scanner depending on where the laser beam falls (e.g. on vegetation or the sediment surface). In each profile in Figure 3.31, the elevation is highest at the sea wall, before it gently slopes downwards towards the centre creek. Changes in elevation occurred across the three profiles over time, except for the first 80 m of the eastern field transect, which only increased in elevation by an average of 1.6 mm between 2000 and 2013, compared to the remaining 140 m where the marsh surface increased by 248.6 mm on average, reaching a peak of 537 mm at 196
m from the sea wall. In the western field area, the sediment surface north of the centre creek is 60 cm lower in 2013 than the marsh south of this, and the increase in elevation was slightly higher: 13.2 mm yr\(^{-1}\) south of the creek and 15.8 mm yr\(^{-1}\) to the north over the 13 year data range. The centre field transect (Figure 3.31 (b)) is more affected by an artefact in the LiDAR data than the other two profiles, so an average of the 2000 and
2001 elevations was taken. In the first 75 m of the transect (the vegetated section), the elevation increase from this average level to 2013 was 241 mm on average, between 76 – 160 m the increase was lower at 207 mm (in the bare mudflat), before rising to 340 mm in the remaining 40 m before the creek edge, where vegetation has been expanding. The changes in the creek beds are shown to have been variable over time, with periods of infilling and deepening, but these may be due to different water levels within the channel being captured by the LiDAR sensor rather than actual bed changes. The marsh surface changes suggest the presence of vegetation is having a fairly mixed effect on sedimentation rates in this region. As described in the centre field, the marsh plants are having a positive effect on elevation change, with the bare section of the transect having the lowest change over time. In the outer areas, however, the change is greatest where no vegetation is present, beyond the north of the creek in the western field, and in the mudflat furthest from the seawall in the eastern segment.

The elevation transect in the northwest corner of the realignment site is shown in Figure 3.32; the LiDAR data was again sampled at 2 m intervals and smoothed with a 10 m moving average. The profile is similar in shape to those from the southern areas, although there is a distinct dip in elevation at the foot of the sea wall after which the marsh surface rises again, before sloping down gently. As before, there has been an overall increase in marsh elevation across the area, an average of 183 mm (excluding the first 30 m of the transect) across the 13 year period. The spatial pattern of change is more interesting than the profiles show at first glance, however, especially when analysed alongside the previously described expansion of vegetation in this region.

![Graph: LiDAR elevation transect: NW corner](image)

**Figure 3.32: LiDAR elevation transect: NW corner**
Figure 3.33 shows the change in elevation per year across the transect between the average height of the 2000, 2001 and 2002 surveys (henceforth referred to as the 2001 average) and the 2011 survey, and between 2011 and 2013. The average of the early surveys was used to remove any small-scale variation in elevation and to reduce the effect of any LiDAR artefacts as much as possible. In 2001, vegetation was already present up to 55 m from the sea wall, with bare mudflat in the rest of the profile. The change in elevation in the established vegetation zone in the 30-55 m section between the 2001 average and 2011 was 4.9 mm yr\(^{-1}\). Between 2001 and 2011, vegetation developed in the 55-112 m section, and as Figure 3.33 shows, the change in elevation increased in this zone, to an average of 17.3 mm yr\(^{-1}\). From 106 m onwards, in the bare mud, the increase falls again, and between 112-130 m, was on average, 7.9 mm yr\(^{-1}\). These differing values in the three sections clearly show a positive relationship between the presence of vegetation and elevation change.

The influence of vegetation on the change in elevation between 2011 and 2013 across the same transect is less clear-cut. In the now mature vegetation between 30-55 m, the rate of elevation change was over 10 times that of the previous period, at 52.7 mm yr\(^{-1}\), despite the consensus that sedimentation slows as marshes age (Pethick, 1981; Allen, 2009). In the 55-112 m section, which at this stage still contained fairly young plants, the change rate fell to 34.0 mm yr\(^{-1}\). Vegetation expanded from this area into the 112-120 m zone, which saw a corresponding rise in elevation change, at 86.6 mm yr\(^{-1}\), whilst the unvegetated outer area beyond 120 m saw an increase in elevation of 68.6 mm yr\(^{-1}\). These figures show that although the area of pioneer vegetation saw the greatest rate of elevation change, which was expected, the second highest increase was

![Figure 3.33: Annual elevation change along NW corner transect](image-url)
found in the bare mudflat, whilst the sediment surface in the mature vegetation rose more quickly than that in the ‘intermediate’ zone which developed between 2001 and 2011. The role of vegetation in surface elevation change will be examined much more fully in chapter 5.

Figure 3.34 shows the transects taken from the remaining vegetated section in the realignment site, that from the central vegetated area which developed between 2001 and 2008. The elevation profiles suggest this colonisation took place after 2002, as there are no major differences between the 2001 and 2002 elevations in this region. The data in these plots has been smoothed with a 5 m moving average to try and preserve more of the variability in elevation due to the development of the minor channels between vegetated sections. As with the previous transects, the change between 2000, 2001 and 2002 fluctuates, so an average of the three elevations was calculated to give the longer term change. The elevation change between the 2001 average and 2011 was highly variable across the transect, with a range of values between -48.4 mm and 371.9 mm. The change was lowest where the two main creeks formed at 26-28 m and 90-108 m in the transect, at 79.2 mm and 31.1 mm. In comparison, where marsh vegetation developed, the corresponding increase in elevation was on average 259.9 mm, with the annual rate of change ranging between 16.4 mm yr\(^{-1}\) and 37.2 mm yr\(^{-1}\). In the minor channels, less sedimentation occurred, with the height of the surface increasing by 223.6 mm on average over the 10 years. Between 2011 and 2013, the elevation at all surface types continued to increase, and at a higher rate than previously: 49.3 mm yr\(^{-1}\) on the raised vegetated segments, 39.9 mm yr\(^{-1}\) on the minor channel separating them. The bed

![Figure 3.34: LiDAR elevation transect: central vegetated area](image)
The transect in 2001 is totally unvegetated. Due to the effects of the LiDAR averaging and the complexity of the vegetation colonisation, it is not possible to show the 2011 vegetation extent.
of the two main creeks also increased more rapidly, at 73.3 mm yr\(^{-1}\), over 2.5 times higher than the previous annual rate. Although these values should be treated with caution due to the LiDAR errors and artefacts outlined above, the trends they show (i.e. slighter higher longer term elevation change in vegetated versus unvegetated areas) can be assumed to be valid.

The final two transects from in the realignment site are shown in Figure 3.35 and Figure 3.36 and are sampled from the unvegetated areas around the main creek near the breach site and both sides of the creek in the NE corner of the site. Both sets of profiles have been smoothed with a 5m moving average to retain detail. To the west of the main creek (covered by the 0-24 m section of the transect in Figure 3.35), the pattern of elevation change was comparable to that for the other areas of the site described above, with the marsh surface increasing at a rate of 8.4 mm yr\(^{-1}\) between the 2001 average and 2011, and 78.7 mm yr\(^{-1}\) between 2011 and 2013. The change to the east of the creek was very different, however. This area is known to suffer regularly from erosion due to scouring from the powerful currents that flow here as the tide floods and ebbs through the breach. Erosion clearly dominated in this region between 2001 and 2011, as the sediment surface decreased in elevation by 7.7 mm yr\(^{-1}\) on average; the area that experienced the most scouring eroded by 148 mm over the 10 year period. This negative elevation change reversed between 2011 and 2013, and the surface recovered by 53.1 mm yr\(^{-1}\). The profiles show the creek bed experienced both periods of erosion and accretion, as did the bank walls, but these are in part due to horizontal and vertical errors in the LiDAR, as well as the lower resolution in the 2000 and 2001 surveys. This in particular explains the considerably higher bed level in these two earlier datasets – 174 mm higher than the 2001 elevation and 421 mm higher than that from 2011.

![Figure 3.35: LiDAR elevation transect: main creek](image)

Figure 3.35: LiDAR elevation transect: main creek
Figure 3.36: LiDAR elevation transect: NE corner

No significant difference in elevation change was found either side of the NE creek, where unconsolidated saturated mud is present and increasing in elevation in both areas. Although the north section was consistently slightly lower in height (122 mm on average over time), the increase in elevation was broadly similar between the 2001 average and 2011: 13.3 mm yr\(^{-1}\) to the south (0-54 m in the transect in Figure 3.36) and 11.2 mm yr\(^{-1}\) to the north (72 m onwards). As with all the other transects, the rate of increase was greater between 2011 and 2013, and in this period was higher in the north than to the south of the creek: 68.8 mm yr\(^{-1}\) and 54.8 mm yr\(^{-1}\). These rates are higher than any of the vegetated marsh areas across the realignment site.

Horizontal and georeferencing errors prevent a similar analysis of elevation transects in the natural marshes surrounding the realignment site. It is known that the elevation of marsh surfaces tends to increase slowly over time, but of more interest is whether there is any evidence for loss of marsh at the seaward edge or expansion of the tidal channel network. Figure 3.37 shows the elevation change between the 2000 and 2013 LiDAR surveys for a section of the natural marsh to the east of the realignment site. Transects were taken across two large channels in this region from these surveys and are shown in Figure 3.38. The elevation change between the datasets is also shown, and shows significant variability in erosion and accretion across both transects. In transect A, the elevation change is negative in the western half of the channel (10-20m and 55-65m) and positive in the eastern half. By looking at the profiles from 2000 and 2013, they seem to suggest there has been a westward shift in the location of the channel of two to four metres. Indeed, Figure 3.37 shows this shift to have occurred...
Figure 3.37: Section of natural marsh elevation change, 2000 – 2013

Figure 3.38: LiDAR elevation transects: natural marsh channels
right across the natural marshes, with accretion to the north and west of all channels and bare mud areas, and erosion to the east and south. Marsh segments are outlined by a thin grey line in Figure 3.37, this is the same data generated for the fragmentation analysis carried out in section 3.2. This pattern of change can be seen clearly along transect B in Figure 3.38, which again suggests a north-westward migration of the meander of between two and four metres. These shifts are of course, due to georeferencing differences or horizontal errors between the LiDAR surveys rather than representing real changes in the locations of the channels and marsh segments. This is particularly clear when the meandering channel covered by transect B is examined in more detail. At the centre of the inside bends, the elevation change switches suddenly from positive to negative by 1.0 – 1.5 m over 10 to 20 m, a pattern that is unlikely to have occurred over such short distances due to natural processes in an environment such as this.

3.3.2 Ground-based Elevation and Bathymetry Surveys

In order to provide a more reliable assessment of the longer-term changes in elevation across the Tollesbury marshes and in the outer estuary channel beds, a series of datasets collected through ground-based techniques has been obtained. These include secondary data from monitoring reports produced by the Centre for Hydrology and Ecology and HR Wallingford, as well as measurements collected from the SETs monitored by Prof J French and Dr H Burningham.

In the natural marshes surrounding the realignment site, sediment accretion was measured from April 1994 until September 2007 by researchers from CEH (Garbutt et al., 2008). Changes in the marsh level were measured using a network of 2 m wide transects (12 locations in the Old Hall marshes, 16 in the Tollesbury marshes) with readings taken at fixed positions along each line relative to an aluminium bar placed across a pair of permanent posts at either end of each transect (Garbutt et al., 2006a). Both marsh areas saw increases in elevation throughout the monitoring period, and although there was some variability in the annual rates, the mean accretion rate was 3.2 mm yr\(^{-1}\) for the Old Hall marshes and 4.5 mm yr\(^{-1}\) in the Tollesbury marshes. There was no significant difference between the rates of change before and after the breaching of the sea wall to create the realignment site in August 1995. These rates are comparable with other saltmarshes in the Greater Thames area (Van der Wal and Pye, 2004), and they suggest the marshes are capable of maintaining their position in the tidal frame.
relative to sea-level rise, currently estimated at ~2.4 mm yr\(^{-1}\) in this area (Woodworth et al., 2009).

In the realignment site, a further 20 transects were positioned at random locations (Figure 3.39, CEH numbers 1-20) and were monitored regularly from September 1995 until 2007. Initially, mean rates of sedimentation were linear and very high: 31.6 mm yr\(^{-1}\) in the year following the breach, as shown in Figure 3.40. Accretion has since slowed consistently over time, with a clear reduction in the rate of increase from 2002. In the period 1995-2001, the mean sedimentation rate was 23 mm yr\(^{-1}\), and in 2002-2007, it was 9.8 mm yr\(^{-1}\). Spatially, deposition ranged between 1.8 mm yr\(^{-1}\) (period 1995–2007) at transect 17 to 29.5 mm yr\(^{-1}\) at transect 11. The overall decline in the annual rate of sediment accretion since 1996 has been on average 1.6 mm yr\(^{-1}\) (ranging between 0.6 – 2.1 mm yr\(^{-1}\)). The falling rate of accretion is likely caused by the increasing elevation over time and the corresponding reduction in inundation frequency. This trend follows the elevation-time model for mudflat/saltmarsh growth developed by Pethick (1981) and Allen (1990) where sediment infilling occurs rapidly during the early stages of marsh formation, but swiftly starts to plateau off. The specific curve of the model depends on the sediment supply, and compaction of the surface under the

![Figure 3.39: SETs and CEH realignment site transect locations](image)
increasing sediment load, but is primarily governed by the tidal regime of the region (Allen, 1999; French, 2006a; Garbutt et al., 2006a). As the surface level rises, tidal inundation is reduced, and there are less opportunities for sediment to be deposited. Other local factors can also be important in determining sedimentation rates; Garbutt and Wolters (2004) note that the reduction in deposition from 2003 coincided with the creation of the Abbotts Hall realignment site (breached in October 2002) to the north of the Tollesbury site. Although it is not possible to link the two events directly, all realignment sites act as a sink for sediment, and it is possible that the formation of repeated realignment sites could impact on the sediment budget within an estuary by removing sediment out of the system. This would reduce the capacity of existing saltmarshes and mudflats to adjust to sea-level rise by increasing their elevation as well as the ability of the surface levels within realignment sites to reach elevational equilibrium with adjacent natural marsh areas (Garbutt and Wolters, 2004).

**Figure 3.40: Average surface level changes in the realignment site**
Data averaged from 20 transects monitored by CEH staff, sourced from Garbutt et al. (2008).
In 1999, five extra sediment transects were installed in the realignment site by the CEH team in areas thought to be eroding (Garbutt et al., 2008). These were located on the high ground fronting the sea wall in the three fields in the south of the site, and near the breach, either side of the main creek (Figure 3.39, CEH numbers 21-25). The change in elevation at these transects between April 1999 and September 2007 is shown in Figure 3.41. The three located in the south of the site (transects 21, 22, 23) all saw small linear increases in elevation of between 1.1 – 2.2 mm yr\(^{-1}\), lower than any of the other transects. Erosion initially occurred at transects 24 and 25 before they started to accrete from 2004/2005 onwards. The erosion was more severe to the east of the creek (transect 25), at 15.2 mm yr\(^{-1}\), compared to 6.4 mm yr\(^{-1}\) to the west. The recovery was also more rapid here, at 25.3 mm yr\(^{-1}\) versus 11.6 mm yr\(^{-1}\).

Surface elevation changes measured by the SETs corresponds with the CEH transect data. The locations of the six SET sites are shown in Figure 3.39: sites 1 and 2 are in the natural marshes to the north of the realignment site; sites 3 and 5 are on low, muddy sediment near the breach; and sites 4 and 6 are on the higher, now vegetated marsh surfaces towards the western and southern sea walls. The mean elevation at each of these sites, as well as the cumulative elevation change over time is shown in Figure 3.42. The SET sites were monitored annually between 1995 and 1999, and then sporadically until March 2014. Sites 3 and 5 could not be located after 2009 due to the elevation of the marsh surface exceeding the installed pipe. Elevation change was greatest at the sites with the lowest elevation: 28.0 mm yr\(^{-1}\) at site 3 and 16.7 mm yr\(^{-1}\) at site 5, over the period 1995-2009. This reflects the higher inundation frequency in these areas (both of which are below MHWN level) as well as the closer proximity to the

![Figure 3.41: Elevation change at the five additional CEH transects](image)
breach and thus the sediment source. Although site 5 is lower in elevation for most of the monitoring period and closer to the breach, the incoming flood tide passes over the central vegetated area before reaching its location, thereby a greater proportion of sediment will have already settled out of the water column compared to that reaching site 3. These values are over eight times greater than the accretion rates for the highest two SET locations, sites 1 and 2 in the natural marshes. The elevation at these sites increased by just 3.6 mm yr\(^{-1}\) and 1.5 mm yr\(^{-1}\) respectively, the latter being lower than the current rate of sea-level rise of \(~2.4\) mm yr\(^{-1}\). Also failing to keep pace with rising sea level is site 6, which fronts the southern sea wall, with an elevation increase of 2.3 mm yr\(^{-1}\) over the full monitoring period, despite its elevation remaining just under MHWS (2.56 m versus 2.60 m OD). Although site 4 also fronts a sea wall, in the western part of the site, its elevation is lower (2.1 m OD in 2014), and has been increasing in elevation at a rate of 10.0 mm yr\(^{-1}\) on average.

Figure 3.42: Surface elevation and elevation change from SETs
In agreement with the CEH data, the surface elevation change from the SETs shows that sedimentation has generally slowed over time, with a clear divide between the periods 1995-1999 and 2003-2014. As described above, the greatest increase in elevation occurred at site 3, but this location also exhibited the greatest difference between these two periods: 41.4 mm yr\(^{-1}\) prior to 1999 and 23.0 mm yr\(^{-1}\) after 2003. The natural marshes saw a comparatively high reduction in the accretion rate: from 6.1 mm yr\(^{-1}\) to 2.5 mm yr\(^{-1}\) at site 1 and from 3.3 mm yr\(^{-1}\) to 1.1 mm yr\(^{-1}\) at site 2, representing a fall of between 60-67% over time. Site 4 saw a much more modest fall, from 11.4 mm yr\(^{-1}\) to 10.8 mm yr\(^{-1}\). Unfortunately, due to the lack of data between 1999 and 2003, it is not possible to determine whether the reduction in accretion coincided with the creation of the Abbotts Hall realignment site in 2002. The two remaining sites (5 and 6), in contrast, experienced an increase in the rate of elevation change between the two time periods. Between 1995-1999, the surface level at the rear of the marsh (site 6) increased by 1.4 mm yr\(^{-1}\), rising to 2.7 mm yr\(^{-1}\) in 2003-2014, whilst the elevation rose from 11.2 mm yr\(^{-1}\) to 20.6 mm yr\(^{-1}\) over the same periods at site 5. As Figure 3.42 shows, the rate of increase appears to be rising here over time. Although there is no clear indication as to the causes of this, one possible explanation at site 5 is the increasing role the minor channel to its immediate west is playing in importing sediment to this area. As the analysis of the aerial photography and LiDAR showed, this channel is widening and deepening and may bring sediment earlier on the flood tide than that which is imported via sheet flow across the central vegetated area from the main creek.

Changes in the bed level of the major creeks outside the realignment site were monitored by HR Wallingford between 1994 and 2001. Depths were measured using a DE719C (210kHz) echo sounder with depth data logged at 10Hz, and positions recorded using a Trimble 4000L receiver and Trimble Probeacon (Spearman and Chesher, 1997). The creeks studied, and the locations of the points surveyed are shown in Figure 3.43. The line spacing for the survey was at 100 m intervals in the widest part of the Tollesbury Fleet and at 50 m intervals in the narrower creeks (Spearman and Semmence, 2002). The average change in bed level for each creek, and across the entire estuary is displayed in Figure 3.44. In general, the whole estuary has deepened over the study period, particularly within Tollesbury Creek and Tollesbury Fleet. The highest erosion occurred in Tollesbury Creek, indicating that the breach has had some effect on creek morphology in the area. Despite this, Spearman and Chesher (1997) report that there was no significant correlation between bed changes and distance from the breach in this creek, when the data were examined on a point-by-point basis. The deepening in
Figure 3.43: Survey points in the outer creeks monitored by HR Wallingford

Figure 3.44: Changes in channel bed level across the Tollesbury region

Old Hall Creek has been attributed to an increase in discharge through the upper end since the breaching took place in order to fill the back of the estuary (Spearman and Semmence, 2002). The greatest change occurred between 1994 and 1996, immediately following the breach, with an average of 10-20 cm of erosion occurring throughout the
estuary; parts of Tollesbury Creek deepened by up to and just over 1 m to adjust to the increased flows in the main channel relative to those prior to the breach (Spearman and Chesher, 1997). Between 1996 and 1999, the change in bed level across the estuary fluctuated, and is primarily linked to natural annual variation in the wave climate. Modelling undertaken by Spearman and Chesher (1997) appears to confirm this as there was no correlation between an increase in modelled bed shear stress caused by the breach and the observed bed level changes. Since 1999 the creek level remained fairly stable, a trend which is likely to have continued. Old Hall Creek in particular appears to have been slowly accreting and returning to a depth similar to that prior to the breach. The average changes for 2000-2001 were lower than those recorded in previous years, with the majority of the bed levels varying by ± 0.1 m (Spearman and Semmence, 2002).

3.4 Summary

All of the data analysed in this chapter show that the realignment site at Tollesbury has developed over time as expected, increasing in elevation and with a steadily increasing area of vegetation. Just over half of the site was vegetated by 2013. There is no evidence to show that the natural marshes surrounding the realignment site are eroding at the marsh edge or through the expansion of the creek network. However, small-scale creek collapses have been observed during field visits. Vertical sediment accumulation is occurring within both marsh systems, with the natural marshes showing average accretion rates of 3.2 – 4.5 mm yr\(^{-1}\). In the realignment site, accretion was initially very rapid, at 31.6 mm yr\(^{-1}\) in the first year after breaching, before reducing to 9.8 mm yr\(^{-1}\) between 2002 and 2007. The next chapter will investigate current rates of sediment deposition over much shorter timescales and the factors controlling any variability.
4. Sediment Accumulation Processes in the Tollesbury Marshes

Despite the considerable research that has been conducted within saltmarshes since the late 19th Century, there remains comparatively little knowledge regarding the geomorphological processes of marshes and mudflats in contrast to our understanding of the botany and ecology in these complex systems (Allen, 2000, 2009). Recently created managed realignment sites offer an excellent opportunity to study the fundamental processes operating in saltmarshes as they are still forming in the present time, and have a broader range of environments (mud flats, pioneer marsh, more mature marsh) than older mature natural marshes (Boorman, 1999; Davy et al., 2009). The Tollesbury realignment site, created in 1995, is surrounded by 240 ha of mature natural marsh dissected by a complex creek network. Together, such systems provide a natural laboratory in which the factors controlling sedimentation can be thoroughly investigated in two contrasting environments where local factors (such as climate, geology and wave and tidal conditions) are identical.

This chapter presents the results from the first field campaign undertaken at the Tollesbury marshes between May 2011 and February 2012. This campaign addressed the first aim of this study regarding the controls on sediment accumulation processes in natural and restored marshes. The analysis is organised as follows: firstly, a summary of the tidal and weather conditions during the four fieldwork periods is given, followed by a description of the variation in sediment flux and retention between these campaigns. Factors influencing this variability, as well as those controlling the spatial variation in these measurements across the marshes are then discussed. Finally, the effect of vegetation on sedimentation is investigated.

4.1 Summary of Tidal and Weather Conditions

To contextualise the sediment trap data, a brief summary of the tidal and weather conditions during each of the seasonal campaign periods is presented. Table 4.1 lists the dates and approximate timings of the two weekly periods that comprise each seasonal campaign. The timings are based on the time of low water, and do not indicate the exact time the sediment traps were deployed; all traps were deployed before the next high water. The water levels recorded by the pressure sensor installed in the main creek running from the breach in the old sea wall in the realignment site are shown in
Table 4.1: Seasonal deployment campaign periods

<table>
<thead>
<tr>
<th>Campaign Period</th>
<th>Week 1</th>
<th>Week 2</th>
<th>Whole</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2011</td>
<td>08/05 08:00 – 15/05 04:45</td>
<td>15/05 05:00 – 22/05 10:30</td>
<td></td>
</tr>
<tr>
<td>August 2011</td>
<td>07/08 12:00 – 14/08 07:15</td>
<td>14/08 07:30 – 21/08 12:00</td>
<td></td>
</tr>
<tr>
<td>November 2011</td>
<td>06/11 16:00 – 13/11 06:45</td>
<td>13/11 07:00 – 20/11 12:30</td>
<td></td>
</tr>
<tr>
<td>February 2012</td>
<td>12/02 08:00 – 19/02 16:45</td>
<td>19/02 17:00 – 26/02 09:30</td>
<td></td>
</tr>
</tbody>
</table>

Figure 4.1. The raw data from the sensor were referenced to metres above Ordnance Datum (OD) using the barometric pressure recorder and measurements from a d-GPS survey conducted during October 2011. The water level appears to flatten off around low water, instead of continuing the traditional sine curve shape; this is because the location of the pressure sensor dries out either side of low water. Also shown on each plot is the tidal surge recorded at the nearest tide gauge at Harwich (35 km from Tollesbury), which is maintained by the National Oceanography Centre, Liverpool and was obtained from the British Oceanographic Data Centre (BODC, 2015). The surge is defined here as the difference between the observed water level and the predicted tidal level and the magnitude of the surge is influenced by wind speed and atmospheric pressure (Gonnert and Sossidi, 2011). A positive surge results in a higher than predicted water level, and may be caused by low atmospheric pressure and winds blowing towards the coast, and a negative surge is when water levels are lower than predicted, caused by the opposite meteorological conditions. In February 2012, the Harwich tide gauge recorded sea level values for the first three days of the study period that were deemed ‘improbable’ by BODC quality control and were excluded from Figure 4.1 as shown.

The mean high water levels of the two one-week deployments from the four seasonal campaigns are listed in Table 4.2, along with the average peak level over the whole period. Although the average high water levels between the seasons are similar

Table 4.2: Mean high water levels (m OD) during the seasonal campaigns

<table>
<thead>
<tr>
<th>Campaign Period</th>
<th>Week 1</th>
<th>Week 2</th>
<th>Whole</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2011</td>
<td>2.33</td>
<td>2.96</td>
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<tr>
<td>August 2011</td>
<td>2.51</td>
<td>2.80</td>
<td>2.66</td>
</tr>
<tr>
<td>November 2011</td>
<td>2.60</td>
<td>2.46</td>
<td>2.53</td>
</tr>
<tr>
<td>February 2012</td>
<td>2.41</td>
<td>2.72</td>
<td>2.57</td>
</tr>
</tbody>
</table>
Figure 4.1: Tollesbury water levels and Harwich tidal residual surges for the seasonal fieldwork campaigns
(with a range of 0.13 m), there are much larger disparities between the two weekly deployments within each seasonal period. The greatest is in May 2011, with a difference of 0.63 m between the mean high water level in week 1 and 2. The campaign with the lowest variability is November 2011, with a difference of just 0.14 m between its two weekly periods. The highest water level reached was also recorded in May, at 3.27 m OD, with the lowest high water in February 2012, 1.92 m OD.

Table 4.3 displays the tidal surge statistics for each campaign period. The ‘calmest’ period, with the smallest residual surge (i.e. closest to zero) was May 2011, with a mean surge of just 0.022 m; this deployment also generally saw the lowest peaks in both positive and negative surges. Both November 2011 and February 2012 were characterised by high surges: November primarily in the negative direction (with the peak negative surge of -0.65 m), whilst February experienced both high positive and negative surges (0.77 m in week 1 and -0.60 m in week 2). This campaign also saw the greatest difference between the two one-week deployments; whether this has a noticeable influence on sediment deposition is investigated in section 4.2.

Meteorological data covering the fieldwork periods were obtained from the MIDAS archive through BADC (2013). Hourly mean wind speed, gusts and directions were taken from the weather station at Walton-on-the-Naze, approximately 32 km from Tollesbury. A time series of mean wind speeds and gusts are shown in Figure 4.2 whilst Figure 4.3 shows wind roses for each period. Summary statistics for the wind speeds are listed in Table 4.4. Both plots incorporate data from three days prior to the deployment periods to give some context to the conditions that may have affected wind-generated waves and associated changes in suspended sediment concentration. Depending on the configuration of the wider estuarine system, strong winds from particular directions can increase the volume of sediment brought into the system.

Table 4.3: Tidal surge statistics during the seasonal campaigns

<table>
<thead>
<tr>
<th>Campaign Period</th>
<th>Mean (m)</th>
<th>Minimum (m)</th>
<th>Maximum (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Week 1</td>
<td>Week 2</td>
<td>Week 1</td>
</tr>
<tr>
<td>May 2011</td>
<td>0.022</td>
<td>-0.015</td>
<td>-0.218</td>
</tr>
<tr>
<td>August 2011</td>
<td>0.031</td>
<td>-0.010</td>
<td>-0.354</td>
</tr>
<tr>
<td>November 2011</td>
<td>-0.068</td>
<td>-0.034</td>
<td>-0.645</td>
</tr>
<tr>
<td>February 2012 *</td>
<td>0.049</td>
<td>-0.095</td>
<td>-0.393</td>
</tr>
</tbody>
</table>

* Data for February 2012 week 1 period commences 15/02/12 00:00
Figure 4.2: Hourly mean wind speeds and gusts at Walton on the Naze during the seasonal campaigns
Figure 4.3: Wind roses for mean wind speeds for the seasonal campaigns

Table 4.4: Wind speed statistics during the seasonal campaigns

<table>
<thead>
<tr>
<th>Campaign Period *</th>
<th>Mean speed (ms⁻¹)</th>
<th>Mean gust (ms⁻¹)</th>
<th>Peak gust (ms⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Week 1</td>
<td>Week 2</td>
<td>Week 1</td>
</tr>
<tr>
<td>May 2011</td>
<td>4.51</td>
<td>5.25</td>
<td>7.02</td>
</tr>
<tr>
<td>August 2011</td>
<td>5.44</td>
<td>4.48</td>
<td>9.05</td>
</tr>
<tr>
<td>November 2011</td>
<td>6.63</td>
<td>5.18</td>
<td>9.23</td>
</tr>
<tr>
<td>February 2012</td>
<td>5.07</td>
<td>5.17</td>
<td>8.38</td>
</tr>
</tbody>
</table>

* Statistics shown exclude the 3 day period prior to deployment
(French et al., 2000; Shi et al., 2000; Van Proosdij et al., 2006a). Variations in wind conditions, which alter wave size and the magnitude of wave-related currents, have been shown to drive repeated shifts between erosional and accretionary regimes in the upper intertidal zone (Allen and Duffy, 1998a). Mudflats are particularly sensitive to wind-wave action (depending on the level of shelter), but wind-wave currents also affect the ability of marsh plants to trap and bind sediment, with some studies showing that sedimentation falls with increasing wave power and the associated increase in bed shear stress (Allen and Duffy, 1998a; Callaghan et al., 2010). Despite this, the increase in suspended sediment concentration caused by strong wave activity has led to higher sedimentation rates during some storm events (Goodbred and Hine, 1995; Maa et al., 1998; Day et al., 1999; Lund-Hansen et al., 1999), so the effect of wind and wind-generated waves on sedimentation processes is clearly variable.

In both May and August 2011, winds were predominately from a south-westerly direction, but there was some variability between the east and west directions, especially in May; the strongest winds mainly originated from the south. The mean wind speeds were also fairly similar, at 4.88 ms\(^{-1}\) and 4.96 ms\(^{-1}\) during the full deployment periods. Despite the generally calm conditions during May, the maximum gust recorded during the four fieldwork campaigns occurred during this time (18.01 ms\(^{-1}\)). In November 2011, there was a steady fall in the wind speed, especially in the maximum daily gust, which decreased from 13.9 ms\(^{-1}\) on 03/11/11 to 6.17 ms\(^{-1}\) on 20/11/11. This is also reflected in the difference between the mean wind speeds between the two deployment weeks, 1.45 ms\(^{-1}\), the greatest difference throughout the entire study period. The wind direction during November was also markedly different from all the other campaigns, as it came from the east and south east for the majority of the time. In February 2012, the wind was more steady, with very little difference between the mean wind speeds between each week (0.10 ms\(^{-1}\)), and no difference between the maximum gusts (both weeks reached 15.95 ms\(^{-1}\)). The predominant wind directions also returned to the traditional south and west sectors.

Daily rainfall totals were sourced from a meteorological station in West Mersea, 6.8 km from the Tollesbury marshes, and are shown in Figure 4.4. Table 4.5 lists the weekly deployment totals. The exact effects of rainfall on marsh sedimentation are not fully known, are hard to quantify and vary between each rainfall event, but precipitation is known to disrupt the sediment surface, lowering erosion thresholds and increasing the erosion rate (Tolhurst et al., 2008). This clearly only occurs when the surface is not inundated by the tide, but rainfall events during this time can redistribute
disproportionally large volumes of marsh sediment (Mwamba and Torres, 2002), but may also wash any sediment trapped on plant stems off onto the marsh surface, possibly leading to a short-term rise in sediment deposition (Stumpf, 1983). The only campaign period rainfall may have influenced the mass of sediment deposited on the filter papers deployed is August 2011, when 18.2 mm was recorded. Hourly precipitation records were obtained from the Writtle, to the west of Chelmsford (28.5 km from Tollesbury). These were compared with the tidal records to determine the amount of rainfall that fell during low water during August. There was a slight difference in the totals between the two stations (16.2 mm at Writtle), but the percentage that fell during low tide is

Table 4.5: Rainfall totals (mm) during the seasonal fieldwork campaigns

<table>
<thead>
<tr>
<th>Campaign Period</th>
<th>Week 1</th>
<th>Week 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2011</td>
<td>0.4</td>
<td>1.0</td>
</tr>
<tr>
<td>August 2011</td>
<td>10.6</td>
<td>7.6</td>
</tr>
<tr>
<td>November 2011</td>
<td>4.0</td>
<td>0.8</td>
</tr>
<tr>
<td>February 2012</td>
<td>3.2</td>
<td>2.2</td>
</tr>
</tbody>
</table>
assumed to be the same. Approximately 46.9% of the rainfall at Writtle fell around low water, equating to 8.5 mm at Tollesbury. As mentioned, it is not clear what effect this would have had on the method of recording sediment deposition used. Much lower amounts of rainfall were measured during the three other fieldwork campaigns, where any impact would have been minimal.

4.2 Temporal Variation in Sediment Deposition and Retention

Figure 4.5 shows the seasonal variation in sediment deposition (vertical settling flux) data measured from every single trap deployed during the two weekly periods at the 33 sampling points covering both the realignment site and the surrounding natural marshes. Three traps were used at each sampling point for each week, so there are six flux measurements per point included in the boxplot for every season. All flux data are presented in terms of grams per square metre per tide (g m$^{-2}$ tide$^{-1}$) for easy comparison. The spread of data across the full period of sampling is broad, ranging from 0.01 g m$^{-2}$ tide$^{-1}$ in November 2011 to 26.27 g m$^{-2}$ tide$^{-1}$ in May 2011. The majority (the central 80%) of the flux measurements fall between 0.51 g m$^{-2}$ tide$^{-1}$ and 7.39 g m$^{-2}$ tide$^{-1}$. A first glance at the boxplots in Figure 4.5 suggests that overall, more sediment was deposited in spring (May), with considerably less in winter (February). The median flux for each season does not support this, however. The median is often regarded as a better measure for comparison purposes as it reduces the effect of any outliers, especially where the data are skewed (French et al., 1995; Spencer et al., 2012). The median varies

![Boxplot showing seasonal variation in vertical sediment flux across the Tollesbury marshes](image)

Figure 4.5: Seasonal variation in vertical sediment flux across the Tollesbury marshes
very little between the four campaigns: 2.04 g m\(^{-2}\) tide\(^{-1}\) in February, 2.44 g in May, 2.61 g in August and 2.88 g in November, indicating that any seasonal impact on sediment deposition is minimal. This is corroborated by the result of the Kruskal-Wallis test which was performed on the data to test for differences between the campaigns: \(H(3) = 6.51, p = 0.089\).

Although this finding demonstrates there is no significant difference between the mass of sediment deposited between the four seasonal periods, there could be variations in flux at shorter timescales. Figure 4.6 displays the flux measurements from the sediment traps separated out into the eight weekly deployments that comprised the four fieldwork campaigns. This chart shows much greater variability both within the previously grouped campaigns, as well as between individual deployments over the entire sampling period. This is particularly evident when examining the two weekly deployments in May 2011. The median flux in the first week was 1.62 g m\(^{-2}\) tide\(^{-1}\), compared to 5.26 g in the second week, whilst the highest flux recorded rose from 12.08 g m\(^{-2}\) tide\(^{-1}\) to 26.27 g. The median mass of sediment deposited during the second week in May was higher than every other deployment week, despite the earlier contention that the November 2011 campaign saw the highest average quantity of sediment flux. The contrast between the two May 2011 periods is further emphasised because the deployment most similar to May week 1 is February 2012 week 1, the week with the lowest median flux, 1.49 g m\(^{-2}\) tide\(^{-1}\).

![Figure 4.6: Weekly variation in vertical sediment flux](image-url)
To analyse the differences between the weekly deployments further, a series of Kruskal-Wallis tests were conducted on the eight sampling periods. The overall test result for all the data groups indicated a significant difference, $H(7) = 88.01$, $p < 0.001$. Table 4.6 lists the probability values from the multiple Kruskal-Wallis tests performed between each data group; those surpassing the 95% confidence interval are highlighted. The results confirm the two May 2011 deployments are significantly different, as are the two from February 2012, whilst the difference between the November 2011 weeks only just fall short of the 95% confidence interval. Each of the eight weekly deployments is different to at least one other week, and the second week of the May 2011 campaign is statistically different to five other data groups. The factors influencing this variability will be discussed in the coming subsections.

The seasonal variation in sediment retention, calculated by dividing the average sediment flux measured from the filter papers deployed for the full two-week period in each campaign by the sum of the average flux from the papers deployed for the two individual weeks and expressed as a percentage, is shown in Figure 4.7. The difference between the two totals is attributed to reworking and resuspension of deposited sediment, and can be caused by wind-driven waves and tidal currents as well as bioturbation (Jordan and Valiela, 1983; Harter and Mitsch, 2003). Across the Tollesbury marshes, sediment retention varied widely between the fieldwork campaigns, and was considerably lower during August 2011 than the other three periods. The average level of retention during this summer deployment was 27.6%, 1.5 – 1.8 times lower than the other seasons. The range in values was also considerably narrower: 21% compared to an average of 58% for May, November and February. These three months saw similar average retention measurements, which ranged between

<table>
<thead>
<tr>
<th>May 2</th>
<th>Aug 1</th>
<th>Aug 2</th>
<th>Nov 1</th>
<th>Nov 2</th>
<th>Feb 1</th>
<th>Feb 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 1</td>
<td>&lt;0.001</td>
<td>0.013</td>
<td>0.446</td>
<td>&lt;0.001</td>
<td>0.738</td>
<td>0.993</td>
</tr>
<tr>
<td>May 2</td>
<td>0.016</td>
<td>&lt;0.001</td>
<td>0.339</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>Aug 1</td>
<td>0.885</td>
<td>0.939</td>
<td>0.594</td>
<td>&lt;0.001</td>
<td>0.997</td>
<td>0.459</td>
</tr>
<tr>
<td>Aug 2</td>
<td>0.202</td>
<td>1.000</td>
<td>0.079</td>
<td>0.459</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nov 1</td>
<td>0.054</td>
<td>&lt;0.001</td>
<td>1.000</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nov 2</td>
<td>0.218</td>
<td>0.175</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feb 1</td>
<td>&lt;0.001</td>
<td>0.175</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feb 2</td>
<td>&lt;0.001</td>
<td>0.175</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Highlighted values indicate the probability has exceeded the 95% confidence interval
166

Figure 4.7 Variation in sediment retention across the seasonal fieldwork campaigns

41.5% in winter to 49.9% in autumn. The Kruskal-Wallis test supports this; although the overall result showed a significant difference between the data groups (H(3) = 54.69, p < 0.001), multiple tests showed the only statistical differences were between the August retention data and the three other deployments (p < 0.001 in each case).

As an initial exploration of the relationships between the sediment flux and retention measurements and a range of environmental variables recorded during the sampling periods, a principal components analysis (PCA; Harper, 1999; Davis, 2012) was undertaken on the entire seasonal campaign dataset. A correlation matrix was used, and the results of this are shown in Figure 4.8, with Table 4.7 listing the full details of the variables included. The PCA biplot indicates which variables are correlated with each other; those vectors pointing towards the same direction are positively correlated, those pointing towards opposite directions are negatively correlated, whilst those perpendicular to each other are not correlated at all. The length of the vectors reflects the contribution of the associated variable in helping to explain the variance (Greenacre, 2010). The biplot in Figure 4.8 confirms the relationship between expected variables, such as the positive correlation between mean high water, water depth and hydroperiod, and the negative relationship between the moisture content of the surface sediment and both the wet and dry bulk density. Regarding the environmental factors influencing the mass of sediment deposited on the marsh surface and the percentage retained, there is no one specific variable that helps to explain the variation in the measurements.
Figure 4.8: PCA results for all seasonal campaign data

Table 4.7: Description of variables included in PCA biplots

<table>
<thead>
<tr>
<th>Label</th>
<th>Full variable name and units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flux</td>
<td>Vertical sediment flux (g m$^{-2}$ tide$^{-1}$)</td>
</tr>
<tr>
<td>Retention</td>
<td>Sediment retention (%)</td>
</tr>
<tr>
<td>SSC</td>
<td>Suspended sediment concentration (mg l$^{-1}$)</td>
</tr>
<tr>
<td>Bare</td>
<td>Indicates no vegetation at sampling point</td>
</tr>
<tr>
<td>Vegetated</td>
<td>Indicates a presence of vegetation</td>
</tr>
<tr>
<td>Bare %</td>
<td>Percentage of bare sediment</td>
</tr>
<tr>
<td>Vegetated %</td>
<td>Percentage abundance of vegetation</td>
</tr>
<tr>
<td>Elevation</td>
<td>Elevation at sampling point (m OD)</td>
</tr>
<tr>
<td>Mean HW</td>
<td>Mean High Water during deployment week (m)</td>
</tr>
<tr>
<td>Hydroperiod</td>
<td>Hydroperiod during deployment week (hours)</td>
</tr>
<tr>
<td>Water depth</td>
<td>Average water depth at high tide (m)</td>
</tr>
<tr>
<td>Tidal surge</td>
<td>Average tidal surge (m)</td>
</tr>
<tr>
<td>NE / SE / SW / NW wind</td>
<td>Median wind speed from relevant direction (m s$^{-1}$)</td>
</tr>
<tr>
<td>Wet density</td>
<td>Wet bulk density of surface sediment (g cm$^{-3}$)</td>
</tr>
<tr>
<td>Dry density</td>
<td>Dry bulk density of surface sediment (g cm$^{-3}$)</td>
</tr>
<tr>
<td>Moisture</td>
<td>Moisture content of surface sediment (%)</td>
</tr>
<tr>
<td>Shear strength</td>
<td>Shear strength of surface sediment (kPa)</td>
</tr>
<tr>
<td>Creek distance</td>
<td>Distance to nearest creek / breach (for realignment sampling points) (m)</td>
</tr>
</tbody>
</table>
recorded. Instead, there are a number of variables with weak correlations. The exception to this is the strongly negative relationship between vertical sediment flux and shear strength, which suggests larger quantities of sediment are deposited in sampling locations where the marsh surface is composed of poorly consolidated mud. Factors revealed to have a positive relationship with sediment flux include the suspended sediment concentration (SSC) of the water column and the distance to the nearest creek. The nature of this latter relationship is unexpected, as sediment deposition is generally greatest nearest to the sources of incoming sediment (Bartholdy et al., 2004; Suchrow et al., 2012), in this case, the creek network in the natural marsh and the breach location in the realignment site. This will be investigated further in Section 4.3.

To determine if there is any temporal variation in the relationship between the measured environmental variables and sediment flux and retention, PCA biplots were created for each of the four seasonal field campaigns. These are shown in Figure 4.9. The need to carry out a PCA for each weekly period is negated by the inclusion of the appropriate tidal and weather variables alongside the sediment trap data for each of the eight deployments. In May 2011 (Figure 4.9(a)), sediment flux appears to be positively influenced by mean high water, hydroperiod, water depth and SSC, all of which are to be expected. There is also a positive relationship between flux and the median wind speed from the NE, SE and SW directions, but a negative relationship with increasing wind speeds from the NW direction. Positive tidal surges seem to reduce the vertical sediment flux, an observation that also applies to November 2011 and February 2012 (Figure 4.9(c) and (d)) but is completely reversed in August 2011 (the tidal surge vector is amongst the four wind speed vectors in Figure 4.9(b)). The influence of the tidal variables is clearly variable: in August, November and February the hydroperiod and water depth have minimal impact on sediment flux. No factor is consistently either positively or negatively correlated with deposition; even mean high water, which is positively correlated in May, November and February, has a negative influence in August. The only constant is the lack of any effect from elevation, the sediment properties or the presence and abundance of vegetation.

The picture is similarly mixed regarding sediment retention. Although the wind speed variables never seem to have any effect, an increasing distance from the nearest creek generally leads to higher retention, except during August when the reverse is true. The same anomaly occurs in a number of variables during the summer sampling period. Retention falls with increasing elevation, vegetation abundance and moisture content.
Figure 4.9: PCA results for individual seasonal periods: May 2011 (a), August 2011 (b)
Figure 4.9: PCA results for individual seasonal periods: November 2011 (c), February 2012 (d)
during this season, but has a positive relationship with these parameters in the three other sampling campaigns. Further analysis will be performed on these variables individually in the coming sections.

In order to further elucidate more about the relationships between sediment flux and retention and the environmental variables, multiple linear regression analyses were performed on the seasonal campaign data. Table 4.8 shows the $r^2$ values of the various models generated when sediment flux was used as the dependent variable. The ‘enter’ method was used for each multiple regression to force all the environmental variables selected to be included; however those that did not meet the minimum tolerance test

Table 4.8: Results of multiple linear regression analysis with sediment flux as the dependent variable

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Model</th>
<th>Variables included</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>May</td>
<td>1: All</td>
<td>Bare %, Creek distance, Elevation, Hydroperiod, Mean HW, Shear strength, SSC, Wet density</td>
<td>0.931</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Bare %, Creek distance, Dry density, Elevation, Hydroperiod, Mean HW, Moisture, SE wind, Shear strength, Vegetation %, Water depth, Wet density</td>
<td>0.506</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Tidal surge, Water depth</td>
<td>0.200</td>
</tr>
<tr>
<td>August</td>
<td>1: All</td>
<td>Creek distance, Elevation, Hydroperiod, Mean HW, Shear strength, SSC, Vegetation %, Wet density</td>
<td>0.796</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Bare %, Creek distance, Dry density, Elevation, Hydroperiod, Mean HW, Moisture, SE wind, Shear strength, Vegetation %, Water depth, Wet density</td>
<td>0.378</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Water depth</td>
<td>0.093</td>
</tr>
<tr>
<td>November</td>
<td>1: All</td>
<td>Elevation, Hydroperiod, Mean HW, Moisture, Shear strength, SSC, Vegetation %, Wet density</td>
<td>0.902</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Creek distance, Dry density, Elevation, Hydroperiod, Mean HW, Moisture, NW wind, Shear strength, Vegetation %, Water depth, Wet density</td>
<td>0.421</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Water depth</td>
<td>0.078</td>
</tr>
<tr>
<td>February</td>
<td>1: All</td>
<td>Creek distance, Elevation, Hydroperiod, Mean HW, Shear strength, SSC, Vegetation %, Wet density</td>
<td>0.792</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Bare, Creek distance, Dry density, Elevation, Hydroperiod, Mean HW, Moisture, NW wind, Shear strength, Vegetation, Water depth, Wet density</td>
<td>0.303</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Tidal surge, Water depth</td>
<td>0.143</td>
</tr>
</tbody>
</table>
were not included as they did not help to explain any further variance in the independent variable. All variables listed in Table 4.7 were available to be included in each regression, (except for sediment flux and retention), excluding the Bare and Vegetated categories. Three regression models are listed for each campaign: 1, where all the environmental variables were included; 2, where the SSC was removed but all other variables included; and 3 which only included the tidal variables (hydroperiod, mean high water, water depth and tidal surge). The actual variables included in each multiple regression are listed; those not listed did not pass the tolerance test. The order of the variables included gives no indication as to the amount of variance explained; they are listed in alphabetical order.

The first model performs well for each campaign, with the $r^2$ value ranging between 0.79 and 0.93. Eight out of the 17 available variables were incorporated in each multiple regression, with elevation, hydroperiod, mean HW, shear strength, SSC and wet density passing the tolerance test and included in each of the four model runs. It is, however, important to note that model 1 includes only the data where SSC samples were obtained alongside the sediment traps. This means that only 36 sets of data were used in each regression, severely reducing the range of sediment flux measurements contained in the full dataset (up to 297 sets of data). To widen the analysis and incorporate as much of the data available as possible, the SSC measurements were excluded and a second set of multiple regressions were run (model 2). In these runs, 11-13 of the available 16 variables were included, yet they only served to account for 30 – 51% of the variability in sediment flux, considerably lower than when SSC was used. In the third set of regressions, only those parameters relating to tidal variations were included, as these were the main factors which changed between each weekly deployment (all other variables except the wind speeds and vegetation abundance are constant between each deployment). These factors accounted for very little of the variation in flux, however, as the $r^2$ values ranged between just 0.08 and 0.20.

Multiple linear regression analysis was also performed using sediment retention as the independent variable; these results are shown in Table 4.9. The same three model groups were used as before, and similar results were found. The use of SSC data reduced the sample data size from 99 down to just 12 per campaign, as sediment retention can only be calculated using the mean sediment trap fluxes for each deployment. Thus instead of three datasets per sampling point per deployment, there is only one. With such a limited set of data when the SSC was included in the first multiple regression, a perfect model was generated which accounted for 100% of the
variability. With the SSC removed from the dataset, the $r^2$ values fell to between 0.16 and 0.45. The results from the third model, using the tidal variables gave similarly poor results as above, accounting for just 1.6 – 10.7% of the variability in sediment retention.

These findings indicate that a linear based model is not appropriate to describe the relationship between either sediment flux or retention and a range of environmental variables. The results of the exploratory PCAs indicate a range of factors have some level of influence over the measurements, whilst some have surprisingly little or no influence. Some of these vary temporally, and some spatially. The following subsections investigate further on an individual basis the relationships between a selection of these variables and the sediment dynamics at Tollesbury.

**Table 4.9: Results of multiple linear regression analysis with sediment retention as the dependent variable**

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Model</th>
<th>Variables included</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>May</td>
<td>1: All</td>
<td>Bare %, Creek distance, Elevation, Hydroperiod, Mean HW, Shear strength, SSC, Wet density</td>
<td>1.000</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Bare %, Creek distance, Dry density, Elevation, Hydroperiod, Mean HW, Moisture, NE wind, Shear strength, Vegetation %, Water depth</td>
<td>0.454</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Water depth</td>
<td>0.016</td>
</tr>
<tr>
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<tr>
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<tr>
<td></td>
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<tr>
<td></td>
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<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Tidal surge, Water depth</td>
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</tbody>
</table>
4.2.1 Suspended Sediment Concentration

Both the results of the PCAs and multiple linear regression analyses conducted on the seasonal fieldwork campaign data suggested that higher volumes of sediment suspended in the water column increased the mass of sediment deposited on the marsh surface. Many similar studies in saltmarshes have shown this relationship, as higher levels of sediment in the tidal creek system results in more sediment being available to be deposited on the marsh surface (Leonard, 1997; Hill et al., 2013). At the Tollesbury saltmarsh system, suspended sediment concentration (SSC) was measured at six sampling points, the locations of which are displayed in Figure 4.10. Water samples were taken for each deployment week at each of these points, and Figure 4.11 shows the variation in SSC across the fieldwork campaign periods. A very broad range of measurements was recorded, with sediment concentrations varying between 37.9 mg l\(^{-1}\) and 729.6 mg l\(^{-1}\). The upper end of the data range is heavily influenced by a small number of very high values recorded during the second week of May; excluding the top 5% of the data lowers the maximum concentration to 414.8 mg l\(^{-1}\). In a similar manner as the sediment flux measurements, the seasons with the greatest variability were spring and winter, with the median SSC being 1.6 times greater in the second week of February than the first, and 5.7 times higher in May week 2 compared to week 1.

**Figure 4.10: Location of suspended sediment concentration sampling points**
Despite these variances, the Kruskal-Wallis test revealed there was no statistical difference between any of the deployments: $H(7) = 9.48, p = 0.220$.

The mass of suspended sediment within the water column is itself mainly controlled by tidal height and velocity, wind regime, biological activity and levels of terrestrial erosion (Temmerman et al., 2003a). Tidal current velocity rises with tidal range, and increases resuspension from channel beds, leading to greater turbidity. The highest sediment concentrations tend to occur during spring tides and within the summer when water temperature and biological activity and bioturbation are at their highest levels (Hutchinson et al., 1995; Temmerman et al., 2003b; Murphy and Voulgaris, 2006). The water samples taken at Tollesbury do not show any significant differences between the summer and winter, though, as the median SSC during August and February are essentially the same, at 108.8 mg l$^{-1}$ and 107.5 mg l$^{-1}$ respectively. There is, however, a strong positive relationship between mean high water and SSC, as Figure 4.12(a) shows. The change in the average high water level during the deployment periods accounts for 75% of the variance in SSC. This is a small sample size, however, as there are only eight data points available, one per deployment week. To expand the dataset and test this relationship further, the average water depth (a function of elevation and mean high water) was compared with SSC for each of the six sampling points. This is plotted in Figure 4.12(b), and has a much lower coefficient of
Figure 4.12: Linear relationship between SSC and mean high water (a), and between SSC and average water depth (b)

determination ($r^2$), at 0.38. There is an overall positive trend in the data, but a linear model between the two variables is a poor predictor. The strength of the relationship varies considerably between the seasons, however. In May 2011, the average water depth accounts for 69.5% of the variability in SSC, but only accounts for 6.5% of the variance in November 2011. Through multiple linear regression analysis, it was found that including the mean high water level and hydroperiod in addition to the average water depth, increased these values to 16.2% in November 2011 and 80.3% in May 2011. The inclusion of the wind speed variables yielded no improvements to the models.

As discussed, the exploratory PCA results indicated a relationship between suspended sediment concentration and sediment deposition during the entire seasonal fieldwork campaign (Figure 4.8). As Figure 4.13(a) indicates, however, this relationship is not particularly strong, as the variability in SSC only accounts for 40.9% of the
Figure 4.13: Linear relationship between SSC and sediment flux during entire campaign period (a), and separated out into seasonal deployments (b)

variability in flux. Suspended sediment concentration is more influential during May 2011 and February 2012, with the coefficient of determination at 0.67 and 0.70, as listed in Figure 4.13(b). Despite these higher values, it is hard to judge the significance of these relationships due to the small number of samples used in the comparison. Only 12 data points are available for each season, and the relationships are subject to multiple influences. During November 2011, high sediment flux values were recorded at site 27 (over 10 g m$^{-2}$ tide$^{-1}$), but the SSC was fairly low at ~155 mg l$^{-1}$. With these data excluded, the relationship between the two variables increases: the $r^2$ rises from 0.110 to 0.506 for this season. The SSC data also only relates to the first tide during each deployment when the bottle sampler would have been filled, whilst the flux measurements refer to the average mass of sediment deposited over 13/14 tides.

As the literature is limited surrounding the percentage of sediment retained within marsh systems, the expected relationship between SSC and sediment retention is
not clear. However, retention is likely to decrease with increasing tidal velocities and wind-driven waves that might cause at least some resuspension of newly deposited sediment from the marsh surface (Fagherazzi et al., 2006), both of which lead to an increase in SSC (Temmerman et al., 2003b), so there is potential for a negative relationship between these variables. Despite this possibility, Figure 4.14(a) shows there is no real relationship between SSC and retention throughout the full campaign period, as the $r^2$ value is just 0.069. With the high retention measurement from site 27 (63.4%) in May 2011 removed, the $r^2$ for the relationship decreases to 0.002. The lack of a relationship is seen within each season, with suspended sediment concentrations accounting for just 0.5 – 19.5% of the variability in sediment retention (Figure 4.14(b)). As before, the lack of a large dataset and the fact that SSC is only recorded for the first tidal inundation makes it hard to draw any significant conclusions.

Figure 4.14: Linear relationship between SSC and sediment retention during entire campaign period (a), and separated out into seasonal deployments (b)
4.2.2 Tidal Factors

The tidal cycle is the dominant forcing factor that drives marsh hydrodynamics and sediment transport, so it is no surprise that many researchers have pointed to the significant role tidal factors play in the mass of sediment deposited in marsh systems (Harrison and Bloom, 1977; Allen, 2000; Voulgaris and Meyers, 2004b). The amount of sediment trapped is positively correlated with tidal height and water depth (Letzsch and Frey, 1980; Allen and Duffy, 1998a), an increase of which leads to rising tidal current velocity, which, as mentioned above, promotes sediment transport (Murphy and Voulgaris, 2006). A greater tidal range also increases marsh inundation frequency, providing more opportunities for sedimentation (French and Spencer, 1993; Leonard, 1997). Figure 4.15(a) shows the relationship between mean high water and sediment flux at the Tollesbury marshes. The median flux from each of the eight deployments over the sampling period has been used, whilst the mean high water has been taken

![Figure 4.15: Relationships between sediment flux and mean high water (a) and between sediment retention and mean high water (b)](image)

Figure 4.15: Relationships between sediment flux and mean high water (a) and between sediment retention and mean high water (b)
from the pressure sensor data (as described in Section 4.1). Overall, the amount of sediment deposited is positively correlated with tidal height, but the strength of the relationship is not particularly strong, as the coefficient of determination is 0.588. This is, however, above the average value of similar studies in the Severn Estuary of 0.440 (Allen and Duffy, 1998b). A weak negative relationship is found between mean high water and average sediment retention, as shown in Figure 4.15(b). As increasing tidal levels increases current velocity, resuspension is also likely to increase, reducing the percentage of sediment retained at the marsh surface.

Figure 4.16 shows the effect of the tidal surge on sediment flux and retention. The surge is the difference between the observed and predicted tide, with a positive surge occurring when the observed water level is greater than that predicted. Both measurements tend to fall with an increasing surge, but the relationship is stronger with respect to sediment retention (Figure 4.16(b)), with an $r^2$ of 0.508, compared to flux (Figure 4.16(a)): $r^2 = 0.227$. This latter figure is influenced by the high sediment flux

![Figure 4.16: Relationships between sediment flux and average tidal surge (a) and between retention and surge (b)](image_url)
recorded in the second May 2011 deployment; excluding it increases the strength of the relationship, as signified by an increase in the coefficient of determination to 0.506. Previous research into the influence of surge conditions on marsh sedimentation has focussed on the effect of storm surges (e.g. Bartholdy and Aagaard, 2001; Yang et al., 2003; Cahoon, 2006; Schuerch et al., 2012) rather than the small-scale surges experienced on a daily basis. During storm events, marshes are flooded to greater depths than normal and are exposed to significant wave activity (Allen, 2000). This can lead to vertical erosion in the seaward portions of exposed open coast marshes (Pethick, 1992) or more commonly, an unusually high rate of sedimentation as higher wave activity increases sediment concentrations in the water column which drives greater deposition (Harrison and Bloom, 1977; Reed, 1989; Ehlers et al., 1993; Williams, 2012).

Surges are driven by meteorological components, primarily pressure and wind, with the latter being most likely to have an influence over sediment dynamics. With this in mind, median sediment deposition and retention have been compared with the mean wind speed recorded during each deployment period, as well as the mean gust. These results are shown in Figure 4.17. Given the negative relationship with both measurements and tidal surge, it is surprising that sediment deposition appears to be associated with increasing wind speeds. The correlations are not particularly strong, but as before, if the high value from May week 2 is removed, the $r^2$ values rise to 0.640 for mean speed and 0.488 for mean gust. With regards to the proportion of sediment retained (Figure 4.17(b)), there is a positive relationship with mean wind speed, but a very weak negative one with the average gust. It is not possible to draw any significant conclusions from this, however, due to the low number of data points. What may be causing differing responses between wind speed and tidal surge with sediment deposition is the direction from which the wind is coming from. According to the seasonal PCA biplots (Figure 4.9), tidal surge is strongly correlated with increasing wind speeds from a north-westerly direction during May and February, and with wind from the south-west during November. But the strongest winds during most of the deployments were from the south-east, the direction which points directly and unimpeded towards the North Sea. These stronger winds increase the mean speeds and gusts recorded, and also originate from a direction that is likely to import more sediment into the Blackwater Estuary and the Tollesbury marshes, thereby increasing rates of sedimentation. What may be complicating the relationships between wind speed, tidal surge and sedimentation, however, is that wind speeds were measured at Walton on the Naze and surges at Harwich, both of which are located 30-35 km from Tollesbury.
Returning to the influence of tidal levels on sedimentation, one of the other main relationships between tidal variations and deposition is that the time the marsh surface is inundated by water increases with rising water level (Cahoon and Reed, 1995; Allen and Duffy, 1998b). The duration of flooding at a particular location within the marsh is known as its hydroperiod, and is related to both elevation and water level. A longer hydroperiod offer a greater potential for deposition as there is more opportunity for the settling out of the sediment suspended in the water column (Temmerman et al., 2003b; Voulgaris and Meyers, 2004b). The relationship between hydroperiod and the mass of sediment deposited at Tollesbury is shown in Figure 4.18. The mean flux for each of the 33 sampling points from each one-week deployment is plotted against the hydroperiod calculated for the same period and location. Figure 4.18(a) shows the data separated out into the four seasonal campaigns, with Figure 4.18(b) splitting them into four spatial groups. The sampling points included in each group are shown in Figure 4.19.
Figure 4.18: Sediment flux compared against hydroperiod: seasonal variation (a), and according to grouped sampling points (b)

Figure 4.19: Groups of sampling points relating to hydroperiod analysis
There appears to be no seasonal variation in the relationship between hydroperiod and vertical sediment flux; a very weak correlation exists throughout the sampling period, with the hydroperiod accounting for 0.04–7.5% of the variability in flux. This is considerably weaker than the 13–26% reported by Cahoon and Reed (1995) in a Louisiana saltmarsh. At Tollesbury, the peak in flux corresponds to a hydroperiod between 35 to 50 hours, but then falls rapidly away as the inundation time increases above this. The relationship between the two variables is so weak because low deposition rates are found throughout the range of hydroperiod values.

To analyse the relationship further, instead of splitting the data into their respective seasons, they were divided spatially (Figure 4.18(b)). Group A includes all the sampling points within the natural marshes; Group B covers those within the realignment that are furthest from the breach site; whilst Group C is made up of sites 27, 28 and 35 which are 100–250 m from the breach. Finally, Group D includes the three sites closet to the breach (32, 33, 34), as shown in Figure 4.19. The direction and strength of the correlation in these four groups differs markedly. There is essentially no relationship between hydroperiod and flux in the natural marshes, as the $r^2$ value is just 0.003. Within the realignment site, at those sites around the sea walls (Group B), there is a more expected positive correlation with an $r^2$ of 0.232. As mentioned above, however, the increasing range of flux measurements as hydroperiod increases is affecting the correlation coefficient. With regards to the sedimentation recorded at Groups C and D, there is in fact a weak negative relationship with inundation time. For Group D, this is not unexpected, as the strong ebb currents that occur in this area have led to scouring and erosion in the past (as discussed in Chapter 3). The hydroperiod is greatest around the breach site as the marsh here is flooded first and drains last, but sedimentation is suppressed due to these strong tidal currents. The sediment dynamics at those sampling points within Group C are more complicated. Although these sites are further from the breach, there are minor channels running adjacent to all three locations, which although they will deliver more sediment, they may also cause more turbulent flows.

Figure 4.20 shows the percentage of sediment retained at each sampling period for each seasonal deployment plotted against the corresponding inundation duration. As the retention is calculated for the whole two-week period in each field campaign, the hydroperiod is approximately double that of the one-week deployments shown in Figure 4.18. Based on the previously described moderate negative relationship between mean high water and retention, a similar correlation between hydroperiod and retention was
expected. Instead an increasing hydroperiod only accounts for 2.9% of the corresponding fall in retention. There is thus essentially no relationship between the two variables. Variations in water depth have not been analysed against sediment flux or retention as the relationship between water depth and hydroperiod is so high ($r^2 = 0.978$), that any relationships will be almost identical to those already examined.

4.3 Spatial Variability in Sediment Deposition and Retention

All of the previously discussed factors refer to variables that vary over time, either seasonally or over shorter timescales, and the results presented have mainly been averaged across the entire marsh system. As described in the last section, however, there is potential for considerable variation both between and within the natural marshes and the managed realignment site. Differences in topography, the distance from sediment source and the properties of the sediment, can all help to play a role in determining the mass of sediment deposited and retained across the marsh system. In this section, a brief description of the spatial variability of these measurements will be presented first, followed by more thorough analyses of the reasons for the patterns found.

The spatial variation in sediment deposition, calculated as the mean flux from each set of three traps per sampling point and averaged across the seasonal deployments, is shown in Figure 4.21. On first inspection, there appears to be a clear difference between the mass of sediment deposited between the natural (points 1-15) and restored (16-35) saltmarshes. Average vertical settling fluxes for the two systems are 2.89 g m$^{-2}$ tide$^{-1}$ and 3.92 g m$^{-2}$ tide$^{-1}$ respectively. Within the natural marshes

Figure 4.20: Sediment retention compared against hydroperiod
themselves, there is also a difference between those sampling points located to the north of the realignment site (points 1-5, henceforth termed the ‘north’ points), and those to the east and south (points 6-15, the ‘south’ points). The mean flux recorded in the south area is 28.3% higher than that in the north, possibly because these marshes are closer to the main outer estuary channel. The highest recorded values are found at points 14 and 15 (4.61 g and 4.16 g m⁻² tide⁻¹), both of which are close to a major creek dissecting this part of the marsh. Sampling points 11 and 12, however, also located on the banks of the main Tollesbury Creek channel and have much lower sedimentation rates. In fact, point 12 saw the lowest average flux recorded, at just 1.05 g m⁻² tide⁻¹. This is lower even than the fluxes measured at points 1 and 3, two of the furthest points from the main channel, which had values of 1.60 g and 1.35 g m⁻² tide⁻¹.

Within the realignment site, the spatial variability of sediment flux is fairly complex and controlled by a number of conflicting factors. The highest average measurement, 8.65 g m⁻² tide⁻¹, was recorded within the central vegetated area near the breach at point 27, whilst the lowest was found at point 32, the nearest sampling location to the breach, at 0.77 g m⁻² tide⁻¹. On average, the three points (32, 33, 34) nearest to the breach (and thus the source of sediment supply) are 3.4 times lower than the mean flux for the realignment site as a whole, due to the assumed higher flows in this region. The three transects around the seawalls (points 18-21, 22-24, 29-31) show similar patterns to each other. The fluxes recorded nearest the sea walls, at the highest
elevations and in the more developed vegetation, are considerably lower than those nearer the breach, ranging between 0.54 g to 0.93 g m\(^{-2}\) tide\(^{-1}\). The highest fluxes in these transects tend to be found at those points located in the bare mud furthest from the sea wall, and average 5.69 g m\(^{-2}\) tide\(^{-1}\). This compares to 5.28 g m\(^{-2}\) tide\(^{-1}\) found at the vegetated points on the vegetation-mud border. The role the distance from the breach plays and the influence of vegetation will be investigated in the coming sections.

In contrast, there is no such similar pattern with regards to the seasonally averaged spatial variation in the percentage of sediment retained at each sampling location (Figure 4.22). Mean retention is very similar between the natural and restored saltmarshes (42.5% and 41.2%), but there is more of a difference between the sampling points in the north compared to the south in the natural system (38.8% vs. 44.4%). The highest values were again recorded at points 14 and 15 (52.8% and 51.2%), with the lowest at point 3 (35.6%). In the realignment site, the lowest retention was also near the breach, this time at point 33 (29.9%), but points 32 and 34 were higher (38.3% on average). Along the three transects, the trend in retention is the reverse that of sediment flux. In the western transect, retention rises from 38.4% in the outer mud at point 18 to 48.45% at point 21, the inner vegetated location. The same pattern is found between point 24 (42.9%) and point 22 (48.4%), but not at the eastern most transect where the border vegetated point (number 30) has the highest retention.

Figure 4.22: Spatial variation in seasonally-averaged sediment retention
4.3.1 Realignment Site versus Natural Marshes

Analysis of the seasonally averaged sediment deposition for each sampling point showed there was a difference in vertical mass flux between the two groups of natural marshes and the realignment site. To explore this further, the data for every trap deployed during the eight one-week deployments have been arranged according to which of the three marsh systems the sampling point is located in; this is shown in Figure 4.23. The range of flux measurements is clearly greatest within the realignment site, and lowest in the northern area of natural marsh, but the minimum values recorded are all very similar: 0.01 g in the south points to 0.13 g m$^{-2}$ tide$^{-1}$ in the realignment site. The median fluxes for the three groups are also surprisingly similar: 2.16 g for the north, 2.47 g for the south and 2.64 g m$^{-2}$ tide$^{-1}$ in the restored marsh. The results of the Kruskal-Wallis test indicate there is a statistical difference between the data, however: $H(2) = 7.32$, $p = 0.03$. Further testing revealed the difference to be significant only between the north sampling points and those in the realignment site ($p = 0.020$). When the two natural marsh areas are combined into one group, the difference between this dataset and that from the realignment site is also statistically different: $H(1) = 4.58$, $p = 0.032$.

Few studies have directly compared rates of sediment deposition between restored saltmarshes and neighbouring areas of natural marsh. In the UK, research has instead focused on the development and diversity of vegetation, soil development and

![Figure 4.23: Variation in sediment flux between sampling points in the north and south natural marshes, and those in the realignment site](image-url)
sediment properties, or the accumulation of nutrients (Garbutt and Wolters, 2008; Kadiri et al., 2011; Mossman et al., 2012a). In the United States, however, where marsh restoration has a longer history, studies comparing sedimentation between natural marshes and restored marshes have been undertaken. The findings are all fairly similar, in that sediment deposition is initially significantly greater in constructed marshes than natural ones, but the difference declines markedly over time (Anisfeld et al., 1999; French, 2006b). Craft (1997) measured sedimentation rates in North Carolina of 21-36 kg m$^{-2}$ year$^{-1}$ in the first 3 years of marsh creation, compared to less than 2 kg m$^{-2}$ year$^{-1}$ of sediment in mature marshes. In the Great Bay Estuary, Maine, Morgan and Short (2002) recorded averages rates of deposition of 6.9 g m$^{-2}$ day$^{-1}$ in six restored marshes (aged 1-14 years old) compared to 2.9 g m$^{-2}$ day$^{-1}$ in 11 natural reference sites. In North Carolina, sedimentation fell dramatically in constructed marshes from 62 kg m$^{-2}$ year$^{-1}$ in the first 11 years after creation to 10 kg m$^{-2}$ year$^{-1}$ in the following 13 years (Craft et al., 2003). This compares to rates in comparable natural marshes of 23 kg m$^{-2}$ year$^{-1}$ and 3 kg m$^{-2}$ year$^{-1}$ over the same time periods. As the marsh surface increases in restored marshes relative to the tidal frame through sediment accumulation, accretion will decline due to less frequent flooding and inundation time (Wolters et al., 2005c; Vandenbruwaene et al., 2011). Sedimentation is most rapid on young marshes as they are flooded more often and for longer, increasing the opportunity for sediment to settle out (Pethick, 1981). At the Tollesbury realignment site, sedimentation rates averaged 31.6 mm per annum in 1996, when the elevation was 1.35 m OD (Garbutt et al., 2008), to a rate now less than 5 mm year$^{-1}$, where the elevation has risen to ~1.79 m OD.

Figure 4.24 shows the percentage of sediment retained at each sampling point grouped according to the three different marsh areas. As described in the previous section, there is not a great deal of difference between the marsh types. This is confirmed by the results of the Kruskal-Wallis test: H(2) = 2.87, p = 0.238. Although the range of values is greatest in the realignment site, retention is, on average, highest in the south area of the natural marsh (44.4%). The mean retention is slightly higher in the realignment site compared to the north marshes (41.2% and 38.8%), but the median percentages are very similar (40.3% and 40.8%). The higher retention recorded at the south points may be due to the influence of the mature vegetation here and lower wave activity; Morgan and Short (2002) hypothesised that higher resuspension (i.e. lower retention) occurs in younger restored marshes due to wind waves developing over larger areas of mudflats. This may be true at Tollesbury, as the realignment site is less sheltered from the wind than the natural marshes, where the lower (mainly vegetated)
4.3.2 Distance from Sediment Source

One of the most frequently quoted controls on the spatial variability of sedimentation within a tidal saltmarsh is the distance from the sampling point to the nearest source of sediment, generally the closest creek (French and Spencer, 1993; Chmura et al., 2001; Temmerman et al., 2003b). The creek network is critical to the continued maintenance of the marsh habitat as creeks are the main medium through which sediment is delivered to the marsh surface, and in some cases provide the original source of sediment through resuspension from the creek bed (Stoddart et al., 1989; Reed et al., 1999). Many studies have shown that rates of sediment deposition fall with increasing distance from the nearest creek (Craft et al., 1993; Leonard, 1997; Chmura and Hung, 2004) as sediment is progressively removed from the water column as it progress over the marsh surface (French et al., 1995; De Groot et al., 2011).

In the Tollesbury realignment site, the channel flowing through the breach in the sea wall acts as the main source of sediment to the marsh within. Figure 4.25 shows the relationship between the Euclidian distance from the centre of the breach to each

![Figure 4.25: Variation in sediment retention between sampling points located in the three different marsh areas](image)

marsh surfaces are protected by the complex anastomosing creek system. The effect of vegetation on sediment flux and retention is investigated more fully later in this chapter.
Figure 4.25: Comparison of distance to breach with sediment flux (a) and sediment retention (b) measured within the realignment site

sampling point and the seasonally-averaged rate of sediment deposition (Figure 4.25(a)) and sediment retention (Figure 4.25(b)) recorded there. When all sampling points are considered, there is no relationship between distance and flux ($r^2 = 0.005$), but when the data from the three points near the breach (sites 32-34) are removed, the relationship improves considerably. A significant negative relationship exists ($p = 0.003$), and the coefficient of determination rises to 0.506. As described above, the areas closest to the breach can be expected to experience scour due to the high current velocities, but in the rest of the realignment site, sediment deposition does generally fall away as the distance from the breach increases. In contrast, there is a weak positive relationship between distance from the breach and sediment retention ($r^2 = 0.162$). This trend does, however, appear to be driven mainly by the low retention values recorded at sites 32-34, caused by the strong tidal currents increasing resuspension. With these data removed, the $r^2$ falls to 0.002, indicating there is no relationship between the distance from the sediment source and sediment retention within the realignment site.
In the natural marshes, the relationship between sedimentation and distance from the nearest creek is likely to be more complex. Major channels nearer the main estuary will have higher suspended sediment loads than those smaller creeks in the more remote interior marsh. Therefore those sampling points closer to larger, higher order channels may experience higher rates of sediment deposition than those close to smaller, low order creeks. To investigate if the distance to creeks of different sizes and orders has a bearing on the mass of sediment deposited, each creek in the natural marshes was first categorised into one of three classes using the aerial photography obtained from the Environment Agency. Large creeks are those branching off from and including the main Tollesbury Creek and measure greater than approximately 10 m in width. Medium creeks are tributaries of the large creeks, typically 2 – 12 m in width, whilst small creeks branch off of these, and measure 0.8 – 5 m. There is some overlap in the width of the creek classes, as it is primarily the order of the channel that determined in which category it was placed. Figure 4.26 shows the categories of the creeks within the north natural marshes. The Euclidean distance between each sampling point and the nearest creek of each class was then measured and compared with the rate of sediment flux (Figure 4.27) and the percentage of sediment retained (Figure 4.28). For those sampling points sited on the banks of a large creek (e.g. point 5, Figure 4.26), the distance to the

Figure 4.26: Categories of creeks in the north natural marshes based on size and hierarchy
Figure 4.27: Relationships between sediment flux and distance to the nearest large creek (a), medium-sized creek (b), and small creek (c)

nearest small and medium creek is the same as to the large creek. This is because the closest lower-order creeks (likely near point 4) play no part in the delivery of sediment to these points.

Somewhat surprisingly, there is no relationship between either sediment flux or sediment retention and distance to any of the different creek categories. The data range
of both sediment measurements is broad at all distances, with the proximity to the nearest creek only accounting for 1.7 – 6.1% of the variance in sediment flux, and just 0.6 – 3.1% of the variability in retention. The highest $r^2$ values are found at the closest creek, irrespective of size (i.e. the nearest small creek), indicating the size of the creek has no bearing on the mass of sediment deposited or retained. Although unusual, these
findings are not without precedent; Van Proosdij et al. (2006a) similarly found there to be no significant relationship between deposition and proximity to the nearest source of sediment. Flow dynamics and saltmarsh sedimentation are highly complex processes, and they speculated that the elevation of a sample point within the tidal frame is likely to be more important than distance from a tidal creek (Van Proosdij et al., 2006a).

4.3.3 Elevation

The elevation of the sediment surface is typically one of the main factors controlling the spatial variation of sedimentation, as areas lower in the tidal frame are inundated more frequently and for longer than those approaching the highest astronomical tide level (Cahoon and Reed, 1995; Chmura and Hung, 2004). However, as previously noted, no relationship was found between hydroperiod (primarily controlled by elevation) and either sediment deposition or retention. Figure 4.29 shows the same observations are

![Figure 4.29: Relationship between elevation and sediment flux (a) and sediment retention (b) across the natural marshes and the realignment site](image-url)
found when comparing elevation itself with both sediment measurements. The coefficient of determination is very low for the sediment flux data, at 0.062, and does not improve when the natural marshes and realignment site are considered independently: \( r^2 = 0.023 \) and 0.049 respectively. In the natural marshes, deposition actually rises with increasing elevation between 1.4 m and 1.8 m OD, before falling as elevation rises beyond this. There is a similar trend in the data from the realignment site as a peak in the flux data occurs around 1.8 m OD. The exception to this general trend are the three data points found between 1.4 – 1.5 m OD with flux measurements over 5 g m\(^{-2}\) tide\(^{-1}\). These data refer to sampling points 27, 28 and 35, all three of which are close to minor channels within the realignment site which help to deliver a higher mass of sediment. In terms of sediment retention and elevation, there is a very weak positive trend in the overall dataset \( r^2 = 0.108 \). Considering the natural marshes independently, however, the relationship between retention and elevation is slightly negative \( (r = -0.063, r^2 = 0.004) \). The relationship is stronger in the realignment site, and here retention rises with increasing elevation \( (r = 0.498, r^2 = 0.248) \).

During the comparison of elevation and the sedimentation measurements in the realignment site, it was observed that the relationship between the variables differed significantly between those sampling points that were vegetated, and those on bare mudflats. Figure 4.30(a) shows there is a fairly strong negative correlation between elevation and sediment flux at the vegetated points, whilst there is a strong positive relationship between the two at the unvegetated, bare mudflat points (Figure 4.30(b)). Both relationships are significant at the 95% confidence level. This is a very interesting result, as although the exploratory PCA results indicated the presence or absence of vegetation had little effect on the mass of sediment deposited on its own, vegetation does seem to be combining with other environmental factors to influence sediment dynamics. At the vegetated sampling points, the correlation conforms to the negative trend typically found in saltmarshes, with those at higher elevations receiving less sediment due to lower inundation times. The reverse trend found at the bare points is likely influenced by the higher flow velocities associated with higher water levels (associated with lower elevations) which suppress deposition. The three lowest bare points are located closest to the breach, where strong tidal flows have been found to lower sedimentation. In contrast, at higher elevations, velocities fall alongside shallower water depths, allowing for more sediment to settle out of the water column.

A similar, although weaker, positive correlation is found between elevation and sediment retention at the unvegetated sampling points within the realignment site.
Figure 4.30: Sediment flux compared with elevation at realignment site sampling points with vegetation (a) and those with no vegetation (b)

(Figure 4.31(b). As with deposition, retention is lowest at points 32, 33 and 34, but increases at those points at higher elevations, which are further from the breach and the associated currents. With these three data excluded, the relationship strengthens even further ($r^2 = 0.945$). Resuspension is lowest at the remaining points due to the lower currents velocities associated with the lower water levels found here (Reed et al., 1999). In contrast, although sediment retention is generally higher at the vegetated sampling points, there is no relationship with elevation, as the coefficient of determination is just 0.011 (Figure 4.31(a)).
4.4 Influence of Vegetation

One of the most surprising outcomes of the PCA results was that they indicated, both with all seasons combined and individually, that both the presence and abundance of vegetation had a minimal influence on the mass of sediment deposited within the Tollesbury marshes. This finding is contrary to the commonly held assumption that saltmarsh vegetation dampens wave energy, reduces erosion and thereby enhances sediment deposition compared to areas of mudflat (French et al., 1995; Möller et al., 2001, 2014). As discussed in Section 1.1.5, marsh plants stabilise the surface by binding mud with their roots, consolidating and strengthening the sediment (Shi et al., 2000). This limits resuspension from vegetated patches, again reducing erosion, and increasing the longer-term elevation change in vegetated areas (Cahoon et al., 2000).

Vegetation abundance and species composition varies from season to season, so the influence of vegetation on sedimentation will also likely vary over time. Figure 4.32
Figure 4.32: Seasonal variation in total vegetation abundance

shows the array of total vegetation abundances recorded for each sampling point in each of the seasonal campaign deployments. Each season has a similar range in abundance; there are a number of points with no vegetation present regardless of the time of year, whilst the maximum abundances are fairly consistent (90-100%), except for February (70%). As would be expected, the season with the highest average abundance is the summer deployment (August – 55.6%), followed by autumn (50.1%), spring (48.0%) and finally winter as the vegetation dies off (35.3%). The median abundances follow the same pattern, and are 5-15% higher than the arithmetic mean due to the lower influence of the unvegetated sites. The ‘quality’ of the marsh plants will also vary over time, but cannot be easily quantified. The vegetation in August is generally very lush and healthy, whilst in February it is mostly dead or dying. It is still standing, however, and providing a barrier to wave action. In fact dead _Salicornia_ stems are rougher and more rigid compared to healthy ones that bend during strong waves (Möller and Spencer, 2002).

Vegetation abundance also varies spatially and is on average higher in the southern natural marshes compared to those to the north of, and within, the realignment site (Figure 4.33). The mean abundance is lowest in the realignment site, but this is heavily influenced by the seven points that are totally unvegetated. When the median abundances are compared, the south marshes and the restored points are identical, both at 57.5%, whilst the north points are lowest (45%). The pattern is more mixed on an individual season basis, as shown in Figure 4.34. In May, the median abundance was
Figure 4.33: Seasonally-averaged variation in vegetation abundance between sampling points located in the north and south natural marsh areas, and the realignment site.

Figure 4.34: Seasonal variation in vegetation abundance between sampling points located in the three marsh areas.
actually lowest in the south marshes, but in November it was highest. In both of these months, the abundance from the natural marshes and the realignment site was very similar (60% for both in May, 50% and 55% in November). The greatest difference between the three marsh areas was found in February 2012, where the median abundance was just 25% in the north marshes, compared to 47.5% and 42.5% in the south and restored marshes. August 2011, by contrast, saw very similar total percentage abundances: all three marsh areas had a median between 60-70%.

The species composition recorded at all the sampling points for each of the seasonal campaigns is shown in Figure 4.35. The list of species found is given in Table 4.10; Table 4.11 shows the Dafor scale used to estimate the cover for each species and the associated numeric rating given for each sampling point as presented in Figure 4.35. *Salicornia* and *Spartina* are the most abundant marsh plants in each season, with a seasonally averaged mean rating of 2.0 and 1.4 on the Dafor scale. *Salicornia* is the

<table>
<thead>
<tr>
<th>Table 4.10: List of species recorded during vegetation sampling</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Label</strong></td>
</tr>
<tr>
<td>Salicornia</td>
</tr>
<tr>
<td>Spartina</td>
</tr>
<tr>
<td>Aster</td>
</tr>
<tr>
<td>Puccinellia</td>
</tr>
<tr>
<td>Spergularia</td>
</tr>
<tr>
<td>Suaeda</td>
</tr>
<tr>
<td>Atriplex</td>
</tr>
<tr>
<td>Enteromorpha</td>
</tr>
<tr>
<td>Ulva</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Table 4.11: Dafor scale for vegetation cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>Source: (Wheater et al., 2011).</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Dafor Scale Rating</strong></th>
<th><strong>Numeric Rating</strong></th>
<th><strong>Vegetation Cover (%)</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Dominant</td>
<td>5</td>
<td>&gt; 75</td>
</tr>
<tr>
<td>Abundant</td>
<td>4</td>
<td>51 – 75</td>
</tr>
<tr>
<td>Frequent</td>
<td>3</td>
<td>26 – 50</td>
</tr>
<tr>
<td>Occasional</td>
<td>2</td>
<td>11 – 25</td>
</tr>
<tr>
<td>Rare</td>
<td>1</td>
<td>&lt; 11</td>
</tr>
<tr>
<td>Absent</td>
<td>0</td>
<td>0</td>
</tr>
</tbody>
</table>
Figure 4.35: Seasonal variation in saltmarsh species composition
more abundant species, found at 23 of the 33 sampling points, whilst *Spartina* is only found at 12, all of which are within the realignment site. When only the sampling points where these two species are found are considered, the mean Dafor scale rating rises to 2.6 for *Salicornia* and 3.5 for *Spartina*, indicating the latter is generally more dense. The other prevalent species found within the marshes is *Enteromorpha*, a macro green algae that grows in mats on the sediment surface. Typically found at 15 sampling sites in August – February, it was present at 30 sites in May 2011. There is little variation in species composition between the seasons, with most plants recorded at least once during each deployment. The overall abundances are lower in February and May, compared to August and November, as expected and described in the total vegetation cover above.

4.4.1 Presence of Vegetation

Figure 4.36 compares the mass of sediment deposited on every trap deployed throughout the entire campaign period at a vegetated sampling point with those deployed at an unvegetated, bare point. A vegetated point is one with a presence of marsh plants, irrespective of total abundance. Sampling points with only *Enteromorpha* or *Ulva* species are not counted as a vegetated point. The boxplot confirms the minimal influence vegetation alone has on the rate of deposition; there is very little difference between any of the descriptive statistics for the datasets listed in Table 4.12. Although

![Figure 4.36: Comparison of sediment flux recorded at vegetated and bare sampling points throughout the seasonal campaign period](image-url)
Table 4.12: Descriptive statistics for sediment flux recorded across the full campaign period at vegetated and bare sampling points

<table>
<thead>
<tr>
<th>Statistic</th>
<th>Vegetated points (g m(^{-2}) tide(^{-1}))</th>
<th>Bare points (g m(^{-2}) tide(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>3.27</td>
<td>3.25</td>
</tr>
<tr>
<td>Median</td>
<td>2.35</td>
<td>2.07</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.01</td>
<td>0.21</td>
</tr>
<tr>
<td>Maximum</td>
<td>26.27</td>
<td>22.27</td>
</tr>
<tr>
<td>25(^{th}) Percentile</td>
<td>1.08</td>
<td>0.83</td>
</tr>
<tr>
<td>75(^{th}) Percentile</td>
<td>4.46</td>
<td>4.72</td>
</tr>
<tr>
<td>Data count</td>
<td>863</td>
<td>275</td>
</tr>
</tbody>
</table>

The maximum rate of sediment deposition was measured at a vegetated point (4.00 g m\(^{-2}\) tide\(^{-1}\) greater than the highest at a bare point), the arithmetic means are almost identical (3.27 g and 3.25 g m\(^{-2}\) tide\(^{-1}\)). The median fluxes are also very similar, with that from the vegetated points just 0.28 g m\(^{-2}\) tide\(^{-1}\) higher than measured at the unvegetated points. The Kruskal-Wallis test confirms the two datasets are not statistically different: H(1) = 1.31, p = 0.2529.

There is some seasonal variation in the differences between sediment flux recorded at the two groups of sampling points (Figure 4.37). In May 2011, barring the maximum rate of flux, the amount of sediment deposited was actually higher at the bare points than those with vegetation. The median flux here was 5.06 g compared to 2.35 g m\(^{-2}\) tide\(^{-1}\) at the vegetated points. These differences are statistically significant, H(1) = 10.97, p = 0.0009. In contrast, the pattern in November was reversed, with significantly higher (H(1) = 20.65, p = 0.00001) fluxes recorded at the vegetated sampling points. The median deposition rate was over double that from the bare points (3.52 g and 1.74 g m\(^{-2}\) tide\(^{-1}\) respectively). In August and February, the differences were much smaller, with the median fluxes at the vegetated points only marginally higher (2.06 g compared to 1.92 g m\(^{-2}\) tide\(^{-1}\) in August, 2.00 g compared to 1.95 g m\(^{-2}\) tide\(^{-1}\) in February). Neither of these differences are significant.

Although these findings are generally contrary to the traditionally held assumption that saltmarsh vegetation significantly enhances sediment deposition, there is a growing body of research that is questioning this assumption (Neumeier and Ciavola, 2004; Silva et al., 2009; Mudd et al., 2010; Gedan et al., 2011). A number of these studies suggest that vegetation instead stabilises the sediment surface and prevents resuspension and erosion from wave activity (Boorman et al., 1998; Shi et al., 2000). To test this at Tollesbury, the percentage of sediment retained at the vegetated sampling
Figure 4.37: Comparison of sediment flux at vegetated and bare sampling points for each seasonal deployment

points during the seasonal deployments was compared with that recorded at the bare points; this is shown in Figure 4.38. Although not statistically significant ($H(1) = 2.4, p = 0.1210$), the sampling points with vegetation present did indeed retain, on average, more sediment that those without plants. The presence of vegetation raised both the mean (43.0% vs. 38.1%) and the median (43.4% vs. 36.5%), as well as the maximum percentage of sediment retained (82.2% vs. 58.0%). The comparison between the two classes did vary between the four seasons, however, as shown in Figure 4.39. The difference between the sediment retained at the vegetated and unvegetated points was greatest in May 2011, the only season where the difference was statistically significant: $H(1) = 7.46, p = 0.0063$. Vegetated points retained on average 51.2% of the sediment deposited during this time, compared to 36.2% at the bare points. A similar pattern, although less pronounced, was observed during November, (51.3% within vegetation, compared to 45.6% in bare mud). During August and February, however, a marginally higher percentage of sediment was retained at the sampling points within the unvegetated mudflat. All descriptive statistics were higher at the bare points, except for the maximum retention, which was greatest within the vegetation during both these deployments. These differences are not statistically significant, however.
Figure 4.38: Comparison of sediment retention recorded at vegetated and bare sampling points throughout the seasonal campaign period

Figure 4.39: Comparison of sediment retention at vegetated and bare sampling points for each seasonal deployment
4.4.2 Vegetation Abundance

It has been shown that the presence of vegetation has different levels of influence on sediment deposition and retention within the Tollesbury marshes. This section now investigates whether there is a relationship between the extent of vegetation cover and sedimentation. Figure 4.40(a) compares the seasonally averaged sediment flux with the

![Figure 4.40: Relationships between seasonally averaged sediment flux and vegetation abundance (a) and for each individual season (b)](image)

\[
y = -0.002x + 3.561 \\
R^2 = 0.001 \\
p = 0.859
\]
average vegetation abundance at each sampling point. There is absolutely no relationship between the two variables, as indicated by the coefficient of determination (0.001). An increase in vegetation cover has no effect on the amount of sediment deposited. Even with the unvegetated points removed, there is still no correlation; in fact the $r^2$ value decreases to 0.0004. Variations in vegetation cover within each season (Figure 4.40(b)) have similarly minimal influence on sediment flux, accounting for just 0.2 – 7.0% of the variance measured. There is a wide variety in flux measurements across the bare sampling points, and at all levels of vegetation cover. As before, removing the unvegetated data from the analysis yields no significant improvements in the relationships.

The wide variation in the amount of sediment deposited across the range of vegetation abundances is shown clearly in Figure 4.41, which groups every flux measurement from across the full campaign period according to 10% increments in vegetation cover in addition to those from the bare sampling points. Although there are a number of statistically significant differences between some of the abundance groups (Table 4.13), there is no trend with increasing cover. The group with the highest median flux is actually the 1-9% abundance (8.54 g m$^{-2}$ tide$^{-1}$), whilst the lowest median flux (0.88 g m$^{-2}$ tide$^{-1}$) is within the >90% cover. This highest cover is in fact, statistically lower than all the other groups except the 20-29% class. The range of flux

![Figure 4.41: Variation in sediment flux from the entire seasonal campaign across the range of vegetation abundance](image)
Table 4.13: Probability results from Kruskal-Wallis tests performed on sediment flux between each vegetation abundance group

<table>
<thead>
<tr>
<th></th>
<th>1-9</th>
<th>10-19</th>
<th>20-29</th>
<th>30-39</th>
<th>40-49</th>
<th>50-59</th>
<th>60-69</th>
<th>70-79</th>
<th>80-89</th>
<th>&gt;90</th>
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<tbody>
<tr>
<td>0</td>
<td>0.006</td>
<td>0.620</td>
<td>0.267</td>
<td>1.000</td>
<td>1.000</td>
<td>0.465</td>
<td>0.000</td>
<td>0.000</td>
<td>0.020</td>
<td>0.000</td>
</tr>
<tr>
<td>1-9</td>
<td></td>
<td>0.000</td>
<td>0.010</td>
<td>0.008</td>
<td>0.067</td>
<td>0.619</td>
<td>0.178</td>
<td>0.156</td>
<td>0.000</td>
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</tr>
<tr>
<td>10-19</td>
<td>0.022</td>
<td></td>
<td>0.836</td>
<td>0.760</td>
<td>1.000</td>
<td>0.678</td>
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<td>1.000</td>
<td>0.000</td>
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</tr>
<tr>
<td>20-29</td>
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<td>0.655</td>
<td>0.461</td>
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<td>1.000</td>
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</tr>
<tr>
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<td>0.000</td>
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<td>40-49</td>
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<td>0.766</td>
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<td>0.158</td>
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<tr>
<td>50-59</td>
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<td>0.961</td>
<td>0.997</td>
<td>0.000</td>
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<tr>
<td>60-69</td>
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<td></td>
<td></td>
<td></td>
<td>0.294</td>
<td>0.303</td>
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<tr>
<td>70-79</td>
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<td></td>
<td></td>
<td></td>
<td>1.000</td>
<td>0.000</td>
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</tr>
<tr>
<td>80-89</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.000</td>
<td></td>
</tr>
</tbody>
</table>

Highlighted values indicate the probability has exceeded the 95% confidence interval

measurements between distinct groups is also similar; the range of the 70-79% group (25.8 g) is most alike that from the unvegetated points (22.3 g), and the 10-19% class is very close to the 60-69% class (ranges 14.6 g and 14.2 g). All these observations simply serve to confirm the lack of any relationship between vegetation cover and sediment deposition.

A direct comparison between the seasonally averaged sediment retention and vegetation abundance is shown in Figure 4.42(a). Although the vegetated sampling points typically retain more sediment, there is no significant trend between the quantity of vegetation and retention, with plant cover only accounting for 27.4% of the variance in retention. When only the vegetated points are included, the strength of the relationship actually falls, with an increase in cover explaining only 13.2% of the measured retention. On a season-by-season basis (Figure 4.42(b)), abundance has a consistently lower influence, accounting for between 0.00% and 20.6% of the variation in retention. The season with the strongest relationship is May 2011, but with the bare points removed, increasing plant cover explains only 2.8% of the retention. None of the remaining seasonal deployments contain any real trend between the two variables, indicating the presence of vegetation is more influential on increasing retention than the amount.

As for sediment flux, the lack of a clear trend between sediment retention and vegetation abundance is attributed to the wide range in retention at all cover levels, as displayed in Figure 4.43. There are no significant differences between the data classes as a whole: $H(10) = 14.0$, $p = 0.172$, with probabilities between individual groups ranging between 0.111 – 1.000. The medians all fall within a fairly narrow range; there
is only a ~20% difference between the highest (53.4%) within the 50-59% class, and the lowest (31.3%) that actually falls within the highest abundance group. The only apparent increase with higher abundance appears to be associated with the maximum retention. The average maximum in the classes between 0 – 40% is 50.3%, whilst that
4.4.3 Species Composition

The previous sections have examined the importance (or lack of) of the presence and abundance of vegetation on sedimentation with no consideration as to the influence of the species of plant at the sampling points. Different species have different physical properties, such as stem height, density and rigidity, leaf arrangement and surface area, and root structure (Adam, 1990). Some plants grow as small single stems from the marsh surface (e.g. *Salicornia perennis*), as tall grasses with upright stems (e.g. *Spartina anglica*), or as shrubs with intricate dense canopies (e.g. *Atriplex portulacoides*). These structures will differ in their ability to attenuate wave energy, encourage deposition, and retain sediment, as well as in their capacity to create turbulence and promote erosion (Silva *et al.*, 2009).

One of the ways of exploring the linkages between species assemblages and environmental factors is through canonical correspondence analysis (CCA). CCA is a direct gradient analysis technique and represents a special case of multivariate regression (Sherrod, 1999). Figure 4.44 shows the CCA triplot for the data from the entire seasonal field campaign. The results are displayed in a similar manner to the PCA.
The environmental variables are represented by vectors, with the length of the vector denoting its importance, with the angle between vectors indicating correlations between the variables. Points represent the species scores (in red), and the location of the scores relative to the vectors indicates the associations between them and the environmental factors (Palmer, 1993). A subset of the environmental variables used in the PCA has been included; those factors excluded (e.g. tidal surge, wind speeds) added little to explaining the variance in the species data. One extra variable has been included to indicate whether sampling points are within the realignment site or the natural marshes.

The CCA for the entire seasonal dataset (Figure 4.44) suggests higher sediment fluxes are found where *Ulva* dominates, but this relationship only exists during May 2011, as indicated by the CCAs carried out for each season separately (Figure 4.45). The presence of *Enteromorpha* is also associated with higher deposition during May (Figure 4.45(a)), but negatively correlated during the three other campaigns. There are no significant positive relationships between any species and flux in these seasons, although *Spartina* appears to have a small influence.
Figure 4.45: CCA results for individual seasonal periods: May 2011 (a), August 2011 (b)
Figure 4.45: CCA results for individual seasonal periods: November 2011 (c), February 2012 (d)
The correlations between sediment retention and plant species are variable throughout the sampling year. Overall, the CCA suggests *Spartina* is associated with higher rates of retention; the same pattern is found during May 2011 and a weak correlation is evident in February 2012 (Figure 4.45(a) and (d)). In August 2011, there is a weak positive association between retention and *Ulva* and *Enteromorpha*, whilst the reverse is seen in November; instead *Spergularia* and *Aster* have more of an influence during this period. It is thus impossible to say that certain species consistently have either a positive or negative effect on either sediment deposition or retention during the sampling period.

4.5 Summary

The results presented in this chapter show the complex nature of the sedimentary processes occurring across the Tollesbury saltmarshes. Both sediment deposition and net retention vary over time and space, but there are no factors that individually explain the variation in either variable with any statistical significance. Instead, it is likely that a number of factors interact synergistically to control marsh sedimentation. What has been demonstrated is that the presence and abundance of vegetation, in itself, has minimal influence over the mass of sediment deposited, a finding that is contrary to the majority of saltmarsh studies. The presence of vegetation does, however, exert a positive influence over the percentage of sediment retained. Table 4.14 provides a summary of the strength of the theorised (i.e. from the literature) relationships between sediment flux and retention and the variables investigated, and the relationship actually found in the field dataset. The full name of the variables listed is given in Table 4.7. The indicators used in Table 4.14 are explained in Table 4.15. For example, an increase in SSC is likely to be associated with a strong positive increase in flux, whilst an increase in distance from the creek edge is theorised to lead to a strong negative impact on deposition. The next chapter investigates further the influence of vegetation over both sediment deposition and retention with a specific reference to the Tollesbury realignment site.
Table 4.14: Summary of theorised and measured relationships between sediment flux and retention and the variables investigated in Chapter 4

<table>
<thead>
<tr>
<th>Theorised Relationship *</th>
<th>Measured Relationship</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Flux</td>
</tr>
<tr>
<td>SSC</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Mean HW</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Tidal surge</td>
<td>↑</td>
</tr>
<tr>
<td>Wind speed</td>
<td>↑</td>
</tr>
<tr>
<td>Hydroperiod</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Creek distance (realignment)</td>
<td>↓↓↓</td>
</tr>
<tr>
<td>Creek distance (natural marsh)</td>
<td>↓↓</td>
</tr>
<tr>
<td>Elevation</td>
<td>↓↓</td>
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<td>Vegetated</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Vegetated %</td>
<td>↑↑↑</td>
</tr>
</tbody>
</table>

* Qualitative estimate of strength of relationship
– Strength of relationship unknown

Table 4.15: Explanation of Indicators used in Table 4.14, with associated qualitative descriptors and r² values

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Descriptor *</th>
<th>r² value *</th>
</tr>
</thead>
<tbody>
<tr>
<td>↓↓↓↓</td>
<td>Very strong negative</td>
<td>64 – 100%</td>
</tr>
<tr>
<td>↓↓</td>
<td>Strong negative</td>
<td>36 – 64%</td>
</tr>
<tr>
<td>↓</td>
<td>Moderate negative</td>
<td>16 – 36%</td>
</tr>
<tr>
<td>↑</td>
<td>Weak negative</td>
<td>4 – 16%</td>
</tr>
<tr>
<td>↑↑</td>
<td>Very weak negative</td>
<td>0 – 4%</td>
</tr>
<tr>
<td>↑↑↑</td>
<td>Very weak positive</td>
<td>0 – 4%</td>
</tr>
<tr>
<td>↑↑</td>
<td>Weak positive</td>
<td>4 – 16%</td>
</tr>
<tr>
<td>↑↑↑</td>
<td>Moderate positive</td>
<td>16 – 36%</td>
</tr>
<tr>
<td>↑↑↑↑</td>
<td>Strong positive</td>
<td>36 – 64%</td>
</tr>
<tr>
<td>↑↑↑↑↑</td>
<td>Very strong positive</td>
<td>64 – 100%</td>
</tr>
</tbody>
</table>

* Descriptors and r² values derived from Evans (1996)
5. The Role of Vegetation in Sedimentation within the Tollesbury Realignment Site

It has long been recognised that halophytic vegetation plays a critical role in the development and maintenance of the saltmarsh system through the attenuation of wave energy, and the associated increase in sediment accumulation compared to areas of bare mudflat (Reed et al., 1999; Möller et al., 2001; Möller, 2003). There is, however, a growing body of research that suggests the influence of vegetation is not as great as previously thought (Silva et al., 2009; Mudd et al., 2010). The results from the seasonal fieldwork campaign undertaken at the Tollesbury realignment site and surrounding natural marshes indicate this to be true (Chapter 4), but more research is required to more fully understand the role of vegetation in sediment accumulation processes.

This chapter presents results from the field campaigns carried out at Tollesbury in May 2012 and March 2014 in order to answer the second aim of this study regarding vegetation and sedimentation. Following a summary of the tidal and weather conditions during these periods, a brief exploration of the factors controlling sediment deposition and retention will be given. The influence of the presence of vegetation, its abundance and species composition on sedimentation and trapping efficiency within the realignment site will then be thoroughly investigated.

5.1 Summary of Tidal and Weather Conditions

Intensive monitoring of sediment deposition and retention and a range of environmental variables was undertaken at Tollesbury over consecutive days in the spring of 2012 and 2014. Table 5.1 lists the dates and approximate timings of the three daily periods for the two deployments. The timings are based on the time of low water, rather than the exact times the sediment traps were deployed; every trap was laid before the next high water and experienced two tidal cycles. Figure 5.1 shows the water levels recorded within the

<table>
<thead>
<tr>
<th>Table 5.1: Deployment periods for intensive monitoring field campaigns</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
</tr>
<tr>
<td>Day 1</td>
</tr>
<tr>
<td>Day 2</td>
</tr>
<tr>
<td>Day 3</td>
</tr>
</tbody>
</table>
realignment site and the weather conditions for the May 2012 deployment obtained from local meteorological stations (wind data from Walton on the Naze; rainfall from West Mersea) from the MIDAS service (BADC, 2015). Water levels were measured every 5 minutes and referenced to Ordnance Datum, and tidal surges were sourced from the tide gauge at Harwich (BODC, 2015). Mean high water levels (Table 5.2) rose
Table 5.2: Tidal and weather statistics for May 2012

| Day  | Mean high water (m OD) | Mean tidal surge (m) | Mean wind speed (ms\(^{-1}\)) | Peak wind gust (ms\(^{-1}\)) | Rainfall (mm) *
|------|------------------------|----------------------|-------------------------------|-----------------------------|---------------------
| Day 1| 1.94                   | -0.025               | 6.56                          | 16.98                       | 0.4                 
| Day 2| 2.06                   | -0.059               | 5.35                          | 15.95                       | 12.8                
| Day 3| 2.40                   | 0.067                | 3.48                          | 9.26                        | 4.2                 

* Rainfall records are for calendar days, i.e. 30/04, 01/05, 02/05, 03/05. Total rainfall on 03/05: 10.6 mm.

between each daily period, the difference between Day 1 and Day 3 being 0.46 m. The tidal surge was generally negative, indicating the observed tidal levels were lower than predicted, but overall the surge was minimal, only ranging between -0.172 m and 0.276 m. The wind data shown incorporate observations from two days prior to the first deployment day to provide details of the antecedent conditions which may have affected suspended sediment concentrations in the water column. Wind speeds were high in the 24 hours prior to deployment (mean: 11.25 ms\(^{-1}\), peak gust: 22.64 ms\(^{-1}\)), but progressively eased during the sampling period. The dominant wind direction for the whole period was from the north east, with 52.6% of the wind readings coming from this direction. Rainfall totalled 28 mm over the full 4-day period.

The tidal and weather conditions for March 2014 deployment are shown in Figure 5.2 and summarised in Table 5.3. Although an automatic weather station was installed at Tollesbury during this field campaign, the wind and rainfall data shown were sourced from the same MIDAS stations as mentioned above for consistency. Water levels during this period were lower than May 2012, averaging 1.90 m OD compared to 2.13 m OD. The tidal surges were greater, ranging between -0.216 m to 0.408 m, but unfortunately missing values from the Harwich tide gauge at the start of

Table 5.3: Tidal and weather statistics for March 2014

| Day  | Mean high water (m OD) | Mean tidal surge (m) * | Mean wind speed (ms\(^{-1}\)) | Peak wind gust (ms\(^{-1}\)) | Rainfall (mm) †
|------|------------------------|------------------------|-------------------------------|-----------------------------|---------------------
| Day 1| 1.92                   | -0.035                 | 5.56                          | 11.32                       | 0.0                 
| Day 2| 1.94                   | 0.161                  | 5.56                          | 10.29                       | 5.4                 
| Day 3| 1.85                   | -0.070                 | 3.89                          | 12.86                       | 2.2                 

* Tidal surge on Day 1 derived from values recorded between 09:15 – 11:25 on 25/03
† Rainfall records are for calendar days, i.e. 24/03, 25/03, 26/03, 27/03. Total rainfall on 27/03: 0.0 mm.
the monitoring period mean the average tidal surge for Day 1 is based on just 10 measurements compared to 100 for the remaining two days. No one wind direction was dominant, with 28 – 35% of readings originating from each of the north east, south west and north west directions; only 0.04% was from the south west. Peak wind speeds were fairly consistent, averaging 11.49 ms$^{-1}$; mean speeds were identical for Days 1 and 2.
and fell slightly in Day 3. Rainfall was low over the deployment, totalling just 7.6 mm. There was, however, a heavy hail shower during the collection of the sediment traps on Day 3. This may have impacted on the amount of sediment recorded on the filter papers (Voulgaris and Meyers, 2004a), but the shower was not prolonged and there were only a small number of traps still deployed at this time.

5.2 Environmental Controls on Sediment Deposition and Retention

5.2.1 Temporal and Spatial Variation in Sedimentation

The mass of sediment deposited on every trap deployed over the three days in May 2012 is shown in Figure 5.3. Two traps were laid per day at each of the 23 sampling points, five of which were located in the south natural marshes, the remainder were in the realignment site in the same locations as during the seasonal fieldwork campaign. The flux recorded on Day 3 is visibly higher than the other two days; a median flux of 7.34 g m$^{-2}$ tide$^{-1}$ was recorded during this time, compared to 4.44 g and 3.82 g on Days 1 and 2. This difference is confirmed by the results of the Kruskal-Wallis test. There is an overall difference between the three deployments: $H(2) = 24.76, p < 0.00001$, but the differences are only significant between Day 3 and 1 ($p = 0.007$) and Day 3 and 2 ($p < 0.001$).

In March 2014, sediment traps were deployed solely within the realignment site, at 36 sampling points across the three days. The fluxes recorded (shown in Figure 5.4)

![Figure 5.3: Daily variation in vertical sediment flux during the May 2012 campaign](image-url)
were significantly lower than during May 2012, with rates varying between 0.88 g to 13.3 g m\(^{-2}\) tide\(^{-1}\), compared to a maximum of 29.23 g m\(^{-2}\) tide\(^{-1}\) for the earlier period. Apart from the maximum rate of deposition recorded in Day 1, there was very little temporal variation in sediment flux during March. The medians were all very similar (2.66 g, 2.38g, 2.70 g m\(^{-2}\) tide\(^{-1}\)), as were the 25\(^{th}\) and 75\(^{th}\) percentiles (averaging 2.07 g and 3.42 g m\(^{-2}\) tide\(^{-1}\) respectively). The similarity between the three deployments is confirmed by the Kruskal-Wallis test: H(2) = 3.13, p = 0.209.

Sediment retention was also lower in March 2014 than May 2012, as shown in Figure 5.5. On average, 21.3% of the sediment deposited over the full three-day deployment in March 2014 was retained, whilst just 15.4% was retained in March 2014. This difference is statistically significant (H(2) = 12.12, p = 0.0005) and is likely related to the lower mass of sediment deposited at the marsh surface. The inclusion of the five sampling points in the natural marshes during 2012 has no significant effect on the average retention during this deployment; without these points the mean retention is 21.8%.

The spatial variation in sediment deposition recorded during May 2012 is shown in Figure 5.6. Deposition was much lower at the sampling points in the natural marshes (numbered 6-11) than in the realignment site (16-35); the two regions had median fluxes of 3.48 g and 6.99 g m\(^{-2}\) tide\(^{-1}\) respectively. Point 11 saw the highest average flux in the natural marshes, at 4.94 g m\(^{-2}\) tide\(^{-1}\); the lowest was at point 10 (2.85 g m\(^{-2}\) tide\(^{-1}\)).

Figure 5.4: Daily variation in vertical sediment flux during the May 2014 campaign

Figure 5.5: Sediment retention in March 2014 and May 2012

Figure 5.6: Spatial variation in sediment deposition during May 2012
Within the realignment site, the sampling points nearest the south and west sea walls (16, 21, 22, 29) recorded much lower rates of sediment deposition than the rest of the site, averaging at 2.60 g m$^{-2}$ tide$^{-1}$, compared to 7.97 g m$^{-2}$ tide$^{-1}$ for the other sampling points. This matches the pattern found during the seasonal campaign, and is attributed to the higher elevations in these areas and the longer distance from the breach, reducing
the inundation time and sediment supply. One of the main differences between this sampling period and the seasonal campaign is the higher than expected fluxes measured at sampling points 32, 33 and 34. These three points, which are closest to the breach and tend to experience strong tidal currents, received a mean rate of sediment deposition of 3.16 g m$^{-2}$ tide$^{-1}$ during the May 2011 field campaign, but one year later recorded an average of 7.05 g m$^{-2}$ tide$^{-1}$. This is despite the mean high water levels being lower in May 2012 (2.13 m compared to 2.65 m OD), which typically lead to lower rates of sedimentation.

As described in Section 2.4.1, in March 2014, the sampling strategy was adjusted to comprise nine transects within the realignment site, eight of which ran perpendicular to the south and west sea walls (Figure 5.7). Each of these transects included sampling points located within the vegetation near the sea walls (points 1, 5, 13 etc.), just within the vegetation on the marsh/mudflat boundary (2, 6, 14 etc.), roughly 2 m into the bare mud from this margin (3, 7, 15 etc.), and finally a sampling point approximately 15 m into the unvegetated area (points 4, 8, 16 etc.). The transect in the centre of the realignment site is arranged in a similar layout, with two of the sampling points within the vegetated area, and two in the mudflat. There is a clear difference in sediment flux between these sampling point types, with those points in the inner vegetated areas receiving significantly less sediment than elsewhere. The average flux at these points was 2.00 g m$^{-2}$ tide$^{-1}$; this compares to 2.63 g m$^{-2}$ tide$^{-1}$ at the border.

Figure 5.7: Spatial variation in average sediment flux, March 2014
vegetated points and 3.14 g m\(^{-2}\) tide\(^{-1}\) in the outer bare areas. The border bare sampling points (those within the mudflat at the vegetation boundary) were generally highest throughout the realignment site, averaging 3.84 g m\(^{-2}\) tide\(^{-1}\). The differences between sediment deposition at these four types of sampling point will be examined more thoroughly later in this chapter.

Figure 5.8 shows the spatial variation in sediment retention during the May 2012 campaign. Retention was slightly higher on average in the realignment site compared to the natural marshes, at 21.8% and 19.2% respectively. In the natural marshes, although sampling point 10 had the lowest average flux (2.85 g m\(^{-2}\) tide\(^{-1}\)), it actually retained the highest mass of sediment, 25.1%. Across both intensive monitoring periods, there is a very weak relationship between the two variables, with sediment flux accounting for only 0.05 – 0.5% of the variance in retention. This is evident within the realignment site too; as described above, sites 32, 33 and 34 received higher than average fluxes, but the retention here was very low, averaging just 13.4%.

There was less of an obvious difference between the four transect types in March 2014 with regards to sediment retention compared to deposition, as Figure 5.9 indicates. However, retention was again on average lower at the inner vegetated points than those vegetated points at the plant/mud border (14.0% compared to 16.5%). The exception to this is point 21, an inner vegetated point, which had the highest retention rate in the whole site of 32.5%. The highest average retention of the four transect types

![Figure 5.8: Spatial variation in sediment retention, May 2012](image-url)
(18.3%) was measured at the border bare sampling points. There are no apparent major differences between the nine transects.

Principal Components Analysis (PCA) was conducted on both sampling periods in order to explore which environmental factors influence the sedimentation processes. Figure 5.10(a) shows the PCA results for May 2012; those for March 2014 are displayed in Figure 5.10(b). The full details of the variables included are listed in Table 5.4. Certain variables are only available for the analysis undertaken for March 2014 due to the new sampling strategy employed for this field campaign. The percentage abundance of vegetation was not recorded during May 2012. Both PCAs confirm expected relationships between several environmental variables, such as the positive correlations between mean high water and surge, hydroperiod and water depth, and negative correlations between elevation and both bare sampling points and hydroperiod. Other relationships differ between the two campaigns: higher suspended sediment concentrations (SSC) are associated with unvegetated areas in May 2012, but there is no relationship between these variables in March 2014. The influence of wind direction is also different; in May 2012 wind from the NE and SE is negatively correlated with mean high water and tidal surge, whilst these directions are positively correlated with the tidal variables in March 2014. One final commonality is that no variables are closely associated with sediment flux in either period. There are weak positive relationships.
Figure 5.10: PCA results for May 2012 (a) and March 2014 (b)
Table 5.4: Description of variables included in PCA biplots

<table>
<thead>
<tr>
<th>Label</th>
<th>Full variable name and units</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flux</td>
<td>Sediment flux (g m(^{-2}) tide(^{-1}))</td>
</tr>
<tr>
<td>Retention</td>
<td>Sediment retention (%)</td>
</tr>
<tr>
<td>SSC</td>
<td>Suspended sediment concentration (mg l(^{-1}))</td>
</tr>
<tr>
<td>Bare</td>
<td>Indicates no vegetation at sampling point</td>
</tr>
<tr>
<td>Vegetated</td>
<td>Indicates a presence of vegetation</td>
</tr>
<tr>
<td>Bare %</td>
<td>Percentage of bare sediment</td>
</tr>
<tr>
<td>Vegetated %</td>
<td>Percentage abundance of vegetation</td>
</tr>
<tr>
<td>Inner vegetated</td>
<td>Indicates sampling point well within vegetated area</td>
</tr>
<tr>
<td>Border vegetated</td>
<td>Indicates sampling point within vegetation at vegetation / mudflat margin</td>
</tr>
<tr>
<td>Border bare</td>
<td>Indicates sampling point within mudflat at vegetation / mudflat margin</td>
</tr>
<tr>
<td>Outer bare</td>
<td>Indicates sampling point well within mudflat area</td>
</tr>
<tr>
<td>Elevation</td>
<td>Elevation at sampling point (m OD)</td>
</tr>
<tr>
<td>Mean HW</td>
<td>Mean High Water during deployment day (m)</td>
</tr>
<tr>
<td>Hydroperiod</td>
<td>Hydroperiod during deployment day (hours)</td>
</tr>
<tr>
<td>Water depth</td>
<td>Average water depth at high tide (m)</td>
</tr>
<tr>
<td>Tidal surge</td>
<td>Average tidal surge (m)</td>
</tr>
<tr>
<td>NE / SE / SW / NW wind</td>
<td>Median wind speed from relevant direction (m s(^{-1}))</td>
</tr>
<tr>
<td>Shear strength</td>
<td>Shear strength of surface sediment (kPa)</td>
</tr>
<tr>
<td>Distance</td>
<td>Distance to nearest creek / breach (for realignment sampling points) (m)</td>
</tr>
</tbody>
</table>

between deposition and water depth, hydroperiod, tidal surge and mean high water in May 2012, but the latter two variables have minimal control over flux in March 2014. The lack of influence from vegetation is apparent in March 2014, whilst higher fluxes are loosely related to the bare sampling points in May 2012. The observed pattern of higher fluxes at the border bare points compared to the outer bare points is confirmed by the PCA. Sediment retention is associated more closely with certain variables, but these are different between the two campaigns. In May 2012, higher retention is related to distance from the sediment source, elevation, shear strength, and the presence of vegetation; SSC has a negative influence on retention. In March 2014, the correlations are totally reversed, as higher retention is associated with water depth (the opposite of elevation), and all four variables related to a lack of vegetation. SSC has almost no influence.

As a way of gaining more understanding of the relationships between the environmental variables and sedimentation, multiple linear regression analysis was carried out on the two datasets. As with the seasonal campaign data, three different
models were used, one with all variables available, one where SSC was excluded, and one that only included the tidal variables. The list of the variables included in the models, as well as the coefficient of determination for each is shown in Table 5.5 (for sediment flux as the dependent variable) and Table 5.6 (for sediment retention). In both campaigns and for both sediment parameters, the model that performed the worst was that which only included the tidal variables, with these factors only accounting for 2.3%

Table 5.5: Results of multiple linear regression analysis with sediment flux as the dependent variable

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Model</th>
<th>Variables included</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2012</td>
<td>1: All</td>
<td>Distance, Elevation, Hydroperiod, SE wind, Shear strength, SSC, Tidal surge</td>
<td>0.392</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Distance, Elevation, Hydroperiod, NE wind, NW wind, SE wind, Shear strength</td>
<td>0.437</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Tidal surge, Water depth</td>
<td>0.325</td>
</tr>
</tbody>
</table>

| March 2014 | 1: All    | Bare %, Distance, Elevation, Hydroperiod, NW wind, SE wind, Shear strength, SSC, Vegetation % | 0.737 |
|            | 2: SSC removed | Bare %, Distance, Elevation, Hydroperiod, NW wind, SW wind, Shear strength, Vegetation % | 0.393 |
|            | 3: Tidal only | Hydroperiod, Mean HW, Tidal surge, Water depth          | 0.259 |

Table 5.6: Results of multiple linear regression analysis with sediment retention as the dependent variable

<table>
<thead>
<tr>
<th>Campaign</th>
<th>Model</th>
<th>Variables included</th>
<th>$r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2012</td>
<td>1: All</td>
<td>Distance, Elevation, Hydroperiod, NE wind, SE wind, Shear strength, SSC</td>
<td>0.271</td>
</tr>
<tr>
<td></td>
<td>2: SSC removed</td>
<td>Distance, Elevation, Hydroperiod, NW wind, SE wind, Shear strength, Tidal surge</td>
<td>0.229</td>
</tr>
<tr>
<td></td>
<td>3: Tidal only</td>
<td>Hydroperiod, Mean HW, Tidal surge, Water depth</td>
<td>0.054</td>
</tr>
</tbody>
</table>

| March 2014 | 1: All    | Bare %, Distance, Elevation, Hydroperiod, NW wind, SE wind, Shear strength, SSC, Vegetation % | 0.976 |
|            | 2: SSC removed | Bare %, Distance, Elevation, Hydroperiod, Mean HW, NW wind, SW wind, Shear strength, Vegetation % | 0.127 |
|            | 3: Tidal only | Hydroperiod, Mean HW, Water depth                      | 0.023 |
32.5% of the variability seen in sediment deposition and retention. The best performing model explaining the variability in sediment flux varied between the two fieldwork periods. In May 2012, the second model faired best, with an $r^2$ of 0.437. This is in contrast to the March 2014 campaign, and each of the previous seasonal periods, in which the model that included SSC achieved the best fit. The SSC model only uses a subset of the entire dataset, as water samples were not taken at every sampling point. In March 2014, the inclusion of these data resulted in 73.7% of the variability in flux being accounted for. The addition of SSC also significantly increased the ability of the model to predict sediment retention in March 2014: the $r^2$ value rose from 0.127 without these data to 0.976 when they were included. The difference was much lower in May 2012 (0.229 vs. 0.271). These results serve to indicate the complexity of the interaction between environmental processes and sedimentation. The following sections will briefly look at the relationships between some of the variables and sediment accumulation and retention individually in more detail.

5.2.2 Tidal Controls on Sedimentation

Although the multiple regression analysis indicated the tidal variables explained only a small proportion of the variance in sedimentation, tidal processes are generally acknowledged as important factors in controlling the rate of sediment deposition in saltmarsh systems. The relationship between the mean high water recorded for each daily deployment and sediment flux over both the intensive monitoring field campaigns is shown in Figure 5.11. The two campaigns are represented separately, but the equation

![Figure 5.11: Relationship between sediment flux and mean high water across both sampling periods](image)

$y = 10.229x - 16.135$

$r^2 = 0.788$  $p = 0.018$
of the regression line and the coefficient of determination refer to the combined dataset. When combined together, there is a strong relationship between the two variables, with mean high water accounting for 78.8% of the variation in sediment flux. Separately, however, the relationships are less strong. In May 2012, there is still a positive correlation, and the water level explains 68.8% of the flux. By contrast, in March 2014, there is essentially no relationship at all, with mean high water accounting for just 2.0% of the variance. The low variance of the water levels in this period (range is just 0.09 m) as well as the comparatively low level of the tidal heights may be reasons for this.

As sediment retention is calculated from the sum of the flux from the three separate daily sediment trap deployments compared to that measured from the traps left in situ over the full three-day period, there is only one measure of retention per sampling point per field campaign. Therefore there is only one average percentage of retention to compare against the mean high water level from each of the May 2012 and March 2014 campaigns. The higher water levels in May 2012 are associated with higher retention (equation of regression line: \( y = 0.042x + 1.247 \)), but no conclusions can be drawn from just two data points. In the seasonal fieldwork programme, higher water levels were related to lower rates of retention.

Figure 5.12 shows the effect of the tidal surge on sediment flux. With both intensive campaigns combined, this effect is minimal, as the regression line and \( r^2 \) value shown indicates. Examined separately, the relationship between surge and deposition during May 2012 and March 2014 is totally different. In May, there is an extremely strong positive correlation, with surge accounting for 99.4% of the variance in

![Figure 5.12: Relationship between sediment flux and average tidal surge across both sampling periods](image)
deposition. During March, although not immediately apparent from the plot, there is a moderate negative correlation, with an increasingly positive surge associated with lower fluxes. The coefficient of determination of the relationship is 0.549; the direction and strength of the correlation is comparable to that from the seasonal period, where the $r^2$ for most of the data (excluding one data point) was 0.506. The link between the two data points for mean surge and retention during the intensive deployments was also negative: $y = -226.93x + 19.979$; a similar gradient of the regression line was found during the seasonal period.

As described in the previous chapter, tidal surges are caused by forces influenced by atmospheric pressure and wind. Higher wind speeds increase wave energy, raising suspended sediment concentration and in turn, sediment deposition (Temmerman et al., 2003a). Although this positive correlation between wind speed and sediment deposition was found in the seasonal fieldwork programme, the relationship between the two variables was negative in Spring 2012 and 2014. As shown in Figure 5.13, both higher mean wind speeds and mean gusts are related to lower sediment fluxes with both deployments combined. This relationship is wholly determined by the wind and flux data from May 2012, however. During this deployment, there is a moderate association between flux and mean wind speed ($r^2 = 0.547$), and a strong link between flux and mean gust ($r^2 = 0.882$). There is essentially no relationship in March 2014 between sediment flux and either mean speed ($r^2 = 0.005$) or gust ($r^2 = 0.034$). The distinction between the two deployments may be caused by the different direction the winds originated. In May 2012, the dominant direction was the north east, whilst the

![Figure 5.13: Sediment flux compared against mean wind speed and mean gust during May 2012 and March 2014](image)
wind came from all directions in March 2014. Mean wind speed and mean gust appear to have had little influence over sediment retention; both deployments had very similar average speeds (5.13 ms\(^{-1}\) and 5.00 ms\(^{-1}\) for May and March), but the average retention was different (21.3% and 15.7% respectively). The average mean gusts for the two field campaigns were also similar (7.84 ms\(^{-1}\) and 8.09 ms\(^{-1}\)).

Tidal variations also influence the sediment load of the water column, with higher water levels generally resulting in an increase in suspended sediment concentrations (Temmerman et al., 2003b). This positive correlation was observed in the seasonal field campaign (\(r^2 = 0.751\)), but no such relationship exists during the intensive deployments (\(r^2 = 0.083\)). This may help to explain the lack of any relationship between SSC and sediment flux during May 2012 and March 2014, as indicated by Figure 5.14(a), which plots the two variables from those sampling points where water samples were collected. With both deployments combined, the coefficient of determination is 0.064; both periods are also consistent in the lack of any

![Plot of SSC vs Sediment Flux and Retention](image)

**Figure 5.14:** Relationships between SSC and sediment flux (a) and retention (b) during May 2012 and March 2014
relationship: \( r^2 = 0.001 \) in May 2012, and 0.009 in March 2014 when the anomalously high SSC value of 532.2 mg l\(^{-1}\) is excluded. A broad spectrum of sediment flux measurements were recorded across the range in SSC values. This was also the case with regards to sediment retention during both deployments, as Figure 5.14(b) shows. With both sampling periods combined, SSC accounts for just 3.2% of the variance in retention. This weak relationship is a positive one, but on an individual deployment basis, the associations are both negative and slightly higher: 21.9% in May 2012, 6.3% in March 2014.

As noted in Section 4.2.2, the other main component of the tidal cycle that typically influences sedimentation is the length of time the marsh surface is inundated, with longer hydroperiods providing more opportunity for a greater mass of sediment to be deposited (Voulgaris and Meyers, 2004b). The relationships between hydroperiod and sediment flux during May 2012 and March 2014, both together and separately, are shown in Figure 5.15. Combined, there is a fairly broad range of fluxes recorded across all the hydroperiod values, especially at inundation times between 4-6 hours. There is, however, an overall moderate positive relationship between the two variables, with increasing hydroperiod accounting for 34.9% of the rise in sediment deposition. The strength of this association is predominately driven by the data recorded in May 2012; on its own, the coefficient of determination between hydroperiod and flux during this period is 0.300, compared to just 0.077 in March 2014. Apart from three data points, the mean measured fluxes for each sampling point were all below 6 g m\(^{-2}\) tide\(^{-1}\) in March, and the range in deposition is very consistent throughout the spectrum of recorded hydroperiods. The March dataset also contains a number of data points with a hydroperiod of zero which is affecting the relationship.

To explore the relationship between hydroperiod and sediment deposition within the two intensive deployments further, the sampling points were grouped into four classes according to their location (May 2012 points) or transect type (March 2014). Figure 5.16 shows the layout of the classes for the two sampling periods that corresponds to the data plotted in Figure 5.15. In May 2012, the correlations between flux and hydroperiod differ significantly between the spatial classes, with the points in the natural marshes (Group A) and the outer points in the realignment site (Group B) exhibiting much stronger relationships than the other two classes. All of the \( r^2 \) values between the two variables for both intensive campaigns are listed in Table 5.7. On average, the coefficient of determination in Groups A and B is 0.527, indicating that hydroperiod does have a moderate influence over sediment deposition for these more
mature marsh areas. These results are significantly higher than that found during the seasonal field campaign, when the $r^2$ for these two classes was just 0.118. The influence of hydroperiod is lower in Groups C and D, as these sampling sites are more affected by other processes. The stronger tidal currents around the breach help to supress
Figure 5.16: Classes of sampling points relating to hydroperiod analysis: May 2012 (a) and March 2014 (b)

deposition, despite the greater length of inundation time. The sampling points in Group C are located near to minor tidal channels running through the realignment site, possibly delivering higher sediment loads than would be expected at these elevations. When the two high fluxes in the 18-22 g m⁻² tide⁻¹ range in this group are excluded, the relationship between flux and hydroperiod is strengthened considerably: the r² rises to
Table 5.7: Strength of relationships between sediment flux and hydroperiod for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th></th>
<th>May 2012 $r^2$</th>
<th>March 2014 $r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.300</td>
<td>All combined</td>
</tr>
<tr>
<td>A</td>
<td>0.518</td>
<td>Inner Vegetated</td>
</tr>
<tr>
<td>B</td>
<td>0.536</td>
<td>Border Vegetated</td>
</tr>
<tr>
<td>C</td>
<td>0.105</td>
<td>Border Bare</td>
</tr>
<tr>
<td>D</td>
<td>0.244</td>
<td>Outer Bare</td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive

0.503, making it more comparable with Groups A and B.

The March 2014 data were separated into the four transect types associated with their location in relation to the vegetation-mud border. None of these groups display any significant link between hydroperiod and flux. The highest coefficient of determination is 0.375 in the inner vegetated points; in the other classes variations in hydroperiod account for less than 6% of the variance in deposition (Table 5.7). In the unvegetated sampling points, higher hydroperiods are actually associated with minor reductions in the mass of sediment deposited.

The relationships between hydroperiod and sediment retention are totally different compared with flux across both sampling periods and all spatial classes, as shown in Figure 5.17 and summarised in Table 5.8. With all data combined, there is essentially no relationship between hydroperiod and retention, as the $r^2$ value indicates (0.032). In May 2012, Groups A and B contain no relationship between the two variables; in Group D, the relationship is moderately negative which would be expected

Table 5.8: Strength of relationships between sediment retention and hydroperiod for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th></th>
<th>May 2012 $r^2$</th>
<th>March 2014 $r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.034*</td>
<td>All combined</td>
</tr>
<tr>
<td>A</td>
<td>0.010</td>
<td>Inner Vegetated</td>
</tr>
<tr>
<td>B</td>
<td>0.002*</td>
<td>Border Vegetated</td>
</tr>
<tr>
<td>C</td>
<td>0.189</td>
<td>Border Bare</td>
</tr>
<tr>
<td>D</td>
<td>0.458*</td>
<td>Outer Bare</td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive
Figure 5.17: Relationships between sediment retention and hydroperiod during May 2012 and March 2014

considering the influence of the strong currents near the breach. There are only three data points in this class and Group C, however, so it is hard to make a conclusive judgement. In March 2014, hydroperiod has little influence over retention in the inner vegetated class and both of the unvegetated groups ($r^2 = 0.001-0.195$). In the border
vegetated class, however, there is a very strong positive correlation, with hydroperiod accounting for 91.9% of the variance in retention. Even excluding the high values from point 11 in the central vegetated area, the association is strong, with an $r^2$ of 0.823. As the border vegetated and border bare sampling points are very close to each other, with very similar elevations and hydroperiods, the only difference between these two groups is the presence or absence of vegetation. The influence of vegetation on sediment deposition and retention will be more fully explored later in this chapter.

5.2.3 Spatial Controls on Sedimentation

As indicated in the previous section, sediment deposition and retention and their potential controlling factors vary spatially both within and between tidal cycles. One of the main controls on the spatial variability of sediment accumulation is the distance from the closest source of sediment, with vertical flux declining with increasing distance from the nearest creek (Stumpf, 1983; French et al., 1995; Temmerman et al., 2005a). Figure 5.18 shows the relationships between the Euclidean distance from each sampling point to the nearest creek (for natural marsh points) or the centre of the sea wall breach (for points in the realignment site) and the mean sediment flux measured at that point. Table 5.9 lists the strength of the associations between the two variables for the two intensive monitoring periods; the spatial groupings in both the plots and the table are the same as those used in the hydroperiod analysis. With all the sampling points grouped together, there is a minimal relationship between flux and distance for both deployments separately, as well as combined, with the coefficients of determination ranging from 0.017–0.122. Even with the sampling points located within

Table 5.9: Strength of relationships between sediment flux and distance from sediment source for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th>May 2012</th>
<th>March 2014</th>
<th>All points $r^2$</th>
<th>Excluding points 9-12 $r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.017</td>
<td>All combined 0.122*</td>
<td>0.163*</td>
</tr>
<tr>
<td>A</td>
<td>0.183*</td>
<td>Inner Vegetated 0.861*</td>
<td>0.153*</td>
</tr>
<tr>
<td>B</td>
<td>0.183*</td>
<td>Border Vegetated 0.360*</td>
<td>0.555*</td>
</tr>
<tr>
<td>C</td>
<td>0.166</td>
<td>Border Bare 0.012</td>
<td>&lt;0.001</td>
</tr>
<tr>
<td>D</td>
<td>0.999</td>
<td>Outer Bare 0.039*</td>
<td>&lt;0.001*</td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive
the central vegetated area in March 2014 (points 9-12) removed, the negative relationship is only marginally strengthened.

Considering the two deployments separately, there is some variation in the correlations between the two variables across the spatial groupings. In May 2012,
Groups A and B exhibit minor negative relationships between flux and proximity to sediment source, with increasing distance accounting for 18.3% of the fall in flux in both groups (Table 5.9). In the natural marshes (Group A), this trend is strongly affected by sampling point 6, located 22 m from the nearest creek. With this point excluded, the strength of the relationship increases dramatically, with increasing distance now accounting for 95.1% of the variance in flux. Distance from the nearest creek is thus a good predictor for sediment deposition in the majority of the sampled natural marsh areas. Within the realignment site, distance is also strongly correlated with flux for those sampling points nearest the breach (Group D). For these points, however, the relationship is positive, with fluxes increasing further from the breach, and its associated turbulent currents. There is a minor positive link between distance and flux in Group C ($r^2 = 0.166$). Analysis of the relationship across different sized creeks was not undertaken for this data set due to the limited number of natural marsh sampling points, and the fact that the seasonal deployment results indicated the strongest association was found between the nearest creek, regardless of size.

In March 2014, the spatial group with the strongest negative link between flux and distance to sediment source is the inner vegetated sampling points, with an $r^2$ of 0.861. This trend is strongly influenced by the sampling point within the central vegetated area, however, which is considerably closer to the breach than the other points. With this point removed, the $r^2$ falls to just 0.153 (Table 5.9), indicating the distance from the breach is not an important influencing factor affecting flux for these points. The same pattern is seen for the border bare and outer bare points, where the removal of the points in the centre area reduces the relationship from a low level to almost zero. Considering just these four points on their own, there is a moderate positive trend ($r^2 = 0.323$), with the mass of sediment deposited generally rising alongside distance from the breach (the exception being point 11, the border vegetated point). The border vegetated points are the only group where removing the point from the centre area increases the strength of the relationship between distance and flux. For this set of sampling points, an increase in the distance from the breach accounts for 55.5% of the variance in deposition.

The relationships between proximity to sediment source and sediment retention for the two intensive deployments, both together and separately, are shown in Figure 5.19. Table 5.10 summarises the strengths of the associations. Overall, distance has no control over retention; with all the data combined it only accounts for 0.2% of the variance in the percentage of sediment retained. Its influence is only slightly higher
Figure 5.19: Relationships between sediment retention and distance from sediment source during May 2012 and March 2014 during May 2012 (7.7%) and March 2014 (0.6%) when the two deployments are examined individually. As the direction of the association is different between the two, no firm conclusions can be drawn. There is some indication of spatial variation in the relationship between the two variables. In May, the strongest correlation was found in
Table 5.10: Strength of relationships between sediment retention and distance from sediment source for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th>Group</th>
<th>May 2012 $r^2$</th>
<th>March 2014 $r^2$</th>
<th>Excluding points 9-12 $r^2$</th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.077</td>
<td>All combined</td>
<td>0.001*</td>
</tr>
<tr>
<td>A</td>
<td>0.115*</td>
<td>Inner Vegetated</td>
<td>0.001*</td>
</tr>
<tr>
<td>B</td>
<td>0.012*</td>
<td>Border Vegetated</td>
<td>0.247*</td>
</tr>
<tr>
<td>C</td>
<td>0.965</td>
<td>Border Bare</td>
<td>0.035</td>
</tr>
<tr>
<td>D</td>
<td>0.003</td>
<td>Outer Bare</td>
<td>0.037*</td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive

Group C (those near the minor channels in the realignment site), where increasing distance from the breach accounted for 96.5% of the variation in retention. The weakest relationship was in Group D, where it could be assumed the retention would increase in a similar manner to flux with increasing distance from the breach. This is clearly not the case, however. The lack of data for both of these groups means definitive conclusions hard to hard to reach. In March 2014, the border vegetated group of sampling points again stands out from the other classes, but for the majority of the transects (excluding that from the central vegetated area), there is still only a very weak relationship between distance from the breach and retention ($r^2 = 0.108$).

The elevation of the sediment surface is one of the other main factors controlling the spatial variation of sedimentation (French and Spencer, 1993; Cahoon and Reed, 1995). Linked to the hydroperiod, areas of lower elevation are submerged by tidal waters more often and for longer periods, thereby increasing the potential for deposition compared to regions of higher elevation (Temmerman et al., 2003b). Figure 5.20 shows the associations between mean sediment flux and measured elevation for the two intensive monitoring deployments. There is a moderate negative correlation between the two variables with all data combined, with rising elevation accounting for 25.2% of the reduction in flux. This relationship rises slightly for both May 2012 (28.7%) and March 2014 (26.6%) when examined separately. Although not high, these values are greater than the strength of the relationship from the seasonal campaign period, where elevation only accounted for 6.2% of the variance in flux.

Within each of the intensive deployments, there is significant variability in the relationship between elevation and flux across the spatial groupings used in the previous sections; these are listed in Table 5.11. In May 2012, the strongest links were within the
Figure 5.20: Relationships between sediment flux and elevation during May 2012 and March 2014

natural marsh sampling points (Group A) and those in the outer areas of the realignment site (Group B). Elevation is indicated to be a good predictor of sediment deposition in these areas, accounting for 66.4 – 72.4% of the variance in settling flux. Flux also falls with increasing elevation within those sampling points nearest the breach, but the
Table 5.11: Strength of relationships between sediment flux and elevation for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th></th>
<th>May 2012 r²</th>
<th>March 2014 r²</th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.287*</td>
<td>0.266*</td>
</tr>
<tr>
<td>A</td>
<td>0.724*</td>
<td>Inner Vegetated 0.852*</td>
</tr>
<tr>
<td>B</td>
<td>0.664*</td>
<td>Border Vegetated 0.114*</td>
</tr>
<tr>
<td>C</td>
<td>0.316</td>
<td>Border Bare 0.098</td>
</tr>
<tr>
<td>D</td>
<td>0.542*</td>
<td>Outer Bare 0.074</td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive.

strength of this relationship may be more due to the increasing distance from the breach associated with the points with higher elevations. Within Group C, there is slight positive correlation between the two variables, but this is influenced by the high deposition at point 28; excluding this point, there is a negative relationship between the other two points in this group. Due to the low number of data points, it is not possible to say if point 28 is an anomaly, or whether it is simply affected by the close proximity to a minor channel within the realignment site.

In March 2014, the relationship between sediment flux and elevation is strongly influenced by the presence of vegetation. Both the inner and border vegetated sampling points are negatively correlated with elevation, with elevation accounting for 85.2% of the variance in flux in the former group of points. Within the unvegetated points, however, there are minor positive correlations, with deposition increasing alongside rising elevation. This may be associated with falling turbulence associated with lower water depths at these higher points.

Elevation is generally less of an influencing factor over sediment retention than it is over deposition. There is essentially no relationship between elevation and retention within the intensive monitoring periods, except for within two of the spatial groupings. As Figure 5.21 and Table 5.12 show, when no spatial classes are considered, the regression line is almost flat when both deployments are considered together, and the strength of the relationship is hardly any better for the periods individually. The lack of influence is also evident across Groups A, B and C from the 2012 dataset, and within the inner vegetated and both of the unvegetated classes from 2014. The strongest relationship was found within the border vegetated group in March 2014, where an increase in elevation was strongly correlated with a fall in retention. It is not clear as to why there is such a strong relationship within these sampling points and no link within
the inner vegetation. In May 2012, those sampling points nearest the breach (Group D) exhibit a moderate positive association with retention and elevation, again more likely attributed to the associated increase in distance from the breach site. A positive relationship was also found when all the unvegetated sampling points from May 2012
Table 5.12: Strength of relationships between sediment retention and elevation for grouped sampling points during intensive fieldwork periods

<table>
<thead>
<tr>
<th></th>
<th>May 2012</th>
<th></th>
<th>March 2014</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>All combined</td>
<td>0.053</td>
<td>All combined</td>
<td>0.023*</td>
<td></td>
</tr>
<tr>
<td>A</td>
<td>0.017*</td>
<td>Inner Vegetated</td>
<td>0.001*</td>
<td></td>
</tr>
<tr>
<td>B</td>
<td>0.017</td>
<td>Border Vegetated</td>
<td>0.866*</td>
<td></td>
</tr>
<tr>
<td>C</td>
<td>0.128*</td>
<td>Border Bare</td>
<td>0.177</td>
<td></td>
</tr>
<tr>
<td>D</td>
<td>0.436</td>
<td>Outer Bare</td>
<td>0.082*</td>
<td></td>
</tr>
</tbody>
</table>

* Indicates a negative correlation; all other relationships are positive

were collated, with increasing elevation accounting for 35.9% of the variance in retention. This compares with just 1.7% of the variance in retention in the vegetated points.

5.3 Influence of Vegetation

Analysis of the data obtained during the seasonal field campaign (Section 4.4) indicated that the role of vegetation in marsh sedimentation processes appeared to be less important than would be expected from the conclusions asserted by some researchers (e.g. Reed et al., 1999; Möller et al., 2001; Möller, 2003; Stammermann and Piasecki, 2012; Möller et al., 2014). This finding also appears to be evident in the data collected during the May 2012 and March 2014 intensive monitoring deployments, as indicated by the principal components analysis. This last fieldwork programme was designed solely to test the influence of vegetation of sediment deposition and retention, with half of the sampling points located within areas of vegetation, and half within bare mudflat. This section presents the analysis undertaken, primarily on the data collected during this period, to more fully examine the role the presence, abundance and composition of vegetation plays in sedimentation within the Tollesbury marshes.

5.3.1 Presence of Vegetation

Figure 5.22 shows the comparison of sediment flux recorded on every trap deployed at the vegetated sampling points with those at the unvegetated points during the May 2012 and March 2014 field campaign periods. In both deployments, deposition appears to be marginally higher at the bare points; this is confirmed by examining the descriptive statistics for both datasets (Table 5.13). Across both periods, all of the values are higher.
Figure 5.22: Comparison of sediment flux recorded at vegetated and bare sampling points during May 2012 and March 2014 deployments

Table 5.13: Descriptive statistics for sediment flux (g m$^{-2}$ tide$^{-1}$) recorded at vegetated and bare sampling points during intensive monitoring deployments

<table>
<thead>
<tr>
<th>Statistic</th>
<th>May 2012</th>
<th>March 2014</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Vegetated</td>
<td>Bare</td>
</tr>
<tr>
<td>Mean</td>
<td>5.07</td>
<td>6.54</td>
</tr>
<tr>
<td>Median</td>
<td>3.30</td>
<td>5.30</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.90</td>
<td>1.76</td>
</tr>
<tr>
<td>Maximum</td>
<td>29.23</td>
<td>18.86</td>
</tr>
<tr>
<td>25$^{th}$ Percentile</td>
<td>2.20</td>
<td>3.82</td>
</tr>
<tr>
<td>75$^{th}$ Percentile</td>
<td>6.18</td>
<td>8.19</td>
</tr>
<tr>
<td>Data count</td>
<td>117</td>
<td>62</td>
</tr>
</tbody>
</table>
at the unvegetated points, except for the maximum flux recorded during May 2012, which is greater in the vegetated points. On average, both the arithmetic mean and median fluxes are approximately 1.5 times greater in the bare points. These differences are statistically significant, as indicated by the results of the Kruskal-Wallis test: \( H(1) = 13.47, p = 0.0002 \) (May 2012); \( H(1) = 39.74, p < 0.00001 \) (May 2014). These results thereby appear to indicate that the presence of vegetation actually has a detrimental effect on sediment deposition. Although not common, this finding has been documented by a small number of researchers, such as Temmerman et al. (2007), who discovered that, although vegetation reduced erosion on a local scale, lower sedimentation rates and increased erosion were found between vegetation patches. Feagin et al. (2009) also found vegetation increased erosion over unvegetated sediments in laboratory studies.

As discussed in Section 1.4.1, it has been proposed by some that instead of increasing deposition within saltmarshes, halophytic vegetation serves to promote the retention of sediment by reducing resuspension and stabilising the sediment surface (Scoffin, 1970; Ward et al., 1984; Brown, 1998; Shi et al., 2000). Sediment retention was marginally higher within the vegetated sampling points during the entire seasonal fieldwork programme, and the same is true for the May 2012 deployment (Figure 5.23). The mean retention was 5% greater in the vegetated points during this time, and all of the statistics were higher in these points compared to those from the bare mudflat areas (Table 5.14). The difference between the two sets of sampling points is not statistically significant, however: \( H(1) = 2.4, p = 0.1213 \). The presence of vegetation was less influential over retention in March 2014; in fact both the mean and median rates of retention were slightly higher in the unvegetated areas. The maximum percentage of sediment retention was recorded within a vegetated point, but all other statistics were either identical to, or marginally lower than those measured in the bare mudflat areas. As expected, the Kruskal-Wallis test indicates there is no statistical difference between the two data groups: \( H(1) = 0.08, p = 0.776 \). The observed data is thus inconclusive regarding the role of vegetation in retaining sediment within tidal saltmarshes.

Although suspended sediment concentration was shown to have almost no relationship with either sediment deposition or retention for both intensive deployments, it will now be explored as to whether there is any difference in SSC between vegetated and unvegetated sampling points which could explain the differences seen in flux and retention at these two groups of points. Figure 5.24 shows the comparison of SSC between the two classes, and it is clear that the trends in the data vary considerably between the two deployments. In May 2012, SSC was, on average, much greater within
Figure 5.23: Comparison of sediment retention at vegetated and bare sampling points during May 2012 and March 2014 deployments

Table 5.14: Descriptive statistics for sediment retention (%) recorded at vegetated and bare sampling points during intensive monitoring deployments

<table>
<thead>
<tr>
<th>Statistic</th>
<th>May 2012</th>
<th>March 2014</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>23.0</td>
<td>15.2</td>
</tr>
<tr>
<td>Median</td>
<td>23.4</td>
<td>13.1</td>
</tr>
<tr>
<td>Minimum</td>
<td>14.1</td>
<td>5.8</td>
</tr>
<tr>
<td>Maximum</td>
<td>41.0</td>
<td>32.5</td>
</tr>
<tr>
<td>25&lt;sup&gt;th&lt;/sup&gt; Percentile</td>
<td>18.3</td>
<td>11.4</td>
</tr>
<tr>
<td>75&lt;sup&gt;th&lt;/sup&gt; Percentile</td>
<td>26.0</td>
<td>18.2</td>
</tr>
<tr>
<td>Data count</td>
<td>15</td>
<td>18</td>
</tr>
</tbody>
</table>
Figure 5.24: Comparison of suspended sediment concentration recorded at vegetated and bare sampling points during May 2012 and March 2014 deployments

the bare sampling points than the vegetated areas. The median SSC in the bare points was 127.5 mg l\(^{-1}\), compared to 74.3 mg l\(^{-1}\) in the vegetated points, whilst the arithmetic mean was 2.3 times greater in areas without vegetation. The difference between the two classes is statistically significant: \(H(1) = 5.75, p = 0.0165\). During March 2014, however, the volume of sediment within the water column was greater within the vegetated sampling points. The descriptive statistics are all higher in this group, although the difference between the two classes is not as pronounced as that from the May 2012 deployment. The median SSC for the vegetated points in 2014 was 52.2 mg l\(^{-1}\), compared to 41.6 mg l\(^{-1}\) in the bare points. This difference is also significant: \(H(1) = \)
4.63, p = 0.0314. These results, because of their opposing nature, show that regardless of the volume of sediment found within the water column, the presence of vegetation appears to have a negative influence on sediment flux. Focussing on the March 2014 dataset, the higher SSC in the vegetated points negates the argument that because the bare points are generally closer to the breach in the sea wall (and thus the source of sediment), due of the layout of the transects, it would automatically be expected that they recorded higher levels of sediment deposition. The effect of vegetation on trapping efficiency, a ratio of the flux deposited to the volume of sediment available, will be investigated at the end of this chapter.

5.3.2 Vegetation Gradient

As detailed in Section 2.4.1, the sampling strategy of the March 2014 field campaign was arranged around nine transects within the Tollesbury realignment site to examine whether the influence of halophytic vegetation varied along the vegetation gradient. Eight transects started around the west and southern sea walls. Each of these contained one sampling point within the mature vegetation fronting the sea wall, termed the inner vegetated point; one within the younger, pioneer plants at the vegetation – mudflat boundary termed the border vegetated point; one in the bare mud at this boundary (border bare), and one 10-20 m into the mud (outer bare). The final transect centred around part of the marsh boundary within the vegetated area in the centre of the realignment site; the same four types of sampling point were also used here.

The total vegetation abundance of each sampling point is shown in Figure 5.25 according to which of the four transect classes in which it is located. Vegetation cover was generally higher at the inner vegetated points than the border vegetated, with the median cover recorded as 75% and 55% respectively. Both classes had one sampling point where vegetation abundance was 100%, but the minimum cover was 10% higher in the inner vegetated areas. The difference between the two vegetated points is not statistically significant. One point in the border bare group (in the central vegetated area) did have a 2% cover of *Salicornia* stems; no halophytic vegetation was present at any of the outer bare points. Any patches of algae have not been included in the abundance totals.

The mass of sediment deposited on every trap deployed during March 2014 at each of the four types of sampling point is shown in Figure 5.26. It has already been found that settling flux is greater within the unvegetated mudflat, but this plot shows
Figure 5.25: Variation in total vegetation abundance across the vegetation gradient, March 2014

how flux varies between the two types of vegetated and two types of bare sampling point across the transect gradient. Although vegetation cover was greater in the inner vegetated points, sediment flux was lower here than within the border vegetated areas. Both the mean and median were approximately 1.4 times greater at the vegetation border; all of the other descriptive statistics were also higher within this transect class (Table 5.15). Due to the low tidal levels within the March 2014 deployment, the

Figure 5.26: Variation in sediment deposition across the vegetation gradient during March 2014
Table 5.15: Descriptive statistics for sediment flux g m\(^{-2}\) tide\(^{-1}\) recorded across the vegetation gradient

<table>
<thead>
<tr>
<th>Statistic</th>
<th>Inner Vegetated</th>
<th>Border Vegetated</th>
<th>Border Bare</th>
<th>Outer Bare</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>1.68</td>
<td>2.31</td>
<td>3.37</td>
<td>2.66</td>
</tr>
<tr>
<td>Median</td>
<td>1.75</td>
<td>2.28</td>
<td>3.18</td>
<td>2.46</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.34</td>
<td>0.60</td>
<td>0.98</td>
<td>0.88</td>
</tr>
<tr>
<td>Maximum</td>
<td>4.34</td>
<td>5.16</td>
<td>13.30</td>
<td>9.81</td>
</tr>
<tr>
<td>25(^{th}) Percentile</td>
<td>0.95</td>
<td>1.70</td>
<td>2.37</td>
<td>1.54</td>
</tr>
<tr>
<td>75(^{th}) Percentile</td>
<td>2.10</td>
<td>2.89</td>
<td>3.78</td>
<td>3.36</td>
</tr>
<tr>
<td>Data count</td>
<td>72</td>
<td>72</td>
<td>72</td>
<td>71</td>
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</tbody>
</table>

Table 5.15 lists the probability values for the multiple Kruskal-Wallis tests performed on vegetation gradient sediment flux data; the overall result on the full dataset indicated a strong statistical difference between the groups: H(3) = 62.38, p < 0.00001. Every group is different from one another, except for the border vegetated and outer bare classes. The strongest differences are between the inner vegetated points and both of the unvegetated groups, this confirms the typically low fluxes recorded at the

hydroperiod and water depth were very low within the inner vegetated areas; this could help to account for the low rates of sediment deposition at these points. At the transect within the central vegetated area, the median flux was marginally higher at the inner vegetated point than at the border point (3.24 g vs. 3.19 g m\(^{-2}\) tide\(^{-1}\)). Within the bare points, flux was again generally higher nearer the vegetation border; in five out of the eight transects around the sea wall, the mean flux was greater at these points than the outer bare areas. This is despite the fact that the elevation of these border points is slightly higher, and they are located 15 m further away from the breach site than those in the outer mud. Across the realignment site, the mass of sediment deposited was 30% higher within the border bare region.

Table 5.16 lists the probability values for the multiple Kruskal-Wallis tests performed between each vegetation group for the flux data; the overall result on the full dataset indicated a strong statistical difference between the groups: H(3) = 62.38, p < 0.00001. Every group is different from one another, except for the border vegetated and outer bare classes. The strongest differences are between the inner vegetated points and both of the unvegetated groups, this confirms the typically low fluxes recorded at the

Table 5.16: Probability results from Kruskal-Wallis tests performed on vegetation gradient sediment flux data

<table>
<thead>
<tr>
<th></th>
<th>Inner Vegetated</th>
<th>Border Vegetated</th>
<th>Border Bare</th>
<th>Outer Bare</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inner Vegetated</td>
<td></td>
<td>0.0009</td>
<td>&lt;0.0001</td>
<td>&lt;0.0001</td>
</tr>
<tr>
<td>Border Vegetated</td>
<td>0.0003</td>
<td></td>
<td>0.6828</td>
<td></td>
</tr>
<tr>
<td>Border Bare</td>
<td>0.0200</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Highlighted values indicate the probability has exceeded the 95% confidence interval
mature marsh areas. The differences are significant to the 99% confidence interval between the two vegetated groups, the two border groups and are significant to 95% between the two bare groups.

Sediment retention is also greatest around the vegetation border compared to those sampling points further away, regardless of the presence of vegetation (Figure 5.27). The median retention is highest within the border bare points (18.1%), followed by the border vegetated points (15.7%). The arithmetic mean in the inner vegetated area is skewed by the maximum retention (the highest across the four groups – 32.5%), but both this area and the outer bare regions have very similar median rates of retention (12.6% and 13.0% respectively). The proximity to the vegetation border thus appears to be more influential than the presence of vegetation itself with regards to the mass of sediment retained, but none of the differences between the four groups are statistically significant: H(3) = 4.96, p = 0.174.

The previous section (5.3.1) showed that suspended sediment concentrations were higher within the vegetated sampling points than the bare areas but, as Figure 5.28 indicates, there is a difference between SSC within the two main vegetation/bare categories. The average SSC of the border vegetated samples is over double that recorded within the inner vegetated area (121.7 mg l\(^{-1}\) vs. 55.7 mg l\(^{-1}\)), but this statistic is heavily influenced by the two anomalously samples, both of which were measured at sampling point 23 on Day 1 of the deployment. The medians, by comparison are much

![Figure 5.27: Variation in sediment retention across the vegetation gradient during March 2014](image-url)
closer, with SSC still greater within the border vegetated group (62.1 mg l\(^{-1}\) compared to 40.4 mg l\(^{-1}\)). Despite the previous finding when both categories of sampling point are combined, the SSC within both vegetated classes is actually lower than that found within the border bare group, which has a median of 65.2 mg l\(^{-1}\). The lowest volume of sediment in the water column is found in the outer bare areas (median = 34.0 mg l\(^{-1}\)), which indicates that SSC actually increases as the proximity to the vegetation border narrows. This can only be attributed to some re-working of the mud surface that resuspends sediment back into the water column just in front of the vegetation margin.

5.3.3 Vegetation Abundance

The presence of vegetation appears to have a negative influence over sediment deposition, with greater settling fluxes recorded within unvegetated sampling points. This section investigates whether the abundance of vegetation had a similar negative effect during the March 2014 sampling period. Vegetation properties were not measured during May 2012, so the data presented here is solely from the more recent deployment.

The relationships between total vegetation cover and sediment flux are shown in Figure 5.29, for the two vegetated classes combined, as well as individually. With all the data together, there is essentially no correlation with abundance and flux, as indicated by the low coefficient of determination (0.029). This lack of relationship is predominately driven by the two sampling points with 100% vegetation cover.
Although these two points have high sediment fluxes compared to the other points (averaging 3.06 g m$^{-2}$ tide$^{-1}$), this positive effect of increased vegetation cover is actually an anomaly within the dataset. Excluding these points, the relationship strengthens negatively, with increasing abundance accounting for 52.5% of the corresponding reduction in deposition. In the inner vegetated areas, there appears at first to be a minor positive correlation between cover and flux. This is also, however, influenced by the point with 100% cover (point 12, in the central vegetated area). Excluding this, the correlation is negative, with an $r^2$ of 0.493. The other sampling point in the central area (point 11) is the data point in the border vegetated class with 100% cover; although the relationship between abundance and flux is negative with its inclusion it is weakening the overall trend. Excluding this data, increasing vegetation cover actually accounts for 73.4% of the reduction in sediment deposition. The strength of this negative correlation is surprising, and is completely in contrast to the majority of
marsh sedimentation studies (Dean, 1978; Knutson et al., 1982; Bouma et al., 2007; Wang et al., 2008).

The influence of vegetation cover on sediment retention is less controversial. There is a very minimal positive relationship ($r^2 = 0.064$) between the two variables within all of the vegetated sampling points; the range in retention data is too broad across the spectrum of cover for any meaningful correlation to be realised, as shown in Figure 5.30. In contrast to the flux data, the two sampling points with 100% cover do not affect the relationship as significantly; excluding them reduces the coefficient of determination to 0.011. In the inner vegetated marsh, the range of retention measurements is very narrow, with all but one of the data points falling within 5-15%. The influence of vegetation abundance is almost non-existent, and reduces even further when the anomalously high retention (32.5% at sampling point 21) is removed: $r^2 = 0.011$. Total cover appears to be moderately more influential in the border bare areas ($r^2 = 0.233$), but the 100% cover point is skewing the trend. Without it, the relationship is

Figure 5.30: Relationships between sediment retention and vegetation abundance within vegetated sampling points
similarly poor as before, and in fact converts to slightly negative ($r^2 = 0.006$).

5.3.4 Species Composition

Despite the ever-increasing number of papers concerning the ability of halophytic vegetation to attenuate wave energy within coastal saltmarshes, only a handful of studies have compared this property between different plant species (Bouma et al., 2005b; Koch et al., 2006, 2009; Bouma et al., 2010). The results from the seasonal fieldwork programme were inconclusive regarding the influence of different species over sediment deposition and retention as associations between plant types and sedimentation varied over time.

The species composition across the vegetation gradient sampled during the March 2014 intensive monitoring deployment is shown in Figure 5.31. The list of species found is given Table 5.17; dead *Salicornia* stems from the previous summer season and new shoots were recorded separately to determine if there was any difference in their ability to trap sediment. Both these forms of *Salicornia* were found within the inner and border vegetated sampling points; the occurrence of new seedlings was more common (found in 5 of the 18 sampling points), but the abundance of the dead stems was greater (25% on average where found, compared to 10% for seedlings). The most commonly occurring and dominant species in the vegetated marsh was *Spartina*, which was observed at all points in both the inner and border vegetated points. The median abundance was identical for both categories (50%), but the mean was marginally greater in the inner points (49.4% vs. 46.7%) due to the higher maximum cover recorded. The inner points showed greater species diversity, with *Aster* and

<table>
<thead>
<tr>
<th>Label</th>
<th>Full species name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salicornia seedlings</td>
<td><em>Salicornia spp.</em>, primarily <em>perennis</em>. Observed as either small new shoots or decaying stems from previous growing season</td>
</tr>
<tr>
<td>Salicornia dead stems</td>
<td></td>
</tr>
<tr>
<td>Spartina</td>
<td><em>Spartina spp.</em>, primarily <em>anglica</em></td>
</tr>
<tr>
<td>Aster</td>
<td><em>Aster tripolium</em></td>
</tr>
<tr>
<td>Puccinellia</td>
<td><em>Puccinellia maritima</em></td>
</tr>
<tr>
<td>Enteromorpha</td>
<td><em>Enteromorpha spp.</em>, primarily <em>prolifera</em></td>
</tr>
<tr>
<td>Ulva</td>
<td><em>Ulva spp.</em></td>
</tr>
</tbody>
</table>

Table 5.17: List of species recorded during March 2014 deployment
Figure 5.31: Variation in species composition across the vegetation gradient
Puccinellia present here, neither of which were observed in the points sampled around the vegetation border. Both the species Enteromorpha and Ulva were present as algal mats at a small number of the vegetated points (2 and 4 points respectively), whilst only the latter was observed at the unvegetated sampling points. Ulva was identified at four points in both the border and outer bare areas (ranging from 10-75% cover where present), whilst a 2% cover of dead Salicornia stems was found at one of the border bare points.

Canonical Correspondence Analysis (CCA) was carried out to investigate the relationships between the species composition and the environmental variables recorded during the March 2014 deployment. Figure 5.32 shows the CCA triplot produced when all sampling points with vegetation (including those in the bare mudflat with Ulva and Salicornia) are included. Figure 5.33 presents the CCA results for data from only the inner vegetated (a) and border vegetated points (b). Certain variables have been excluded (e.g. mean high water and wind speeds) as they did not contribute to explaining the variance in the species data. What is apparent from all three plots is that there is no consistent relationship between either sediment flux or retention and any of the plant species. With all the data combined, there is a moderate positive link between

![Figure 5.32: CCA triplot for all vegetated points](image)

Includes all inner and border vegetated points, and bare points where either Ulva or Salicornia are present.
Figure 5.33: CCA results for inner vegetated points (a) and border vegetated points (b)
flux and *Ulva*, but this is mainly an indicator that flux is greater in the bare sampling points (Figure 5.32). The strongest relationship appears to be between flux and *Spartina*, with its presence negatively associated with sediment deposition. In the inner vegetated points *Ulva* has no influence over flux, but has a negative effect in the border points. The role of *Spartina* seems to vary between slightly negative in the inner points to slightly positive around the vegetation border (Figure 5.33). In the inner marsh areas, the species with the strongest positive link with flux are both forms of *Salicornia* and the presence of *Enteromorpha*, however, these either have a strong negative effect (*Salicornia*) or almost no influence at all (*Enteromorpha*) within the border vegetated points.

The picture is equally mixed with regards to sediment retention. In the inner vegetated areas, *Ulva* is most closely associated with higher rates of retention, but has essentially no influence at the vegetation border or when the data from the bare sampling points is also included. In the border area, *Enteromorpha* is indicated as having the greatest positive influence, but has no effect at all in the inner marsh points. *Spartina* plays only a small role in controlling retention, but varies between positive in the inner points, and negative in the border area. Overall, therefore, species composition appears to have no conclusive influence over either sediment deposition or retention within the Tollesbury realignment site.

5.3.5 Trapping Efficiency

All of the analysis undertaken thus far has explored the variability of and factors controlling sediment deposition and retention without reference to the actual supply of sediment available to the marsh system. Although the influence of suspended sediment concentration was investigated, only a simple linear regression was carried out, without linking sediment supply to the hydroperiod or water depth at each sampling point. Very few field studies have compared how much of the sediment delivered on the tide is actually deposited on the marsh surface over short-term timescales. The limited number of papers that do consider the efficiency at which sediment is trapped in saltmarshes tend to either compare the suspended sediment concentration on the flood tide with that on the ebb tide or determine the sediment budget for the marsh system as a whole (e.g. Stumpf, 1983; Stevenson *et al.*, 1988; French, 1993; Van Proosdij *et al.*, 2006b). As such, they do not shed any light on the spatial variation of trapping efficiency within the marsh and any factors influencing it. The exception to this is the work carried out by
Moskalski and Sommerfield (2012), who investigated the trapping efficiency in a saltmarsh system in Delaware, USA. They developed formulas for calculating trapping efficiency for every sampling point where sediment deposition and SSC measurements were available. Their formulas have been amended and applied to the data collected for this study from the May 2012 and March 2014 deployments in the Tollesbury realignment site. Trapping efficiency is calculated as follows:

\[
\text{Efficiency} = \frac{\text{Flux}}{\text{Inventory}}
\]  

(5.1)

where Flux is the average mass of sediment recorded on the filter paper sediment traps and Inventory is the mass of sediment available in the water column. Both are converted to kg m\(^{-2}\) per day. The inventory was calculated for each point by multiplying the SSC from the siphon samplers by the water depth at high water at that point:

\[
\text{Inventory} = \text{SSC} \times h_{\text{HW}}
\]  

(5.2)

SSC was converted to kg m\(^{-3}\) and \(h_{\text{HW}}\) is the total water depth from the two high waters per deployment day (in metres), taken by comparing the water level from the pressure sensor installed in the realignment site with the elevation for each sampling point. The efficiency ratio was then converted to a percentage and calculated for each point for each of the three days for both deployment periods. Efficiencies less than 100% indicate a proportion of the sediment available was not deposited on the marsh surface, flowing out on the ebb tide or possibly adhering to the leaves and stems of plants (Li and Yang, 2009); values greater than 100% indicate more sediment was deposited than theoretically available in the water column.

It is hypothesised that the presence and abundance of vegetation is likely to influence the efficiency at which sediment is trapped, due to its wave-attenuating properties. However, based on the prior analysis indicating the lack of control over sedimentation, it is unclear as to how efficiency will vary between the vegetated and unvegetated sampling points. Figure 5.34 shows the comparison between the two classes from the data measured in the two intensive monitoring campaign periods. It is immediately apparent that the influence of vegetation varies totally between the two deployments, as the presence of vegetation is linked to greater efficiencies than the bare points in May 2012, but related to lower efficiencies in March 2014.

During both deployments, sediment deposition was marginally higher in the bare sampling points, but in May 2012 the vegetated points were much more efficient at
trapping what sediment was available in the water column. Efficiency was on average four times higher where vegetation was present, as 19.2% of the suspended sediment was deposited compared to 4.7% in the bare areas. This difference is statistically significant: $H(1) = 11.01, p = 0.0009$. In May 2014, efficiencies were typically higher, and more of the available sediment was trapped in the mudflat areas (averaging 35.6% vs. 23.8% in vegetated points). These values compare with 45% trapping efficiency in Chesapeake Bay, USA, (Jordan et al., 1986), 54% in Norfolk (French, 1989) and 80% in Delaware (Stumpf, 1983), all of which were calculated using the sediment budget approach. In contrast, Moskalski and Sommerfield (2012) found that up to five times
more sediment was deposited on the marsh surface than was available in the water column, but it was unclear as to where this additional sediment could have originated.

The sampling strategy of the March 2014 field campaign allows the variation in trapping efficiency to be examined across the vegetation gradient, from the inner vegetated points through to those in the outer bare areas; this is shown in Figure 5.35. Median efficiency increases consistently throughout the gradient, from 16.6% in the inner most vegetation, 22.9% in the border vegetated area, to 24.2% in the border bare points and 31.5% in the outer bare mudflat. The mean efficiencies are all higher, but are heavily influenced by the maximum values, especially within the outer mud points. The overall Kruskal-Wallis test indicates none of the differences are significant: $H(3) = 4.61, p = 0.2023$. The trend in efficiency varies slightly compared to both flux and SSC. For these two variables, values were greater in the border bare points, but trapping efficiency is highest at the outer bare areas.

There are a few possible reasons why the trapping efficiency was higher during March 2014, and why it appears to be greater within the unvegetated mudflat than the vegetated sampling points. Firstly, SSC was considerably lower during March 2014 (53.4% lower), reducing the inventory available to be deposited. Flux was also lower during this time, but the comparison between the earlier period was less stark. This thereby increases the efficiency of the trapping process. Water levels were likewise reduced in March 2014 (0.23 m lower); French (2006) showed that trapping efficiency
was higher in marshes with lower tidal range. Although this finding was determined between micro- and meso-tidal systems compared to macro-tidal systems, it can be assumed a similar difference exists over tidal variations in the same marsh.

The other main factor affecting the above results is the location of the sampling points used for the trapping efficiency analysis. Figure 5.36 shows the points used from the May 2012 (a) and March 2014 (b) deployments; only one point is in the same

Figure 5.36: Spatial variability in trapping efficiency: May 2012 (a), March 2014 (b)
location: point 27 in 2012 is the same as point 12 in 2014. Although the scale used in the two maps is different, the trapping efficiency at this location is very similar in both periods (19.0% in May 2012, 16.0% in March 2014). Excluding this one point, it is thus unreasonable to compare the two datasets with each other. The low efficiency of the bare points from the 2012 deployment is attributed to their location near the breach site, and its associated strong currents. Only points 32 and 34 are sited in unvegetated locations, no other bare points are available for comparison. In comparison, SSC was sampled in more points in the mudflat areas in 2014 than within vegetation. Although an equal number was planned for this deployment, only three points in the vegetated areas could be used due to the low water levels not filling up the siphon samplers. Looking at the efficiency within the transects themselves, there is little variation between the vegetated and unvegetated points. In the transect in the central vegetated area, efficiency ranged between 19.0% – 22.6%, and between 30.2% – 31.7% in the western transect (points 6, 7, 8). The exception is between points 23 and 24, but the high efficiency at the latter (averaging 73.7%) is likely an anomaly or due to localised influences. It is thus likely that vegetation has very little control over trapping efficiency and other factors, such as distance to sediment source, are more important. A greater number of water samples spread across the realignment site would be required before firm conclusions can be made.

5.4 Summary

The results from the two intensive monitoring periods presented in this chapter confirm the complexity of sedimentation processes operating in the Tollesbury marshes. In a similar manner to the seasonal field campaign, the relationships between environmental factors and sediment deposition and net retention vary between deployments. The conclusions drawn depend how the data is separated and analysed, with patterns varying between the natural marshes and the realignment site, vegetated and unvegetated sampling points, and spatial groups based on distance from the nearest sediment source. The data presented does, however, add further weight to the smaller than expected role vegetation plays in controlling sedimentation. The presence and abundance of vegetation has no influence, and at times is indicated to have a negative effect, on settling flux and retention, and there is no evidence to support its hypothesised contribution to trapping efficiency. Table 5.18 provides a summary of the strength of the theorised and measured relationships between sediment flux and retention and the variables studied. The strength of the measured relationship refers to that of the
combined dataset from both the May 2012 and March 2014 field campaigns. Table 5.19 explains the indicators used. The next chapter uses numerical modelling to explore further the role of vegetation in the hydrodynamics and sediment transport within the Tollesbury realignment site.

Table 5.18: Summary of theorised and measured relationships between sediment flux and retention and the variables investigated in Chapter 5

<table>
<thead>
<tr>
<th>Theorised Relationship</th>
<th>Measured Relationship</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flux</td>
<td>Retention</td>
</tr>
<tr>
<td>Mean HW</td>
<td>↑↑</td>
</tr>
<tr>
<td>Tidal surge</td>
<td>↑</td>
</tr>
<tr>
<td>Wind speed</td>
<td>↑</td>
</tr>
<tr>
<td>SSC</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Hydroperiod</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Creek distance</td>
<td>↓↓</td>
</tr>
<tr>
<td>Elevation</td>
<td>↓↓</td>
</tr>
<tr>
<td>Vegetated</td>
<td>↑↑↑</td>
</tr>
<tr>
<td>Vegetated %</td>
<td>↑↑↑</td>
</tr>
</tbody>
</table>

* Qualitative estimate of strength of relationship
– Strength of relationship unknown. For measured relationships, this is due to a lack of data

Table 5.19: Explanation of Indicators used in Table 5.18, with associated qualitative descriptors and r² values

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Descriptor *</th>
<th>r² value *</th>
</tr>
</thead>
<tbody>
<tr>
<td>↓↓↓↓</td>
<td>Very strong negative</td>
<td>64 – 100%</td>
</tr>
<tr>
<td>↓↓</td>
<td>Strong negative</td>
<td>36 – 64%</td>
</tr>
<tr>
<td>↓</td>
<td>Moderate negative</td>
<td>16 – 36%</td>
</tr>
<tr>
<td>↓↑</td>
<td>Weak negative</td>
<td>4 – 16%</td>
</tr>
<tr>
<td>↑↑↑↑</td>
<td>Very weak negative</td>
<td>0 – 4%</td>
</tr>
<tr>
<td>↑↑↑↓</td>
<td>Very weak positive</td>
<td>0 – 4%</td>
</tr>
<tr>
<td>↑↑</td>
<td>Weak positive</td>
<td>4 – 16%</td>
</tr>
<tr>
<td>↑↑</td>
<td>Moderate positive</td>
<td>16 – 36%</td>
</tr>
<tr>
<td>↑↑↑</td>
<td>Strong positive</td>
<td>36 – 64%</td>
</tr>
<tr>
<td>↑↑↑↑</td>
<td>Very strong positive</td>
<td>64 – 100%</td>
</tr>
</tbody>
</table>

* Descriptors and r² values derived from Evans (1996)
6. Hydrodynamic Modelling in the Tollesbury Realignment Site

The results obtained from both the seasonal and intensive field campaigns at the Tollesbury managed realignment site are in agreement with a growing body of research (e.g. Neumeier and Ciavola, 2004; Mudd et al., 2010; Gedan et al., 2011) that suggests the influence of vegetation on sediment deposition is less significant than previously assumed. However, the patterns of vertical settling flux and sediment retention and their relationship between vegetation presence, abundance and composition vary spatially and temporally, so it is difficult to determine conclusively from field measurements alone that vegetation plays only a secondary role in the accumulation process at Tollesbury. By their very nature, field campaigns can only last for short periods and cover a limited spatial extent at low resolution. These unresolved questions can be followed up by undertaking physically-based hydrodynamic and sedimentation modelling. Numerical models can provide vital insights into the morphodynamics of mudflats and saltmarshes over larger areas and at higher spatial resolutions than can be achieved through field studies alone (Lakhan, 2003; Allen, 2009).

This chapter presents the results of a series of model runs undertaken with a 2D hydrodynamic model (Telemac; Hervouet, 2000a). Various simulations have been undertaken to determine if the presence and increasing roughness of vegetation alters flow patterns in the realignment site, which would be expected to influence sedimentation. Following an introduction to the Telemac software suite and its selection for this study, the full details of the modules used and their fundamental equations are given. Details of model implementation and validation are then presented, followed by the key hydrodynamic model results.

6.1 Modelling Approach

Numerical models are increasingly being used as tools with which to study the processes in coastal and estuarine environments, including hydrodynamics, surface wave processes, sediment transport and bed evolution (Brush and Harris, 2010; Ganju et al., 2016). Models are primarily used for three purposes: to increase understanding of either a real, physical system or a logical system; to predict the future of a state that is currently unknown; or to control, constrain or manipulate a system to produce a desirable condition (Haefner, 2005). Numerical solutions to estuarine hydrodynamics
have existed since the 1960s, when pioneers such as Hansen and Rattray (1965) wrote a set of partial differential equations to describe tidally-driven estuarine circulation. Since then, the complexity of hydrodynamic models has increased significantly alongside the advances in computational capacity such that a range of two- and three-dimensional models can resolve wetting and drying, wave-current interaction and sediment transport over complex bathymetries at high spatial and temporal resolutions (Ganju et al., 2016).

Despite the increasing capabilities of models and the advances in computing power over the last 10–20 years, there remain only a few studies (e.g. Arega and Sanders, 2004; D’Alpaos et al., 2006, 2007) that have modelled hydrodynamics and sediment transport within saltmarsh and mudflat systems. Saltmarshes are some of the most challenging environments to model successfully at high spatial resolutions due to their topographical complexity (Pye and French, 1993d). This is caused by the extensive presence of shallow, narrow and anastomosing channel networks which also serve to generate highly nonlinear tidal constituents (Blanton et al., 2002; Robins and Davies, 2011). Creating a detailed model of an existing saltmarsh system which combines processes on the marsh surface, in the tidal channels as well as the fronting mudflats requires knowledge of a wide variety of physical, chemical and biological processes and their interactions (Stammermann and Piasecki, 2012). The general approach taken in this study is to follow that of Temmerman et al. (2005b), who used a numerical model to study the impact of vegetation on current velocities in a tidal marsh over single tidal cycles.

6.1.1 The Telemac Modelling System

There is a range of numerical model packages available that can be used to study both the hydrodynamics and sedimentation processes with saltmarsh environments. Most models are comprised of a suite of modules, which can be run either individually, or dynamically coupled to allow for the effect of bed evolution on water circulation to be evaluated (Hostache et al., 2014). Temmerman et al. (2005b) used the Delft3D software (Lesser et al., 2004; Delft Hydraulics, 2007), a two- or three-dimensional, finite difference scheme. Other widely used three-dimensional sediment transport models include the Community Sediment Transport Model (CSTM), which is embedded in the Regional Ocean Model System (ROMS) (Warner et al., 2008); the Estuarine Coastal Ocean Model with Sediment Transport (ECOMSED) (Blumberg, 2002); and the quasi 3D MIKE21 suite, comprising the MIKE21 MT mud transport module and the MIKE21
HD flow module (DHI, 2006, 2007). The discretisation method employed by both CSTM-ROMS and ECOMSED is a finite difference approach, whereby the mesh nodes at which the model equations are solved during each time step are within a structured grid (Robins, 2009; Amoudry and Souza, 2011). By contrast, MIKE21 employs a finite volume approach, which provides much greater geometric flexibility during the mesh generation to better fit complex, irregular coastal boundaries as the mesh grid can be modified locally throughout the model domain (Chen et al., 2007). The open-source Telemac modelling system was initially developed by the Laboratoire National d’Hydraulique at Electricité de France (Hervouet, 2000a, 2007) and has been widely used to model coastal and estuarine hydrodynamic problems (e.g. French, 2008; Marks and Bates, 2000; Fox and Aldridge, 2000; Spearman, 2011). Telemac uses either a finite difference or finite element formulation on an unstructured triangular mesh and is well suited to 2D and 3D problems that involve complexity morphologies and/or wetting and drying (Robins and Davies, 2011).

Open-source software is always preferable for research applications and the Telemac system (version 7.1) was chosen here on account of its proven performance for intertidal wetting/drying problems, and excellent computational efficiency when run in parallel on multiple CPUs (Hervouet, 2000b). The Telemac suite includes both two- and three-dimensional flow modules (Telemac-2D and -3D), a spectral wave propagation module (Tomawac) and a sediment transport model (Sisyphe) (Peltier et al., 1994; Hervouet, 1999, 2000a). Each module can be run individually; Sisyphe can also be internally coupled to the main flow model (Villaret et al., 2013). Telemac has been used in a variety of situations and for different purposes, including computing wind-induced circulation in the Irish Sea (Jones and Davies, 2006), wave propagation off the East Anglia coast (Dawson et al., 2009), river floodplain flow (Marks and Bates, 2000) and within a variety of coastal and estuarine settings throughout the UK and beyond (e.g. Fox and Aldridge, 2000; Fernandes et al., 2002; Brière et al., 2007; French, 2008; Robins, 2009; Verschoren et al., 2015). Sisyphe has been used as part of the Telemac system in a range of riverine, estuarine and saltmarsh applications (e.g. Brown and Davies, 2009; Robins et al., 2011; Robins and Davies, 2011; Luo et al., 2013; Villaret et al., 2013; Hostache et al., 2014). Telemac has already been used to run simulations at Tollesbury; Spearman (2011) developed a model to predict the evolution of the realignment site over longer-term timescales (80 years). The present study differs in that the modelling is undertaken within the site over much shorter, tidal timescales.
6.1.2 Model Formulation

The Telemac-2D depth-averaged flow module is used here to simulate a range of hydraulic variables, including water depth and fluid velocity across the Tollesbury realignment site. The following paragraphs present the main mathematical features of the model that are relevant for this study. Further details are available in the Telemac-2D user manual (Lang et al., 2014).

Telemac solves the non-conservative Saint-Venant equations (conservation of mass and momentum, Equations 6.1–6.3) at every node on the triangular mesh.

Continuity:

\[
\frac{\partial h}{\partial t} + \vec{u} \cdot \text{grad}(h) + h \text{div}(\vec{u}) = 0 \tag{6.1}
\]

Momentum:

\[
\frac{\partial u}{\partial t} + \vec{u} \cdot \text{grad}(u) = -g \frac{\partial Z}{\partial x} + F_x + \frac{1}{h} \text{div}(hv_t \text{grad} u) \tag{6.2}
\]

\[
\frac{\partial v}{\partial t} + \vec{u} \cdot \text{grad}(v) = -g \frac{\partial Z}{\partial y} + F_y + \frac{1}{h} \text{div}(hv_t \text{grad} v) \tag{6.3}
\]

where \( u \) and \( v \) are flow velocities in the \( x \) and \( y \) directions, \( h \) is the water depth, \( t \) is time, \( Z \) is water surface elevation, \( F_x \) and \( F_y \) are source terms to represent bottom friction, \( v_t \) is eddy viscosity, and \( g \) is gravitational acceleration. The \( F_x \) and \( F_y \) source terms in the momentum equations represent the force induced by bottom friction in the \( x \) and \( y \) directions.

In the model simulations presented here, friction is parameterised with a Manning friction coefficient, \( n \). Different \( n \) values were used for the unvegetated mudflats and for the areas of saltmarsh. This is discussed in more detail in sections 6.1.3 and 6.2. For turbulence, an Elder-type turbulence model was used (Elder, 1959; Fischer et al., 1979), whereby eddy viscosities in the streamwise and span-wise dimensions, respectively, are given by Equations 6.4 and 6.5:

\[
K_1 = \beta_1 u_* h \tag{6.4}
\]

\[
K_t = \beta_t u_* h \tag{6.5}
\]
where $\beta_1$ and $\beta_t$ are dimensionless dispersion coefficients and $u^*$ is the friction velocity. This scheme allows spatial and temporal variation in eddy viscosity to be incorporated simply and with little computational penalty.

Surface wave processes can be modelled using the Tomawac module. However, surface wave simulations are omitted here as there was very little wind-wave activity in the realignment site during the study period. Therefore all simulations presented hereafter do not take into account the effect of surface waves.

Telemac is able to model sediment transport and deposition, using the Sisyphe module, but due to technical difficulties and time constraints, it has not been possible to implement this part of the model at this stage.

6.1.3 Model Implementation

The first step in implementing Telemac to simulate the hydrodynamics and sediment transport within the Tollesbury realignment site was to create a bathymetry file of the model domain. This was primarily based on the 2013 LiDAR product obtained from the Environment Agency (1 m resolution, $\pm$5 cm quoted vertical and horizontal accuracy; see Section 2.3.2 for full details). All heights below -0.35 m OD were removed as it was apparent the LiDAR instrument was capturing standing water at this level. Additional data for elevations in the main channel outside the realignment site was obtained from a bathymetric survey carried out in November 2001 by HR Wallingford (Spearman and Semmence, 2002). For the final bathymetry, however, the area outside the breach was lowered to ensure the model boundary was deep enough never to dry out. In addition, the bed levels of the creek system within the realignment site were lowered as well as widened, as they were not fully captured by the LiDAR. Clearly defined channels are needed to provide flow paths for the flood and ebb tidal currents. The final bathymetry used in the model simulations is shown in Figure 6.1.

The model mesh was generated using the Surface Water Modelling System (SMS), marketed by Aquaveo. This system allows a higher degree of control over the spatial variation in mesh properties than is offered by most other software, making it especially suited to modelling saltmarsh environments (French, 2003). The boundary of the model domain, the major areas of vegetation, and all main creeks in the realignment site were digitised in SMS using the 2013 aerial photograph obtained from the EA (Figure 3.1(e)). The spacing of the node points along the boundary lines was varied in size according to different regions in the study area, with the breach area and creeks
Elevations were primarily derived from 2013 LiDAR product obtained from EA. Tidal creeks and area outside the breach were lowered and widened. Having a much finer resolution than the vegetated areas nearest the sea walls. The adaptive tessellation method was used to fill the modal domain with a triangular mesh. Figure 6.2 shows the computational mesh generated, with (a) showing the complete mesh, and (b), (c) and (d) highlighting the finer resolutions used around the breach area and the creek network compared to the marsh surface. Throughout the domain, the mesh comprises 54,913 triangular elements and 27,691 nodes. The nodes closest to the locations of the deployed instruments monitoring water level and velocity (used for the validation process and discussed in the next section) are also shown. The bathymetry data were interpolated onto the mesh nodes using a linear inverse distance weighting approach.

In order to test the effect of vegetation on the hydrodynamics within the realignment site, a set of material types were created within the SMS system, and each node was assigned a material. Three material types were produced: saltmarsh, mudflat and one to represent the channel network and breach area. A simplified outline of the main areas of vegetation was obtained using the 2013 aerial photograph; the resulting material type map is shown in Figure 6.3. As previously mentioned, the presence of...
Figure 6.2: Computational mesh generated for the Tollesbury realignment site
The full domain is shown in (a); (b) and (c) show finer detail around the breach area and the creek to the west of the domain. The junction of the main creek entering the site with the major east-west creek is shown in (d), highlighting the finer mesh used here compared to the marsh surface. Marked points show the location of the closest nodes to the deployed instruments.
Figure 6.3: Material type zones used in model simulations
Main saltmarsh areas where sourced from the 2013 aerial photograph obtained from the EA. Different friction coefficients (Manning’s n) were assigned to each zone to simulate the presence or absence of vegetation.

vegetation is being simulated through a higher friction coefficient (Manning’s n) than that given to the fronting mudflats and the channels. Therefore each material zone was assigned a different coefficient value; the details of the values chosen for each model scenario are given in the coming sections (6.2.1 and 6.2.3).

The model is run using four main input files. The steering file contains the configuration of the computation, with the various parameters and settings chosen (e.g. turbulence model, number of time steps) listed within it. The geometry file contains the mesh and elevation data, whilst the final two files are related to the boundary conditions. The model boundary of the simulations run here is the pre-defined edge of the channel entering the realignment site through the breach in the sea wall (highlighted in Figure 6.4). Water can enter and leave the model domain through this boundary. The liquid boundary file drives the model simulations, and contains the observed water levels monitored using a self-recording pressure sensor deployed in the outer Tollesbury Fleet. Water levels were recorded every 5 minutes, and were interpolated to 60-second intervals before being used to drive the model. The location in the Fleet (Figure 6.4) was chosen to ensure the water level dataset did not suffer from drying out at the lowest
tides, which would have caused errors in the model. Although the location of the pressure sensor is 2.4 km from the model boundary, no corrections were applied to the data as the recorded levels were in very good agreement with those recorded by the ADCP deployed just outside the domain boundary (0.014 m average difference between the two datasets). The water levels within the liquid boundary file used to run the model are shown in Figure 6.5. The complete dataset covers six weeks of tidal cycles, beginning at 16:00 on 23/03/14 and running until 15:00 on 06/05/14.

Figure 6.5: Observed water levels within Tollesbury Fleet used to drive model runs
Dates refer to day and month in 2014. Data commences at 16:00 on 23/03/14, ending at 15:00 on 06/05/14. Time at marked dates is 12:00.
6.1.4 Validation and Sensitivity Analysis

Before any scenarios could be undertaken to investigate the influence of vegetation on hydrodynamics, the model first had to be validated against observed datasets. As described in Chapter 2, various instruments were deployed within the realignment site to provide observations of water levels and flow velocity. The locations of the nodes closest to the instruments are shown in Figure 6.2. An Acoustic Doppler Current Profiler (ADCP; RDI 1200 KHz Workhorse Sentinel) was placed just outside the breach in the main Tollesbury Creek (22.0 m from the domain boundary). This monitored both water levels and velocity, but due to the distance from the domain, only the water levels were compared with model results. Water velocities are more easily affected by differences in water depth and channel width than water level measurements. Water levels were also recorded by self-recording pressure sensors (referred to as divers, after the make and model of the instrument: Schlumberger Mini-Diver) installed in the Main Creek and the West Creek (Figure 6.2 (b) and (c)). Velocity measurements were obtained from two flow meters (Valeport model 105) deployed at the breach in the sea wall and in the West Creek.

The model was validated by adjusting the Manning’s n friction coefficient and comparing the simulated water levels and velocities with the observed instrument data. For the purpose of this process a constant friction was applied to the entire model domain. Nine simulations were run, with the Manning’s n values varying between 0.01 and 0.05. The validation period comprised the first 48 hours of the liquid boundary file, from 16:00 on 23/03/14 to 15:55 on 25/03/14. As well as visually comparing the modelled and observed datasets, a quantitative method of evaluating the model performance was required. The Nash-Sutcliffe Efficiency (NSE) (Nash and Sutcliffe, 1970) was used for this purpose and is calculated as follows:

\[
NSE = 1 - \frac{\text{sum}(Q_{obs} - Q_{mod})^2}{\text{sum}(Q_{obs} - \text{mean } Q_{obs})^2}
\]

(6.6)

where \(Q_{obs}\) is the observed data and \(Q_{mod}\) is the modelled data. Possible values for the NSE range from \(-\infty\) to 1. Efficiencies equal to or less than 0 indicate the model is predicting no better than using the average of the observed data, whilst an NSE value of 1 means a perfect fit between the two datasets.
The model proved straightforward to validate with regards to water levels, and a Manning’s n value of 0.035 was adopted throughout the model domain. This value proved to be a sensible compromise of the three separate comparisons. Figure 6.6 shows the comparison of the observed and modelled water levels for this friction value at the

Figure 6.6: Comparison of modelled and observed tidal water levels at the three instrument stations
The time commences at 16:00 on 23/03/14. These simulations were run with a constant friction coefficient (Manning’s n) of 0.035.
three instrument locations where this variable was monitored: the ADCP, the Main Creek and the West Creek (see Figure 6.2 for locations). The modelled levels at the ADCP and Main Creek match the observed data very well, with NSE values of 0.999 and 0.997 respectively. This indicates the model replicates the levels here almost perfectly. The comparison is slightly less favourable at the West Creek as there is a small lag in the modelled water levels on both the flood and ebb tide. Despite this, the magnitude of the high water is well matched. This lag is likely caused by the effect of the constant friction applied throughout the model. In reality, the marsh surrounding this channel is densely vegetated and this exerts a control on water levels and velocity (French and Stoddart, 1992). The model result is still valid, however, as the NSE value is 0.929.

The comparison of modelled velocity with the observed data from the flow meters was less simple. As Figure 6.7 shows, the two datasets are not well matched, with considerable differences throughout the validation period. This dissimilarity is

![Figure 6.7: Comparison of modelled and observed flow velocities](image)
The time commences at 16:00 on 23/03/14. These simulations were run with a constant friction coefficient (Manning’s n) of 0.035.
more likely attributable to the quality of the data recorded by the flow meters than the model results, however. The Valeport flow meters used in the instrument deployment can suffer from fouling which impacts on the movement of the impeller; this may be responsible for the drop-outs seen in the observed data, e.g. at 10 hours at the Breach site. The adjustments to the bathymetry during the model implementation are also impacting on the comparisons. The channels throughout the realignment site were widened and deepened in order for them to be fully represented in the model. Increasing the dimensions of the channels results in lower velocities; this is visible when comparing the modelled and observed flow at the West Creek. Although the comparisons result in poor efficiency values (-1.108 and 0.434) the model appears to be generating more plausible speeds than the observed data. The model is overall a valid representation of the realignment site.

A simple visual check was also undertaken to confirm the water levels and velocities were reasonably simulated across the rest of the model domain. Figure 6.8 shows the maximum water levels modelled during the validation period. Simulated water levels appear to be sensible throughout the site, with all the main features represented, e.g. greater depths in the main channels, and shallow depths nearer the sea walls. The minor creeks branching off the main channels are also visible. There are no

Figure 6.8: Maximum simulated water level (m OD) across the model domain during the validation period
obvious anomalies or areas of hydrodynamic instability generated by the model. Maximum velocities simulated during the validation period are shown in Figure 6.9 for both the flood (a) and ebb tide (b). Again, there are no apparent errors, and sensible

Figure 6.9: Maximum simulated velocity (ms\(^{-1}\)) during the validation period on the flood tide (a) and ebb tide (b)
speeds have been calculated both within the channel network and on the marsh surface. A comparison of the velocities between the flood and ebb tide shows the main channels are ebb dominated, with greater speeds at this time than during the flood tide, especially within the main channel leading from the breach site and on its immediate banks. These areas are known to be suffering from scouring and erosion, so this ebb-domination is expected. In contrast, most of the mudflat and vegetated marsh areas are flood dominated, helping to aid deposition on these surfaces.

As part of the validation process, a simple sensitivity analysis was conducted to determine how sensitive modelled water level and velocity are to changes in the constant friction factor applied to the model domain. A series of simulations were run, each with a different Manning’s n value, ranging from 0.010 to 0.050 in increments of 0.005. The observed variables were then compared with the modelled results for each simulation and the Nash-Sutcliffe Efficiency calculated for each instrument location as before. Figure 6.10 shows the NSE values for each simulation plotted against the

![Figure 6.10: Sensitivity of modelled water levels and flow velocities to variations in constant bottom friction](image-url)
friction coefficient applied for both the water level data and the flow velocities. It is clear from both graphs that neither variable is sensitive to changes in bottom friction. With regards to the water levels, at the ADCP location the NSE value is constant throughout all the simulations at 0.9990, whilst the values at the Main Creek only range from 0.9927 to 0.9973. As previously noted, the efficiency is overall lower at the West Creek location, and the modelled water levels are slightly more sensitive here too, ranging between 0.9218 and 0.9378. Across the three locations, however, the sensitivity is very low.

Although the comparison of the modelled and observed flow velocities has already been shown to be a poor measure of the model’s performance, it is still worth examining the sensitivity of this variable to changes in friction. Compared to the water level, velocity is generally more sensitive to varying levels of friction. This is not unexpected, as a number of studies have shown the impact of vegetation (and thus friction) on water velocities and wave energy within saltmarshes (French and Stoddart, 1992; Möller et al., 1999; Möller and Spencer, 2002). The NSE values for the Breach site varied between -1.2270 and -1.0278 (a difference of 0.199), whilst those for the West Creek indicated velocity was less sensitive, ranging between 0.4160 and 0.4449, a difference of just 0.0289.

6.2 Hydrodynamic Model Results

6.2.1 Effect of Vegetation on Flow Velocity

As discussed in Chapter 1, it has been widely observed that the presence of saltmarsh vegetation can significantly increase attenuation of wave energy compared to unvegetated mudflats (Knutson et al., 1982; Brampton, 1992; Yang et al., 2008; Möller et al., 2014). Attenuation is increased by the higher bed roughness found within saltmarshes, a result of the greater topographic complexities as well as vegetation geometry (Dijkema, 1987; Pethick, 1992; Möller, 2006). In addition to the presence of vegetation, variations in plant height, density and complexity of the structure all affect drag on water passing through the vegetation canopy (Leonard and Croft, 2006). For the present modelling study, vegetation is simulated by the application of a higher friction factor compared with the unvegetated mudflat areas in the realignment site. In order to test the influence of vegetation on flow velocity, a series of simulations were run with a range of friction coefficients applied to the saltmarsh zones as defined in Figure 6.3. The first scenario had a constant friction across the entire domain, using a Manning’s n
of 0.035. Five further scenarios were run, in which a different friction value was applied to the vegetated areas. The Manning’s n values applied to these areas ranged from 0.040 to 0.120, as detailed in Table 6.1. The scenarios were run over two different time periods taken from the liquid boundary file. As water depth is a major factor controlling velocity, with higher velocities occurring during greater depths (Möller and Spencer, 2002), the two time periods comprised a low flow period and a high flow period. The tidal conditions during these periods are shown in Figure 6.11. The average high water level during each period was 1.67 m OD and 2.61 m respectively, whilst the maximum high water level was 2.03 m and 2.68 m. A total of 12 model scenarios were thus run to investigate the effect of vegetation on flow velocity. In order to compare each scenario, the maximum velocity on the flood tide was extracted from the two flow periods that were simulated.

Table 6.1: Model scenarios used to test effect of vegetation on flow velocity

<table>
<thead>
<tr>
<th>Scenario</th>
<th>Friction coefficients (Manning’s n)</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>Mudflat</td>
</tr>
<tr>
<td>Constant</td>
<td>0.035</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.035</td>
</tr>
<tr>
<td>Zoned, n = 0.06</td>
<td>0.035</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.035</td>
</tr>
<tr>
<td>Zoned, n = 0.10</td>
<td>0.035</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.035</td>
</tr>
</tbody>
</table>

Figure 6.11: Tidal conditions during low and high flow test periods
Water levels as recorded at the boundary diver in Tollesbury Fleet. Average high waters: 1.67 m OD and 2.61 m; Maximum high waters: 2.03 m and 2.68 m.
As an initial exploration of the influence of vegetation, Figure 6.12 shows the maximum flood velocity across the realignment site from the constant bottom friction scenario (a) and the highest zoned friction scenario (n = 0.12) (b) during the low flow period. Flow velocity is expressed in units of metres per second (ms⁻¹). Both simulations capture the main features that would be expected, primarily the highest velocities in the main Tollesbury Creek channel outside the breach area, which decrease slightly throughout the creek network within the realignment site. From the creek edges, the velocity generally decreases rapidly in the form of sheet flow across the marsh surface as the roughness of the bed reduces the tidal energy. The presence of the minor channels within the marsh, such as those within and to the north of the central vegetated area near the breach and those that branch off from the major west-east creek can be seen as areas of greater flow than the surrounding marsh. In this low flow period, the areas closest to the sea walls in the west and south of the site are not inundated by water, hence the very low velocities found here. Although the results of the two scenarios presented here are fairly similar in form, there are some subtle differences. The most obvious is the appearance of what can be described as ‘ripples’ in the high friction scenario (Figure 6.12 (b)), which seem to emanate from the breach and fan out in a circular fashion. These ripples indicate the velocity is falling and rising, whereas in the same areas in the constant friction scenario (Figure 6.12 (a)), the velocity is higher overall.

The same rippling pattern is seen in the high friction scenario (n = 0.12) during the high water period (Figure 6.13 (b)) and is absent from the constant friction scenario (a). It is not clear as to what is causing this repeating surge effect, as it is present in both the bare mudflat and the vegetated marsh areas. As would be expected, water speeds are higher throughout the realignment site during this period, and a similar pattern of falling velocity throughout the channel system and across the marsh surfaces can be seen. One difference between this period and low flow simulation is the greater reduction in velocity in the west-east channel. Speeds are proportionately much lower at the far extremities of this channel compared to that at the breach in the high flow period than in the low flow period. The higher water levels across the site in this latter period have resulted in water penetrating a much greater area of the realignment site, with water within the vegetated areas close to the sea walls. Although the velocity within these marsh regions does appear to be lower in the higher friction scenario than that with no vegetation, the difference is not as large as might be expected. Velocities are also lower within the channel network in the higher friction scenario.
Figure 6.12: Maximum flood velocity (ms$^{-1}$) from the constant friction scenario (a) and highest zoned friction scenario ($n = 0.12$) (b) during the low flow period.
Figure 6.13: Maximum flood velocity (ms$^{-1}$) from the constant friction scenario (a) and highest zoned friction scenario ($n = 0.12$) (b) during the high flow period.
Although the broad patterns in velocity across the model domain can be seen using these visualisations, it is not easy to determine the specific influence of vegetation. A series of transects were thus created throughout the site which span across the mudflat / marsh interface. Figure 6.14 shows the locations of these transects; each one starts at the edge of a creek and traverses the marsh surface. Data from each scenario were extracted along each transect at 10 m intervals. Transect 1 is characterised by a fairly long stretch of bare mudflat (134 m) before continuing as vegetation to the southern sea wall. Transect 2 contains a much shorter mudflat area (33 m), followed by a vegetated segment (105 m in length) before returning to mudflat. Transects 3 and 4 are characterised by vegetated areas directly adjacent to the creek, then a region of mudflat before vegetation is present again. Finally, transect 5 is fully vegetated along the length of the transect; this is used to compare the difference between a totally unvegetated area (using the constant friction scenario) and the increasingly rougher vegetation scenarios.

The maximum flood velocity at each sampling point along transect 1 for all the scenario runs during both the low flow and high flow periods are shown in Figure 6.15. What is initially clear is that the presence of vegetation in the realignment site, regardless of its friction factor, has an impact on the velocity in the mudflat areas,
Figure 6.15: Maximum flood velocities across transect 1 for all scenario runs for both low flow and high flow periods

despite these having no vegetation themselves. If the vegetation had no influence in the bare areas, the velocity transects would be identical until they traversed the vegetation margin; that is not the case in any of the forthcoming plots. During both flow periods, the reduction in velocity across transect 1 is not a smooth, gradual process, but is instead marked by a number of intermittent peaks and troughs; these can be seen at 80 m, 110 m, 130/140 m and 170 m (in the high flow period) from the creek edge. Within the mudflat section of both transects, velocities are generally highest in the lowest zoned scenario ($n = 0.04$) and lowest in the highest zoned scenario ($n = 0.12$) with the other scenarios ranged in order between them, as seen between $0 – 70$ m and $100 – 120$ m. The constant scenario most closely follows the trajectory of the zoned, $n = 0.06$ scenario.

At the vegetation boundary, the comparison between the scenarios is more complex. In the low flow period, the velocity in both the constant friction and the lowest
zoned friction (n = 0.04) scenarios increases slightly between the two material types. It should be noted here that although the presence of vegetation is not directly simulated in the constant friction run, the increase in elevation associated with the marsh plants is present in the bathymetry used for all the scenarios. This is because the LiDAR product used to create the bathymetry is a Digital Surface Model and incorporates the ground surface and all vegetation on it. In the higher friction scenarios (n = 0.06 to 0.12), velocity falls at the vegetation boundary. These trends are not replicated in the high flow period, however. Velocity only falls in the constant friction and zoned, n = 0.08 scenario. Velocity increases across the vegetation margin in all other scenarios. The vegetation boundary has been shown to lead to an increase in kinetic energy in some previous studies; Leonard and Croft (2006) found an increase in turbulence production in association with wake formation as tidal flows first encounter plant structures. Their observations were in fairly similar velocity conditions (0.25 ms$^{-1}$ in mudflat, 0.12 ms$^{-1}$ after vegetation margin), and this production of microturbulence was thought to inhibit sediment deposition, a finding that is relevant to the present study and could be examined further using sediment transport modelling.

Although the high flow period is typified by higher velocities than the low flow period, the overall trajectory of velocity reduction in the mudflat appears to be fairly similar, and is characterised by a rapid fall in speed in the first 10 m, followed by a more gradual decrease. Within the vegetated area, however, the velocity is more rapidly attenuated during the low flow period. Table 6.2 lists the reduction in velocity across the transect in terms of a percentage per metre (% m$^{-1}$). Taking the velocity at the first transect sampling point as 100%, the difference between the velocity here and at the last mudflat point (130 m) was calculated and converted into a percentage change, which

<table>
<thead>
<tr>
<th></th>
<th>Mudflat</th>
<th>Vegetated area</th>
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<tbody>
<tr>
<td></td>
<td>Low Flow</td>
<td>High Flow</td>
</tr>
<tr>
<td>Constant</td>
<td>0.69</td>
<td>0.57</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.70</td>
<td>0.56</td>
</tr>
<tr>
<td>Zoned, n = 0.06</td>
<td>0.59</td>
<td>0.60</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.56</td>
<td>0.57</td>
</tr>
<tr>
<td>Zoned, n = 0.10</td>
<td>0.57</td>
<td>0.60</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.59</td>
<td>0.60</td>
</tr>
</tbody>
</table>
was then divided by the distance between the two points. The same calculations were carried out within the vegetated area, using the velocity at the first point (140 m) as 100% and 160 m as the final point during the low flow period, and 200m in the higher period. Within the mudflat, velocity was attenuated at a similar rate across to the two flow periods, averaging 0.62% m⁻¹ in the low flow and 0.58% m⁻¹ in the high flow period. In the vegetated area, kinetic energy is also lost at a slightly faster rate during the lower period, at 4.58% m⁻¹ on average, compared to 4.31% m⁻¹ in the higher flow period. This may be more due to the differences in the length of vegetation inundated rather than the differences in the water levels. As already mentioned, the reduction trajectory is more complex when comparing the different friction scenarios in the vegetated area. In the low flow period, increasing the vegetation roughness results in a faster reduction in velocity (4.36% m⁻¹ rising to 4.70% m⁻¹). However, the scenario with no vegetation actually has the most efficient rate at 4.74% m⁻¹. There is no clear trend in the high flow period between the vegetation scenarios, but the constant friction is much less effective at reducing kinetic energy (3.86% m⁻¹ compared to 4.40% m⁻¹ on average for the vegetation scenarios).

Figure 6.16 shows the reduction in maximum velocity across transect 2 for the same series of scenarios. This transect starts at the edge of the creek just south of the breach area, and spans a short area of mudflat, before encountering 100 m of vegetation followed by another 80 m of mudflat. The pattern of energy attenuation is very similar for the low and high flow periods. In the first stretch of mudflat, velocity increases from the creek edge to the first sampling point, before falling sharply at 20 m. Between this point and the point immediately before the vegetation margin, velocity increases in all scenarios in the low flow period, but an increase is only seen in the rougher vegetation scenarios in the high flow period (n = 0.08 – 0.12). In the smoother vegetation runs, as well as the constant friction, velocity continues to fall. Across this mudflat stretch as a whole, velocity reduction in the low period varies between -0.25% m⁻¹ (i.e. an increase in velocity) in the roughest scenario to 0.10% m⁻¹ in the constant run. In the high period, attenuation generally increases alongside roughness, ranging from 0.28% m⁻¹ to 0.43% m⁻¹, but the constant scenario is between these extremes at 0.39% m⁻¹.

In contrast to transect 1, there is a clear reduction in flow velocity across the vegetation margin (between 30 m and 40 m). In the low flow period, 30.5% (on average) of the kinetic energy that passed through the first few metres of marsh vegetation was lost, whilst 22.4% was attenuated in the high flow period. These figures only refer to the zoned friction scenarios; the constant friction run has not been
Figure 6.16: Maximum flood velocities across transect 2 for all scenario runs for both low flow and high flow periods

included. Other studies have found that attenuation is inversely related to water depth (e.g. Fonseca and Cahalan, 1992; Tschirky et al., 2001; Koftis et al., 2013; Anderson and Smith, 2014), so the results presented here are in agreement with previous work. There is also a clear positive relationship between increasing friction and attenuation in the two periods, as shown in Table 6.3. In the vegetated scenarios, the lowest velocity reduction is found in the smoothest surfaces (18.9% and 17.8% in the two flow runs), increasing consistently to the roughest surface simulated (41.6% and 27.2%). In the remaining section of the vegetated area (40 m to 130 m), velocity attenuation is inversely related to roughness with the rate falling slightly between the smoothest and roughest zoned scenarios (Table 6.3). The constant scenario is most similar to the mid-level zoned results (n = 0.08).

Across the final area of mudflat in transect 2, velocities were marked by a series of increases and falls, but across the stretch as a whole speeds increased in all the
Table 6.3: Reduction in flow velocity across the vegetation margin and the rest of the vegetated area in transect 2 (%)
The reduction across the vegetation margin is calculated as the percentage change in velocity between sampling points at 30 and 40 m. Rest of vegetated area is defined as 40 m to 130 m.

<table>
<thead>
<tr>
<th></th>
<th>Vegetation margin</th>
<th>Rest of vegetated area</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low Flow</td>
<td>High Flow</td>
</tr>
<tr>
<td>Constant</td>
<td>20.46</td>
<td>16.72</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>18.86</td>
<td>17.84</td>
</tr>
<tr>
<td>Zoned, n = 0.06</td>
<td>24.93</td>
<td>19.89</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>30.45</td>
<td>22.33</td>
</tr>
<tr>
<td>Zoned, n = 0.10</td>
<td>36.88</td>
<td>24.96</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>41.57</td>
<td>27.21</td>
</tr>
</tbody>
</table>

vegetated scenarios in the high flow period. This is likely caused by the influence of the creek running west-east in the realignment site, which this stretch of transect is now closer to than that at the 0 m sampling point. Interestingly, however, velocities continued to fall in the unvegetated constant scenario during the low flow period (18.1%) but increased in the high flow period (17.3%).

As an exploratory analysis to determine potential influences on flow velocity along the transect, the water level at each data point has been extracted from the model simulations and compared with the water speed. It is possible that changes in the velocity could be due to varying water levels, for example the higher currents speeds simulated following the transition from vegetation to mudflat at 150 m (Figure 6.16) may be due to a surge in the flow as the vegetation thins out, and linked to higher tidal levels. Figure 6.17 shows the maximum velocities on the flood tide during the low flow and high flow periods alongside the simulated water levels for the same data point on transect 2. Only the results from the constant friction scenario run and the highest zoned friction (n = 0.12) run are shown for ease of examination. The water levels plotted are the mean heights from the two-hour window immediately before high water at 18:45 on 25/03/14 for the low flow period, and prior to high water at 00:50 on 01/04/14 for the high flow period. These tides were selected as they are approximately in the middle of each test period (see Figure 6.11 for the tidal conditions for both periods). The maximum water levels from each period were not plotted, as they are essentially identical throughout the transect as there is little variation in tidal height at high water. It was not possible to extract the water level from the model results at the exact time the maximum velocity was simulated.
Flow velocities are the maximum simulated speed during the flood tide for each flow period. Water levels are the mean tidal height in the two-hour time period prior to high water at 18:45 on 25/03/14 for the low flow period and at 00:50 on 01/04/14 for the high flow period.

As Figure 6.17 shows, the pattern of variation in simulated water levels across transect 2 is very similar between the low and high flow periods. The first mudflat section is characterised by lower heights, which rise to a peak at the end of the vegetated area, before falling again and then stabilising in the second mudflat section. The magnitude of variation is different, however, with the range between the minimum and maximum water levels calculated as 0.63 m during the low flow period, and only 0.17 m during the high flow period. The average levels are 1.53 m and 2.07 m respectively. There is very little variance between the two friction scenarios, with an average difference of 0.002 m between the constant friction and the zoned, \( n = 0.12 \) run in the low flow period. In the high flow period, the mean difference across the transect is just 0.0004 m, although the highest friction scenario is on average 0.004 m higher than the constant simulation in the first half of the transect, and 0.006 m lower in the second half.
From a visual examination of Figure 6.17, there appears to be a negative relationship between simulated flow velocity and water level. As velocity falls across the transect, water level tends to increase, but these trajectories are marked by a series of peaks and troughs. To explore this relationship further, the water velocities and levels are plotted against each other; this is shown in Figure 6.18. The trend lines confirm the negative association between the two variables for both the low and high flow periods and for both friction scenarios. The strength of the relationship is greatest for the constant friction simulations: \( r^2 = 0.717 \) and 0.781 for the two flow periods, compared to 0.510 and 0.634 for the zoned friction scenario. These results are contrary to those hypothesised above, and as found by Murphy and Voulgaris (2006), who noted that water levels are positively correlated with velocity. Although all the relationships found in this study are significant at the 95% confidence interval, they do appear to be heavily influenced by a set of points at low water levels (between 1.0 – 1.2 m in the low flow period).

**Figure 6.18:** Relationships between simulated water levels and flow velocity for selected friction scenarios for both low flow and high flow periods
period and 1.96 – 2.00 m in the high flow period). When these points are removed, the $r^2$ of the relationship falls to 0.262 for the constant friction scenario and 0.145 for the higher roughness run for the low flow period, and 0.264 and 0.054 respectively for the high flow period. It is thus concluded that this analysis fails to provide any further insights into the causes of the variation of flow velocity along transect 2.

Transects 3 and 4 are both characterised by two short sections of vegetated saltmarsh, separated by a longer stretch of unvegetated mudflat. The pattern of kinetic energy attenuation as water flows across each transect are very different, however, as indicated in Figure 6.19 and Figure 6.20. In the first short stretch of vegetation (40 m in length for transect 3, 20 m for transect 4), attenuation increases with the presence of and greater roughness of vegetation in the low flow scenario (Table 6.4) along transect 3 (0.48% m$^{-1}$ rising to 1.70% m$^{-1}$). However, increased roughness reduces attenuation in transect 4 (3.86% m$^{-1}$ falling to 1.86% m$^{-1}$). In the high flow period, there is no clear trend, with the attenuation rate peaking at the zoned, $n = 0.08$ scenario for both transects.

In the mudflat section of both transects, attenuation is overall very poor and marked by a number of peaks and troughs in velocity, particularly in transect 4. Across both transects, velocity is reduced more quickly in the low flow scenarios, and actually increases in the high flow period in transect 5. Increasing the roughness of the vegetation surrounding these bare areas has no conclusive influence except during the low flow period within transect 4 where the higher zoned scenarios reduce velocity more effectively.

To further add to the complexity, the velocity reduction profiles within the second area of vegetation are different from those seen in the first vegetated section and between the two transects. Due to the low water levels during the low flow period, no statistics can be calculated for transect 3, but there is a clear negative relationship between roughness and attenuation during the high flow period, as shown in Table 6.4. This is in contrast to the lack of influence seen over the high flow period in the 0 – 40 m vegetated area. In transect 4, increasing roughness clearly aids energy attenuation in both flow periods, with greater rates again recorded in the lower water levels.

Figure 6.21 shows the maximum flow velocity plotted alongside the simulated water levels for transects 3 and 4 for the constant friction and zoned, $n = 0.12$ scenarios during the high flow period only. As before, the extracted water levels shown are the mean heights from the two-hour window immediately before high water at 00:50 on 01/04/14. There is very little difference between the water levels from the two friction
Figure 6.19: Maximum flood velocities across transect 3 for all scenario runs for both low flow and high flow periods

Table 6.4: Velocity attenuation (% m$^{-1}$) for selected scenarios along transects 3 and 4

<table>
<thead>
<tr>
<th></th>
<th>1st Vegetated area</th>
<th>Mudflat</th>
<th>2nd Vegetated area</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Low Flow</td>
<td>High Flow</td>
<td>Low Flow</td>
</tr>
<tr>
<td><strong>Transect 3</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>0.48</td>
<td>1.28</td>
<td>0.59</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.88</td>
<td>1.32</td>
<td>0.63</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>1.70</td>
<td>1.38</td>
<td>0.76</td>
</tr>
<tr>
<td><strong>Transect 4</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>3.86</td>
<td>2.29</td>
<td>0.49</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>3.73</td>
<td>2.03</td>
<td>0.59</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>1.86</td>
<td>2.09</td>
<td>0.69</td>
</tr>
</tbody>
</table>
scenarios, which rise fairly steadily from the creek edge at the start of each transect to reach a peak at the end of the second vegetated area. The peaks and troughs in the velocity data do not appear to correspond with any variations in the water levels. When the two variables are directly compared with one another, there are weak negative relationships for both friction scenarios in both transects. As listed in Table 6.5, the $r^2$ values range between 0.224 and 0.472. All relationships are statistically significant at the 95% confidence interval, except that between water velocity and levels from the zoned friction scenario ($n = 0.12$) for transect 4. In a similar manner as transect 2, the results from this comparison show that water levels, in both the vegetated and unvegetated areas, have little influence over flow velocity within the realignment site.
Figure 6.21: Flow velocities and water levels across transects 3 and 4 for selected scenario runs during the high flow period
Flow velocities are the maximum simulated speed during the flood tide. Water levels are the mean tidal height in the two-hour time period prior to high water at 00:50 on 01/04/14.

Transect 5 is different from the others in that it is located in an area of the realignment site that is totally vegetated. This is used to see if the constant scenario (i.e. no vegetation zones) is significantly different from the other scenarios or simply acts as another scenario with a smoother surface than the zoned, n = 0.04 simulation. As Figure 6.22 and Table 6.6 show, however, neither of these are applicable, as the constant

<table>
<thead>
<tr>
<th></th>
<th>Transect 3</th>
<th>Transect 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant</td>
<td>0.408</td>
<td>0.472</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.415</td>
<td>0.224</td>
</tr>
</tbody>
</table>

All relationships are negative in nature
Figure 6.22: Maximum flood velocities across transect 5 for all scenario runs for both low flow and high flow periods

Table 6.6: Velocity attenuation (% m$^{-1}$) across various sections of transect 5

<table>
<thead>
<tr>
<th>Section</th>
<th>0–10 m Low Flow</th>
<th>0–10 m High Flow</th>
<th>10–30 m Low Flow</th>
<th>10–30 m High Flow</th>
<th>30–40 m Low Flow</th>
<th>30–40 m High Flow</th>
<th>30–80 m Low Flow</th>
<th>30–80 m High Flow</th>
</tr>
</thead>
<tbody>
<tr>
<td>Zoned, n = 0.04</td>
<td>9.83</td>
<td>8.07</td>
<td>-247.84</td>
<td>-6.60</td>
<td>9.95</td>
<td>0.89</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zoned, n = 0.06</td>
<td>9.86</td>
<td>8.05</td>
<td>-287.74</td>
<td>-20.92</td>
<td>9.85</td>
<td>1.55</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>9.81</td>
<td>7.78</td>
<td>-100.11</td>
<td>-12.93</td>
<td>9.83</td>
<td>1.43</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zoned, n = 0.10</td>
<td>9.95</td>
<td>7.81</td>
<td>-171.25</td>
<td>-19.39</td>
<td>9.88</td>
<td>1.53</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>9.96</td>
<td>7.53</td>
<td>-242.50</td>
<td>-7.42</td>
<td>9.90</td>
<td>1.18</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
scenario tends to be in the mid range of the other simulations. The overall profile of velocity across the transect is again marked by a series of peaks before eventually falling to zero in the marsh interior. In both the low and high flow period, velocity is almost totally attenuated in the first 10 m. On average, 98.4% and 97.5% is lost over this short distance. There is no clear influence of increasing friction in either period. There is then a sharp increase between 10 and 30 m from the creek in water speed with velocities almost reaching, and in some cases surpassing (in the higher flow period) the original value at the creek edge. In the low flow period, the highest rise is generally seen in the lower friction scenarios, whilst there is no conclusive trend in the high flow period. After 30 m, velocities fall away rapidly in all scenarios in the low flow period as water levels do not penetrate far into the marsh interior. The high flow period sees some further smaller peaks in velocity along the transect, with no clear influence of increasing friction.

6.2.2 Comparison of Flood and Ebb Velocities

As well as comparing the maximum velocities on the flood tide between each scenario, it is also interesting to compare these values with those from the ebb tide to determination which areas of the site are flood or ebb dominated and whether the vegetation helps to retard water as tidal levels fall. Figure 6.23 shows the maximum velocity on the flood tide (a) and ebb tide (b) for the constant friction scenario during the high flow period. As mentioned during the validation process, the main channel network within the realignment site is strongly ebb dominated with velocities much higher in the channel leading to the breach and on its surrounding banks. Velocities are also stronger on the ebb tide in the minor channels in the central vegetated area to the south west of the breach, in the NE corner, and in the central and eastern section of the main west-east channel in the centre of the site. In the western section of this channel, domination switches to the flood tide, with slightly higher current speeds on the incoming tide. Across most of the mudflat area flow velocity appears to be higher on the flood tide, suggesting conditions more favourable for sediment deposition. Velocities are also clearly lower on the ebb tide within the vegetated areas around the sea walls. Although this scenario does not simulate the presence of vegetation directly, elevation is higher in these areas which would have an effect on velocity.

To visualise the influence of vegetation on the flood and ebb tides, Figure 6.24 shows the maximum velocities on these tides during the high flow period from the
Figure 6.23: Maximum velocity (m s\(^{-1}\)) for the flood tide (a) and ebb tide (b) from the constant friction scenario during the high flow period
Figure 6.24: Maximum velocity (ms$^{-1}$) for the flood tide (a) and ebb tide (b) from the highest zoned friction scenario ($n = 0.12$) during the high flow period.
highest zoned friction scenario, where the areas of vegetation have been given a Manning’s n value of 0.12. Velocity is lower across the majority of the site compared to the constant friction scenarios; the reduction is especially visible around the breach site where a much smaller area of the marsh surface experiences the highest current speeds. The effect of vegetation appears to be more significant on the ebb tide as there are greater differences between the constant scenario and the rougher friction scenario on both the mudflat and vegetated areas than when comparing the two flood tide maps. To more fully determine the influence of vegetation across the two tides, data was extracted from the same transects as before, the locations of which are shown in Figure 6.14. In the forthcoming plots (Figure 6.25 – 6.28), only the data from the high flow period is examined. These data show a greater difference between the flood and ebb tides compared to the low flow period, where much of the vegetated areas around the sea walls were not fully inundated. Each plot only shows the velocities from the constant scenario, and those from the zoned, n = 0.04, 0.08 and 0.12 scenarios to improve visualisation.

Figure 6.25 compares the flow velocities on the flood and ebb tide during the high flow period within transect 1. Current speeds at the creek edge are significantly higher over the ebb tide; on average flow on the outgoing tide is 2.6 times greater than that on the incoming tide. The rest of the mudflat section as a whole is also characterised by higher ebb velocities, as shown in Table 6.7. This is most likely attributable to the presence of a minor creek in this area that branches off the main west-east channel; this channel can be seen be its higher ebb speeds in Figure 6.24(b). The effect of this creek reduces after approximately 50 m from the creek edge as defined in

![Figure 6.25: Maximum velocities on the flood and ebb tide across transect 1 during the high flow period](image)
Table 6.7: Average maximum velocities (ms⁻¹) on the flood and ebb tides during the high flow period within the mudflat and vegetated area of transect 1  

<table>
<thead>
<tr>
<th></th>
<th>Mudflat Flood Tide</th>
<th>Mudflat Ebb Tide</th>
<th>Vegetated Flood Tide</th>
<th>Vegetated Ebb Tide</th>
</tr>
</thead>
<tbody>
<tr>
<td>Constant</td>
<td>0.170</td>
<td>0.256</td>
<td>0.054</td>
<td>0.037</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.177</td>
<td>0.272</td>
<td>0.064</td>
<td>0.040</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.156</td>
<td>0.230</td>
<td>0.061</td>
<td>0.040</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.149</td>
<td>0.199</td>
<td>0.073</td>
<td>0.043</td>
</tr>
</tbody>
</table>

transect 1, and ebb velocities fall below flood velocities after 70 m. Higher friction factors in the vegetated scenarios result in lower average speeds over the mudflat, with the constant scenario positioned between the lowest and middle scenario. In contrast, greater vegetation roughness increases both flood and ebb maximum flow speeds in the vegetated area after 140 m from the creek edge. The unvegetated scenario reports the slowest speeds in this area, averaging at 0.037 ms⁻¹ on the ebb tide, compared to 0.043 ms⁻¹ in the highest vegetation run (n = 0.12). Across all the scenarios listed, velocities are on average 36.6% lower on the ebb tide than the flood tide in the vegetated area. This difference should aid sediment deposition in the saltmarsh.

Within the first mudflat section of transect 2, ebb velocities are also over double those simulated on the flood tide, as shown in Figure 6.26 and Table 6.8. Mean ebb velocities are 1.095 ms⁻¹ across all scenarios, 2.1 times higher than those on the incoming flood tide. Increasing roughness in the vegetated areas around the marsh

Figure 6.26: Maximum velocities on the flood and ebb tide across transect 2 during the high flow period
Table 6.8: Average maximum velocities (ms\(^{-1}\)) on the flood and ebb tides during the high flow period within the mudflat and vegetated areas of transect 2

<table>
<thead>
<tr>
<th></th>
<th>1st Mudflat</th>
<th>Vegetated area</th>
<th>2nd Mudflat</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Flood Tide</td>
<td>Ebb Tide</td>
<td>Flood Tide</td>
</tr>
<tr>
<td>Constant</td>
<td>0.524</td>
<td>1.125</td>
<td>0.258</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.559</td>
<td>1.182</td>
<td>0.278</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.536</td>
<td>1.078</td>
<td>0.269</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.490</td>
<td>0.993</td>
<td>0.254</td>
</tr>
</tbody>
</table>

reduces both flood and ebb currents in this unvegetated area. Velocities remain higher on the ebb tide within the central vegetated section of transect 2, but the difference between the two tides is less than in the outer mudflats, with the ebb speeds on average 11.1% higher than those on the flood. This is likely due to the proximity of this vegetated area to the main channel running from the breach, but speeds will also be affected by the ridge-furrow topography in this region. Velocities are assumed to be higher in these minor channels, and it is possible this transect spans one of these areas, especially around 80 m from the creek edge. Velocities are inversely related to friction factors across both flood and ebb tides in this region. Only in the second mudflat stretch of the transect do ebb velocities fall below those on the flood tide, averaging 11.1% lower. This corresponds with the overall observation that the majority of the mudflat areas are flood dominated, with increasing roughness surrounding them reducing flow velocities.

Despite the similar composition of transects 3 and 4, both of which are comprised of two vegetated sections separated by mudflat, there are significant differences between the flood and ebb maximum velocities, as indicated in Figure 6.27. In the first vegetated area, current speeds are higher on the ebb tide in both transects, but in transect 3 the average flow is 59.3% higher than the flood tide, compared to just 4.3% in transect 4 (Table 6.9). This is due to the closer proximity of transect 3 to the main breach channel; as mentioned previously, the strength of the ebb tide fades considerably along the west-east channel. Transect 4 is only located ~150 m to the west along this channel than transect 3, but the maximum velocity at the creek edge is less than half, at 0.202 ms\(^{-1}\) compared to 0.566 ms\(^{-1}\). The mudflat sections of both transects are flood dominated, with the flood velocities on average 27.1% higher than those on the ebb tide. The overall pattern of intermittent peaks and troughs in current speed on the flood tide
Figure 6.27: Maximum velocities on the flood and ebb tide across transects 3 and 4 during the high flow period

Table 6.9: Average maximum velocities (ms$^{-1}$) on the flood and ebb tides during the high flow period within the mudflat and vegetated areas of transects 3 and 4

<table>
<thead>
<tr>
<th>transect</th>
<th>1st Vegetated area</th>
<th>Mudflat</th>
<th>2nd Vegetated area</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Flood Tide</td>
<td>Ebb Tide</td>
<td>Flood Tide</td>
</tr>
<tr>
<td>Transect 3</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>0.222</td>
<td>0.376</td>
<td>0.126</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.229</td>
<td>0.390</td>
<td>0.147</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.199</td>
<td>0.300</td>
<td>0.127</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.184</td>
<td>0.263</td>
<td>0.127</td>
</tr>
<tr>
<td>Transect 4</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>0.138</td>
<td>0.160</td>
<td>0.137</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.136</td>
<td>0.165</td>
<td>0.134</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.132</td>
<td>0.124</td>
<td>0.136</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.122</td>
<td>0.101</td>
<td>0.136</td>
</tr>
</tbody>
</table>
is generally mirrored in the ebb tide along each transect. Ebb speeds are also lower within the second vegetated area in both transects, with increasing friction resulting in increased flow retardation.

The maximum velocities on the flood and ebb tide across transect 5 are shown in Figure 6.28. This transect is completely covered by saltmarsh vegetation. The whole transect is flood dominated, but the difference between the flood and ebb velocities varies across the marsh surface. The transect has been divided into three sections to highlight these differences: 0-10 m, 20-80 m and 80-100 m. In the first 10 m, peak velocities are very similar, decreasing rapidly from 0.23 ms\(^{-1}\) to 0.13 ms\(^{-1}\) on the flood tide, and 0.22 ms\(^{-1}\) to 0.04 ms\(^{-1}\) on the ebb, a difference of only 8.3% between the two tides. Increasing vegetation roughness increases velocities on the flood tide, but reduces them on the ebb (Table 6.10). In the central section (20-80 m) the difference is considerably greater, with the flood velocities 47.4% higher on average. There is no

![Figure 6.28: Maximum velocities on the flood and ebb tide across transect 5 during the high flow period](image)

Table 6.10: Average maximum velocities (ms\(^{-1}\)) on the flood and ebb tides during the high flow period across various sections of transect 5

<table>
<thead>
<tr>
<th></th>
<th>Flood Tide</th>
<th>Ebb Tide</th>
<th>Flood Tide</th>
<th>Ebb Tide</th>
<th>Flood Tide</th>
<th>Ebb Tide</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0-10 m</td>
<td>20-80m</td>
<td>80-100 m</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Constant</td>
<td>0.126</td>
<td>0.133</td>
<td>0.116</td>
<td>0.058</td>
<td>0.031</td>
<td>0.023</td>
</tr>
<tr>
<td>Zoned, n = 0.04</td>
<td>0.136</td>
<td>0.145</td>
<td>0.085</td>
<td>0.059</td>
<td>0.029</td>
<td>0.024</td>
</tr>
<tr>
<td>Zoned, n = 0.08</td>
<td>0.150</td>
<td>0.128</td>
<td>0.119</td>
<td>0.057</td>
<td>0.029</td>
<td>0.023</td>
</tr>
<tr>
<td>Zoned, n = 0.12</td>
<td>0.154</td>
<td>0.113</td>
<td>0.111</td>
<td>0.053</td>
<td>0.032</td>
<td>0.023</td>
</tr>
</tbody>
</table>
clear trend in the roughness scenarios regarding the flood tide, but rougher vegetation is indicated to increase the retardation of the ebb flow is this central stretch. As velocities approach zero in the rest of the transect (80-100 m), the trajectory of the flood and ebb tide aligns again, with the difference between them reducing to 22.6%. Variations in the friction factor applied to the marsh surface have very little effect in this inner area, with velocities ranging between 0.029 – 0.032 ms\(^{-1}\) on the flood tide and 0.023 – 0.024 ms\(^{-1}\) on the ebb.

### 6.2.3 Influence of Species Composition

In order to further integrate the hydrodynamic modelling with the field studies undertaken at Tollesbury, the vegetated areas used in Sections 6.2.1 and 6.2.2 have been sub-divided to account for the different species and vegetation characteristics and a different Manning’s n friction coefficient applied to each. The modelling undertaken so far assumes that the vegetation within the realignment site is identical, with all areas of vegetation assigned the same Manning’s n value. The more detailed approach is very original and may help to improve the accuracy of the model runs. Very few saltmarsh modelling studies incorporate areas of different vegetation species, and no recent work could be found that used different Manning’s n values to represent the different characteristics. Miller (1988) used a two-dimensional long-wave numerical model to simulate mean flows and water levels in a test estuary and varied the friction coefficient of the vegetated area. One value was applied to the whole test area per model scenario, however, rather than individual vegetation zones. A Manning’s n value of 0.06 was used to represent *Spartina alterniflora* and 0.125 for *Juncus roemerianus*. Neither of these species are found in the Tollesbury marshes. Temmerman *et al.* (2005b) did account for different halophyte species and plant structure in their 3D modelling study of a saltmarsh in the Netherlands, but used a very different method. They created a vegetation map of the marsh, and harvested plant matter from each species zone in the field. From these samples, they calculated the average diameter and number of cylindrical plant structures (stems and leaves) per unit area, in vertical layers of 0.1 m. The frontal surface area of leaves was also measured. All of these parameters were added to the model, and represent a highly complex method of modelling the influence of vegetation on flow in saltmarshes which could not be achieved in the current study.

Figure 6.29 shows the vegetation zones created for the Tollesbury realignment site; Table 6.11 lists the vegetation structure of each zone and the Manning’s n friction
Figure 6.29: Vegetation zones and transect locations to test influence of different plant species across the realignment site

Table 6.11: Vegetation structure and Manning’s n friction coefficients used for each material zone

<table>
<thead>
<tr>
<th>Material Zone</th>
<th>Vegetation structure</th>
<th>Manning’s n</th>
</tr>
</thead>
<tbody>
<tr>
<td>Channels</td>
<td>–</td>
<td>0.02</td>
</tr>
<tr>
<td>Mudflat</td>
<td>–</td>
<td>0.02</td>
</tr>
<tr>
<td>1</td>
<td>Predominately <em>Salicornia</em>, some <em>Spartina</em></td>
<td>0.055</td>
</tr>
<tr>
<td>2</td>
<td>Medium density <em>Spartina</em>, some <em>Atriplex</em> fronting the sea wall</td>
<td>0.065</td>
</tr>
<tr>
<td>3</td>
<td>Predominately taller, dense <em>Spartina</em>, some <em>Atriplex</em> fronting the sea wall</td>
<td>0.075</td>
</tr>
<tr>
<td>4</td>
<td><em>Spartina</em> (low density)</td>
<td>0.06</td>
</tr>
<tr>
<td>5</td>
<td><em>Salicornia</em> (low density)</td>
<td>0.045</td>
</tr>
</tbody>
</table>
coefficient assigned. The zones were divided according to the broad species composition that is visible in the 2013 aerial photograph (Figure 3.1) and using knowledge gained from the various field visits undertaken at the site. It was not possible to sub-divide the vegetation zones further or into smaller zones, as there is considerable small-scale variation across the marsh areas, and delineating the exact boundary of each species would require considerable surveying in the field. The vegetation structure listed in Table 6.11 provides the two main species present in each zone and an indication of their abundance and density. As mentioned, there are no previous saltmarsh modelling studies from which to determine the appropriate Manning’s n friction coefficient. However, the United States Geological Survey (USGS) have produced guidance to assist in the estimation of surface roughness values for non-tidal freshwater floodplains; this is detailed by (Arcement and Schneider, 1989) and was adapted by (Shafer and Yozzo, 1998) for use in tidal marsh applications. The calculation is based on determining a base roughness factor for the soil surface and adding further factors for marsh surface irregularities, and for the marsh vegetation. The vegetation factors include choices for characteristics such as density, structure, height and seasonality. The Manning’s n values listed in Table 6.11 were reached based on this guidance, and with advice from Prof Jon French. These material zones were then added to the model as previously described in Section 6.1.3.

Figure 6.30 shows the maximum simulated velocity on the flood tide (a) and the ebb tide (b) during the low flow period based on the vegetation zones shown in Figure 6.29; Figure 6.31 shows the same flow data from the high flow period. The spatial pattern of flow is similar to that shown in Figures 6.23 and 6.24 which show the maximum velocities for the constant friction and highest zone friction simulations. The magnitude of the velocity is roughly half that of the maximum scenario, as the friction coefficients used here (Table 6.11) are roughly half way between the two former simulations. As expected, maximum velocities from the new vegetation zones scenarios are higher than those in the constant friction scenario (Figure 6.23), and less than the highest zoned scenario (Figure 6.24). As discussed before, the main channel network is strongly ebb-dominated; in both the low and high flow periods, maximum velocities are higher on the ebb tide in the channels and marsh surfaces nearest the breach in the sea wall. The spatial variability between the two flow periods is similar across the flood and ebb tides, except the magnitude of the velocity during the high flow period is approximately double that of the low flow period.
Figure 6.30: Maximum velocity (m s\(^{-1}\)) for the flood tide (a) and ebb tide (b) from the vegetation zone scenario during the low flow period.
Figure 6.31: Maximum velocity (ms$^{-1}$) for the flood tide (a) and ebb tide (b) from the vegetation zone scenario during the high flow period.
To explore the results of these simulations further, a new set of transects has been created within the model domain which span two different vegetation zones. The locations of these transects are shown on Figure 6.29. As before, velocity data from each transect was extracted at 10 m intervals. Transect 6 is located in the south-eastern part of the realignment site, and comprises a 114 m stretch of mudflat that begins at the edge of the main west-east creek, followed by 24 m of low density *Spartina* (Zone 4). A short section of mudflat (17 m) separates this vegetated area and the remainder of the transect – Zone 3 vegetation (taller, denser *Spartina*). Transect 7 is towards the north west of the site, and is characterised by 90 m of Zone 1 vegetation (*Salicornia*), 55 m of unvegetated mudflat and finally 24 m of Zone 2 vegetation (*Spartina*). Transect 8 is comprised of the same two vegetation zones, but without the centre mudflat section (20 m of Zone 1, followed by 50 m of Zone 2). Finally, transect 9 spans two zones either side of the main west-east channel; more detail about this transect will be given later.

The maximum velocity at each sampling point along transect 6 for the flood and ebb tides during both the low and high flow periods is shown in Figure 6.32. In the first mudflat section (0 – 110 m), there is an overall reduction in velocity over both flood tides, but the trajectory is marked by a series of small peaks and troughs. Although the transect is unvegetated in this area, it passes through a narrow corridor with plants either side from 50 m from the creek edge (as shown in Figure 6.29). This does not seem to have affected the velocity within the transect, however, as the velocity profile does not appear to change significantly after the 50 m data point for either the low or high flow period, or the flood and ebb tides. Table 6.12 lists the reduction in velocity across the

![Graph showing maximum velocities](image)

**Figure 6.32:** Maximum velocities (ms⁻¹) on the flood and ebb tides in the low and high flow periods across transect 6

<table>
<thead>
<tr>
<th>Transect</th>
<th>Zone 1</th>
<th>Zone 2</th>
<th>Zone 3</th>
<th>Mudflat</th>
</tr>
</thead>
<tbody>
<tr>
<td>Transect 6</td>
<td>0.35</td>
<td>0.25</td>
<td>0.30</td>
<td>0.20</td>
</tr>
</tbody>
</table>

---

316
Table 6.12: Maximum flood velocity attenuation across transect 6 (% m\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>1st Mudflat</th>
<th>Zone 4</th>
<th>2nd Mudflat</th>
<th>Zone 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flow</td>
<td>0.72</td>
<td>4.44</td>
<td>-0.24</td>
<td>9.11</td>
</tr>
<tr>
<td>High Flow</td>
<td>0.36</td>
<td>3.70</td>
<td>-7.47</td>
<td>1.43</td>
</tr>
</tbody>
</table>

transect over the two flood tide simulations in terms of a percentage per metre (% m\(^{-1}\)). In the first mudflat section, kinetic energy was lost at a higher rate during the low flow period, at 0.72% m\(^{-1}\), compared to 0.36% m\(^{-1}\) in the high flow period. This observation is repeated throughout the remainder of transect 6, with velocity falling more rapidly during the low flow period. This pattern is the same as that found in transect 1, which is 50 m to west of transect 6. In the Zone 4 vegetated area, comprising low density *Spartina* plants, velocity was attenuated more rapidly than in the fronting mudflat, at 4.44% m\(^{-1}\) and 3.70% m\(^{-1}\) in the two flow periods. Maximum speeds then rose between 130 and 140 m from the creek edge at the vegetation / mudflat boundary and kept rising across the second mudflat section, up to 7.47% m\(^{-1}\) during the high flow period. Velocities fell away again in Zone 3. It is not easy to compare between the two different vegetated zones, because Zone 4 is only 20 m in width, compared to 80 m for Zone 3.

In a similar manner to most of the previous profiles, velocities are higher along transect 6 closest to the creek edge on the ebb tide than on the flood, as shown in Table 6.13. In the first mudflat section, ebb velocities are 1.23 times (low flow period) and 1.46 times (high flow period) greater than on the incoming tide. Although ebb velocities are lower in Zone 4, they increase again in the second mudflat section, before being at their lowest in Zone 3, furthest from the creek. In both the flood and ebb tide simulations, current speeds are lowest in the vegetated areas than the intervening mudflats, indicating the presence of vegetation does have a positive influence in reducing wave energy along this transect.

Table 6.13: Average maximum velocities (ms\(^{-1}\)) on the flood and ebb tides during the low and high flow periods for transect 6

<table>
<thead>
<tr>
<th></th>
<th>1st Mudflat</th>
<th>Zone 4</th>
<th>2nd Mudflat</th>
<th>Zone 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flood</td>
<td>0.069</td>
<td>0.026</td>
<td>0.068</td>
<td>0.015</td>
</tr>
<tr>
<td>Low Ebb</td>
<td>0.087</td>
<td>0.029</td>
<td>0.044</td>
<td>0.015</td>
</tr>
<tr>
<td>High Flood</td>
<td>0.160</td>
<td>0.078</td>
<td>0.152</td>
<td>0.047</td>
</tr>
<tr>
<td>High Ebb</td>
<td>0.234</td>
<td>0.063</td>
<td>0.071</td>
<td>0.031</td>
</tr>
</tbody>
</table>
Along transect 7, there is less of a difference between maximum velocities on the flood and ebb tides. As Figure 6.33 indicates, and Table 6.14 confirms, the average current speeds for the two tide types are very similar. In the vegetated Zone 1, closest to the creek edge, the mean velocities on the ebb tide are just 0.001 m s\(^{-1}\) faster than those on the incoming tide. The difference varies over the rest of the transect between the two flow periods, with the low flow period retaining the closeness between the two tides (especially during the mudflat section), whilst the flood velocity is almost twice as high than the ebb during the high flow mudflat section (0.112 m s\(^{-1}\) compared to 0.221 m s\(^{-1}\)).

In terms of wave attenuation on the flood tides, the overall trajectory in velocity reduction is again punctuated by a number of fluctuations, especially around the vegetation / mudflat boundary at 90 m from the creek edge. In both the low and high flow periods, velocity falls between 60-90 m within the marsh plants, before rising again in the following mudflat area. As in the previous transect, attenuation rate along transect 7 (Table 6.15) is greatest in the low flow period in both vegetated zones.

![Figure 6.33: Maximum velocities (m s\(^{-1}\)) on the flood and ebb tides in the low and high flow periods across transect 7](image)

<table>
<thead>
<tr>
<th></th>
<th>Zone 1</th>
<th>Mudflat</th>
<th>Zone 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flood</td>
<td>0.119</td>
<td>0.059</td>
<td>0.032</td>
</tr>
<tr>
<td>Low Ebb</td>
<td>0.120</td>
<td>0.054</td>
<td>0.017</td>
</tr>
<tr>
<td>High Flood</td>
<td>0.226</td>
<td>0.221</td>
<td>0.060</td>
</tr>
<tr>
<td>High Ebb</td>
<td>0.227</td>
<td>0.112</td>
<td>0.047</td>
</tr>
</tbody>
</table>
Table 6.15: Maximum flood velocity attenuation across transect 7 (% m⁻¹)

<table>
<thead>
<tr>
<th>Zone 1</th>
<th>Mudflat</th>
<th>Zone 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flow</td>
<td>0.71</td>
<td>3.83</td>
</tr>
<tr>
<td>High Flow</td>
<td>0.57</td>
<td>-3.17</td>
</tr>
</tbody>
</table>

Velocities in the intervening mudflat section rise sharply during the high flow period (3.17% m⁻¹), but continue to fall slowly (0.64% m⁻¹) during the low flow period. Comparing the two vegetated zones, current speeds are attenuated much more rapidly in Zone 2 than Zone 1; across the two flow periods, velocity is lost 5.4 times more quickly in the second marsh area, although the magnitude of the velocities are 3.7 times lower here.

Transect 8 spans the same two vegetated zones as transect 7 (Zones 1 and 2), but does not contain the centre area of mudflat. The only transition is thus from predominately *Salicornia* plants (Manning’s n = 0.055) to denser *Spartina* plants (n = 0.065) 20 m from the creek edge. The maximum velocities on the flood and ebb tides during both flow periods along transect 8 is shown in Figure 6.34; Table 6.16 lists the average velocities for each tide category. Although the mean current speeds on the flood tide are lowest in the zone furthest from the creek during both flow periods, the change in velocity across the transect is markedly different between the two periods. During the low flow period, the profile follows the typical pattern of a continual reduction in velocity, with the transition to the higher friction zone marked by an increase in the rate of attenuation (2.47% m⁻¹ in Zone 1, 5.00% m⁻¹ in Zone 2; Table 6.17). In contrast,
Table 6.16: Average maximum velocities (ms\(^{-1}\)) on the flood and ebb tides during the low and high flow periods for transect 8

<table>
<thead>
<tr>
<th></th>
<th>Zone 1</th>
<th>Zone 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flood</td>
<td>0.054</td>
<td>0.006</td>
</tr>
<tr>
<td>Low Ebb</td>
<td>0.044</td>
<td>0.005</td>
</tr>
<tr>
<td>High Flood</td>
<td>0.122</td>
<td>0.091</td>
</tr>
<tr>
<td>High Ebb</td>
<td>0.098</td>
<td>0.067</td>
</tr>
</tbody>
</table>

Table 6.17: Maximum flood velocity attenuation across transect 8 (% m\(^{-1}\))

<table>
<thead>
<tr>
<th></th>
<th>Zone 1</th>
<th>Zone 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low Flow</td>
<td>2.47</td>
<td>5.00</td>
</tr>
<tr>
<td>High Flow</td>
<td>1.84</td>
<td>-0.60</td>
</tr>
</tbody>
</table>

tidal velocity increases in Zone 2 during the high flow period, at a rate of 0.60% m\(^{-1}\) across the whole zone. This rate increases to 8.21% m\(^{-1}\) when only the transect section between 50-70 m from the creek edge is considered. The vegetation transition at 20 m is also characterised by a change in attenuation; speeds increase in the 10 m immediately after the increase in friction roughness. The maximum flow on the ebb tides is much less unusual, with a steady increase in velocity towards the creek edge, only marked by a small dip at the vegetation transition in the high flow period.

Transect 9 covers two vegetation zones in the centre of the realignment site separated by the main creek running in a west-east direction (Figure 6.29). The data from each point on the transect is plotted in Figure 6.35, with the section of the transect covering Zone 3 (the south section) plotted back-to-back with that covering Zone 4 (the north section). The velocities at the creek edges are shown at the 0 m distance on each plot, with the data from the centre of the creek plotted at -2.7 m. Although both transects thus begin with the same data, the change in maximum flood velocity across the profiles is very different. In the north section, there is a fairly steady reduction in current speeds during the high flow period, only marked by three small increases (at 0 m, 30 m, 70 m). Across the full transect, 30.46% of the original speed (the centre of the creek, -2.6 m) is attenuated by the 70 m data point. In contrast, the south section is characterised by a sharper fall from -2.6 m to 30 m, before rising again across both mudflat and vegetated sections. Despite this rise, 61.96% of the original velocity is lost over the entire transect. It is not possible to calculate the percentage attenuation over
Figure 6.35: Maximum velocities (ms\(^{-1}\)) on the flood and ebb tides in the low and high flow periods across transect 9

each stretch of the transects (mudflat, Zone 3 etc.) due to the short distances (only 10 m for some stretches) they cover. A similar pattern is visible in the low flow period, although the overall reduction in velocity is very different between the two sections: 75.95% in the south section, but only 0.99% in the north section. With regards to the ebb velocities, both low and high flow periods in the south section are characterised by a gradual increase from 70 m down to 0 m, whilst the profile in the north section roughly follows the more turbulent trajectory as seen in the flood tide. No definitive remarks can be made regarding the differences between the two vegetated zones (Zones 1 and 3) or the influence of the mudflat sections as both transect sections experienced both increases and decreases in velocity in both the vegetated and unvegetated areas.

6.3 Summary

A two-dimensional numerical model has been used to simulate the water levels and tidal velocities within the Tollesbury realignment site. Bathymetry data was determined by combining LiDAR measurements with a bathymetric survey, and a high-resolution finite-element triangular mesh was generated within which a series of hydraulic variables are calculated. The model was validated and found to be in close agreement with observed water levels at three instrument locations; observed flow velocities were not well matched by the model, but this was attributed to the quality of the measured data. Neither water levels nor flow velocities were sensitive to changes in constant friction across the entire model domain. The effect of vegetation on velocity was simulated by running a series of scenarios with varying friction factors applied to the
areas of the site known to be vegetated. These results indicated the majority of the mudflat and saltmarsh areas were flood dominated, whilst the main channel network and its immediate banks, especially closest to the breach site are ebb dominated. Close examination of the simulated flow velocities across a number of transects revealed the pattern to be more complex. In some cases, current speeds increased as they traversed the vegetation boundary, and increasing vegetation roughness was less effective at reducing kinetic energy than lower friction values. Overall, varying the roughness of the vegetated areas did not conclusively result in conditions more favourable for sediment deposition. However, areas of vegetation did generally tend to retard ebb currents more effectively than mudflats, but increasing friction had mixed results. It would have been useful to couple these results with a sedimentation model, but technical difficulties and time constraints prevented this at this time.
7. Discussion

Much of the data analysed in this thesis has led to conclusions that are contrary to those presented in the majority of other saltmarsh studies. No evidence has been found for erosion in the natural marshes surrounding the Tollesbury managed realignment site, despite the significant erosion reported across Essex between 1973 and 1998 by Burd (1992) and Cooper et al. (2001). The processes controlling saltmarsh sedimentation and retention across the Tollesbury marshes are complex and vary over time, with none of the typical environmental factors (e.g. elevation, distance from sediment source) having any significant influence over sediment deposition or resuspension. Most surprisingly, the small role vegetation appears to play in controlling sediment flux at Tollesbury is a total contradiction to the previously widely held view that marsh vegetation significantly increases deposition. Results from a series of hydrodynamic model runs also indicate the presence and increasing roughness of vegetation has both positive and negative effects on attenuating current velocities.

This chapter looks at the implications of these findings and compares and contrasts the results with a range of other published studies. Starting with an examination of the longer-term changes in the natural marshes at Tollesbury in the broader perspective of saltmarsh degradation in Essex and southeast England, the performance of the realignment site is then evaluated against other schemes around the country. The sedimentation processes at Tollesbury are compared with other research, and the role of vegetation in sediment accumulation is discussed.

7.1 Longer-Term Changes in the Tollesbury Marshes

In Chapter 3 of this study, a series of aerial photographs were examined to quantify the colonisation of vegetation across the Tollesbury realignment site and investigate any changes in the surrounding natural marshes. The loss of coastal saltmarshes in the southeast of England to historical land claim and development, and more recently sea-level rise, coastal squeeze and the expansion of internal creek networks has been recognised as a significant problem for years (Pye and French, 1993f; Doody, 2004; Airoldi and Beck, 2007; Wolanski et al., 2009; Pye and Blott, 2014). The most recent major study, carried out by Cooper et al. (2001) found that between 1973 and 1998, over 196.6 ha of marsh was lost to erosion in the Blackwater Estuary alone, with saltmarsh throughout Essex being lost at a net rate of 40 hectares per year. In response to this dramatic loss, a variety of methods to restore and recreate marsh habitats have
been examined and trialled, such as managed realignment (Dixon et al., 1998). Although the realignment site at Tollesbury was an experiment into the performance of the technique, and not a direct response to marsh degradation and erosion in the Blackwater Estuary, the site was created in this broader context of concern regarding saltmarshes, and is situated within 130 hectares of natural marsh that was shown to be eroding by Burd (1992). It is thus important to compare the results of this study with others and examine the overall evolution of the realignment site.

7.1.1 Natural Marshes

The data presented earlier in this study (Section 3.1.2), obtained by inspecting a series of aerial photographs taken between 1997 and 2011, shows no evidence of erosion within the natural marshes at Tollesbury, either at the seaward marsh edge or through the expansion of the tidal channels in the marsh interior. In fact, both of the test areas showed a slight gain in marsh area, although these are small enough to be considered within the error margins of the technique. It was thus concluded the marsh as a whole was stable during the period examined. Although the time period studied was fairly short (14 years), it is comparable with earlier reports, and the conclusion was surprising considering the widely reported figures of significant saltmarsh erosion throughout Essex.

Despite the widespread concern into saltmarsh erosion in the southeast of England, there are actually very few independent, reliable systematic surveys that provide quantitative estimates of loss (Pye and French, 1993f). The majority of researchers tend to refer back to previous reports, or investigate the causes of the erosion, rather than measuring it themselves. Until the Environment Agency carried out a major survey of saltmarsh extent in England and Wales in 2009 (reported in Phelan et al., 2011), the last complete survey of saltmarshes in the UK was undertaken by the Nature Conservancy Council (NCC) between 1981 and 1988. The results of this survey are found in Burd (1989): ‘The Saltmarsh Survey of Great Britain’, and have been extensively referred to in the literature, prior to the publication of the EA report in 2011. For most of the country, this involved conducting field surveys and drawing sketch maps of marsh areas, often without close reference to an OS map or aerial photograph (Foster et al., 2013). This information was then transferred by hand to base maps at a scale of 1:25,000 or 1:50,000 (Pye and French, 1993f). Needless to say, the methods used were very basic, had numerous limitations and are assumed to have considerably
underestimated saltmarsh extent. The EA survey utilised high-resolution digital areal photography, and areas of saltmarsh were digitised directly within GIS software, typically taking place at a screen scale of 1:500, resulting in much greater accuracy (Phelan et al., 2011). Table 7.1 compares the estimates of extent derived from the 1989 and 2011 reports for England and Wales; saltmarsh in Scotland was measured by Burd (1989), but falls outside the EA’s remit so has not been included here. Although the EA report does not make any observations regarding the comparison between the two surveys, it is shown that the recent survey recorded an extent (across both England and Wales) that is 3.1% higher than the NCC survey, despite the 21-year difference between the two periods. Across England, the difference between the two surveys varied between -12.9% in the South West EA region through to +34.5% in the North East (Phelan et al., 2011). The Anglian region, in which Essex is located, has an extent 10.3% higher than was reported by Burd (1989). It is highly unlikely the extent of saltmarsh across the country increased by such significant levels, so the NCC survey is considered a considerable underestimation across much of the country.

This underestimation becomes an issue when the NCC survey is compared with earlier estimates of marsh extent, primarily in the form of OS maps and Admiralty charts. These themselves suffer from a number of limitations, depending on the survey method used, and on the completeness of any revisions undertaken for subsequent editions (Pye and French, 1993b). In one of their reports for the Ministry of Agriculture, Fisheries and Food (MAFF, now Defra), Pye and French (1993b) compared historic saltmarsh extent throughout England, derived primarily from these sources, with the NCC survey and found saltmarsh was being lost rapidly over time. Their calculated rate of loss, estimated at 100 hectares per year across England, has been used by government agencies, local authorities and academics for determining habitat restoration and coastal defence policy (Phelan et al., 2011). However, Pye and French (1993c) themselves recognise that marsh extent derived from OS maps contains significant discrepancies in some areas, and the results should be treated with caution.

**Table 7.1: Comparison of estimated saltmarsh extent between the NCC and EA surveys**

<table>
<thead>
<tr>
<th></th>
<th>Estimated saltmarsh extent (ha)</th>
<th>Difference</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>NCC (Burd, 1989)</td>
<td>EA (Phelan et al., 2011)</td>
</tr>
<tr>
<td>England</td>
<td>32,500.13</td>
<td>33,571.57</td>
</tr>
<tr>
<td>Wales</td>
<td>6,747.74</td>
<td>6,950.16</td>
</tr>
</tbody>
</table>
In order to explore the accuracy of estimates of saltmarsh extent calculated from historic OS products, a series of maps were obtained from Digimap (2016) that cover the Tollesbury natural marshes. Available maps were originally produced in 1875, 1897, 1922, 1973 and 2016; all historic maps were drawn at a scale of 1:2,500, the 2016 map is at a scale of 1:2,000. Due to time constraints, a small test area of natural marsh at the eastern-most seaward edge was chosen (similar to the outer test area used in Chapter 3). This region allows for the detection of changes at both the marsh edge, as well as the internal creeks. The mapped extents are shown in Figure 7.1 for 1875, 1922, 1973 and

Figure 7.1: OS maps and digitised saltmarsh areas from 1875 (a) and 1922 (b)
2016 with the total area of marsh in each map listed in Table 7.2. Marsh extent was digitised in ArcGIS, at a screen scale of 1:200. A map of the extent from 1897 has not been shown as it was essentially identical to that from 1875; the slight gain in marsh area between the two surveys (0.03 ha) is attributed to digitising error or map georeferencing issues, rather than an increase in marsh. There are significant losses of marsh between 1897 and 1922; 4.25 ha were lost, equating to 16.6% of the 1897 extent. This loss is partly explained by a loss of marsh at the eastern seaward edge, shown at point A on Figure 7.1(b), as well as a reduction in the northern extent, point B.
Table 7.2: Marsh extents derived from OS maps for section of Tollesbury marshes

<table>
<thead>
<tr>
<th>Year</th>
<th>Map scale</th>
<th>Marsh area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1875</td>
<td>1:2,500</td>
<td>25.54</td>
</tr>
<tr>
<td>1897</td>
<td>1:2,500</td>
<td>25.57</td>
</tr>
<tr>
<td>1922</td>
<td>1:2,500</td>
<td>21.32</td>
</tr>
<tr>
<td>1973</td>
<td>1:2,500</td>
<td>14.19</td>
</tr>
<tr>
<td>2016</td>
<td>1:2,000</td>
<td>14.14</td>
</tr>
</tbody>
</table>

It is likely part of the calculated total loss must also be attributed to a difference in the level of detail captured between different surveys. A number of creeks in the marsh interior appear to have extended significantly between 1875 and 1922, including those highlighted at points C and D in Figure 7.1(b); the creek at point D is shown to have lengthened by ~350 m between the two surveys. Equating to a rate of 7.4 m yr$^{-1}$, this rapid expansion seems improbable and is more likely due to a greater level of detail in marsh creeks captured in the latter survey. Comparing the maps from 1922 and 1973 further highlights this point. According to the extents calculated for the two periods (Table 7.2), 33.4% of the marsh was lost, but the level of detail captured in the 1973 survey is vastly greater than that from 1922. Instead of the five large marsh segments surveyed in 1922, a total of 104 individual marsh segments were recorded in 1973, all separated by bare mud that must previously have been incorporated into the calculated extent. This assertion can be made, as by comparing the 1973 and 2016 surveys (which capture the same high level of detail) only 0.35% of the marsh area was lost. If the rate of marsh loss between 1922 and 1973 were to be extrapolated to 2016, only 9.5 hectares would be present today. Regardless of the differences in the level of detail captured over time, the reliability of OS maps as a source of accurate information is called into doubt by the fact that the realignment site at Tollesbury is still shown as agricultural land in the 2016 map, with the sea wall intact and fronted by marsh, as shown in Figure 7.2, despite it having been breached over 20 years ago. In their comparison of historic maps and aerial photography, Baily and Inkpen (2013) came to similar conclusions, and suggested that map data should be treated with a great deal of caution. As historic OS maps have been shown to display significant overestimates of saltmarsh extent, and the NCC’s 1988 estimate is considered an underestimate, the calculated rate of loss of 100 ha per year by Pye and French (1993c) must now be considered to be a gross overestimation of the erosion rate of saltmarshes across the UK.
Another frequently cited report into saltmarsh erosion is Burd (1992), again for the NCC, entitled ‘Erosion and vegetation change on the saltmarshes of Essex and north Kent between 1973 and 1988’. The 1973 marsh extent was obtained from the results of a study carried out by the Institute of Terrestrial Ecology (ITE, now CEH), which generated a series of panchromatic vertical aerial photographs of Essex at a scale of 1:10,000. Saltmarsh boundaries were traced by hand from the photographs and then traced back onto OS base maps, from which area calculations were made (Cooper et al., 2001). The 1988 study again generated a new series of panchromatic aerial photographs, but at a more detailed scale of 1:5,000. The same tracing method from the photographs was used, but these tracings were scanned into a GIS and compared with the tracings from the 1973 study (Burd, 1992). Across the Essex and north Kent coast, it was determined that 1213.7 ha of marsh were lost, equating to a rate of 1.46% per year.

Although a loss was calculated for each estuary over the study period (200.2 ha for the Blackwater Estuary), separate figures for the Tollesbury marshes were not given. However, a series of maps accompanied the report that showed the losses and gains in marsh area for each sub-region; those covering Tollesbury were scanned and digitised in ArcGIS. The resulting change map is shown in Figure 7.3, with the area of each class shown in Table 7.3. The majority of the marsh is shown to be stable, with 79.5% of the
Figure 7.3: Change detection in saltmarsh area across the Tollesbury marshes, 1973 – 1988
Areas of marsh digitised from map contained within Burd (1992).

area in 1973 also present in 1988. A net loss of 22.83 ha was found to have occurred, mainly through the expansion of tidal channels into the marsh interior; from a visual analysis of the change map (Figure 7.3), only a small proportion of the recorded loss was at the seaward edges of the marsh. Indeed, there was a small gain of marsh area in some of these regions. In a similar manner as the change detection process from the OS maps showed, however, most of the recorded loss of marsh area is actually attributed to

Table 7.3: Change detection statistics derived from Burd (1992) saltmarsh extents for Tollesbury natural marshes, between 1973 and 1988

<table>
<thead>
<tr>
<th>Class</th>
<th>Area (ha)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stable</td>
<td>101.54</td>
</tr>
<tr>
<td>Loss</td>
<td>26.12</td>
</tr>
<tr>
<td>Gain</td>
<td>3.29</td>
</tr>
<tr>
<td>Net Loss</td>
<td>22.83</td>
</tr>
</tbody>
</table>
differences in the level of detail captured by the different surveys. In the NCC report itself, Burd (1992) notes the ITE survey used a ‘broad-brush’ approach in which the mapping of the marsh outline was approximated with no significant detail regarding the creek networks captured. To save time, most creeks were only roughly indicated or simply included in the marsh extent. The large scale of the photographs used (1:10,000) also prevented a high level of detail during the tracing process. In contrast, the photographs captured in 1988 were at a higher scale (1:5,000) and the aim of the study required a more detailed mapping approach, so more tidal channels were omitted from the marsh extents. Both surveys also used unrectified aerial photographs, so any calculated areal extents likely contain errors (Cooper et al., 2001). Considering the saltmarshes in Essex and Kent comprise highly fragmented systems with complex creek systems and significant areas of bare mud within the marsh interior, the variation in detail captured by the two reports has thus again led to significant overestimates in marsh loss across the region.

Despite these considerable issues, both NCC reports continue to be used as baselines for later studies investigating more recent saltmarsh erosion. This is in spite of the clear warning Burd (1989) gives in the earlier report: “it is not recommended that the method be repeated or used as a baseline for monitoring”. Cooper et al. (2000) did exactly that, however, and presented rates of marsh loss between 1988 and 1998. Although they found marsh loss had slowed since 1973-1988, they reported a loss in the Blackwater Estuary of 54.9 ha or 7.4%. It is not clear from the paper what level of detail was captured in terms of the internal creek networks as no maps are presented, but the use of large scale 1:10,000 photographs and the method of calculating marsh extents from tracing paper likely resulted in significant errors in marsh loss.

A more recent study carried out by Thomson et al. (2011a) for Natural England broadly corroborates with the findings presented in this thesis and the criticisms of the earlier reports. Using high-resolution orthorectified aerial photographs (similar to those used in Chapter 3), they mapped saltmarsh changes throughout Essex SSSIs between 1997 and 2008 directly into a GIS system. Although their method of digitising extents captured less detail than that used in this thesis, it was consistent between the two survey periods. Across their entire study area, they reported only a net loss of 1.04 ha over the 11-year period, equating to an erosion rate of 0.09 ha yr\(^{-1}\). This compares with a rate given by Cooper et al. (2000) of 32.78 ha yr\(^{-1}\). Similarly low rates of erosion were found by the same team of researchers in neighbouring Suffolk (Boyes and Thomson,
In the Blackwater Estuary, saltmarsh was lost at just 0.13 ha yr\(^{-1}\), a rate considerably lower than that reported in previous studies.

As the 2011 Natural England report contains a breakdown of the marsh areas for each SSSI management unit, the change in the Tollesbury region can be examined. Figure 7.4 shows the change detection map obtained directly from the report (Thomson et al., 2011b); the summary statistics are listed in Table 7.4. The Tollesbury marshes are covered by two units, with the northern marshes incorporated into Unit 18 (which also includes a small region to the east of the main marsh area), and the southern marshes comprising Unit 19. Due to the availability of aerial photography, the change in marsh area is calculated for two different time periods: 2000-2008 for Unit 18 and 1997-2008 for Unit 19. In contrast to the earlier studies (Burd, 1992; Pye and French, 1993b; Cooper et al., 2000, 2001), approximately 93% of the Tollesbury natural marshes are found to be stable. A small loss occurred in the northern marshes (1.95 ha, equating to

![Figure 7.4: Change detection in saltmarsh area at Tollesbury, 1997 – 2008](image)

Source: Obtained directly from Thomson et al. (2011b). Unit 18 includes a small area of marsh in the east of image not included in earlier figures. Marsh areas to the immediate east of Marina (below solid black line) are not included in Unit 19. Image was not of sufficiently high resolution to be digitised in ArcGIS.

Unit 18 change is for period 2000 – 2008; Unit 19 change is for period 1997 – 2008. Cream indicates areas of stable marsh, green indicates gain, red indicates loss.
Table 7.4: Change detection statistics derived from Thomson et al. (2011a) saltmarsh extents for Tollesbury marshes, 1997 – 2008
Please see Figure 7.4 and caption for details of marsh areas and study periods.

<table>
<thead>
<tr>
<th>Class</th>
<th>Unit 18</th>
<th>Unit 19</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Stable</td>
<td>75.17</td>
<td>49.17</td>
</tr>
<tr>
<td></td>
<td>91.3%</td>
<td>95.4%</td>
</tr>
<tr>
<td>Loss</td>
<td>4.56</td>
<td>1.68</td>
</tr>
<tr>
<td></td>
<td>5.5%</td>
<td>3.3%</td>
</tr>
<tr>
<td>Gain</td>
<td>2.61</td>
<td>0.68</td>
</tr>
<tr>
<td></td>
<td>3.2%</td>
<td>1.3%</td>
</tr>
<tr>
<td>Net Change</td>
<td>-1.95</td>
<td>+1.01</td>
</tr>
<tr>
<td></td>
<td>2.4%</td>
<td>2.0%</td>
</tr>
</tbody>
</table>

0.24 ha yr\(^{-1}\)), but the southern natural marshes were found to have a slight increase in area (1.01 ha, 0.09 ha yr\(^{-1}\)). These findings are comparable with those reported for the two test areas in Chapter 3 of this thesis, although there is a difference in the level of detail included.

Discrepancies in the geomorphological complexity captured by different surveys were highlighted in Section 3.2 of this thesis when the level of fragmentation in the natural marshes at Tollesbury was investigated. The saltmarsh extent digitised by the EA from the 2008 aerial photograph contained many inconsistencies, and both overestimates and underestimates were found in neighbouring sections of the marshes, as was shown in Figure 3.19. Across the Tollesbury marshes, the extent measured as part of this thesis is 86.11 ha, whilst the EA extent is 89.51 ha, 3.95% higher. Some of these errors can be attributed to differences in GIS analysts’ diligence, but the majority are likely caused by the guidelines by which the digitising was undertaken. The EA operators digitised significant areas of algae that fronted the seaward extent of marsh that was excluded in this study. The expanse of algae present on mudflats can change widely from year to year, and should never be included in estimates of marsh extent. Although algal mats and biofilms can act to reduce wave energy (Sutherland et al., 1998; Austen et al., 1999), they are in no way comparable to the function provided by raised marsh platforms colonised by halophytic vegetation. More important, however, is the way in which creeks were mapped. According to Phelan et al. (2011), creeks narrower than 1.5 – 2 m in width were merged into the marsh extent. For this study, all creeks were mapped individually and excluded from the overall extent. It is likely that most of the 3.4 ha difference between the EA extent and that produced from this study can be attributed to this difference. It is argued that, for complex drainage networks like those found in the natural marshes at Tollesbury, a much more detailed technique
should be applied for digitising areas of saltmarsh and tidal creeks. Although this method takes longer to achieve, increasing the geomorphological complexity in the mapping process significantly improves estimates of marsh extent and how it changes over time. As an example, a creek with a width of 1.9 m in one photograph would be incorporated into the saltmarsh extent by the EA, but, if that creek suffered minor erosion before the next photograph was taken and widened to 2.1 m, it would be mapped in its own right and excluded from the marsh extent. In this scenario, the actual erosion of 0.2 m of marsh would be calculated as being 2.1 m by the EA, a gross overestimate of the genuine change over time. The errors highlighted by the EA and others (Phelan et al., 2011; Foster et al., 2013) of previous calculations of marsh extent and overestimates of erosion would thus perpetuate further into the future.

The examination of a variety of estimates of saltmarsh extent since the 1970s has highlighted the inaccuracy of reported levels of marsh erosion, all of which have been shown to be significant overestimates. In fact, some areas of marsh (such as the southern area of marshes at Tollesbury) have been shown to have increased in area. Other small-scale studies note an assortment of estimates of marsh changes, ranging from a 40% loss in the Solent from 1971 – 2001 (Cope et al., 2008) to rapid lateral expansion (62-73 ha yr\(^{-1}\)) reported in the Wash since 1950 (Phelan et al., 2011). Despite some losses around England, they have been far short of the 100 hectares per year erosion estimate produced by Pye and French (1993b) and used by government in coastal management plans. The surprise at the lack of erosion within the Tollesbury natural marshes found in Chapter 3 should thus not have been a surprise at all.

7.1.2 Realignment Site

The 21-hectare realignment site at Tollesbury was created in 1995 by breaching a section of sea wall that previously protected reclaimed agricultural land. The existing terrestrial vegetation within the site was quickly killed off with the introduction of tidal inundation (Garbutt et al., 2006b). Since then, sediment has steadily accumulated on the former soil surface, and saltmarsh vegetation has colonised parts of the mudflat that formed. A supervised classification of a series of aerial photographs covering the realignment site was used to quantify the expansion in vegetation over time (Section 3.1.1). After 18 years, just over half of the site (9.4 ha) was colonised by pioneer, low and mid marsh halophytic vegetation, double the 4.7 ha estimate that was predicted before the site was restored (Pendle, 2013). Other researchers found that 12 years after
tidal inundation at the site, the vegetation communities in the realignment site were similar to the adjacent natural saltmarsh (Hughes et al., 2009). This compares to only 5 years at the nearby Abbotts Hall realignment site, also in the Blackwater Estuary and created in 2002. Development of vegetation within the Tollesbury site was characteristic of tolerance type successions, as opposed to facilitated succession as little sediment had accreted (only 1 cm over 8 years in the inner parts of the site; Garbutt et al., 2008). The early opportunistic species (e.g. *Salicornia* spp.) had no apparent facilitative or preventative effect on the perennial species that arrived later and out-competed them (Hughes et al., 2009). However, as part of the main monitoring programme at Tollesbury carried out by CEH, Garbutt et al. (2006b) recorded only 32% of the regional species pool had established within the realignment site after 8 years. Based on personal observations whilst undertaking fieldwork at Tollesbury between 2010 and 2014, the species composition within the realignment site is very different from that found in the natural marshes. *Salicornia* spp. is found in the lower areas of the site, particularly on the creek edges and part of the central vegetated area near the breach, whilst *Spartina anglica* totally dominates the vegetated areas nearest the sea walls. Only a few pockets of the mid marsh shrub *Atriplex Portulacoides* are found fronting the sea walls. The natural marshes, by contrast, are primarily composed of *Aster tripolium*, *Puccinellia maritima* and *Atriplex*. No significant patches of *Spartina* were found here.

The rate of vegetation colonisation at Tollesbury has been slower than that which has occurred at other realignment sites both in the UK and the rest of Europe. Table 7.5 lists the saltmarsh cover at a number of schemes at different times after the surrounding sea walls were breached and tidal inundation restored. Although there are approximately 50 realignment sites in the UK, and 100 throughout northwest Europe (Mazik et al., 2010), there are little quantitative data available on the coverage of saltmarsh vegetation over these sites as a whole. Tollesbury is one of the oldest and most intensively studied sites and the reports by the CEH staff (Reading et al., 2002, 2006, 2008b) from the main monitoring programme are very detailed. Comparable data is unfortunately lacking for most other sites; instead vegetation growth and species composition is typically measured using individual quadrats so a comparison cannot be made. There are some sites where this information is available, such as Abbotts Hall and Brancaster, where vegetation developed rapidly and colonised 30% of the restored marsh after two years and 62% after four years at the respective locations (Table 7.5). This compares with only 7.1% at Tollesbury after two years, and 9.8% after four (by averaging the cover after two and six years).
Table 7.5: Vegetation colonisation in realignment sites around the UK and Europe

<table>
<thead>
<tr>
<th>Realignment site</th>
<th>Period embanked</th>
<th>Time after breaching</th>
<th>Saltmarsh cover</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tollesbury Blackwater Estuary, Essex, UK</td>
<td>~150 years</td>
<td>2 years</td>
<td>7.1%</td>
<td>This study</td>
</tr>
<tr>
<td></td>
<td></td>
<td>6 years</td>
<td>12.5%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>13 years</td>
<td>36.2%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>16 years</td>
<td>45.0%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>18 years</td>
<td>50.2%</td>
<td></td>
</tr>
<tr>
<td>Abbots Hall Blackwater Estuary, Essex, UK</td>
<td>&gt;200 years</td>
<td>2 years</td>
<td>30%</td>
<td>Mossman (2007)</td>
</tr>
<tr>
<td>Brancaster North Norfolk, UK</td>
<td>4 years</td>
<td>62%</td>
<td>Davy et al. (2011)</td>
<td></td>
</tr>
<tr>
<td>Freiston Shore The Wash, UK</td>
<td>19 years</td>
<td>1 year</td>
<td>69.8%</td>
<td>Freiss et al. (2012)</td>
</tr>
<tr>
<td>Wallasea North Crouch Estuary, Essex, UK</td>
<td>&gt;400 years</td>
<td>1 year</td>
<td>&lt;1%</td>
<td>ABPmer (2011)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>2 years</td>
<td>6%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>3 years</td>
<td>60%</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>4 years</td>
<td>~100%</td>
<td></td>
</tr>
<tr>
<td>Sieperda marsh Scheldt Estuary, The Netherlands</td>
<td>24 years</td>
<td>5 years</td>
<td>90%</td>
<td>Eertman et al. (2002)</td>
</tr>
<tr>
<td>Karrendorfer Wiesen Baltic Sea, Germany</td>
<td>~165 years</td>
<td>5 years</td>
<td>75%</td>
<td>Bernhardt and Koch (2003)</td>
</tr>
</tbody>
</table>

The Freiston Shore site in The Wash developed even more quickly, achieving almost 90% of marsh coverage in just four years. The main reason for this rapid rate of plant growth was the very recent reclamation, which only took place in 1982. When the site was restored just 19 years later, the elevation of the surface within the sea walls was only <0.3 m less than that in the external marshes (Friess et al., 2012) and was at a height more suitable for plant establishment than the majority of other realigned areas.

A similar recent restoration in the Scheldt Estuary in The Netherlands also enabled vegetation to colonise rapidly, as site elevation was on average 2.3 m above the high water mark (Eertman et al., 2002). At Tollesbury, reclamation took place approximately 150 years before it was later restored, and the soil within the sea walls had been compacted and subsided to a level almost 1 m lower than the natural marshes, resulting in much greater inundation times that prohibit the germination of halophytic vegetation.
Early colonisation here was thus limited to those areas fronting the secondary sea walls. The realignment site on the north bank of Wallasea Island in the Crouch Estuary shows that a long period of embankment does not have to preclude a fast rate of vegetation development, however. At this 115 ha site, almost the entire site was vegetated after 4 years, despite the fact it was reclaimed over 400 years ago (ABPmer, 2011). Little is known about the use of the site over this time, but a series of measures were taken in order to aid marsh development prior to the breaching of the sea wall in 2006. Over 550,000 m$^3$ of maintenance dredge material from the port of Harwich were placed just outside the site as a source of sediment to be carried into the restored area. Seven island features were created in the site to vary the topography, which acts to promote drainage (Brooks et al., 2015). Dykes and field drains already in use within the site also promoted effective filling and draining of the entire area on each tide. The Karrendorfer Wiesen area in Germany also had a well-established dyke system that resulted in fairly quick colonisation despite a 165 year period of embankment. As former grassland, the site had not suffered from heavy compaction with agricultural machinery in the same way as Tollesbury (Bernhardt and Koch, 2003).

There are a number of other possible reasons for the slow rate of marsh development at Tollesbury. One is the closeness of a viable seed bank and the easy dispersal of plant propagules into the site, without which vegetation development will naturally be hindered (Wolters et al., 2005b). However, this is unlikely to be an issue at Tollesbury due to the immediate proximity of the natural marsh which will provide seeds. The development of Spartina in the realignment site when no large areas of this species are present in the surrounding marshes would also indicate this not to be problematic. Paramor and Hughes (2002) proposed the lack of vegetation in large parts of this site and others is due to the presence of burrowing invertebrates, primarily the ragworm Nereis diversicolor, which disturbs the sediment and consumes seeds before plants can become established. There is little evidence to support their concern, however, due to the colonisation of over half of the Tollesbury site and other realignment sites in Essex where this invertebrate has been found. Its activities appear to be insufficient to restrict plant succession of exposed sediment (Wolters et al., 2005a).

The primary reason for the continued presence of large areas of unvegetated mudflat within the Tollesbury realignment site has been attributed to the lack of effective drainage that has resulted in water-laden unconsolidated sediments (Boorman et al., 2002; Crooks et al., 2002). These soils are more anoxic than those in the surrounding natural marshes, which limits plant colonisation or restricts abundance to
only the pioneer species (Davy et al., 2011; Brooks et al., 2015). Instead, large areas of the mudflat are regularly covered with algal mats, in particular Enteromorpha, which is linked to the poorer than expected establishment of Salicornia (Garbutt et al., 2002). The poor drainage in the site is primarily due to the development of an impervious layer of over-consolidated material that acts as an aquaclude (Hazelden and Boorman, 2001). Sediment accumulating within the realignment site builds up on top of the old soil surface that was created when the site was originally reclaimed ~160 years ago. This soil has compacted and ripened over time, and although its moisture content increases over time due to regular inundation, the ripening process is irreversible and the soil will never return to the same physical state it was in before reclamation (Brooks et al., 2015). As the tidal water cannot drain down through the sediment, it remains water logged, with low bulk densities and low resistance to resuspension and erosion (Brown et al., 2007).

Topographic homogeneity, in the form of large expanses of flat muddy sediments, also precludes vegetation colonisation. The natural marshes are characterised by large variations in topography over short distances, but the realignment site, due to its former use in arable farming was almost level when tidal inundation was restored (Brooks et al., 2015). This is especially true of the north east corner of the site, where the surface is very flat and totally devoid of vegetation as water can only drain very slowly due to a lack of any channels or creeks. In contrast, the area to the west of the breach in the sea wall (termed the central vegetated area in Chapter 3), has a clearly defined ridge-furrow topography, with lush vegetation growing on the ridge surfaces, and water is drained effectively through the furrows.

More recent realignment schemes, such as the Wallasea North and Karrendorfer Wiesen sites (mentioned above) have learnt from experiences gained at sites such as Tollesbury and taken advantage of existing creek systems and excavated new ones to facilitate better drainage before breaching existing defences. As Hazelden and Boorman (2001) note, the drainage patterns established before realignment will control those that become established in the developing saltmarsh, and trying to excavate creeks or break up over-consolidated sediments after flooding is extremely difficult (Brooks et al., 2015).

Looking to the future, the further development of the Tollesbury realignment site is unclear. Deposition is expected to continue, with a gradual infilling of the site up to a level similar to that of the surrounding natural marshes (Garbutt et al., 2008). This does assume, however, that there will be no reduction in sediment supply to the region,
a prediction that cannot be guaranteed and could be affected by coastal protection measures, dredging and harbour expansion (Nicholls et al., 2000). If sediment does continue to accumulate, the sustained colonisation and succession of vegetation is not guaranteed either. Garbutt et al. (2006b) predict two potential future states, the first of which sees the development of a marsh system fronted by mudflats, with a developed creek system draining them. Alternatively, if an adequate creek system fails to develop, a larger area of waterlogged mudflat will persist, fringed by a relatively static monoculture of halophytes. A continued long-term monitoring programme is thus needed to see if the full potential of the scheme, both in terms of habitat restoration as well as flood defence will be realised.

7.2 Sedimentation Processes at Tollesbury

Chapters 4 and 5 of this thesis examined the controls on the sedimentation processes in both the natural and restored saltmarshes at Tollesbury. An extensive dataset was collected through a series of field campaigns between May 2011 and February 2012, and in May 2012 and March 2014. Data were collected on the spatial variation in sediment deposition, retention, suspended sediment concentration, elevation, tidal conditions and wind, as well as sediment properties and vegetation characteristics. The role of vegetation was explored further in Chapter 5, with additional data collected from the realignment site. The effect of vegetation on the hydrodynamics within the site was simulated using a two-dimensional numerical model, and these results were presented in Chapter 6.

7.2.1 Controls on Sediment Deposition

The results presented from both the seasonal and intensive monitoring field campaigns show considerable spatial and temporal variation in both sediment deposition and sediment retention across the Tollesbury marshes. No one specific environmental variable accounts for the variance on its own; instead there are a series of weak relationships, such as between mean high water level, water depth, suspended sediment concentration and proximity to the nearest source of sediment. The strength of these relationships differs considerably between the field campaigns, however, with some correlations even switching from positive to negative across deployments, or from strong associations to no influence at all.
In some studies, just one or two environmental variables control most of the spatial variation in sediment deposition. Elevation has traditionally been viewed as the dominant factor influencing rates of deposition (Steers, 1948, 1959; Pethick, 1981; Wolaver et al., 1988); elevation is directly linked to the frequency and length of tidal inundation and longer hydroperiods provide greater opportunity for sedimentation (French and Spencer, 1993; Cahoon and Reed, 1995). Therefore, marsh areas lower in the tidal frame should experience higher rates of deposition than those at higher elevations. Sediment deposition is rarely as simple as this, however. Marion et al. (2005) found significant variability in sedimentation at sampling points at the same elevation, whilst French et al. (1995) recorded the highest sedimentation values within the middle of the elevation range at their field site at Hut Marsh, Norfolk, as other factors overrode the influence of topography. At Tollesbury, elevation on its own has essentially no relationship with vertical sediment flux in the seasonal field campaigns and only a very poor negative correlation was found in the intensive monitoring periods.

The supply of sediment to different areas of saltmarshes is one of the main factors that has the potential to modulate the control of elevation on deposition. Tidal creeks deliver tidal flows and sediment within saltmarsh systems, and areas in close proximity to these creeks tend to receive higher volumes of sediment, an observation recorded in many studies (French et al., 1995; Leonard et al., 1995; Leonard, 1997; Reed et al., 1999; De Groot et al., 2011). The effect of creek distance is highly dependent on local geomorphology and hydraulic conditions, however. In the Tollesbury realignment site, the source of all sediment entering the marsh system is the breach in the sea wall, the location from which distance to each sampling point was calculated in this study and compared with sediment deposition. Both the seasonal field data and the later intensive periods saw no correlation between distance to the breach and deposition, with some of the closest sampling points recording the lowest vertical sediment fluxes. However, it has been shown in both Chapter 3 and Chapter 6 that the areas closest to the breach are strongly ebb dominated and experience strong flows that act to inhibit deposition. Indeed, sediment flux rose with increasing distance away from the breach at these sampling points.

It is clear that sedimentation on saltmarsh surfaces is a very multifaceted process that is controlled by the interactions of a complex set of variables and forcing agents. As well as the factors mentioned above influencing spatial patterns, temporal variability can be caused by variations in water levels, suspended sediment concentration, wind and wave patterns, vegetation abundance, biological parameters and storm activity.
(Leonard et al., 1995; Brown, 1998; Marion et al., 2009). Therefore the simplest models relating accretion to frequency of submergence and proximity to tidal channels may not be applicable to all marsh types at all times of the year (Brown, 1998). This study has demonstrated the high degree of variability associated with sedimentation and the synergistic relationship of a number of factors that control sediment deposition in a very small tidal marsh system. In order to more fully understand the patterns and processes of sediment transport and deposition, an extensive sampling programme of biological, physical, sedimentological and geomorphological parameters would be required.

7.2.2 Role of Vegetation

For more than a century, saltmarshes and other coastal wetland habitats have been recognised for their ability to stabilise shorelines and protect coastal communities against flooding and erosion (Shaler, 1886; Yapp et al., 1917). The widely-held traditional view of marsh vegetation is that it acts as a baffle to tidal currents and wave energy, dampening flow, which in turn leads to less erosion and higher accretion rates on the sediment surface (Kobayashi et al., 1993; Leonard and Luther, 1995; Woolnough et al., 1995; Möller et al., 1999, 2001; Leonard and Reed, 2002; Morris et al., 2002; Langlois et al., 2003; Mudd et al., 2010; Stammermann and Piasecki, 2012). The presence of vegetation increases drag forces on the currents that pass through them, with stems and leaves and even roots resulting in higher friction compared to bare, unvegetated mudflats (Neumeier, 2007; Koch et al., 2009). The reduction in flow velocity and turbulence limits vertical mixing, significantly increasing sediment deposition and reducing resuspension in marsh areas (Langlois et al., 2003; Gedan et al., 2011). Greater stem density, plant height, and a more complex structure (e.g. shrubs vs. grasses) increase the friction factors and further reduce kinetic energy (Gleason et al., 1979; Leonard and Croft, 2006).

The data collected from the Tollesbury realignment site and natural marshes and presented in Chapters 4 and 5 of this thesis are not fully in agreement with these assertions. Across the seasonal field campaign, both the presence and increasing abundance of vegetation had a minimal influence on the mass of sediment deposited, with both the mean and median vertical settling flux being almost identical between the vegetated and unvegetated sampling points. In the intensive monitoring periods in May 2012 and March 2014, sediment fluxes recorded in the bare mudflat areas were actually statistically higher than those obtained from the marsh regions. Figure 7.5 shows the
vertical settling flux recorded on every sediment trap deployed throughout the fieldwork programme, separated into whether the sampling point was vegetated or unvegetated. The associated descriptive statistics are listed in Table 7.6. A vegetated point is one with a presence of marsh plants, irrespective of total abundance. Sampling points with only Enteromorpha or Ulva species are not counted as a vegetated point. Although there is considerable variability in both categories of sampling points, sediment flux is on average higher in the unvegetated areas, with a median of 2.75 g m$^{-2}$ tide$^{-1}$ compared to

![Figure 7.5: Comparison of sediment flux recorded at vegetated and bare sampling points throughout the entire field campaign period](image)

All data collected from both the seasonal and intensive monitoring campaigns is shown, and includes sampling points from within the realignment and in the natural marshes at Tollesbury.

Table 7.6: Descriptive statistics for sediment flux recorded across the entire field campaign period at vegetated and bare sampling points

<table>
<thead>
<tr>
<th>Statistic</th>
<th>Vegetated points (g m$^{-2}$ tide$^{-1}$)</th>
<th>Bare points (g m$^{-2}$ tide$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>3.29</td>
<td>3.61</td>
</tr>
<tr>
<td>Median</td>
<td>2.33</td>
<td>2.75</td>
</tr>
<tr>
<td>Minimum</td>
<td>0.01</td>
<td>0.21</td>
</tr>
<tr>
<td>Maximum</td>
<td>29.23</td>
<td>22.27</td>
</tr>
<tr>
<td>25$^{th}$ Percentile</td>
<td>1.19</td>
<td>1.27</td>
</tr>
<tr>
<td>75$^{th}$ Percentile</td>
<td>4.31</td>
<td>4.66</td>
</tr>
<tr>
<td>Data count</td>
<td>1124</td>
<td>480</td>
</tr>
</tbody>
</table>
2.33 g m\(^{-2}\) tide\(^{-1}\) in the vegetated marsh points. Although this difference is small, as a whole there is a statistically significant difference between the two datasets, as indicated by the results of the Kruskal-Wallis test: \(H(1) = 5.25, p = 0.0219\). Therefore it can be asserted that sediment deposition is higher on the unvegetated mudflats at Tollesbury than the vegetated marsh areas.

Although this finding is contrary to many other studies, there is a growing body of research that has called into question the traditional view of vegetation playing a key role in increasing sedimentation, and it is therefore not without precedent. Studies by Harper (1979), Brown (1998) and Boorman \textit{et al.} (1998) all found that there was no evidence to suggest that vegetation enhanced deposition in saltmarshes, or that it only played a very minor role in accretion. Boorman \textit{et al.} (1998) actually collected data at the Tollesbury natural marshes and found no correlation between either the presence of vegetation or vegetation height and deposition. Pethick and Burd (1996) found deposition was inversely related to vegetation density, with lower rates recorded under dense vegetation compared to areas of less dense vegetation and open mudflat areas. Silva \textit{et al.} (2009) measured higher sediment fluxes in unvegetated areas indicating that, at times, vegetation can have a negative influence on sedimentation processes.

Saltmarsh hydrodynamic studies have shown that vegetation has the potential to not just attenuate tidal currents, but also to increase turbulence and kinetic energy both at the edge of a vegetation canopy and further within it (Leonard and Luther, 1995; Leonard and Croft, 2006; Maynard \textit{et al.}, 2011). In Brown’s (1998) paper, scouring occurred around \textit{Spartina} stems at the vegetation boundary, whilst laboratory flume studies have shown that complex velocity profiles and associated shear stresses develop as water flow encounters vegetation (Pethick \textit{et al.}, 1990). The results from the modelling study carried out in this thesis have also shown vegetation to have both a positive and negative effect on flow velocity across the realignment site at Tollesbury. Further work should be carried out to see what impact this has on sediment deposition.

Instead of acting to increase particle accumulation, it is thought that vegetation is more effective at retaining sediment already deposited on the marsh surface (Pethick and Burd, 1996). Although a small number of studies have found that marsh plants do not reduce resuspension and erosion (e.g. Feagin \textit{et al.}, 2009), the majority observe that binding by roots consolidates and strengthens the sediment surface, increasing sediment retention (Scoffin, 1970; Ward \textit{et al.}, 1984; Shi \textit{et al.}, 2000). This would appear to be the case at Tollesbury, as indicated by the comparison of the percentage of sediment retained at vegetated and unvegetated sampling points throughout the entire field
campaign, shown in Figure 7.6. Despite the range in values for both categories, retention is overall higher in the vegetated points, with a median of 40.4% compared to 32.8% in the bare points (Table 7.7). The difference between the two sampling points is statistically significant: $H(1) = 15.93, p = 0.00007$. It is worth noting, however, that this observation was not found in every individual field campaign.

Following on from the field campaigns undertaken at Tollesbury, a series of model simulations were undertaken with various scenarios examining the effect of the presence of vegetation and varying degrees of roughness on surface water velocities.

![Figure 7.6: Comparison of sediment retention recorded at vegetated and bare sampling points throughout the entire field campaign period](image)

All data collected from both the seasonal and intensive monitoring campaigns is shown, and includes sampling points from within the realignment and in the natural marshes at Tollesbury.

<table>
<thead>
<tr>
<th>Statistic</th>
<th>Vegetated points (g m$^{-2}$ tide$^{-1}$)</th>
<th>Bare points (g m$^{-2}$ tide$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>40.57</td>
<td>33.47</td>
</tr>
<tr>
<td>Median</td>
<td>40.40</td>
<td>32.82</td>
</tr>
<tr>
<td>Minimum</td>
<td>5.75</td>
<td>8.78</td>
</tr>
<tr>
<td>Maximum</td>
<td>82.18</td>
<td>57.97</td>
</tr>
<tr>
<td>25th Percentile</td>
<td>26.84</td>
<td>25.24</td>
</tr>
<tr>
<td>75th Percentile</td>
<td>52.58</td>
<td>43.66</td>
</tr>
<tr>
<td>Data count</td>
<td>333</td>
<td>122</td>
</tr>
</tbody>
</table>

Table 7.7: Descriptive statistics for sediment retention recorded across the entire field campaign period at vegetated and bare sampling points.
within the realignment site. These results were presented in Chapter 6 and added a further layer of complexity regarding our understanding of the influence of vegetation at Tollesbury. A series of transects located throughout the site allowed for a more detailed examination of the attenuation of surface velocity away from the creek edges. Some of the transects showed the expected gradual reduction in current speeds along the profile, with the presence of and increasing roughness of vegetation acting to positively lower the maximum speeds simulated. In other cases, however, velocities increased as they traversed the vegetation boundary, or as friction increased from a less rough vegetation section to a rougher vegetated area. At times, speeds increased as a transect passed through areas with vegetation to unvegetated mudflats, and although these changes could not be directly linked to changes in water surface elevation, there was a generally negative relationship between water levels and maximum velocities. Overall, it was found that increasing the roughness of vegetation did not conclusively result in conditions more favourable for sediment deposition, but vegetated marsh areas did generally tend to retard ebb currents more effectively than mudflats.

Very few high-resolution modelling studies have been undertaken to investigate the effect of vegetation on saltmarsh hydrodynamics and sedimentation processes, so it is not easy to compare the results from this thesis with other work. Simulations carried out by Temmerman et al. (2005b) showed that at the edge of large vegetation areas, flow velocities were reduced and sediments rapidly deposited and trapped due to the increased friction caused by the saltmarsh plants. The results presented in this thesis do not definitively corroborate or contradict their findings regarding velocity reduction at the vegetation margin. The high variability, and at times, increased velocity in mudflat sections between vegetated areas (e.g. transects 3 and 4, Figures 6.19 and 6.20) found at Tollesbury is, however, similar to that found by Temmerman et al. (2005b) in their Netherlands case study site. They found that between vegetation patches, flow velocities were enhanced leading to reduced sediment deposition, which would have likely led to increased erosion (although this was not modelled specifically).

An attempt was made to integrate field observations regarding the species zonation in the Tollesbury realignment site into the model simulations by defining different vegetation zones within the model domain, and relevant Manning’s n friction coefficients were applied to each one. Although this could be assumed to increase the accuracy of the model runs, and improve the key findings from this work, such an approach is rather reductionist and simplistic. Representing vegetation through the indirect use of higher roughness coefficients does not account for the influence of
vegetation over the whole water depth and does not allow for the study of the three-dimensional influence of the plant canopy on the flow (Temmerman, et al., 2005b). A better approach would be to utilise a 3D model and gather more detailed data regarding vegetation zonation and characteristics.

It has been shown that the results from the point-based field data and extraction of data points along transects from the modelling simulations tend to show that vegetation has no significant influence over sediment deposition or tidal current attenuation. This is contrasted, however, with the broader, site-wide model output maps (e.g. Figure 6.13), and by comparing the maximum velocities from the constant (i.e. unvegetated) friction scenario and the highest roughness scenario (Figures 6.23 and 6.24). These maps show that, when examined at a larger spatial scale, both the presence of and increasing roughness of vegetation does have a significant effect on reducing flow velocity. In the high friction, high flow scenario (Figure 6.24(b)), velocities throughout the realignment site are reduced compared to the unvegetated simulation (Figure 6.23(b)). The areas of highest flow around the breach and main channels are smaller and restricted to locations closer to the creek edges, whilst zones of low velocities around the sea walls encroach further towards the mudflat areas. This could have resulting benefits on sedimentation in both vegetated and bare areas, and thus overall marsh dynamics. Further work would be beneficial to address these issues of different results depending on the scales investigated, especially between field data and modelling studies.

Although the results from this study have indicated the role of vegetation to be less important than many previous studies have shown, in a similar manner as when discussing the other processes controlling sedimentation, there is significant variation both spatially and temporally and patterns cannot always be explained or predicted (Temmerman et al., 2003b). This has been a fairly short-term study and at only one case study site, and the longer-term rates of elevation change may well be different due to processes such as compaction (Marion et al., 2009). Further research, with a more extensive data collection, would assist with making firmer conclusions. What this thesis does do, is to build on the work undertaken by others examining the reciprocal linkages between biological communities and geomorphic processes and landforms in the field of biogeomorphology (e.g. Phillips, 2006; Corenblit et al., 2007). In saltmarshes, biological processes are controlled by, and in turn, control geomorphological processes, and it has been believed for some time that halophytic vegetation always leads to a reduction in water velocity and therefore an increase in sedimentation rates. As
mentioned in Section 1.2.2, however, these complex non-linear linkages can be both positive and negative in nature, and experience dynamical instability. The results presented here add to the growing body of research that demonstrates that it cannot be assumed that saltmarsh vegetation is critical to the sedimentation process.

7.3 Summary

This chapter has sought to compare the results obtained from this thesis with those from other studies, and examine their implications. No evidence of erosion was found in the natural marshes surrounding the Tollesbury realignment site, and it has been shown that this is not without precedent, as former estimates of marsh extent and rates of erosion were erroneous. The realignment site itself has not performed as well as other sites around the UK and Europe, with only half of the site being colonised by saltmarsh vegetation after 18 years. This has been attributed to a lack of proper drainage, resulting in large areas of water-laden sediments that inhibit plant growth. The patterns and processes controlling sedimentation are highly complex, with none of the typical associations being found between sedimentation and a range of environmental variables. The vegetation at the Tollesbury marshes has little effect on sediment deposition, but has a greater influence over the prevention of resuspension, a finding that contributes to a growing volume of research into saltmarsh biogeomorphology.
8. Conclusions

This thesis has investigated the biogeomorphological development of the Tollesbury realignment site in the Blackwater Estuary, Essex, examined the processes of sedimentation within the site and the influence of vegetation on sediment accumulation. Saltmarshes are critically significant environments, both within the UK and around the world as they provide a range of ecosystem functions and services (Frey and Basan, 1985; Luisetti et al., 2011). Saltmarshes provide important habitats for a range of resident and migrating birds, fish, invertebrates and other small animals (Doody, 1992b; Brown and Atkinson, 1996; Tinch, 2003; Mander et al., 2007). They also support a considerable diversity of vegetation species (Adam, 1981; 1990). Marshes play a key role in the nutrient cycle, transforming and recycling nutrients and supplying organic matter to the wider coastal ecosystem, as well as acting as a sink for heavy metals and other contaminants (Hughes, 2004; Barbier et al., 2011). Saltmarshes and other wetlands around the globe sequester significant amounts of carbon, and contain up to a quarter of the world’s soil carbon, despite only occupying five per cent of the land area (Chmura et al., 2003; Cundy et al., 2005). One of the most important benefits provided by saltmarshes is their ability to attenuate wave energy and protect against coastal flooding and erosion (Brampton, 1992; Dean, 1978; Möller et al., 2001). The increased friction associated with the plant structures retards tidal currents and waves (Shi et al., 2000; Neumeier, 2007; Coulombier et al., 2012); saltmarshes fronting sea walls significantly reduce their construction and maintenance costs due to the protection they afford (Toft et al., 1995; King and Lester, 1995).

Throughout history, saltmarsh extent around the world has risen and fallen due to both natural influences and human interference (Doody, 1992b; Allen, 2000; Singh Chauhan, 2009). Over the past 200 years, however, there has been a continued trend towards the deterioration and erosion of marshes, caused by land reclamation, sea-level rise, increased storminess and wave activity and the enlargement of tidal creek systems within marsh interiors (Harmsworth and Long, 1986; Shennan and Horton, 2002; Temmerman, et al., 2004; Crowther, 2007; Kirwan et al., 2008). Most concerning is the interaction between rising sea levels and the presence of fixed coastal defences, causing coastal squeeze (Dixon et al., 1998; Cooper et al., 2001). Across England, 100 hectares of saltmarsh is estimated to be lost per year (Pye and French (1993b). Predicted sea level rises of up to a metre by the end of the century will only increase the rate of marsh
loss (Hartig et al., 2002; FitzGerald et al., 2008; Church et al., 2013). Various management actions are being considered to protect and restore these habitats. One such option is managed realignment, which involves breaching sea defences to create intertidal habitats on formerly reclaimed land, allowing the tide to inundate the surface and return it to saltmarsh (Morris et al., 2004; Alexander et al., 2012; Mossman et al., 2012b).

Although saltmarshes have been studied by researchers for over 100 years (e.g. Shaler, 1886; Yapp et al., 1917), fundamental questions still remain regarding the processes of sedimentation and there have been recent doubts over the influence of vegetation on sediment accumulation (Allen, 2009; Silva et al., 2009; Mudd et al., 2010). Traditionally, halophytic vegetation has been recognised for its ability to reduce erosion and enhance deposition through the dissipation of tidal and wave energy (French et al., 1995; Allen and Duffy, 1998a; Cahoon et al., 2000; Möller, 2003). Recent research, however, has found that vegetation is not always as important as previously thought (Coulombier et al., 2012; Moskalski and Sommerfield, 2012), and in some locations has a minimal or even a negative effect on sedimentation processes (Feagin et al., 2009; Temmerman et al., 2007). This thesis has aimed to examine the controls on sediment accumulation processes and investigate the role of vegetation in deposition and retention. The research has been conducted via an intensive case study of a managed realignment site and adjacent natural marshes at Tollesbury in the Blackwater Estuary, Essex, in eastern England.

To provide context to a series of field campaigns, the evolution of the realignment site was first examined through a series of aerial photographs. A supervised classification was carried out to quantify the rate at which the site was colonised by vegetation. Plant growth has been fairly slow, and at a much slower rate than most other realignment sites in the UK and Europe. After four years of tidal inundation, only one tenth of the site had been vegetated; this compares with 62% – 100% of saltmarsh cover at other sites in East Anglia (Davy et al., 2011; Freiss et al., 2012; ABPmer, 2011). In 2013, 18 years after tidal inundation was restored, 50.2% of the 21-hectare site was vegetated. The main reason for this slow development is likely to be the lack of effective drainage in the site due to the presence of an impervious aquaclude, formed from the hard soil layer of the former agricultural field (Boorman et al., 2002; Crooks et al., 2002; Brooks et al., 2014). It is unclear as to how the site will continue to develop in future, and it is likely a significant proportion of the site will remain unvegetated.
(Garbutt et al., 2006b). Rates of sedimentation were initially very high immediately following the breaching, at 31.6 mm yr⁻¹ between 1995 and 1996, and fell to 9.8 mm yr⁻¹ in the period 2002-2007 (Garbutt et al., 2008).

Despite the suggested context of rapid rates of saltmarsh loss within southeast England (Burd, 1992; Cooper et al., 2001), no evidence for erosion was found between 1997 and 2013 within the natural marshes surrounding the realignment site at Tollesbury. As this result was unexpected, former estimates of marsh extent and erosion were re-examined. These were found to have been calculated using very simple methods, and thus tended to be over-estimates. A more recent survey by the Environment Agency provides much more accurate estimates of marsh extent, and can be used as a baseline survey for the future (Phelan et al., 2011). Further work is required, however, to increase the accuracy of this product throughout the country. The EA’s estimate of natural marsh extent at Tollesbury was 4% higher than that produced as part of this thesis, partly down to operator error. Significant inconsistencies also exist in the EA extent, and adding geomorphological complexity in how the internal creek system is digitised will improve calculations of saltmarsh erosion over time.

The results of a series of seasonal field campaigns at Tollesbury, undertaken between May 2011 and February 2012, showed high variability in sediment deposition and retention over both time and space. None of the environmental variables measured, such as mean high water, suspended sediment concentration, hydroperiod, elevation and distance to sediment source, had strong relationships with measured sedimentation. Instead, sediment accumulation is controlled by complex interactions between all of these variables and others. Simplistic models relating accretion to frequency of inundation and proximity to the nearest tidal channel are clearly not applicable to all marsh locations at all times. What was clearly demonstrated, however, was that the presence and abundance of vegetation, in itself, has minimal influence over the mass of sediment deposited in both the realignment site and the natural marshes.

The influence of vegetation was explored further during two intensive field campaigns that centred mainly on the realignment site in May 2012 and March 2014. In both of these periods, a greater mass of sediment was deposited in the bare, unvegetated mudflats than at sampling points within the vegetated marshes. In the 2014 field programme, sediment flux was found to peak just in front of the boundary between the vegetated marshes and the bare mudflats. Sediment retention was also highest in these locations. Vegetation abundance was inversely correlated with deposition, and only a
marginally positive relationship existed between abundance and retention. Species composition was indicated to have very little effect on either parameter. The trapping efficiency of the vegetated areas was shown to be higher than the mudflats in May 2012, but this trend was reversed in March 2014, adding weight to the complex nature of the processes operating at Tollesbury.

With all the field data combined, sediment deposition was statistically higher at the unvegetated sampling points. Although unexpected, this small role that vegetation appears to have over the sedimentation processes is not without precedent. An ever-increasing number of studies are overturning the previously held assertion that vegetation always results in increased accumulation (Harper, 1979; Brown, 1998; Boorman et al., 1998). Instead, it has been shown that the presence of vegetation can increase turbulence and shear stress in the water column, providing conditions that are unfavourable to particle settling (Leonard and Luther, 1995; Leonard and Croft, 2006; Maynard et al., 2011). Sediment retention was shown to be statistically higher within the marsh areas, and thus it is asserted that vegetation predominately acts to reduce the resuspension of particles already deposited on the sediment surface. The results from the modelling study undertaken within the realignment site show the presence and increasing friction of vegetation has both a positive and negative effect on flow velocity depending on the sampling points examined and the depth of water within the site. The various simulations do suggest, however, that vegetation may play a role at a larger spatial scale in mediating flow patterns on both bare and vegetated surfaces, which may have resulting benefits on sedimentation within the overall mudflat / saltmarsh system. This work adds to the wider field of biogeomorphology, whereby the non-linear interactions between biologic and geomorphic processes can be both positive as well as negative in nature, making complex behaviour common in saltmarsh systems (Phillips, 2006).

In conclusion, the results from this thesis add weight to the growing body of research that asserts that the influence of vegetation in facilitating sediment deposition is not as critical as has been traditionally assumed. However, it is clear that there are significant variations over time and between the case studies examined and there are likely important local factors at play. Vegetation in the managed realignment site at Tollesbury has colonised slowly and its future development is uncertain. Restored marshes are not generally equivalent to natural marshes in either form or process, and are unlikely to meet the requirements of the national and international legislation which
requires marsh habitats lost to coastal erosion to be replaced. However, considering it has been shown that saltmarshes in the UK, and Essex in particular, are not being lost as rapidly as previously thought, further research should be carried out to determine if the managed realignment of large stretches of coastal defence is actually necessary.
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