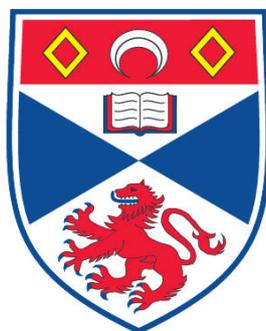


**MEASURING AND UNDERSTANDING BIOGENIC INFLUENCES
UPON COHESIVE SEDIMENT STABILITY IN INTERTIDAL
SYSTEMS**

James Edward Saunders

**A Thesis Submitted for the Degree of PhD
at the
University of St. Andrews**



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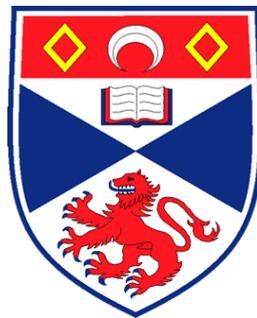
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Measuring and understanding biogenic influences upon cohesive sediment stability in intertidal systems

James Edward Saunders



A thesis submitted in partial fulfilment of the requirements for the degree of
Doctor of Philosophy, at the University of St Andrews

School of Biology
University of St Andrews
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Vardy, S., Saunders, J.E., Tolhurst, T.J., Davies, P.A. & Paterson, D.M. (2007). Calibration of the high-pressure cohesive strength meter (CSM). *Continental Shelf Research*. **27**(8): 1190-1199.

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Abbreviations

Km	Kilometres
m	Metres
cm	Centimetres
mm	Millimetres
nm	Nanometres
μm	Micrometres
g	Grams
mg	Milligrams
μg	Micrograms
l	Litres
ml	Millilitres
μl	Microlitres
Nm^{-2}	Newtons per square metre
$^{\circ}\text{C}$	Degrees centigrade
<i>Ab</i>	Sample Absorbance
rpm	Revolutions per minute
psi	Pounds per square inch
CSM	Cohesive Strength Meter
π	Pi
n	Number of replicates
s.e.	Standard error
d.f.	Degrees of freedom
r_s	Spearman rank correlation
R	ANOSIM similarity value
H'	Shannon-Weaver Diversity
S	Bray-Curtis Similarity Index
n-MDS	non-metric Multidimensional Scaling
ANOSIM	Analysis of Similarity
SIMPER	Similarity Percentage
PCA	Principle Component Analysis
MPB	Microphytobenthos
EPS	Extracellular Polymeric Substances
SPM	Suspended Particulate Matter

Abstract

Intertidal cohesive sediment systems are found throughout the world in areas of low hydrodynamic energy. These systems are ecologically and economically important but are under pressure from global warming, sea level rise and other anthropogenic influences. To protect and conserve these systems it is important to understand the sediment dynamics, especially the erosional properties of the sediment. The study of sediment erosion and transport is complex, encompassing biological, chemical and physical properties of the ecosystem. This thesis contributes towards this area of research, firstly in regard to the methods used to measure sediment erosion on exposed and submerged sediments and secondly with respect to assessing influences upon sediment stability through changes in the ecosystem, comprising of both the sediment environment and the macrofaunal community.

Chapter 3: In partnership with Sediment Service a thorough re-evaluation of the Cohesive Strength Meter (CSM), a commercially available device used to measure surface sediment strength, was performed. New components, deployment method and calibration protocol were devised and tested. The new design was not effective, but the deployment and calibration have improved the ease of use and interpretation CSM data.

Chapter 4: The study of intertidal sediment stability was conducted during the submerged period of the tidal cycle. Protocols and methods were devised or modified to sample submerged sediments with the aim to determine how sediment properties are affected by submersion and the resulting effect on sediment stability. Sediment stability increased with submersion. The existence of a fine layer of sediment on the surface, similar to the fluff layer found in submerged sediments, is given as a suggested explanation as it may be removed by the incoming tide. However, no other changes in sediment properties were detected. This may be due to flaws in the methods used in detecting fine scale changes in the sediment surface. *In situ* and laboratory experiments revealed contrasting effects of submersion on sediment

stability with disturbance from the sampling and movement of sediment from the field to the laboratory given as an explanation for this.

Chapter 5: The influence of the ecosystem engineering polychaete *Arenicola marina* on sediment properties was examined with an exclusion experiment. *A. marina* was excluded from five 20m² plots on an intertidal mudflat on the German island of Sylt. A holistic approach was used to measure the ecosystem, including a range of biotic and abiotic sediment properties as well as the macrofauna community. It was hypothesised that *A. marina*'s exclusion would alter the macrofaunal community and increase sediment stability. However, there was no consistent change in the macrofauna community or sediment environment with the exclusion of *A. marina* and subsequently no change in sediment stability.

Chapter 6: The impact of bait digging for *A. marina* was examined with six 5m² plots dug up and *A. marina* removed, the plots were then monitored over a three month period. Bait digging disturbance was expected to have an impact upon the sediment environment and macrofauna community, resulting in a reduction in both sediment stability and microphytobenthic abundance. However, bait digging had minimal impact on the macrofauna community and caused no change in the sediment environment, despite the removal of a large proportion of the *A. marina* population. No change was recorded in the sediment stability or biomass of the microphytobenthos, indicating that with the exception of removing *A. marina*, bait digging of this nature was not detrimental to the sediment ecosystem. However, the consequences of larger, longer term digging operations can not be determined from this work and further studies are suggested.

The study of intertidal sediment stability was progressed with advances made in methods and protocols. The work highlighted the importance of studying sediment stability as an ecosystem function through a holistic ecosystem approach rather than isolating individual variables.

Chapter One

General Introduction

1.1 Intertidal cohesive sediment systems

Intertidal cohesive sediment systems are common throughout the world and found in estuaries, lagoons and sheltered marine areas where the hydrodynamic regime is dominated by slow moving, low energy currents (Townend, 2002). These conditions allow fine sediment particles to fall out of suspension and be deposited on the bed, creating an environment of mudflats, salt marshes and mangroves, each defined by characteristic flora and fauna. In more exposed areas, with greater hydrodynamic energy, finer sediments rarely come out of suspension or are quickly eroded, leading to the creation of sandy beaches where only larger particles remain or rocky shores from which all sediment is removed. Intertidal cohesive sediment systems are mainly comprised of fine sediment particles, predominantly clays such as illite, kaolinite, chlorite and montmorillonite (Dyer, 1973; Whitehouse *et al.*, 2000). This creates a habitat with very different physical and chemical properties to systems composed of larger, non-cohesive sediment such as sandy beaches.

1.2 The importance of intertidal cohesive systems

1.2.1 Primary production and productivity

Despite their barren appearance intertidal coastal sediment systems are amongst the most highly productive ecosystems in the world with carbon production rates of between 29 and 324 gCm⁻² (MacIntyre *et al.*, 1996; Hagerthey *et al.*, 2002). This is comparable to the productivity of temperate forests (Bertness, 1999). This productivity supports a trophic web of organisms, including important fish nurseries and many permanent and migratory bird populations (Lee, 2001). As a result many mudflats and salt marshes within the UK have existing legal protection as Special

Areas of Conservation (SACS), Special Protected Areas (SPA) or Ramsar site designations (Lee, 2001). Indeed, Townend (2002) found that 76% of UK estuaries have some kind of environmental protection order.

1.2.2 Coastal defence

Land adjacent to the intertidal zone is often economically important for agriculture, industry or rural development. Protection of this land from coastal erosion is highly important, often involving extensive system management plans. Producing these plans requires an understanding of sediment transport, budgets and flux (Townend & Whitehead, 2003) since the construction of inappropriate developments can change the hydrodynamic regime and result in the erosion of the systems that are under protection (e.g. Ryu, 2003). Historically, the protection of land has been based upon the building of “hard” engineered systems such as sea walls (Lee, 2001). However, the increasing maintenance cost of hard defences means natural or “soft” options are becoming more favourable (Watts *et al.*, 2003). Soft coastal defences utilise natural sediment systems, including increased efforts to protect and conserve established mudflats and salt marshes and the development of new systems. A method which is increasingly being used to promote new systems is that of “managed realignment”, where existing sea defences are deliberately breached and the land behind sacrificed to the sea, allowing it to develop as a salt marsh or mudflat (Watts *et al.*, 2003; Reading, *et al.*, 2004; Paramor & Hughes, 2005). Soft systems can absorb and dissipate the energy of the sea before it reaches valuable land. As these are natural systems they are self sustaining, requiring little or no maintenance, they are often a more economically viable option for coastal defence. Additionally there are many environmental and economic benefits of creating a natural wetland habitat in contrast to destroying such habitat for hard systems (Lee, 2001; Winn *et al.*, 2003; Martin *et al.*, 2005). The regeneration of such habitats is often seen as a great benefit to local wildlife and so is popular and possibly beneficial to local communities. However, such an approach is only viable where land is available for sacrifice, or where the economic benefits of managed realignment outweigh the loss of commercial value of the land.

1.2.3 Pollution sequestering

The riverine input of water into estuaries is often highly contaminated by industrial and municipal waste from upriver sources including heavy metals, complex organic molecules and radioactive elements. Intertidal sediment systems can act as sinks for these pollutants as they can adsorb onto individual suspended sediment particles and be incorporated into the mudflat or salt marsh through deposition and bioturbation (Petersen *et al.*, 1998; Rasmussen *et al.*, 1998; Spencer, 2002; Cundy *et al.*, 2003; Edgar *et al.*, 2003; Lansard *et al.*, 2006; Gerbersdorf *et al.*, 2007). Such pollutants are often highly toxic and non-biodegradable so can remain a threat to the environment for long periods of time (Edgar *et al.*, 2003). Sequestering of a pollutant into the sediment reduces the possibility of it entering a food chain and consequently affecting the health of the marine environment (Cundy, *et al.*, 2003). However, for sediment systems to act as reliable sinks for pollutants requires long-term stability within the system. If the sediment is disturbed or resuspended then the pollutants may re-enter the water column and subsequently the marine environment (Cundy, *et al.*, 2003).

1.3 Threats to intertidal systems

1.3.1 Climate change and sea level rise

The prospect of global warming is a major threat to the existence of mudflats. Average global temperatures are predicted to rise by 1.8-4.0⁰C by the end of the century, resulting in sea level rises of between 0.18 and 0.59m (IPCC, 2007). Increasing sea levels should result in increased deposition of sediments in the upper intertidal zone, allowing the systems to migrate up the shore. However, this is often not possible due to the construction of hard sea defences, resulting in “coastal squeeze” where intertidal systems are squashed between rising sea levels and a permanent land barrier (Kaiser *et al.*, 2005). However, this concept is increasingly being questioned (e.g. Hughes & Paramor, 2004) as some systems should be able to increase in height with increased deposition even without moving further up shore.

Global warming is also predicted to result in an increase in the occurrence and severity of storm events (IPCC, 2007), both of which will increase the frequency and

strength of wind driven turbulent currents, increasing the annual erosional pressure on intertidal systems (Brown & McLachlan, 2002).

1.3.2 Human development

Estuaries in which mudflats and salt marshes are common have been hubs of human settlement and development for centuries. As the world's population has grown over hundreds of years, estuaries have been used for an increasing variety of needs including industry, agriculture, rural development, conservation and recreation. The majority of goods transported around the world are, and will continue to be, moved via the seas and oceans, requiring the building of ports and harbours. These have historically been built within rivers as they provide naturally sheltered areas, with the location of ports progressing downriver into estuaries as ships have increased in size (Townend, 2002). Associated with ports and harbours are secondary businesses and population centres, which add a major pressure on the land around estuaries and coastlines, and with the population predicted increase by 50% by the middle of the century (United Nations Population Division) this pressure will increase dramatically. Agriculture has historically been one of the major land uses supplying food to the increasing population. In the last two hundred years salt marshes and mud flats have been drained with the resulting land protected from the sea and reclaimed for agricultural uses. However, with developing farming practices, transport of crops from overseas and increasing maintenance costs of defences there is less need or economic viability in maintaining this land and some has been returned to the sea.

1.4 Macrofauna community of intertidal cohesive sediment systems

1.4.1 Species composition and diversity

The macrofaunal community within intertidal cohesive sediment systems is relatively simple, based on *in situ* primary production from micro and macro algae, and externally derived organic material from terrestrial and marine sources (Kaiser *et al.*, 2005). The diversity and number of species, trophic levels and life strategies are limited in comparison to most other systems. Molluscs, crustaceans and many different types of worms are the most abundant permanent residents, while fish and bird species are common but their presence is related to the tidal cycle. This lack of

diversity within the system makes them ideal for experimental manipulation (Kaiser *et al.*, 2005), but also relatively simple to define and summarise. An example of this is the common division of animal species into four main functional groups based upon their feeding behaviour (Bertness, 1999; Pearson, 2001; Bolam, *et al.*, 2002), although some species can change their feeding behaviour in relation to the environmental conditions (Gerdol & Hughes, 1994).

Surface deposit feeders

These species live and feed on the sediment surface or create burrows with access to the sediment surface for feeding. Organic debris, microphytobenthic algae (MPB), extracellular polymeric substance (EPS), meiofauna and bacteria constitute the main food sources. This functional group includes gastropods, crustaceans and polychaete worms such as *Hydrobia ulvae*, *Corophium volutator* and *Pygospio elegans*, respectively. Some members of this group can also be sub-classified as grazers depending upon their feeding habits (Bolam *et al.*, 2002).

Subsurface deposit feeders

The majority of these species are polychaete or annelid worms such as *Arenicola marina* and Oligiochaete spp. Their elongated body shape allows easy movement through the compact sediment. The composition of the food sources for these species is essentially the same as that of surface deposit feeders.

Filter feeders

Predominantly bivalve species, these species feed when submerged by using either modified gills or feeding apparatus to filter the passing water for food. This is either a passive process or assisted by active pumping of water across the gills. Bivalves in this group can form extensive beds on the sediment surface (e.g. *Mytilus edulus*) or survive individually by burrowing into the upper sediments for protection and feeding through extending appendages to the submerged sediment surface (e.g. *Cerastoderma edule* and *Macoma balthica*).

Predators

This group can be divided into two sub groups, animals that permanently inhabit the sediment, and those that frequent the system to feed. Permanent residents are mostly polychaete worms such as *Eteone flava* and *Nephtys hombergii* that are themselves

rarely much larger than some of their prey. Additionally some species of crab generally feed from the sediment surface during submersion and bury themselves during exposure. Larger predators are more transitory, either birds during exposure or fish species during submersion. Some of these predators, specifically birds, are highly specialised feeders, feeding off only one species or type of organisms (e.g. the oystercatcher (*Haematopus ostralegus*), while other species are less selective (e.g. the redshank (*Tringa tetanus*) or curlew (*Numenius arquata*)).

1.4.2 Factors shaping community structure

During low tide cohesive sediments retain a large volume of water within the inter particle spaces. Although this may drop during an exposure period (Perkins *et al.*, 2003) it is a gradual process and sediments will very rarely dry out to the level that sandy sediments do. This means the environment within the sediment is relatively stable, certainly in comparison to sandy and rocky intertidal systems, with only minimal changes in temperature and water level (Bertness, 1999). On rocky intertidal zones changing environmental conditions are a major driving force of species zonation, often dictating the upper boundary of a species range (Nybakken, 1997; Bertness, 1999). However, as cohesive sediment environments are more stable, there is little zonation due to abiotic pressures related to exposure duration. Equally, inter-species competition for space and predation that usually influence the lower reach of a rocky shore species (Nybakken, 1997; Bertness, 1999) are less important in cohesive sediments. The three-dimensional nature of the sediment environment (species can live under the surface unlike rocky shores) means space is not such a limiting factor while predation pressure, which is dominated by submerged species in rocky shores, is roughly equally divided between periods of submersion (fish and crabs) and exposure (birds) (Bertness, 1999).

Although zonation does occur in intertidal cohesive sediment systems it is usually very gradual and often dictated by changes in sediment size, which can vary across and along the shore, rather than abiotic factors related to submersion and exposure duration or biotic pressures of competition and predation. Instead, general patterns of species distribution are found related to sediment grain size. Deposit feeders tend to dominate in finer sediment environments, benefiting from high levels of deposited organic material, and filter feeders dominate areas with larger sediments where they

benefit from being able to feed in the water column as the larger sediment contains less organic material (Nybakken, 1997). By extension from this it can be said that the most significant force in shaping the community structure of intertidal cohesive sediment systems is the location and orientation of the system and its dominant hydrodynamic regime, as this largely dictates the sediment grain size distribution (Soulsby, 1997).

At a local level, small scale factors will also influence the composition of the community, these include localised inputs of organic material (Levinton & Kelaher, 2004), fresh water (Zipperle & Reise, 2005), pollution and disturbance (Peterson & Rosenberg, 1978), and variation in sedimentation rates (Anderson *et al.*, 2004). The influence of these factors will promote heterogeneity within a system which is relatively independent of shore position or submersion/exposure duration.

1.5 Sediment particle properties

1.5.1 Cohesive and non-cohesive sediments

Sediment particles are generally classified by size gradings such as the Wentworth scale: gravels (>2mm); sands (2mm-62.5 μ m); and silts and clays (<62.5 μ m). Below the boundary of 62.5 μ m sediments display cohesive properties, and hence this is the division of cohesive and non-cohesive sediments.

Cohesive particles have a plate-like structure with large flat negatively charged faces and small positively charged edges. In fresh water the attraction of opposing charges on the edges and faces of adjacent particles forms a weak electrostatic van der Waals interaction, resulting in a low level of cohesion between the two particles. This situation changes in salt water as soluble anions and cations are attracted to the particle faces and edges respectively, these form an electrical double layer around each particle. This causes cohesion between the particles based on ionic interactions of the electrical double layer, rather than the electrostatic interactions. The ionic bonds are stronger than the existing electrostatic attractions and therefore the strength of cohesion between particles is greater. Subsequently it can be seen that the strength of cohesion between particles increases with increasing salinity (Whitehouse *et al.*,

2000). Larger, non cohesive, sediment particles have a different composition without strong ionic surfaces (Soulsby, 1997) and have a smaller relative surface area (Jickells & Rae, 1997) and as such do not possess the cohesive properties of smaller sediments.

Natural sediments are rarely well-sorted and mostly comprise of a mixture of grain sizes. For a mixture of sediments to display cohesive properties only 10% (by mass) of the sediment needs to be cohesive (Whitehouse *et al.*, 2000).

1.5.2 Deposition and erosion of sediment particles

Within any sediment based system individual particles are involved in erosion, transport, deposition and consolidation (the ETDC cycle). The balance of which will be dictated by many biotic and abiotic variables.

Sediment deposition

Sediment accumulates on a submerged sediment surface through deposition of particles from the water column. All particles will naturally settle at the bottom of a water column due to gravity (Soulsby, 1997; Whitehouse *et al.*, 2000). The speed at which they do this is called the settling velocity and is dependant upon the size, shape and characteristics of the particle as well as the density of the medium (Gibbs *et al.*, 1971; Allen, 1985). For a particle to be maintained within the water column, hydrodynamic conditions must be sufficiently energetic for turbulent forces created by water movement to counteract the settling velocity of the sediment particle. If the level of water movement drops these turbulent forces may become insufficient to maintain the suspended particle and it will settle out of suspension and be deposited upon the sediment surface (Whitehouse *et al.*, 2000).

Within the water column cohesive sediment particles can not be considered as individual particles, but as part of the suspended particulate matter (SPM) often expressed as a concentration. Cohesion between particles within the water column can result in the creation of a floccule, a larger particle composed of individual sediment particles. As cohesion between particles is largely related to salinity, flocculation rate and the internal strength of a floccule both increase with salinity, hence the increased flocculation within estuaries compared to the source river (Whitehouse *et al.*, 2000). A floccule is larger than the individual sediment particles

that constitute it and therefore its settling velocity and likelihood of being deposited will both be greater (Gibbs *et al.*, 1971). Flocculation occurs with a collision of particles, so higher SPM concentrations will increase flocculation by increasing collision rates (Dyer, 1973).

Sediment erosion

Water flowing over a sediment surface is slowed down through the friction between the moving water particles and the stationary sediment. At the point of contact between the sediment and water, the water molecules adjacent to the surface are stationary; this is called the no-slip condition. Directly above this layer, water will flow but will be slowed down by the friction with the stationary water below. This effect of friction between layers of water continues into successive layers of water, each moving increasingly quickly as the effect of the sediment surface diminishes until the flow reached the free stream velocity. The increase in speed with each layer is determined by the free stream velocity, with the rate of change termed the velocity gradient. At a given height, depending upon the speed of the water, the influence of the sediment surface will cease and water will flow will reach the free stream velocity with no further increases in velocity with distance from the bed. The zone in which the sediment surface exerts this influence on the flow is called the boundary layer. The effect of water moving over a stationary surface, or of one layer of water moving over a different layer with a different speed is to create shear stress between the two.

The shear stress at the sediment surface is related to the velocity gradient in the boundary layer. In natural systems it is unusual for perfectly laminar flow to occur (low shear), instead flow is usually turbulent reducing the extent of the boundary layer and increasing the shear stress at the bed. If this shear stress contains sufficient energy then it may overcome the forces of gravity and cohesion between particles and result in the entrainment of the particle within the water column, effectively eroding the particle (Soulsby, 1997; Brown 1999). Under highly turbulent flow conditions the boundary layer breaks down and flow is dominated by complex eddies and vortices, although usually a prevailing direction of flow will exist over the water body as a whole. Within these conditions the sediment surface will again be exposed to shear stresses and erosion occurs in a similar fashion. The energy required to erode a sediment particles from the sediment surface is called the critical erosion threshold of

the sediment. This energy is commonly expressed as the value of the shear stress required in Newtons per square metre (Nm^{-2}) (Allen, 1985; Soulsby, 1997; Brown *et al.*, 1999; Whitehouse *et al.*, 2000).

For sand particles, the critical erosion threshold is related to the size and weight of the individual sediment particles with less energy required to erode smaller, lighter particles. This trend continues until the particle diameter decreases below $62.5\mu\text{m}$ where inter-particle cohesive forces start to develop. The critical erosion threshold then increases with further reductions in sediment size, with more energy required to counteract the increasing cohesive forces between smaller particles (Fig. 1.1) (Morgan, 1995; Soulsby, 1997; Whitehouse *et al.*, 2000). Natural sediments are composed of a mixture of sediment sizes, these mixtures of cohesive and non-cohesive sediments are often more stable than well sorted sediment of any size due to complex matrices of interactions (Allen, 1985; Soulsby, 1997).

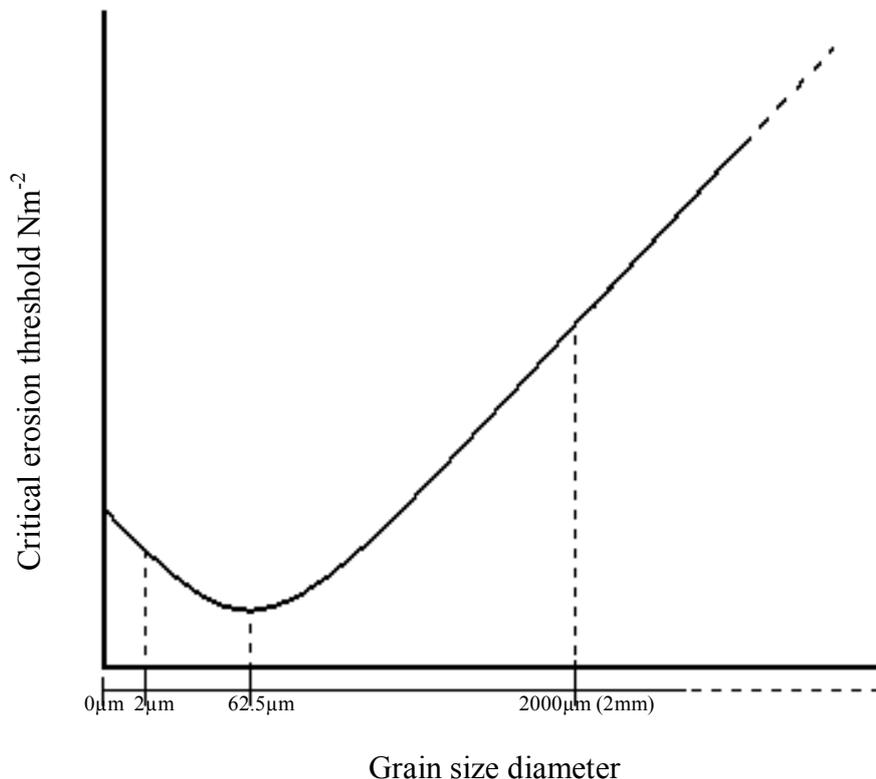


Figure 1.1. The Shields Diagram of changing critical erosion threshold with sediment grain size. Gravels ($>2\text{mm}$) and sands ($62.5\mu\text{m} - 2\text{mm}$) are non-cohesive and critical erosion threshold decreases with reducing grain size. Silts ($2\mu\text{m} - 62.5\mu\text{m}$) and clays ($<2\mu\text{m}$) are cohesive and attractive interactions between particles results in an increase in critical erosion threshold with decreasing grain size.

1.6 General trends in erosion and deposition

The movement of sediment particles within an intertidal system is a highly dynamic process, with multiple physical variables relating to their erosion and deposition.

1.6.1 Hydrodynamic conditions

In sheltered areas that promote the creation of cohesive sediment systems, hydrodynamic forces are produced by tidal currents and wind driven wave action (Le Hir *et al.*, 2000). In normal calm conditions weak tidal currents dominate, waves are small and contain little energy, a situation that is reversed with increasing wind speed (Bell *et al.*, 1997; Janssen-Stelder, 2000). Shear stresses produced by tidal currents are relatively low and rarely exceed the critical erosion threshold of the sediment, allowing deposition of SPM. However, in severe weather conditions wave induced shear stresses increase beyond those produced by the tidal currents and may surpass the critical erosion threshold of the sediment, promoting erosion over deposition (Christie *et al.*, 1999, Janssen-Stelder, 2000; Andersen & Pejrup, 2001).

The sediment composition of intertidal systems is rarely homogenous with gradients of grain sizes occurring across and along the systems. This is a result of varying hydrodynamic conditions. Although not universal, there is a general pattern of decreasing grain size with increased shore height, this is because in the lower shore hydrodynamic conditions are both stronger and last for longer, eroding finer particles. In the upper shore very slow currents occur for only short periods so that fine particles are deposited while the water rarely contains sufficient energy to contain suspended larger particles. Along an intertidal system, especially estuaries, hydrodynamic gradients are common. Grain size trends to increase with exposure, so that the inner estuary is commonly comprised of the finer sediments. Given that grain size has a bearing on the erosion potential of the sediment surface these gradients in grain size will invariably result in varying stabilities of the sediment surface.

1.6.2 Atmospheric conditions

The largest impact of weather and atmospheric conditions on sediment erosion and deposition is indirectly through its influence upon hydrodynamic conditions, largely through wind driven waves (Bell *et al.*, 1997; de Brouwer *et al.*, 2000; Andersen &

Pejrup, 2001; Mitchell *et al.*, 2003; Amos *et al.*, 2004). However, sediment stability and erosion are also directly affected by the weather with prolonged atmospheric exposure, related to increased height on the shore, resulting in increased desiccation and an increase in sediment stability (Widdows *et al.*, 2000a). The effect of rain is often to reduce sediment stability, either through physical disturbance or dilution of stabilising chemicals (Paterson *et al.*, 2000; Tolhurst *et al.*, 2006b), although quick recovery of stability levels after the rain stops has been observed (Paterson *et al.*, 2000).

In temperate regions the conditions promoting erosion or deposition of sediments often follow a seasonal pattern with high deposition rates during calm spring and summer months and erosion occurring with an increase in severe weather in the winter (Andersen & Pejrup, 1999; O'Brien *et al.*, 2000; Andersen *et al.*, 2005). This pattern seems to be site specific as opposing patterns have been observed (de Brouwer *et al.*, 2000). Large storms will have different consequences for each system, Andersen & Pejrup (2001) found landward suspended sediment increased dramatically for several days after a particularly large storm off the Danish coast. This movement of sediment was estimated to account for 40% of the annual deposition at the site. Such differences between sites are often a result of their orientation to the changing atmospheric and hydrodynamic conditions (Ryu, 2003).

1.6.3 Particle and water composition

Within deposited sediment, individual sediment particles will be distributed with varying degrees of compaction or density. This is expressed as the bulk density, a measurement of weight of sediment in a given volume. Related to this are the spaces between sediment particles, which may be filled with water, air and organic components (Tolhurst *et al.*, 2005). Sediments with high bulk density tend to hold less water and are more stable (Underwood & Paterson, 1993 *a, b*; Christie *et al.*, 2000), with a high water content increasing the fluidity of sediments and decreasing their erosion threshold (Fernandes *et al.*, 2006). However, such trends are not universal and often site specific (Christie *et al.*, 2000).

1.7 Biological influences upon sediment stability

The high productivity of intertidal cohesive sediment systems supports a large number of organisms, the activities of which have a large influence upon the overall sediment stability.

1.7.1 Micro-organisms

Mudflats are very biologically active systems with high numbers of bacteria and micro-algae (MacIntyre *et al.*, 1996). Their high concentrations are due to the levels of resources available such as nutrients, light and space. Microphytobenthic algae and bacteria strongly influence sediment stability, primarily through the production of extracellular polymeric substances (EPS), organic materials comprised mainly of a complex mix of carbohydrates (Underwood & Smith, 1998; Taylor *et al.*, 1999; Decho, 2000; Yallop *et al.*, 2000; de Brouwer *et al.*, 2003). Explanations for EPS production vary but it is most often coupled in diatoms with the movement and migration of cells through the sediment (Smith & Underwood, 1998). EPS binds surrounding particles and stabilises the local environment, reducing concentration gradients of environmental conditions such as water content that occur during a tidal cycle and limiting stresses on MPB such as desiccation (Decho, 2000). Diatoms and bacteria produce large quantities of EPS which can result in the formation of biofilm mats on the sediment surface (Decho, 2000) a mixture of cells, sediment particles and EPS. The binding characteristics of biofilms can cause an increase in the critical erosion threshold and stability of the sediment (Holland *et al.*, 1974; Dade *et al.*, 1990; Underwood & Paterson., 1993b; Yallop *et al.*, 1994; Sutherland *et al.*, 1998; Austen *et al.*, 1999; Tolhurst *et al.*, 1999; Paterson *et al.*, 2000; Decho, 2000; Yallop *et al.*, 2000; Andersen, 2001; Staats *et al.*, 2001; Lelieveld *et al.*, 2003; Lucas, 2003; Mason *et al.*, 2003; de Brouwer *et al.*, 2005; Tolhurst *et al.*, 2006c; Widdows *et al.*, 2006). The long history and extensive research in this area reflecting the importance of microbiological stabilisation within intertidal sediments. Changes in sediment stability related biofilm distribution are variable with increases in stability found to double (de Brouwer *et al.*, 2000), treble (Mason *et al.*, 2003) or increase by an order of magnitude (Austen *et al.*, 1999).

EPS quantification is usually based on the carbohydrate content within the sediment (Underwood & Paterson, 1993 *a, b*). Diatom abundance or concentration is rarely measured directly but assessed by the proxy measurement of chlorophyll *a* concentration, which is often in negative correlation with erosion threshold (e.g. Underwood & Paterson, 1993*a*; Austen *et al.*, 1999). The binding properties of EPS are affected by the sediment properties, with small grain sizes with larger more active surfaces increasing the binding influence of EPS (de Brouwer *et al.*, 2000; de Brouwer *et al.*, 2003) and high water content possibly diluting many of the EPS fractions, reducing stability (Paterson *et al.*, 2000).

1.7.2 Macrofauna

Almost every organism inhabiting intertidal sediments will interact with their environment, directly changing the characteristics of that sediment. The consequences of such actions on sediment stability are highly variable, and research on the influences of individual species is extensive (Table 1.1). Widdows & Brinsley (2002) divided macrofauna species into two functional groups; sediment stabilisers or destabilisers (bio-stabilisers and bio-destabilisers). Possibly included within stabilisers could be bio-depositors, species that increase deposition rates of suspended particles (Jie *et al.*, 2001). The methods through which a species interacts with the sediment are highly varied but can usually be placed into one of several categories;

Bioturbation

Bioturbation involves the moving of sediment particles, either vertically or horizontally by an organism. This will be done by almost every organism within the sediment to some extent through their movement, but some species actively move or disturb the sediment as part of their feeding mechanism (Cadée, 2001; Reise, 2002; Widdows & Brinsley, 2002). Bioturbation will disrupt the sediment, usually leaving it less consolidated and increasing its surface roughness, generally reducing its critical erosion threshold (Graf & Rosenberg, 1997; Black *et al.*, 2002*a*).

Burrow and tube building

Many macrofauna species live within tubes or burrows, which are permanent, semi-permanent or temporary. While the formation of temporary burrows and tubes could be considered as bioturbation, more permanent burrows act as physical structures

which can stabilise the surrounding sediment (Black *et al.*, 2002a) or can change properties of the sediment by increasing drainage, again increasing sediment stability (Jones & Jago, 1993; Black *et al.*, 2002a). In their construction, an organism will often secrete mucus to form the tube, which will in turn stabilise the surrounding sediment (Reise, 2002). Tubes and burrows often result in a structure on the sediment surface, either a depression or object protruding from the surface, both of which can increase turbidity and erosion (Graf & Rosenberg, 1997; Reise, 2002; Black *et al.*, 2002a)

Mucilage production

Locomotion within and along the sediment is often accompanied by the production of a mucus trail. This trail can act as a binding agent directly increasing stabilisation or creating large floccules of bound sediment, both resulting in an increase in stability (Reise, 2002; Black *et al.*, 2002a).

Faecal pellet production

Faecal pellets of macrofauna are often larger than the surrounding sediment and can constitute 87% of the upper 5mm of sediment (Austen *et al.*, 1999). Pellets are easily eroded and are removed from the surface before the sediment particles (Minoura & Osaka, 1992). This can have the result of decreasing sediment erosion thresholds, although sediment with a high faecal pellet concentration has a faster settling velocity so may be redeposited soon after erosion, possibly reducing the effect of faecal pellets on the system wide movement of sediment (Andersen & Pejrup, 2002; Andersen *et al.*, 2002; Black *et al.*, 2002a; Andersen *et al.*, 2005).

Biodeposition

Many macrofauna species either actively or passively promote the deposition of sediment. Filter feeding species may capture suspended sediment and deposit it upon the surface while the physical presence of some species, especially when forming extensive structures (e.g. mussel beds), can lead to a reduction in hydrodynamic forcing above the sediment surface promoting sediment deposition (Graf & Rosenberg, 1997; Reise, 2002; Black *et al.*, 2002a; Widdows & Brinsley, 2002).

Table 1.1. Examples of the varying influence of macrofauna species on sediment stability. * Experimental study, ** Correlative observation, *** Review

Species	Behaviour	Influence	References
<i>Corophium volutator</i> (Amphipod)	Deposit feeder, burrowing	Destabiliser	Gerdol & Hughes, (1994)*; Grant & Daborn (1994)**; de Deckere <i>et al.</i> , (2000)*; Defew <i>et al.</i> , (2002)**; Widdows <i>et al.</i> , (2006)**; Huxham <i>et al.</i> , (2006)*
<i>Hydrobia ulvae</i> (Gastropod)	Deposit feeder and grazing. Faecal pellet production	Destabiliser	Austen <i>et al.</i> , (1999)**; Andersen, (2001)**; Andersen & Pejrup, (2002)*; Andersen <i>et al.</i> , (2002)*; Biles <i>et al.</i> , (2003)*; Andersen <i>et al.</i> , (2005)*; Orvain <i>et al</i> (2006 <i>a*</i> , <i>b*</i>); Widdows <i>et al.</i> , (2006)**; Huxham <i>et al</i> (2006)*
<i>Nereis diversicolor</i> (Polychaete)	Deposit feeder Tube builder	Bioturbator, destabiliser and stabiliser	de Deckere <i>et al.</i> , (2001)*; Fernandes <i>et al.</i> , (2006)* Widdows <i>et al.</i> , (2006)**
<i>Ruditapes philippinarum</i> (Bivalve)	Burrowing Suspension feeder	Destabiliser	Sgro <i>et al.</i> , (2005)*
<i>Leptochelia dubia</i> (Crustacean)	Tube building	Stabiliser	Kransnow & Taghon (1997)*
<i>Scrobicularia plana</i> (Bivalve)	Suspension feeder	Destabiliser	Orvain (2006)*
<i>Cerastoderma edule</i> (Bivalve)	Filter feeder	Depositor	Widdows <i>et al.</i> , (2000 <i>a*</i>); Ciutat <i>et al.</i> , (2007*)
<i>Ruditapes philippinarum</i> (Bivalve)	Filter feeder	Depositor	Jie <i>et al.</i> , (2001)*
<i>Crassostrea virginica</i> (Bivalve)	Filter feeder	Depositor	Porter <i>et al.</i> , (2004)*
Oligochaetes (Annelid worm)	Tube builder	Stabiliser	Widdows <i>et al.</i> , (2006)**
<i>Arenicola marina</i> (polychaete)	Deposit feeder	Destabiliser	Defew <i>et al.</i> , (2002)**
<i>Mytilus edulis</i> (Common mussel)	Filter feeder	Depositor	Widdows & Brinsley (2002)***
<i>Pygospio elegans</i> (Polychaete)	Deposit feeder Tube builder	Stabiliser	Bolam & Fernandes (2003)**
<i>Arenicola marina</i> (Polychaete)	Deposit feeder	Destabiliser	Defew <i>et al.</i> , (2002)**
<i>Neomysis integer</i> (Mysid shrimp)	Deposit feeder	Destabiliser	Roast <i>et al.</i> , (2004)**
<i>Macoma balthica</i> (Common Clam)	Deposit feeder and grazer, burrowing behaviour	Bioturbator	Willows <i>et al.</i> , (1998)*; Widdows <i>et al.</i> , (2000 <i>a*</i> , <i>b*</i>); Huxham <i>et al.</i> , (2006)*
Chironomid larvae (Polychaete)	Tube builder	Stabiliser	Ólafsson & Paterson (2004)*

1.7.3 Macrophytic algae

The large algae *Entromorpha spp.* grows on intertidal sediment surfaces in mats and smothers the sediment underneath. The effect of this is to lower the microphytobenthic community in the sediment but increase macrofauna abundance (Defew *et al.*, 2002). The stability of the sediment surface below the *Entromorpha spp.* is usually higher than adjacent bare sediment (Friend *et al.*, 2003a; Romano *et al.*, 2003), but has also been found to be lower (Defew *et al.*, 2002). The presence of the *Entromorpha spp.* will have an influence on the hydrodynamic conditions above the bed, potentially slowing the flow and allowing increased deposition of sediment particles (Defew *et al.*, 2002; Romano *et al.*, 2003), however, the effects of this will be highly dependant upon the density and structure of the *Entromorpha spp.* bed.

1.7.4 Large fauna

Large animals tend to be periodic visitors to intertidal systems, either seasonal migrations or tidally dependant. The impact on sediment stability of such organisms feeding on intertidal sediments is usually negative, with the size of the organism resulting in large disturbance to the sediment surface (Cadée, 1990; Reise, 2002; Cadée, 2001).

1.7.5 Indirect influences on stability

In addition to directly interacting with the sediment, all organisms will interact to some extent with other species, indirectly effecting sediment stability through these actions. Some species of macrofauna are very common on mudflats in high numbers, in some cases abundances of 300 000 and 13 000 individuals per square meter can occur for the gastropod *Hydrobia ulvae* and the amphipod *Corophium volutator*, respectively (Grant & Daborn, 1994; Andersen & Pejrup, 2002). At these densities species will exert a large predatory influence on the microphytobenthos that constitute their food source (Smith *et al.*, 1996; Hagerthey *et al.*, 2002) reducing their abundance within the sediment. Some macrofauna species have been found to feed directly on EPS (Hoskins *et al.*, 2003). Both of these feeding strategies will reduce EPS quantities and therefore its stabilising influence upon the sediment. However, each species is also part of a trophic chain and a species that feeds upon the microphytobenthos will themselves be preyed upon, reducing their impact and allowing higher MPB levels to stabilise the sediment (Daborn *et al.*, 1993). Large

bivalve beds will promote deposition of sediment from the water column, which can result in an increase in the light penetration, increasing MPB abundance and stabilisation (Porter *et al.*, 2004).

1.8 Variation within intertidal systems

By their nature intertidal sedimentary systems are constantly changing with patterns of tidal exposure/inundation, sediment movement, biotic influences and changes in the properties of the sediment. This leads to a highly heterogeneous habitat with changes rarely happening in isolation but as part of a dynamic biotic and abiotic system, with the consequences for sediment stability complex. This has a large bearing on the study of sediment stability, with temporal and spatial variation needing to be accounted for in the design and analysis of any experiment/observations (Cadée, 2001).

1.8.1 Spatial heterogeneity

Influences on sediment stability vary on many scales, with comparisons between such scales often complicated. On a depth scale, sediment is highly variable with the majority of microphytobenthic activity and a significant portion of macrofauna activity occurring within the upper 2mm (Blanchard *et al.*, 2000; Consalvey *et al.*, 2004; Consalvey *et al.*, 2005), although gradients in sediment properties can occur even within this scale (Taylor & Paterson, 1998).

Differences in horizontal scale are common when comparing research, although measurements are often dictated by the aims and scale of the study. For example the influence of algae has to be considered on different scales depending upon the species and the study. Single celled microphytobenthic algae produce EPS that can be shown between individual sediment particles by electron microscopy, and are an integral part of small (2-3cm) patchy biofilms. These biofilms can increase stability by an order of magnitude compared to adjacent bare sediment, although bare sediment is usually more homogenous (Tolhurst *et al.*, 1999; Tolhurst *et al.*, 2006c). While patches of the large algae *Entromorpha spp.*, which usually increased sediment stability (Friend *et*

al., 2003a; Romano *et al.*, 2003) can be measured within a meter but equally can stretch across a whole shore (Defew *et al.*, 2002).

Within a relatively small area of sediment, physical structures or bedforms will influence stability. Structures of biological origin such as tubes protruding from the sediment or faecal casts may have different stabilities to the adjacent sediment (Reise, 2002). While hydrodynamic conditions may create varied bedforms such as ridges and troughs, with ridges tending to have higher stability, possibly due to lower water contents (Widdows *et al.*, 1998; Paterson *et al.*, 2000; Blanchard *et al.*, 2000; Christie *et al.*, 2000).

Sediment conditions will vary within a mudflat related to position on the shore and duration of atmospheric exposure, most often associated with sediment properties such as water content (Christie *et al.*, 2000; Blanchard *et al.*, 2000; Andersen *et al.*, 2002). However, biological variation across the shore is also important with zonation of macrofauna species resulting in contrasting sediment properties and stability based upon their influences, such as mussel beds stabilising low shore sediments (Widdows *et al.*, 1998; Widdows & Brinsley, 2002).

1.8.2 Temporal heterogeneity

Sediment properties will change over a single exposure period. These general changes are often related to sediment dewatering, which leads to higher bulk densities which increases sediment stability (Perkins *et al.*, 2003; Tolhurst *et al.*, 2006a, c).

Biogenic influences upon stability often operate on a cycle or fluctuate in time. Diatoms tend to migrate to the sediment surface shortly after exposure and rapidly increase stability (Consalvey *et al.*, 2004) while in comparatively bare sediment stability will increase with exposure time due to sediment dewatering and increasing bulk density (Paterson *et al.*, 2000; Tolhurst *et al.*, 2003; Perkins *et al.*, 2003; Tolhurst *et al.*, 2006 a, c). However, this pattern is varied and changes will occur between exposure during day or night, with stability in areas of high microbial activity unexpectedly being highest during night sampling, possibly due to degradation of EPS (Friend *et al.*, 2003b).

Seasonal differences in stability are often related to changing weather conditions, calm summer months often resulting in an increase in the deposition of cohesive sediments, possibly stabilising the sediment. Seasonal changes will also have a large influence on the biological influence upon sediment stability. Species abundance and activity will invariably be highest in warm weather, leading to an increase in the biological influence on stability in the spring and summer (de Brouwer *et al.*, 2000; Cadée, 2001; Widdows & Brinsley, 2002; Friend *et al.*, 2003a; Ysebaert *et al.*, 2005).

1.9 Understanding sediment erosion

With the variety of temporal and spatial heterogeneity in the interacting biotic and abiotic factors that dictate sediment stability it is difficult to estimate how stable an area of sediment will be at a given time. The overall effect of these variables has even been described as idiosyncratic by Tolhurst *et al.*, (2003). However, with the high environmental and economic importance of intertidal cohesive systems there has been a great deal of work undertaken towards quantifying the influences of conditions upon the overall sediment stability. This research is based on the desirability of two outcomes, firstly, the simplification of assessing sediment stability, especially over large areas, and secondly to quantify and parameterise the factors affecting stability into a predictive equation.

1.9.1 Rapid measurement of sediment stability

Directly measuring the stability of sediment over a large area requires a large investment of time and resources. To solve this problem, attempts have been made to find rapid or large scale proxy measurements of sediment properties, including the use of remote sensing to assess sediment stability. Generally such attempts have proved to be unsuccessful (Houwing, 1999; Riethmuller *et al.*, 2000; de Brouwer *et al.*, 2000; Paterson *et al.*, 2000) or highly site specific (Christie *et al.*, 2000; Defew *et al.*, 2003; Friend *et al.*, 2003a; Smith *et al.*, 2004).

1.9.2 Predictive models of sediment stability

Being able to predict sediment movement and transport is vitally important for making decisions on the development or conservation of sediment systems (Townend,

2002, Black *et al.*, 2002a; Ryu, 2003; Gleizon, *et al.*, 2003). To be able to predict the critical erosion thresholds of sediment surfaces is vital for this goal. Attempts to create a predictive equation have been made with models based on a purely physical approach using sediment and hydrodynamic properties and conditions (e.g. Lumborg & Windelin, 2003), belying the importance of the biotic influence on sediment deposition and erosion. Unsurprisingly, such models are often relatively inaccurate. However, the need to include biogenic influences in such models is becoming accepted with some successful predictions of sediment stability being made (Willows *et al.*, 1998; Widdows *et al.*, 2002; Uncles, 2003; Wood & Widdows, 2003; Lumborg *et al.*, 2006; Orvain *et al.*, 2006).

1.10 The study of sediment stability

Sediment stability is studied worldwide by many different institutions using a variety of methods and approaches. Studies include measurements of sediment stability at and between specific locations, influences of bioturbation, pollution sequestering within sediment, habitat productivity and nutrient flux. Such a varied field of study reflects the many different important functions and properties of the intertidal ecosystem.

Methods used in these studies are as varied as the aims of the studies themselves leading to issues of comparability and intercalibrations between machines (Tolhurst *et al.*, 2000a; Jonsson *et al.*, 2006). Numerous erosional devices have been developed and used, which can be categorised mostly into one of several groups of machines, each with advantages and disadvantages.

1.10.1 Flumes

Laminar flumes consist of a straight chamber through which water is pumped, creating flow conditions. Sediment is either placed into the base of the machine or the flume is designed with an open bottom and placed directly onto a sediment surface. The flow of the water is variable and controllable. The usual operating procedure for these machines is to incrementally increase the flow speed until erosion occurs. Erosion is measured as an increase in the suspended sediment concentration within

the water either through water samples or a variety of optical methods. Laminar flumes range in size from portable machines that can be used *in situ* to very large permanent laboratory based machines (Houwing, 1999; Lelieveld *et al.*, 2003; Orvain *et al.*, 2006).

Annular flumes work in a similar manner to laminar flumes, but instead of a straight chamber they have a race track chamber which water flows around, usually driven by a paddle system positioned in the top of the flume. Sediment erosion is produced and detected in the same fashion as laminar flumes. These flumes also come in a variety of sizes and can be used *in situ* and in the laboratory (Widdows *et al.*, 1998; Austen *et al.*, 1999; Widdows *et al.*, 2000a, b; Andersen *et al.*, 2002; Amos, *et al.*, 2004; Andersen *et al.*, 2005; Bale *et al.*, 2006).

1.10.2 The Cohesive Strength Meter (CSM)

The CSM is a comparatively novel erosional device devised by Paterson (1989) and now commercially built by Sediment Services and used in several laboratories. Instead of a horizontal flow, the CSM fires a series of vertical jets of increasing pressure onto the sediment surface within a flooded test chamber. Erosion is detected optically by an increase in suspended sediment inside the test chamber (Tolhurst *et al.*, 1999; Christie *et al.*, 2000; de Deckere *et al.*, 2001).

1.10.3 Other erosional devices

Several other devices exist that have a more engineering background but can be applied to biological study of sediment strength as a proxy for stability. The shear vane measures the cohesive strength of subsurface sediments. By rotating a vane within the sediment the force required to shear the sediment is measured (Hauton & Paterson, 2003; Bassoullet & Le Hir, 2007). The fall cone penetrometer allows a metal cone to fall onto the sediment surface, from the depth of penetration of the cone the strength of the sediment surface can be measured (Watts *et al.*, 2003; Fernandes *et al.*, 2006). The pole penetrometer measures the force required to push a pole into the substratum, from this the strength of the sediment on a depth profile can be obtained up to a depth of about 55cm (Reading *et al.*, 2004).

1.11 Aims of Thesis

The study of intertidal sediment stability has been performed for many years but despite this no comprehensive understanding of the many principles involved in determining the critical erosion threshold of a sediment surface has been reached. This work covers two areas of the study of intertidal sediment stability. Firstly an assessment and development of methods and protocols used in such studies and secondly an assessment of sediment stability as a product of the ecosystem.

Chapter 3; Calibrating and investigating improvements in the Cohesive Strength Meter (CSM)

The CSM has been used to measure intertidal surface sediment stability since its conception by Paterson (1989). For the first time since then a thorough evaluation of the machine was performed with several intentions.

- Establishing a methodology to calibrate the erosional force produced by each machine following the discovery of discrepancies between different models.
- Determine the source of irregularities in output at apparently random jet pressures in all models.
- Investigate the feasibility of new parts for the CSM to improve accuracy and lower manufacturing costs.

Chapter 4; Intertidal sediment stability from tidal exposure to submersion

The vast majority of previous research on intertidal sediment stability is based on measurements taken on exposed sediment. This is mostly due to the logistical difficulties of sampling submerged sediments in comparison to exposed sediments. However, as sediment erosion occurs during submersion this omission needs to be rectified. Accurate methods of measuring sediment stability and properties during submersion are needed for *in situ* and laboratory based studies. Additionally, the validity of replicating *in situ* tidal submersion in the laboratory using both stationary and flowing water was assessed. Sediment properties that influence sediment stability need to be measured to determine if they are affected by submersion and any subsequent effects this may have on stability.

Chapter 5; The influence of Arenicola marina on intertidal sediment stability

The influence of the ecosystem engineering polychaete worm *A. marina* on sediment stability is unknown. Given the importance of a marina in influencing many biotic and abiotic elements of the ecosystem a holistic ecosystem approach is required to determine its influence. A large scale a marina exclusion experiment on the German island of Sylt allowed the sediment stability of two parallel intertidal ecosystems, with and without the influence of a marina, to be compared. Measurements of biotic and abiotic properties of the sediment ecosystem will determine which elements of the ecosystem are effected by a marina and may have a bearing upon the overall sediment stability.

Chapter 6; The impact of bait digging on the intertidal sediment ecosystem

The impact of commercial bait digging in intertidal systems has been studied extensively in respect to both target and non target macrofauna species. However there is very little understanding of the impact of bait digging on the sediment environment, especially the potentially important elements of sediment stability and primary productivity. These elements need to be assessed in reference to both the disturbance of bait digging and the removal of a commercially viable species from the ecosystem.

Chapter Two

Materials and Methods

Throughout the following thesis, numerous methodologies have been used in multiple chapters. These common methods are detailed within this chapter.

2.1 Study Site - The Eden Estuary

The Eden Estuary, Scotland ($56^{\circ}22'N$; $02^{\circ}51'W$) opens into the North Sea about 4km north of St Andrews, extending inland for about 8km, past the village of Guardbridge, Fife (Fig. 2.1). The Eden Estuary covers an area of 10.41km^2 containing 9.37km^2 of intertidal mudflats (Davidson & Buck, 1997). The estuary is considered significant for conservation and research and has been given Special Area of Conservation status (SAC) as well as containing two Sites of Special Scientific Interest (SSSI). Dominant hydrodynamic conditions within the estuary are slow tidally driven currents, with wave driven currents and extreme conditions rare due to the relatively small and protected opening of the estuary to the North Sea and the low frequency of storm events from the East. The Estuary has been the main focus of continuous research by the Sediment Ecology Research Group (SERG) in St Andrews University since the early 1990s and its characteristics are well known, making it an ideal site for progressing research. Two main sites within the estuary are used with contrasting sediment conditions. On the South shore the “Golf Course” site is composed mostly of muddy sand while the “Papermill” site further inland on the North shore of the estuary is composed of fine muds.

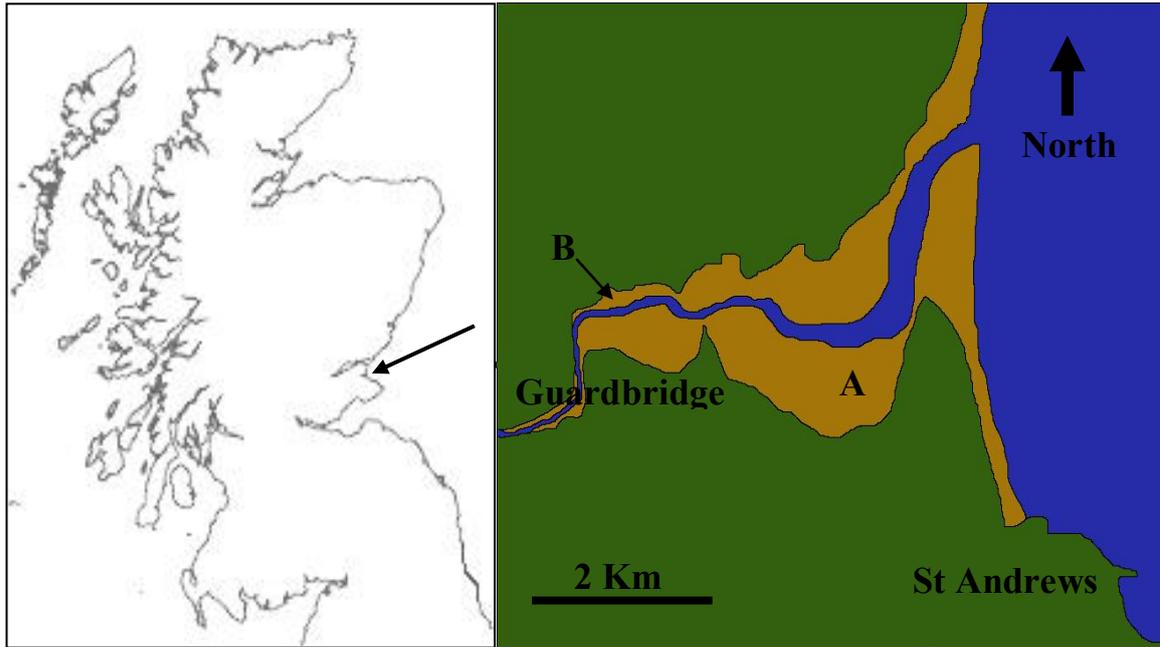


Figure 2.1. The Eden Estuary (Scotland) labelled with the Golf Course site (A) and Papermill site (B).

2.2 Sediment Sampling – The Contact Core

In situ sampling of sediment was done through the use of a contact core as described in the HIMOM (2003) protocols guidelines. The core is used to sample the upper 2mm of sediment using liquid nitrogen to freeze the sediment and preserve its properties *in situ*. The contact core is a small metal chamber with an internal diameter of 4.85cm (Fig. 2.2). There are two sections of the contact core, the bottom section is 2mm deep and the top 10mm. The core is pushed into the sediment to a depth of 2mm then the top section filled with liquid nitrogen. After a given time (between 2 to 5 minutes depending on the amount of water in the sediment) the sediment around the bottom section is frozen into the core. The core can then be lifted from the sediment and the excess sediment scraped away from the bottom until it is flush with the base of the contact core, producing a disk of frozen sediment from the surface. This disk is then wrapped in labelled tin foil and transferred to liquid nitrogen storage *in situ* prior to storage in a -80°C freezer for subsequent analysis.

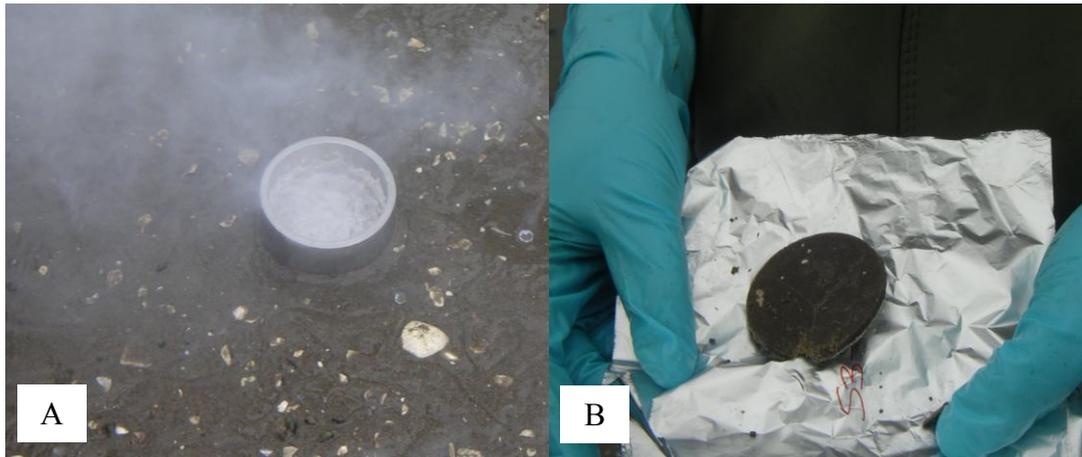


Figure 2.2. The contact core on a sediment surface with added liquid nitrogen (A) and the disk of frozen sediment produced (B).

2.3 Analysis of Sediment Properties

2.3.1 Content versus concentration

It has been customary to measure variables associated with the sediment matrix as a “content” of the sediment (e.g. levels of water, organic material, and carbohydrates), this is essentially the weight of the respective variable divided by the total weight of the sample and thus expressed as a percentage or as weight per weight. However, there are problems associated with this approach which have been highlighted as potentially misleading and possibly in error in some studies (Flemming & Delafontaine, 2000; Perkins *et al.*, 2003; Tolhurst *et al.*, 2005). These authors presented the case that the weight of the whole sample is mostly dependant upon the sediment weight, which is itself a variable that will change depending upon the size, composition and density of the sediment particles. This is a measurement best described by the dry bulk density of the sample (sediment weight per sample volume) which is a concentration. Dividing by such a variable property can potentially lead to a confounding of the result or covariance between the dry bulk density of the sample and the content of other variables.

Flemming and Delafontaine (2000) presented an example from Taylor and Paterson (1998) where carbohydrate concentration was stated to decrease with increasing depth within the upper 2mm of sediment. However, Flemming and Delafontaine (2000) realised that carbohydrate content was being measured rather than concentration as

stated, and also that with increasing depth dry bulk density increased due to increased compaction of the sediment. Consequently, what was probably a constant amount of carbohydrates within the sediment was being divided by increasingly higher values of sediment weight with increasing depth, hence reducing the final content value with increasing depth. Equally Perkins *et al.*, (2003) argued the case that increases in chlorophyll *a* content in intertidal sediments over an emersion period were mostly due to increases in the dry bulk density of the sediment caused by dewatering. In both examples the measurement of sediment properties as concentrations is suggested as a better alternative (weight of property per volume of sample) as it removes the dry bulk density variable from equations and treats each property in isolation.

In addition to the reasoning presented by the previously stated authors it is considered that the behaviour of an individual organism will be more related to the amount of a substance within an area, rather than the amount per weight of sediment, and therefore concentration is the most appropriate measurement to relate to the biology of the system. For example, the feeding behaviour of *Corophium volutator* (mud shrimp) on organic material is more related to the area it needs to cover to consume a set volume, rather than the weight of sediment it has to travel over. Equally the grain size of sediment will have a large influence on its stability. As grain size will also influence the weight of sediment, and therefore be a variable in the calculation of sediment properties expressed as a content, it is possible that influences of the properties on sediment stability are being masked by changes in the grain size distribution.

For these reasons, measurements of water, organic material, carbohydrates and chlorophyll *a* are given as concentrations per unit volume rather than the customary contents. Equally, after many years of study into sediment stability few clear relationships have been expressed relating stability to a sediment property expressed as a content, and possibly a change to concentration may reveal a previously hidden pattern.

2.3.2 Water concentration and dry bulk density

Although two different properties, the methods for measuring water concentration and dry bulk density are integrated together.

Firstly the volume of the sample was obtained, this is based on the size of the contact core sample. This is not a constant as the depth of the core varied, effected by sediment properties (especially water content/concentration) and time spent freezing the core. The depth of the frozen core was therefore measured using callipers (n=3 with an average taken to represent the core). If the core was intact then it was circular with a diameter of 4.85cm (set by the internal diameter of the contact core). Using these values in the equation for the volume of a cylinder (equation 2.1) the volume of the core was obtained.

Volume of sample (cm³) =

$$(\pi \times (4.85 \text{ (cm)} / 2)^2) \times \text{Depth of core (cm)}$$

Equation 2.1

If the core was broken or segmented then its volume was obtained by drawing around the segment(s) on graph paper and measuring the flat surface area which was then multiplied by the average depth of the segments.

The core was then placed into a pre-weighed labelled plastic bag, and their combined weight taken, before freeze drying the sample in the dark for 12 hours to remove any water. Freeze drying in the dark is preferred to the older technique of oven drying as it does not damage or degrade the organic substances within the sediment that may need to be measured (Honeywell *et al.*, 2001). The plastic bag and sediment were then reweighed.

From these values the following equations were used to obtain the dry bulk density (equation 2.2) and water concentration (equation 2.3);

Dry bulk density (g cm⁻³) =

$$\frac{\text{Freeze dried sediment and bag (g)} - \text{bag (g)}}{\text{Sample volume (cm}^3\text{)}}$$

Equation 2.2

Water concentration (g cm^{-3}) =

$$\frac{\text{Wet sediment and bag (g)} - \text{Freeze dried sediment and bag (g)}}{\text{Sample volume (cm}^3\text{)}}$$

Equation 2.3

2.3.3 Organic concentration

The measurement of organic material in the sample was performed through the loss on ignition technique (HIMOM, 2003). The method required a sub sample ($\approx 2\text{g}$) of the freeze dried sediment produced during the water concentration/dry bulk density procedure (section 2.3.2). This sub sample was placed into a pre-weighed crucible and the two weighed together. They were then placed into a muffle furnace at 450°C for 4 hours. The sediment and crucible were then reweighed after being allowed to cool in a desiccator to ensure the sediment did not absorb water from the atmosphere during cooling.

As the procedure is based on a sub sample of the core of unknown volume, the organic concentration of the whole core can not be directly calculated. Instead the organic content of the sub sample is obtained (equation 2.4) which is then used with the known volume of the entire core to calculate the organic concentration of the sediment (Equation 2.5).

Organic Content (g g^{-1}) =

$$\frac{\text{Crucible and sediment pre furnace (g)} - \text{Crucible and sediment post furnace (g)}}{\text{Crucible and sediment pre furnace (g)} - \text{Crucible (g)}}$$

Equation 2.4

Organic concentration (g cm^{-3}) =

$$\frac{\text{Organic content (g g}^{-1}\text{)} \times \text{Sample weight (g)}}{\text{Sample volume (cm}^3\text{)}}$$

Equation 2.5

2.3.4 Colloidal-S Carbohydrate Concentration

As with organic concentration (2.3.3) the method used for obtaining colloidal-S carbohydrate concentration is based on the HIMOM (2003) procedure but again an

additional calculation was used to convert content into concentration. A sub-sample ($\approx 50\text{mg}$) of the freeze dried sample was weighed and placed in a test tube with 5ml of distilled water and centrifuged at 1500rpm for 15 minutes. 1ml of the supernatant was removed to a second test tube into which 1ml of 5% w/v phenol and 5ml of concentrated sulphuric acid was added. The samples were then vortexed and left for 35 minutes prior the absorption being read with a Cecil 3000 spectrophotometer at 486.5nm.

In every set of samples tested a selection of glucose solutions of known concentrations (0, 1, 5, 10, 50, 100 and $200\mu\text{g l}^{-1}$), were also run. The absorbance of these glucose solutions were used to construct a standard curve from which the calibration equation was obtained (equation 2.6).

$$y = mx + c$$

Equation 2.6

Where y = absorbance, m = gradient, x = glucose concentration and c = intercept on the axis.

These figures were used to construct equation 2.7 which allows sample absorbance (Ab) to be converted into an equivalent glucose content before colloidal-S carbohydrate concentration was obtained for the whole sample (equation 2.8).

Colloidal-S carbohydrate content ($\mu\text{g g}^{-1}$) =

$$\frac{((Ab-c) / m) \times 5}{\text{Sub sample (g)}}$$

Equation 2.7

Colloidal-S carbohydrate concentration ($\mu\text{g cm}^{-3}$) =

$$\frac{\text{Colloidal-S carbohydrate content } (\mu\text{g g}^{-1}) \times \text{Freeze dried sediment (g)}}{\text{Sample volume (cm}^3\text{)}}$$

Equation 2.8

2.3.5 Chlorophyll *a* concentration

Chlorophyll *a* concentration within the sediment was determined by using High Performance Liquid Chromatography (HPLC) on extracted chlorophyll *a* from the sediment.

Preparation of the extractant

A weighed sub sample ($\approx 50\text{mg}$) of freeze dried sediment was placed into a pre-weighed eppendorf into which 1ml of acetone (90% acetone buffered with 10% saturated sodium carbonate) was added and reweighed. The weighing allowed the exact volume of acetone to be obtained. The eppendorf was then sonicated in -4°C seawater for 90 minutes prior to being stored in a -80°C freezer for 48 hours including 1 minute of vortexing after 24 hours. The samples were then centrifuged for 3 minutes at 1300rpm before the extractant was removed and filtered through $0.2\mu\text{m}$ glass filter into a HPLC vial ready for analysis. Samples were stored in a -80°C freezer until they were loaded into the HPLC machine.

High Performance Liquid Chromatography

Samples were run through a HPLC machine comprising of a quaternary high pressure pump (Perkin-Elmer 410), an autosampler with temperature set to 4°C (Walters 910), a column oven set to 25°C containing a reverse phase Nucleosil C18 column (Capitol HPLC Ltd.) and a Photo-diode Array Detector (PDA; Walters 910).

The HPLC machine was run continuously with samples added in batches of 6-8, ensuring that no sample was out of the -80°C freezer for more than 7 hours. This was to prevent degradation of the sample prior to analysis.

Results were given as the concentration of chlorophyll *a* within the extractant. This was calculated into chlorophyll *a* concentration per sample through equations 2.9 and 2.10;

Sub sample chlorophyll *a* content ($\mu\text{g g}^{-1}$) =

$$\frac{\text{HPLC derived Chlorophyll } a \text{ concentration } (\mu\text{g ml}^{-1}) \times \text{Volume of acetone (ml)}}{\text{Weight of sub sample (g)}}$$

Equation 2.9

Sample chlorophyll *a* concentration =

$$\frac{\text{Chlorophyll } a \text{ content } (\mu\text{g g}^{-1}) \times \text{Freeze dried sediment (g)}}{\text{Sample volume (cm}^3\text{)}}$$

Equation 2.10

2.3.6 Grain size distribution

Grain size analysis was performed using a Coulter Laser Particle Sizer (LS230). The machine required small amounts (≈ 3 grams) of freeze dried sediment which was treated with Calgon to promote separation of individual particles. Unlike using nested sieves the coulter machine allows the sample to be divided into any number of predetermined size fractions. The boundaries of these fractions changed relating to the needs of the work, but constantly used throughout were the boundaries of $<63\mu\text{m}$ for cohesive sediment and between $63\mu\text{m}$ and 2mm for sand. Three sets of results were produced from each test which were averaged to give a single grain size distribution per sample.

2.4 Measurement of sediment strength

2.4.1 The Cohesive Strength Meter (CSM)

The Cohesive Strength Meter measures the critical erosion threshold of the sediment surface. It is used comprehensively throughout this work and is covered in Chapter 3.

2.4.2 The Shear Vane

A Shear Vane (Fig. 2.3) was used to measure the shear strength of sediment at known depths. The vane is inserted into the sediment to the desired depth and the circular disk turned clockwise at a speed of 1 rotation a minute. The vane within the sediment is attached to the disk through a spring and turning the disk while the sediment resists the rotational force tightens the spring. Resistance is measured through a dial on the top of the disk. Once the sediment fails the spring rotates the vane leaving the arm on the measure of torque force at which the sediment failed. This force is reported as the shear strength (Nm^{-2}) of the sediment on the calibrated scale of the meter.



Figure 2.3 The Shear Vane

Chapter Three

Calibrating and investigating improvements in the Cohesive Strength Meter (CSM)

Abstract

The Cohesive Strength Metre (CSM) is a commonly used machine to test the surface stability of exposed intertidal sediment. After its conception by Paterson (1989) the machine has been commercially manufactured by Sediment Services and many machines are now used in laboratories and research institutes across the world. In consultation with Sediment Services a comprehensive review of the CSM was undertaken in an attempt to improve its performance and reduce manufacturing costs. Although these were unsuccessful, several flaws in the current operating procedure were identified. These flaws were investigated and rectified through the creation of a new calibration method. New component parts for the erosion chamber of the CSM were trailed as was a new method of deployment. The new components failed to increase the accuracy or reduce the manufacturing costs of the CSM, while the new deployment method was found to be equally accurate as the existing method but was generally considered easier to use.

Foreword: Sections of research and analysis from this chapter were devised, performed, analysed and written by myself and contributed to the journal “Calibration of the High-Pressure Cohesive Strength Meter (CSM)” Vardy *et al.*, (2007), (appendix 1). This work is covered in the following sections;

The methods used to isolate the cause of discrepancies between the firing efficiency of different CSM models, or the same model over time, are covered in sections 3.2.2.1-3 of this thesis and section 2.1 of the journal. With the results shown and analysed in sections 3.3.1 and 3.2 of this thesis and the journal respectively. The methods and results used in identifying and determining the cause of erroneous outputs of the CSM are covered in sections 3.2.2.6 and 3.3.1 of this thesis and sections 2.1 and 3.2 of the journal.

3.1 Introduction

3.1.1 The Cohesive Strength Metre (CSM)

The CSM was introduced by Paterson (1989) as a method for testing the critical erosion threshold of sediment surfaces *in situ* and in the laboratory. Since this work the CSM has become a commercially available product manufactured by Sediment Services, a company specialising in building machines for use in marine sediment environments. This has led to the CSM becoming a common tool in sediment stability research, within both research (e.g. Defew *et al.*, 2002; Friend *et al.*, 2003) and commercial fields (e.g. Watts *et al.*, 2003).

How a CSM works

The CSM has changed little from its original conception, and despite changes to the exterior and peripheral parts the internal workings of all CSMs are based on exactly the same principles. A CSM induces erosion by firing a series of increasingly pressurised vertical jets of water (salt or fresh water depending upon requirements) onto a sediment surface within a small water filled chamber placed directly onto a sediment surface (Fig. 3.1). Once the jet has reached a sufficient pressure the critical erosion threshold of the sediment is passed and sediment will be eroded from the surface and become suspended in the water within the chamber. Inside the erosion chamber a UV transmitter and receiver measure the transmission across the chamber after every jet every 0.1 seconds for a set period of time depending upon the test. Once sediment is eroded the suspended particles will reduce the transmission across the chamber. The transmission value for each jet is taken as the average value between 0.2 and 1.2 seconds after the jet, measurements of transmission can then be plotted against pressure for each jet creating an erosion profile. The critical erosion threshold is deemed to have passed once the transmission drops below 10% of the maximum transmission taken prior to the test (usually between 70-110%, dependant upon machine) (Tolhurst *et al.*, 1999) which can be worked out from the erosion profile (Fig. 3.2). A CSM has about 40 programmed tests, each with different settings for the incremental increases in firing pressure, the duration of each jet and the length of time which the transmission is recorded depending upon the requirements of the experiment or sediment to be tested (Table 3.1).

Using the CSM

The CSM has many advantages for measuring sediment stability in both the laboratory and field. The machine is relatively small, compact and light compared to some flumes so easy to use *in situ*, especially in difficult muddy intertidal systems. A rapid test time means many replicates can be taken quickly, allowing improved statistical analysis of results. The erosion chamber is not part of the main body of the machine and is very small allowing it to be positioned on very specific patches of sediment such as biofilms (Defew *et al.*, 2003), ridges and troughs on a sediment surface (Paterson *et al.*, 2000) or between physical obstructions such as salt marsh plants (Paterson & Black, 1999). However, especially in salt marshes and upper intertidal sediments, the original CSM (the Mark III CSM) has been exposed as having insufficient power to erode some sediment surfaces (Friend *et al.*, 2003). In response to this problem a new CSM (the Mark IV CSM) was developed with a maximum working pressure of 60psi, double that of the existing model.

3.1.2 Calibrating the results of the CSM into horizontal shear stress

A major disadvantage to the CSM is the applicability of the results to real life situations and comparisons to other erosional devices. The vertical jet produced by the CSM, while being a quick and efficient method of eroding sediment, does not replicate the horizontal flow which occurs *in situ* or in flume based machines. Such machines express results of critical erosion thresholds as shear stress (Nm^{-2}), a format which is applicable to the natural environment and can be placed into models of sediment erosion and transport. As such, data produced by the CSM is only comparable with other CSM data and could not be used in actual sedimentary models incorporating shear stress and velocities.

As an attempt to solve this problem Tolhurst *et al.* (1999) devised a practical calibration to convert the pressure of the vertical jet into an equivalent horizontal shear stress. The calibration was based on comparing the jet pressure at which the CSM eroded samples of sieved clean sediment and the theoretical critical erosion threshold (expressed as shear stress) of that sediment. This calibration resulted in an equation for converting CSM firing pressure into horizontal shear stress.

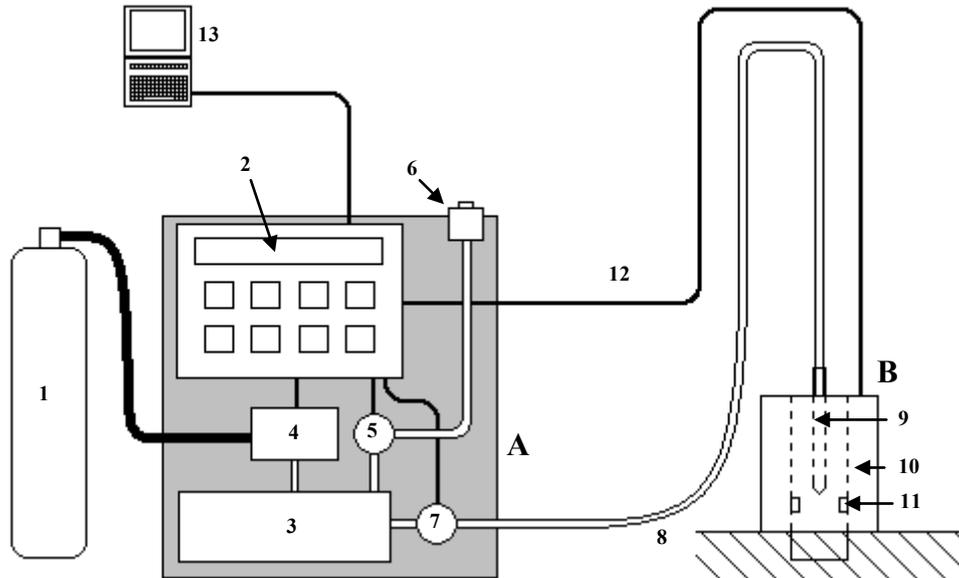


Figure 3.1. The basic mechanism and component parts of the CSM. The CSM consists of a main body (shaded area (A)) and an external erosion chamber (B) which is placed directly onto the sediment surface. Within the CSM there are high pressure air hoses (thick black line), low pressure air and water hoses (thick open lines) and electronic cables (thin black lines). The CSM requires filtered water and a compressed air source to fire a jet onto the sediment surface. Compressed air is supplied from a small (0.5l) air cylinder (232bar max pressure) (1). The operation of the CSM is run from the central consol (2), into which a test setting can be programmed before each test. Prior to firing a jet the central consol increases the pressure within the water container (3) by opening the air intake valve for a short period (4) through which pressurised air is filtered into the water container. Should the pressure within the chamber be too high the central consol can release some pressure by opening the venting solenoid (5) for a short period, releasing pressurised air out of the CSM (6) until it is returned to the required level. Once the desired pressure is reached the central consol opens the firing solenoid (7) firing the jet through the water hose (8) and out of the nozzle (9) contained within the erosion chamber (10). After each jet a UV transmitter and sensor (11) measure the transmission across the erosion chamber, which is then relayed to the central consol (12) where it is stored prior to being downloaded onto a PC (13) for analysis.

Table 3.1. Examples of different test settings available for selection from the central consol of the CSM. The maximum pressure of a CSM is either 30.0 or 60.0psi depending upon model. * Indicates tests that were used throughout the work on the CSMs.

Test setting	Increase in pressure between jets (psi)	Jet Duration (sec)	Data logging time (sec)
Fine 1*	0.1 increments from 0 to 3.0 0.3 increments from 3.0 to 6.0 2.0 increments to 30/60	1.0	3.0
Sand 1	0.3 increments from 0 to 12.0	0.3	3.0
Sand 3*	0.5 increments from 0 to 5.0 1.0 increments from 5.0to 30/60.0	0.3	3.0
Sand 7*	0.3 increments from 0 to 12.0	1.0	3.0
Sand 8*	0.5 increments from 0 to 20.0	1.0	3.0
Sand 9*	0.5 increments from 0 to 5.0 1.0 increments from 5.0to 30/60.0	1.0	3.0
Sand 16	2.0 increments from 0 to 30/60.0	0.5	3.0
Sand 18*	2.0 increments from 0 to 30/60.0	2.0	3.0
Mud 1	0.3 increments from 0 to 12.0	0.3	30.0
Mud 7	0.3 increments from 0 to 12.0	1.0	30.0
Mud 9	0.5 increments from 0 to 5.0 1.0 increments from 5.0to 30/60.0	1.0	30.0

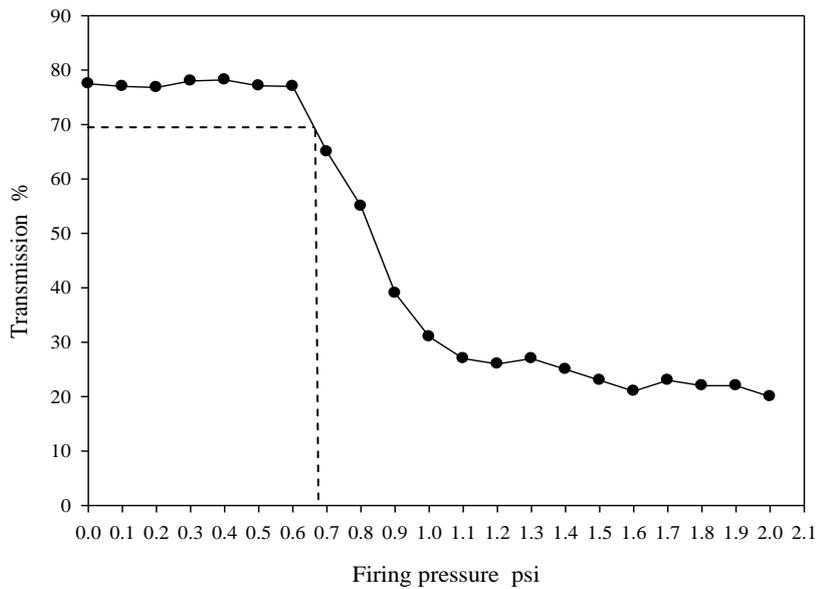


Figure 3.2. An erosion profile from a test run on Fine 1. Maximum transmission at the start of the test is 77.5%, therefore a 10% drop in transmission means the critical erosion threshold has been passed when transmission drops below 69.75% (dashed line). Hence the critical erosion threshold is passed with a jet of 0.7psi.

The calibration equation devised by Tolhurst *et al.* (1999) was based upon the Mark III CSM, and therefore restricted to a maximum jet pressure of 30psi. With the development of the Mark IV CSM, Vardy *et al.* (2007) devised a similar method to calibrate the higher pressure system. During this work errors were found in the method used by Tolhurst *et al.* (1999) to calculate the theoretical erosion thresholds of their sediment samples. A corrected equation is given by Vardy *et al.*, (2007) that should have been used, however Vardy *et al.* (2007) did not redo the calibration using this equation stating that the theory behind it use was questionable;

“Further work needs to be undertaken to understand the flow within the CSM chamber before a satisfactory relationship between the CSM pressures and critical suspension thresholds can be developed.”

Therefore the calibration devised by Vardy *et al.*, (2007) gives the erosional force as a stagnation pressure (Nm^{-2}), a measurement of the impinging force of the bed to cancel out the energy contained within the jet.

3.1.3 Discrepancies between CSM machines

As part of the calibration experiments Vardy *et al.* (2007) used three different CSMs on a variety of grain substrata to obtain their critical erosion thresholds. During these calibration experiments it was noted that there were discrepancies between the three CSMs (Fig. 3.3), the different machines achieving the erosion of the same substrate with jets of different pressure. These discrepancies could potentially be a result of unknown differences between individual CSMs or between CSM models, as such this required further study as it had been assumed all CSMs were identical and results have been compared directly based upon that assumption.

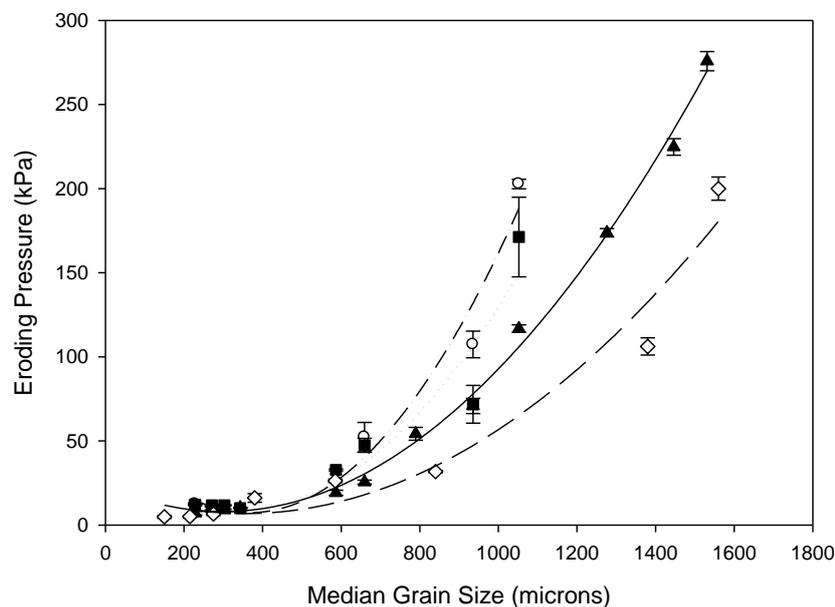


Figure 3.3. Taken from Vardy *et al.* (2007). Discrepancies between three models of CSM discovered while performing calibration experiments on a Mark IV CSM. Mark IV CSM prototype using garnet (○), Mark IV CSM using garnet (▲), Mark III CSM using garnet (■) and Mark III CSM using quartz (◇).

3.1.4 Reassessing component parts of the CSM

Sediment Services were interested in improving the CSM from both a functional and economical perspective. This meant improving the accuracy in detecting the critical erosion threshold of sediments or finding new parts that would allow a CSM to be manufactured more economically. To this aim the whole CSM was assessed to identify components that could be redesigned for inclusion in a new model of the machine.

It was decided that changing the internal workings and firing mechanisms of the CSM would be costly and unlikely to yield positive results. Therefore work focused on the erosion chamber, where simple changes to nozzle and erosion detection mechanism, in addition to an appraisal of a new deployment device, were considered as possibilities for improving the CSM and worthy of further study.

The jet nozzle

Not changing the internal workings of the CSM meant that jets would be fired at equal pressures as before. However, improving the accuracy of detecting the critical erosion threshold would be possible by having finer increments of increasing jet velocity. To achieve this without changing the internal workings nozzles with larger diameters were considered. For a given pressure a jet being fired from a larger diameter nozzle would have a lower velocity, and therefore hit the sediment surface with less energy. This would allow smaller increments in the erosional force on the sediment surface given the same firing pressure and allow more accuracy in detecting the critical erosion threshold.

The UV transmission detection device

Currently the detection of suspended sediment in the erosion chamber is based on a reduction in the UV transmission from a firing node to a sensor. This has two major problems, firstly the system is not waterproof and therefore has to be contained within the walls of the chamber. Secondly the firing node and sensor have to be lined up very accurately when the chamber is made which is difficult with errors common. As this can only be tested once the chamber is completed any error means the chamber is defunct and a new chamber is required. The possibility of replacing the UV transmission system with a cheaper fibre optic sensor (Keyence Dual Digital Fibre Sensor) was considered. A fibre optic system detects backscatter rather than transmission, allowing the transmitter and sensor to be combined within a system that is also waterproof, allowing the entire erosion chamber to be redesigned, potentially reducing their production costs (Fig. 3.4). As detection of suspended sediment is dependant upon the concentration of sediment different sizes of erosion chamber are also considered as a simple mechanism to change concentrations without affecting the eroded sediment. However, before it could be considered the accuracy of the system needed to be assessed.

Deployment of the CSM erosion chamber

Positioning the erosion chamber directly onto a sediment surface using the suggested technique of a clamp stand (Tolhurst *et al.*, 1999; Black *et al.*, 2002b) (Fig. 3.5) is difficult. Errors can occur in the height of the nozzle from the sediment if the chamber is incorrectly positioned or if the chamber is set up with the nozzle at the wrong height within the chamber. To reduce this potential error a new disk was designed to be connected to the erosion chamber to give it more stability on the sediment surface (Fig. 3.5).

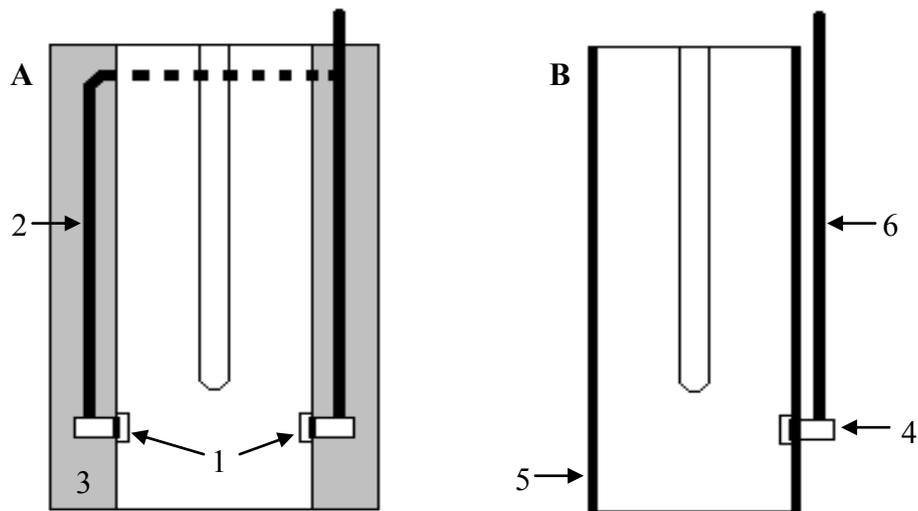


Figure 3.4. The current CSM erosion chamber (A) with UV transmitter and sensor (1) and connecting cables (2) enclosed within a waterproof plastic casing (3), and the proposed CSM chamber based on the requirements of the fibre optic system (B) with the combined transmitter and sensor (4) positioned through a single thin plastic tube (5) and single fibre optic cable (6).

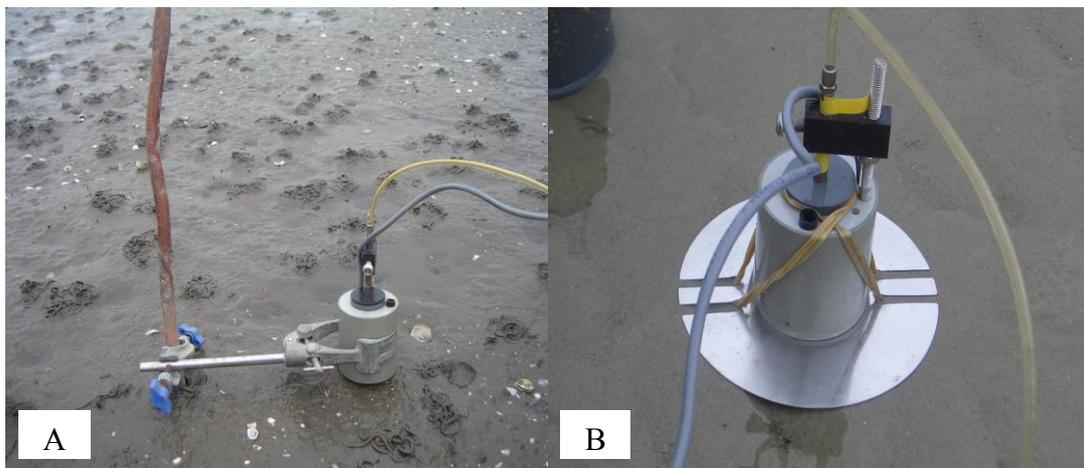


Figure 3.5. The recognised method of deploying the erosion chamber with a clamp stand (A), and deployment with a new disk attachment (B).

3.2 Methods

3.2.1 The CSMs

Three types of CSM were tested, an original Mark III (Fig. 3.6a) with a maximum working pressure of 30psi, a Mark IV prototype (Fig. 3.6b), which has the same components of the Mark III CSM with a maximum working pressure of 30psi but is built in a more compact style similar to the Mark IV high pressure CSM which has a maximum working pressure of 60psi (Fig. 3.6c). All three CSMs are owned and operated by the Sediment Ecology Research Group in The University of St Andrews. In addition to these, two other Mark IV CSMs were borrowed for testing from Sediment Services and Silsoe Research Institute (referred to as the Mark IV CSM-SS and Mark IV CSM-SRI respectively).

3.2.2 Comparing the operational efficiency of different CSMs

3.2.2.1 Internal pressures

The Mark IV CSM-SS was manufactured with an additional outlet port, onto which a Digitron 2022P manometer was attached to measure the internal pressure during a test. Actual internal pressure was compared to the stated firing pressure on the central consol.

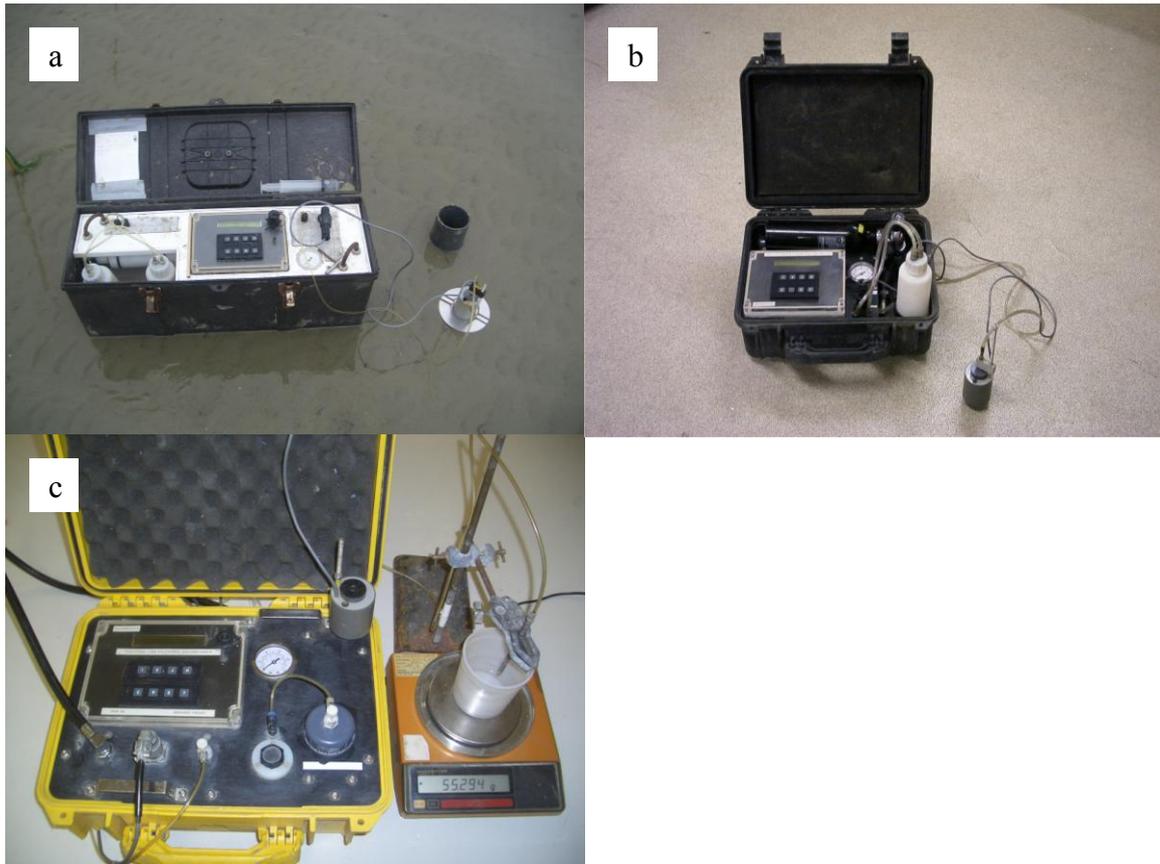


Figure 3.6. Three models of the CSM, (a) Mark III, (b) Mark IV prototype and (c) Mark IV.

3.2.2.2 Jet volumes

The volume of each jet fired during a CSM test was measured by weighing the water fired from the nozzle. The nozzle was removed from the test chamber and positioned over a beaker placed onto scales (0.000g accuracy). After each jet the total weight of water was recorded and the volume of individual jets was calculated as being the increase in total weight from one jet to the next (n=5).

3.2.2.3 Changes over time

The performance of different CSMs was tested over a period of eight months to assess if there was a change in output over time.

3.2.2.4 Component parts of the CSMs

Component parts of the CSMs were assessed to determine if differences within their construction could account for the differences in machines.

The nozzle

The nozzle from the Mark IV CSM was attached to the Mark IV prototype machine and tested on Fine 1 specifically to look at jets fired at lower pressures.

Filters

The Mark IV CSM differs from the previous models in that the filter can be removed and replaced. Therefore three filters of varying state of use (new, moderate and heavy) (Fig. 3.7) were tested.



Figure 3.7. Three Mark IV CSM filters, (a) new, (b) moderately used and (c) heavily used.

Hose length

It is impossible to measure the internal hose length of a CSM without taking the machine apart, however to simulate changes in total length the hose connecting the main machine to the nozzle was replaced with hoses of 70 and 310cm on the three Mark IV CSM models. Additionally the original lengths of hoses varied between these three machines with lengths of 120cm, 170cm and 240cm on the Mark IV CSM-SS, Mark IV CSM and Mark IV CSM-SRI respectively.

3.2.2.5 Statistical comparisons of CSM efficiencies

Comparisons of CSM output was performed by placing the jet weights from each jet fired (n=5) into an ANOVA with pressure nested within CSM and also used as a covariate. A Tukey post hoc test was performed where more than two CSMs (or CSM set ups) were being tested.

To simplify comparisons between CSM tests the output of each test is expressed as a percentage of the results from the Mark IV CSM in April 2004. The selection of that data as a reference in no way indicates that it is a more valid or reliable set of data than other tests. The jet weight at each pressure was compared to the reference and expressed as a percentage of the reference weights. These values were then averaged over the entire test to give an overall percentage value on how the output from that test compared to the reference CSM. Due to the precise nature of the CSM, and its consistency in firing accurate jets over many tests any difference in jet output is considered unsatisfactory as it would invalidate any comparison between data obtained between different machines.

3.2.2.6 Jet duration

Each jet was timed by filming the firing jet with a 25 frame per second digital camera (n=3). Timing of the jet was based on counting the frames in which the jet was being fired. This gave a resolution of 0.04 seconds for each jet. Sand 3, Sand 9 and Sand 18 test settings were used to give different jet durations of 0.3, 1.0 and 2.0 seconds respectively. Calculations were performed to obtain the possible level of timing error based upon expected and actual jet weights.

3.2.3 Improving the erosion chamber of the CSM

3.2.3.1 Erosion trials on new component parts

The new nozzles, detection device and chamber sizes were tested on sediment collected from the Eden Estuary (see Chapter 2) in February 2004 and sieved through a 500 μ m sieve to remove macrofauna and large particles before being homogenised. Trays of sediment were prepared by forming a smooth surface and leaving to dry for three days to create a sufficient level of stability for trails. Tests were performed randomly throughout the trays to reduce the impact of differences in stability. The

Mark IV CSM, set to Fine 1 for maximum resolution, was used to supply the pressurised jets of water.

Nozzles

Three different nozzles were provided by Sediment Services for trials, these had internal diameters of 1mm (same as current machines), 3mm and 4.2mm. The nozzle from the Mark IV CSM was removed and replaced with the new nozzles. Each new nozzle was positioned in the tray 20mm from the sediment surface inside a flooded chamber. Visual observations were made of the critical erosion point with the related pressure noted from the CSMs display (n=3).

Fibre optic sensors and chamber size

The hardware and software needed to run the fibre optic system was supplied by Sediment Services. The hardware comprised of a sensor and cable attached to a data amplifier which in turn connected to a PC. The software was specifically written to allow the gain and offset of the sensor to be adjusted (see note). The sensor works by detecting levels of backscatter, therefore increases in the level of suspended sediment should increase backscatter values. The result is inverted by the software so an increase in backscatter produces a drop in the value output, this is purely so data is displayed in the same format as the current CSMs. Tests were run with the sensor positioned 10mm from the sediment surface in four different sized chambers (80, 50, 40 and 30mm diameter). The critical erosion point was timed manually through observation and plotted on the data retrieved by the software for comparison.

NB. The gain is a degree of amplification of the signal from the sensor, increasing the gain increases the scale of results caused by the same levels of detection. An increase in the gain does not increase the sensitivity of the sensor, only the scale of the output data. Offset value allows the reading to be set at any level, for these tests the offset was set to give a reading of 100 prior to running the test. Offset values varied between tests depending upon the level of residual backscatter caused by the chamber, sediment surface and nozzle.

3.2.3.2 Deployment method of erosion chamber

Calculation of errors in deploying the erosion chamber

The calibration by Vardy *et al.*, (2007) devised equation 3.1 to express the erosional force of a fired jet as stagnation pressure (Nm^{-2}) upon the sediment surface;

$$P = \frac{1}{2} \frac{\rho_w (7.0)^2 4Q^2}{d^2 (z - z_0)^2 \pi} \text{ for } z > z_0$$

Equation 3.1

Where;

P = Stagnation pressure (Nm^{-2})

ρ_w = Density of the fluid (kgm^{-3})

Q = Volume of the flux / jet fired (m^3/s)

d = orifice diameter (m)

z = vertical distance of the bed from the source of the jet (m)

Under normal calibration and operating procedures all values within this equation except Q are constants, therefore the equation can be simplified to equation 3.2;

$$P = (7.79859 \times 10^{13}) \times Q^2$$

Equation 3.2

Given that Q is obtained from measuring the weight of water produced by an individual jet over one second this equation can be converted into equation 3.3;

$$P = 7.79859 \times 10^{13} (0.000001 \times Qg)^2$$

Equation 3.3

Where Qg equals the jet volume in grams per second, allowing the jet weight in grams per second to be inputted directly into the equation.

In the original equation the value of z (the height above the sediment surface of the nozzle) is a constant of 0.02m (2cm). However, by manipulating this constant the degree of error that occurs through incorrectly setting up the CSM chamber can be calculated.

Assessment of a new erosion chamber deployment method

The Mark III CSM was deployed *in situ* on the Papermill and Golf Course mudflat sites within the Eden Estuary (see Chapter 2) on the 24th and 30th November 2006 respectively. The CSM chamber was positioned using the recognised and suggested technique of a clamp stand (Tolhurst *et al.*, 1999, Black *et al.*, 2002b) (Fig. 3.5a) and a novel new disc base (Fig. 3.5b) and stability tests performed (n=8). Potential errors in erosion based upon errors in height were calculated by placing the height into the calibration equation from Vardy *et al.*, (2007) and assuming all sediment eroded at a Qg value of 1 gram per second. Comparisons of the critical erosion thresholds obtained from the tests were also performed. Both methods using a two-way ANOVA based on deployment method and site.

3.3 Results

3.3.1 Discrepancies between different CSM machines

3.3.1.1 Different models of CSM

The three models of CSM were tested on Sand 9 and Fine 1 settings. The weights of water produced by the jets were different for all the machines (Sand 9, $F_{2, 517} = 10.47$, $p < 0.001$; Fine 1, $F_{2, 144} = 249.88$, $p < 0.001$) (Fig. 3.8a and 3.8b). The Mark III CSM firing 86.5% of the Mark IV CSM on Sand 9 and 96.4% on Fine 1, and the Mark IV prototype only firing at 60.1% on Sand 9 and 92.2% on Fine 1.

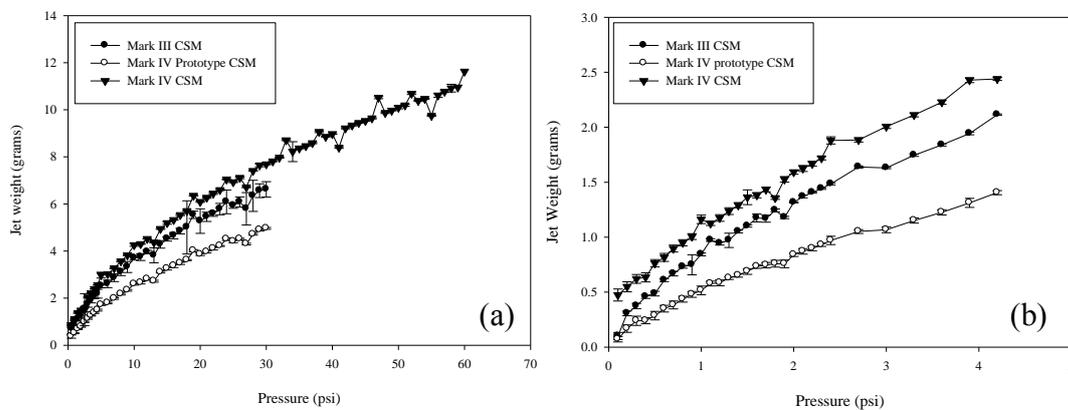


Figure 3.8. Jet weights from three different CSM models on (a) Fine 1 and (b) Sand 9.

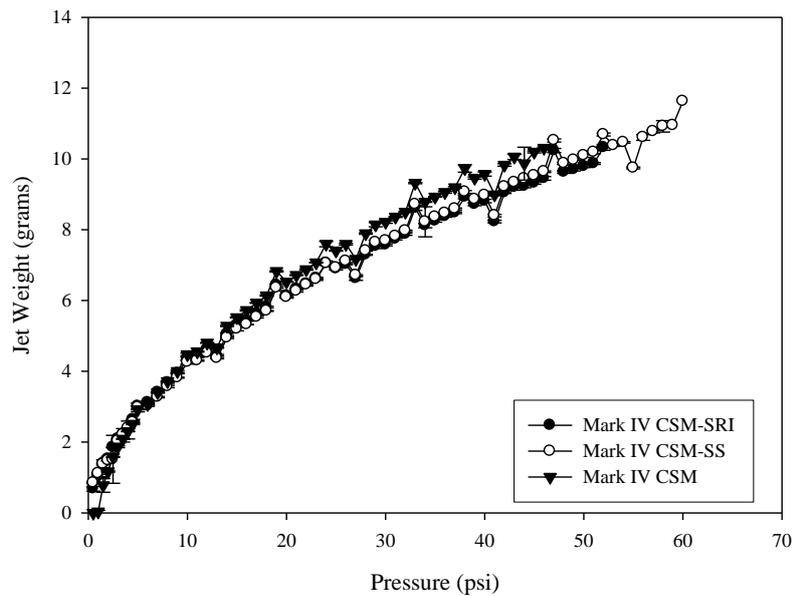


Figure 3.9. Jet weights of three different Mark IV CSMs on Sand 9.

3.3.1.2 Comparison of three CSMs of the same model

Three different Mark IV CSMs were tested on Sand 9 (Fig. 3.9). The Mark IV CSM-SRI operated at 105.4% and was different from the other two machines. The Mark IV CSM-SS operated at 99.8% and was not significantly different from the Mark IV CSM (which is used as the reference values) ($F_{2, 144} = 10.47, p < 0.001$).

3.3.1.3 Changes over time

Four machines were tested 8 months after first testing, during which time they were subjected to different levels of use. The Mark III CSMs output dropped significantly from 86.5% to 82.9% (Fig. 3.10a) ($F_{1, 346} = 12.20, p = 0.027$), while the Mark IV prototype had a large drop from 61.8% to 47.7% (Fig. 3.10b) ($F_{1, 346} = 40.05, p < 0.001$). Two Mark IV CSMs were tested and they registered insignificantly small changes in performance, 100% to 97.8% for the Mark IV CSM ($F_{1, 574} = 2.45, p = 0.118$) and 105.4% to 105.8% for the Mark IV CSM-SRI ($F_{1, 496} = 0.51, p = 0.476$) (Fig. 3.10c and d).

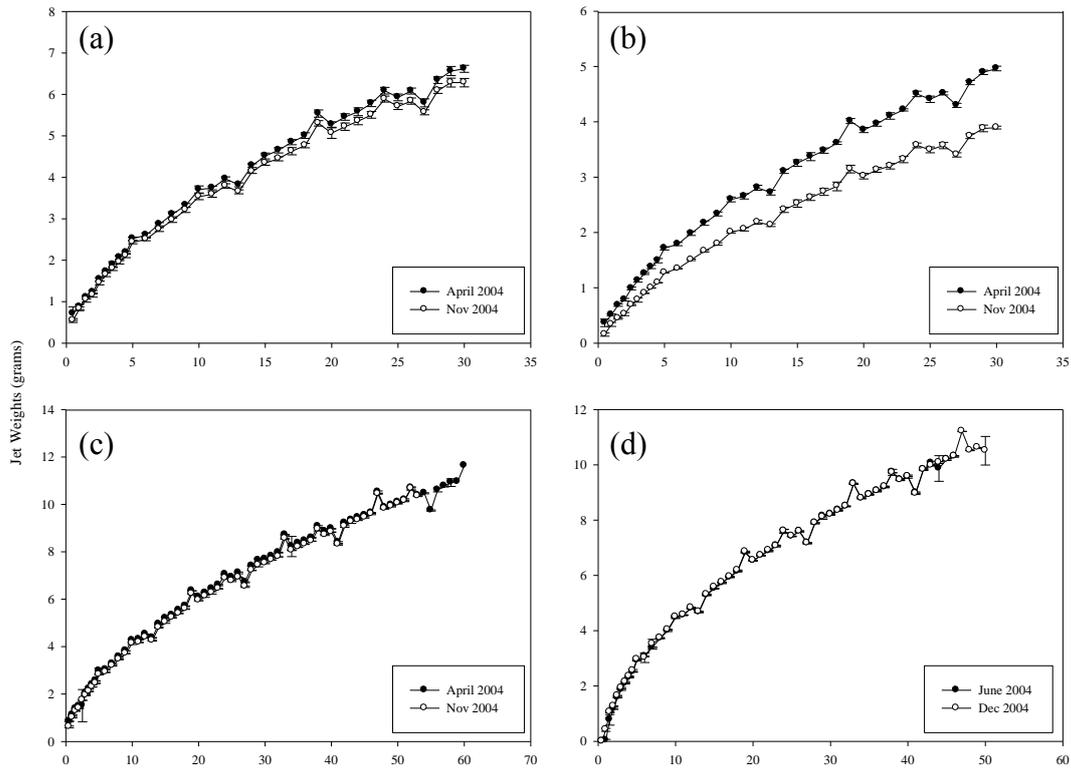


Figure 3.10. Changes in jet weights after 8 months; (a) Mark III CSM, (b) Mark IV prototype CSM, (c) Mark IV CSM and (d) Mark IV CSM-SRI.

3.3.1.4 Causes of the difference between CSM machines

The nozzle

The nozzle from the Mark IV CSM was attached to the Mark IV prototype CSM and tested on Fine 1. This did not cause a significant change ($F_{1, 176} = 2.76, p = 0.098$), with an increase of only 0.1% in output, although the difference in outputs does appear to be increasing with increasing pressure (Fig. 3.11).

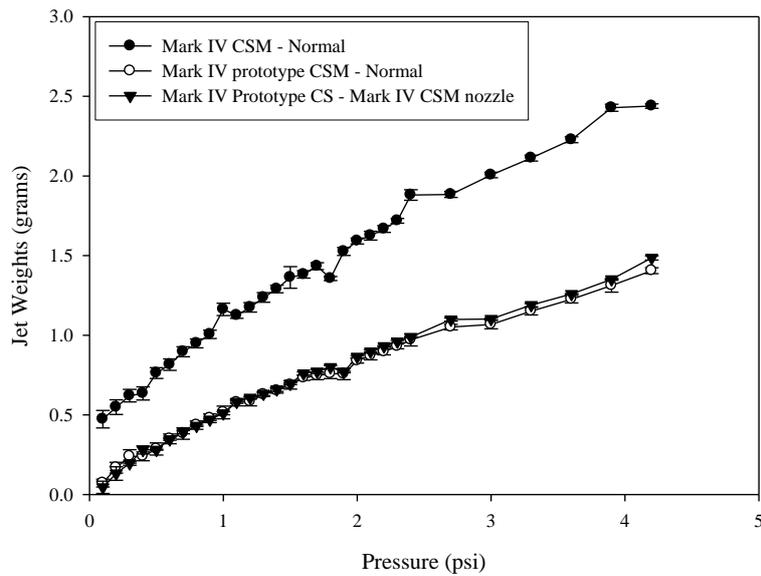


Figure 3.11. Effect of placing the nozzle from the Mark IV CSM onto the Mark IV prototype CSM.

The filter

The use of three different filters caused no change in the output of the Mark IV CSM ($F_{2, 849} = 0.30, p = 0.738$) (Fig. 3.12).

Internal hose length

Large changes in output occurred on all three machines with the different hose lengths, each showing a drop in output with increasing hose length (Fig. 3.13a-c). With increasing hose lengths the Mark IV CSM gave outputs of 109.6%, 100.0% and 87.8% ($F_{2, 444} = 28.92, p < 0.001$), the Mark IV CSM-SS gave 103.8%, 102.4% and 85.8% ($F_{2, 444} = 23.23, p < 0.001$ (short and medium length hoses were not significantly different)) and the Mark IV CSM-SRI gave 110.8%, 104.2% and 89.7% ($F_{2, 444} = 7.83, p < 0.001$).

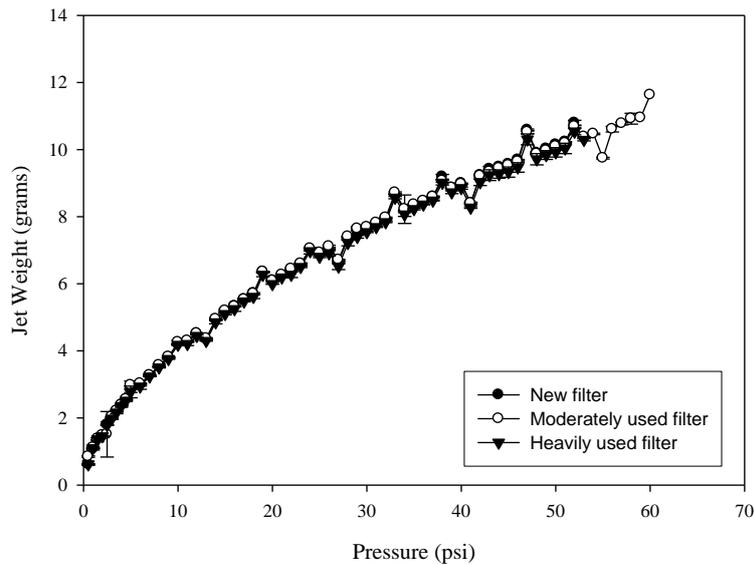


Figure 3.12. Output from different filters on the Mark IV CSM on Sand 9 setting.

3.3.2 Peaks and dips in output

3.3.2.1 Occurrence of the peaks and dips

Irregularities in output were noticed from all CSMs on all tests. Instead of producing a straight or smooth curve on a graph of increasing pressure against jet weight output, irregular results are either higher or lower than expected (Fig. 3.14). These irregularities occurred on all machines at the same points on each test. A comparison of the individual replicates used to create figure 3.14a shows that the peaks and dips are consistent between every run and not a result of erroneous results on a single run (Fig. 3.15). The cause for this was unknown but if the CSM is not firing the programmed jet it could have serious implications, e.g. an output higher than programmed would result in a stronger jet and may cause erosion but the CSM would record the jet as being fired at the programmed pressure and a false (lower) measurement of the critical erosion threshold would be taken.

The expected jet weight for the peaks and dips was obtained by averaging the 2 jet weights before and after the peak or dip, the error is then expressed as a percentage of the expected jet weight. Differences are relatively consistent within a test regardless of pressure but decrease between tests related to an increase in jet duration (Table 3.2).

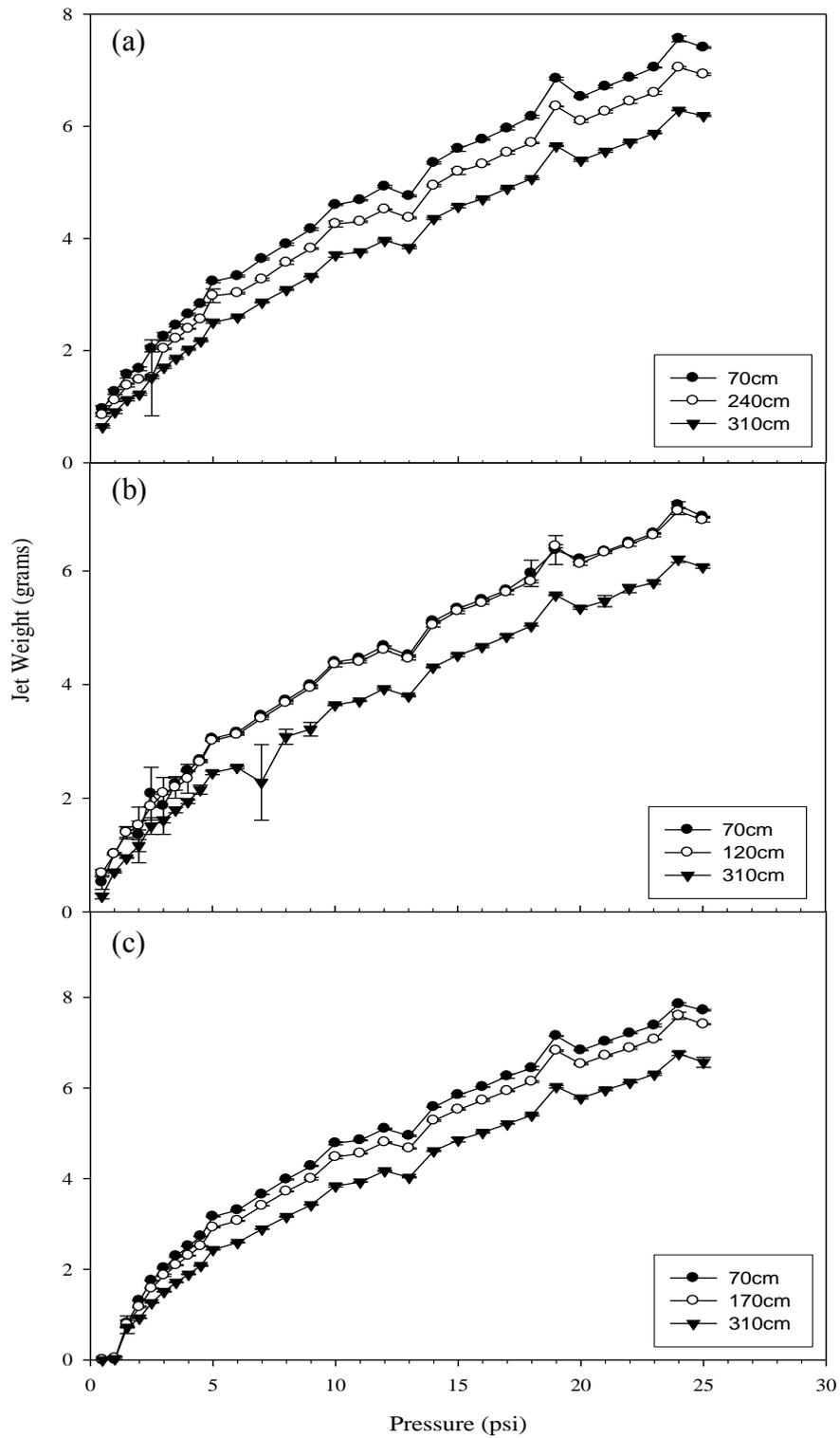


Figure 3.13. Effect of changing hose length between the CSM and nozzle, tested on Sand 9; (a) Mark IV CSM, (b) Mark IV CSM-SS and (c) Mark IV CSM-SRI.

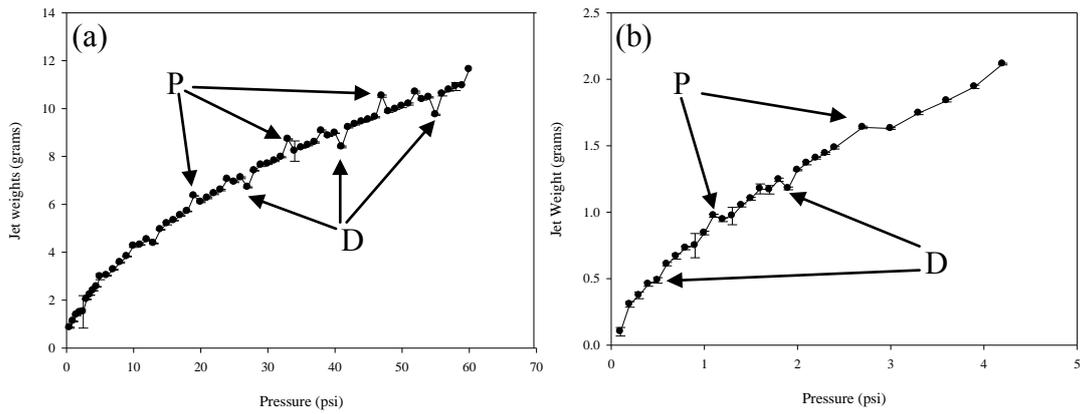


Figure 3.14. Examples of unexpected peaks (P) and dips (D) in output; (a) Mark IV CSM on Sand 9 (b) Mark III CSM on Fine 1.

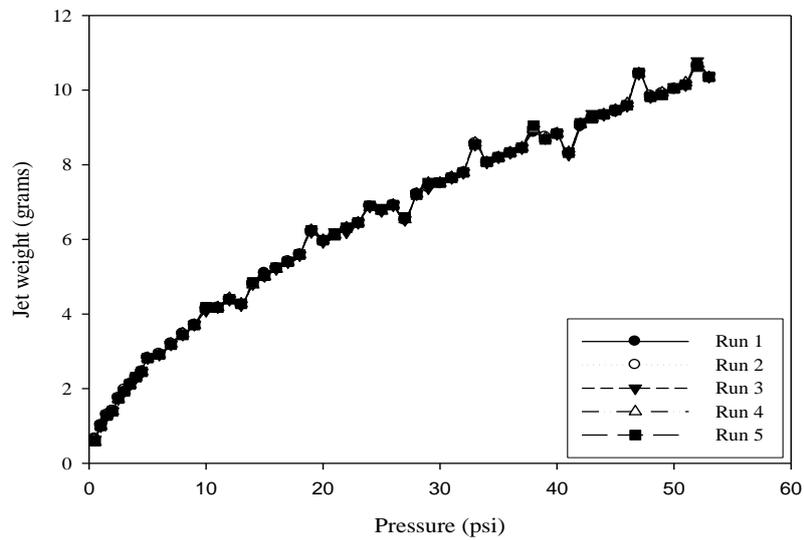


Figure 3.15. Consistency of the peaks and dips in output over 5 replicates of Fine 1.

Table 3.2. Calculated % errors of discrepancies in output.

	Peak		Dip	
	Average error	se	Average error	se
Sand 3	19.0	3.1	-26.5	0.8
Sand 9	5.7	1.6	-8.1	3.3
Sand 18	3.2	1.1	-2.6	n/a

To identify a pattern of when these peaks and dips were occurring jet weights were compared between three tests with a variety of firing pressures and pressure increments (Sand 3, 8 and 18) on the Mark IV CSM. The discrepancies did not appear to be related to specific pressures as a peak or dip may occur at a certain pressure on one test but not another (Fig. 3.16). However, when the three tests were compared based upon their jet number within the test a large number of the peaks and dips correlated between tests (Fig. 3.17). This would indicate that they are a result of a programming error rather than mechanical problem.

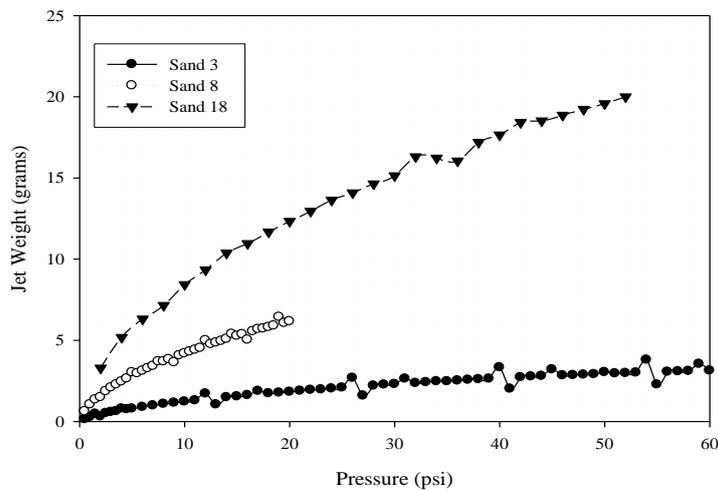


Figure 3.16. Comparing discrepancies in output of three test settings based upon firing pressure from the Mark IV CSM.

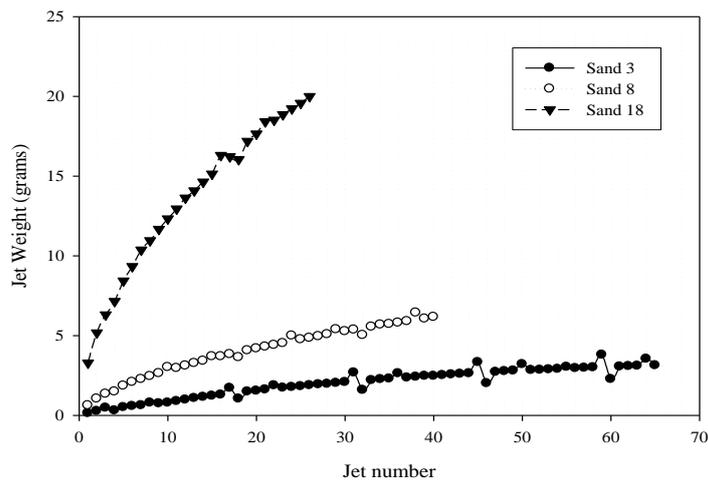


Figure 3.17. Comparing discrepancies in output of three test settings based on jet number in sequence from the Mark IV CSM.

3.3.2.2 Cause of the irregularities in output

Pressure

Internal pressure and stated firing pressure displayed a near linear relationship that does not appear to have any variations at specific pressures that could result in a change in output (Fig. 3.18). The error at 4.5psi appears to be an erroneous result based on reading error.

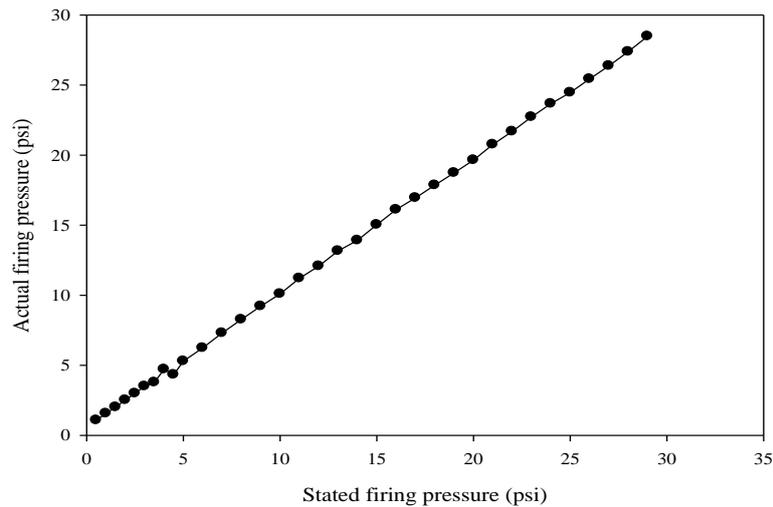


Figure 3.18. Actual firing pressure of the Mark IV CSM-SS compared to stated pressure.

Jet duration

Test settings Sand 3, 9 and 18 were selected for examining the jet duration as they had programmed firing times of 0.3, 1.0 and 2.0 seconds respectively. Differences in the length of the jet did occur in these tests, with shorter jets coinciding with lower than expected outputs and longer jets with higher outputs (Fig. 3.19). The difference in the scale of the discrepancy in jet output changes with test, with the shortest test (Sand 3) having the largest fluctuation while Sand 18 (the longest test) only showing a small level of fluctuation, however all timing errors appear consistently to be between 0.05 and 0.08 seconds. When the percentage errors in output (Table 3.2) are converted into errors in timing that would be needed to produce the peaks and dips the results indicate the error is consistently about 0.06 to 0.08 seconds regardless of the programmed jet duration (Table 3.3).

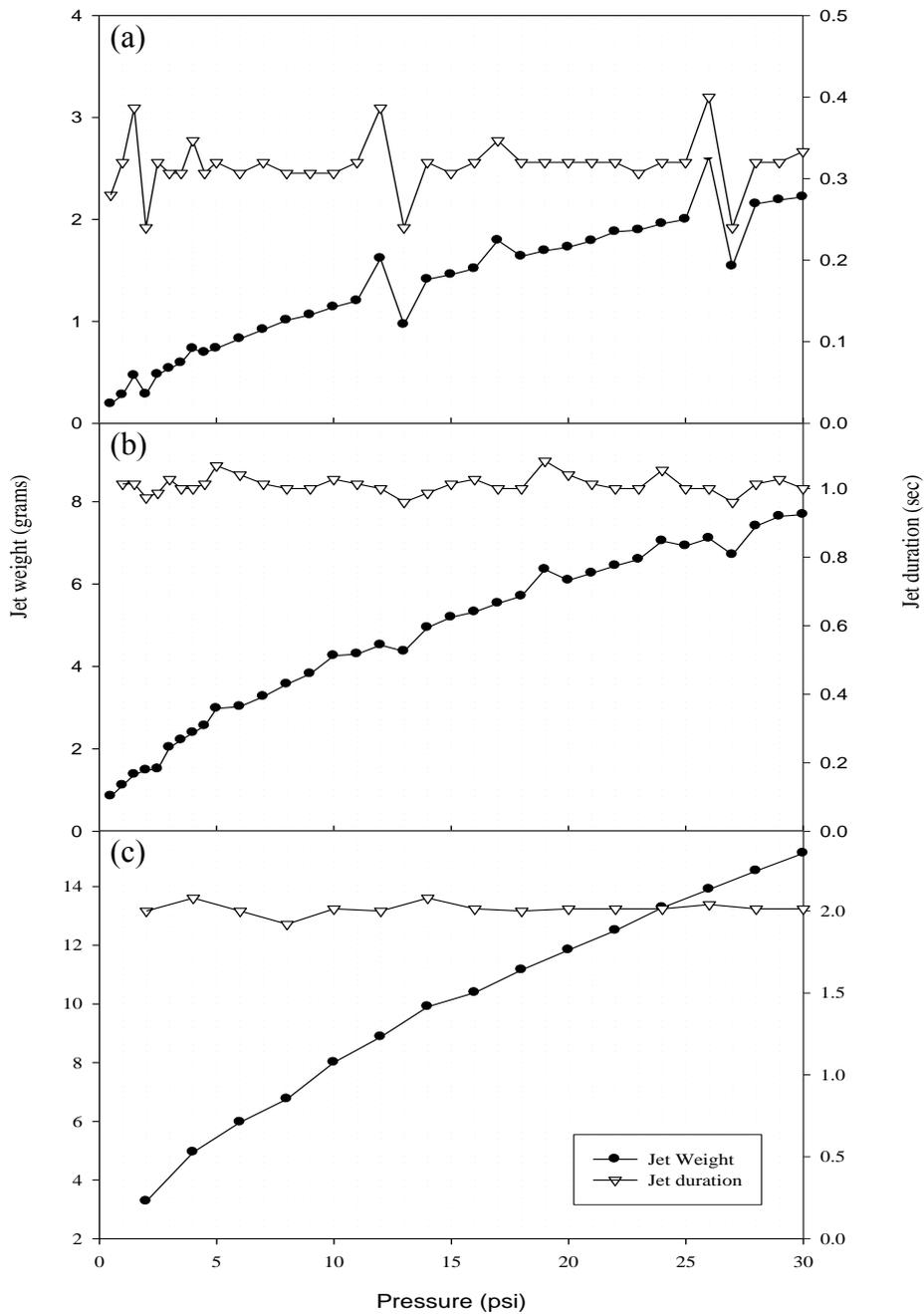


Figure 3.19. Jet output compared to duration; (a) Sand 3, (b) Sand 9 and (c) Sand 18.

Table 3.3. Calculated timing errors in jet duration.

	Peak		Dip	
	Average error	se	Average error	se
Sand 3	0.06	0.01	-0.08	0.00
Sand 9	0.06	0.01	-0.08	0.02
Sand 18	0.07	0.02	-0.05	n/a

3.3.3 Improving the erosion chamber of the CSM

3.3.3.1 Assessment of new components parts of the CSM

Nozzle size

As expected the larger internal diameters of the nozzles result in slower jets at specific pressures (Table 3.4), meaning the sediment erodes at higher pressures with increasing nozzle size. However, the degree of change is considerably larger than expected with a 3mm diameter nozzle requiring roughly 75 times the pressure of a 1mm nozzle to produce a jet capable of eroding the sediment. The 4.2mm diameter nozzle was unable to produce sufficient velocity in the jets to erode the sediment. To produce a CSM capable of running a nozzle of a diameter of 3mm would require increasing its maximum working pressure well above the existing 60psi limit and the water reservoir size would need to increase considerably to supply the volume of water a test would require.

Table 3.4. Jet pressure at which sediment erosion was observed with different nozzles. N/A indicates that erosion did not occur before the maximum 60psi pressure was obtained.

1mm	3mm	4.2mm
0.75	55.0	N/A
0.74	57.0	N/A
0.69	54.0	N/A
0.79	57.0	N/A
0.75	N/A	N/A
0.69	55.0	N/A
0.79	58.0	N/A
0.81	59.0	N/A
0.69	55.0	N/A

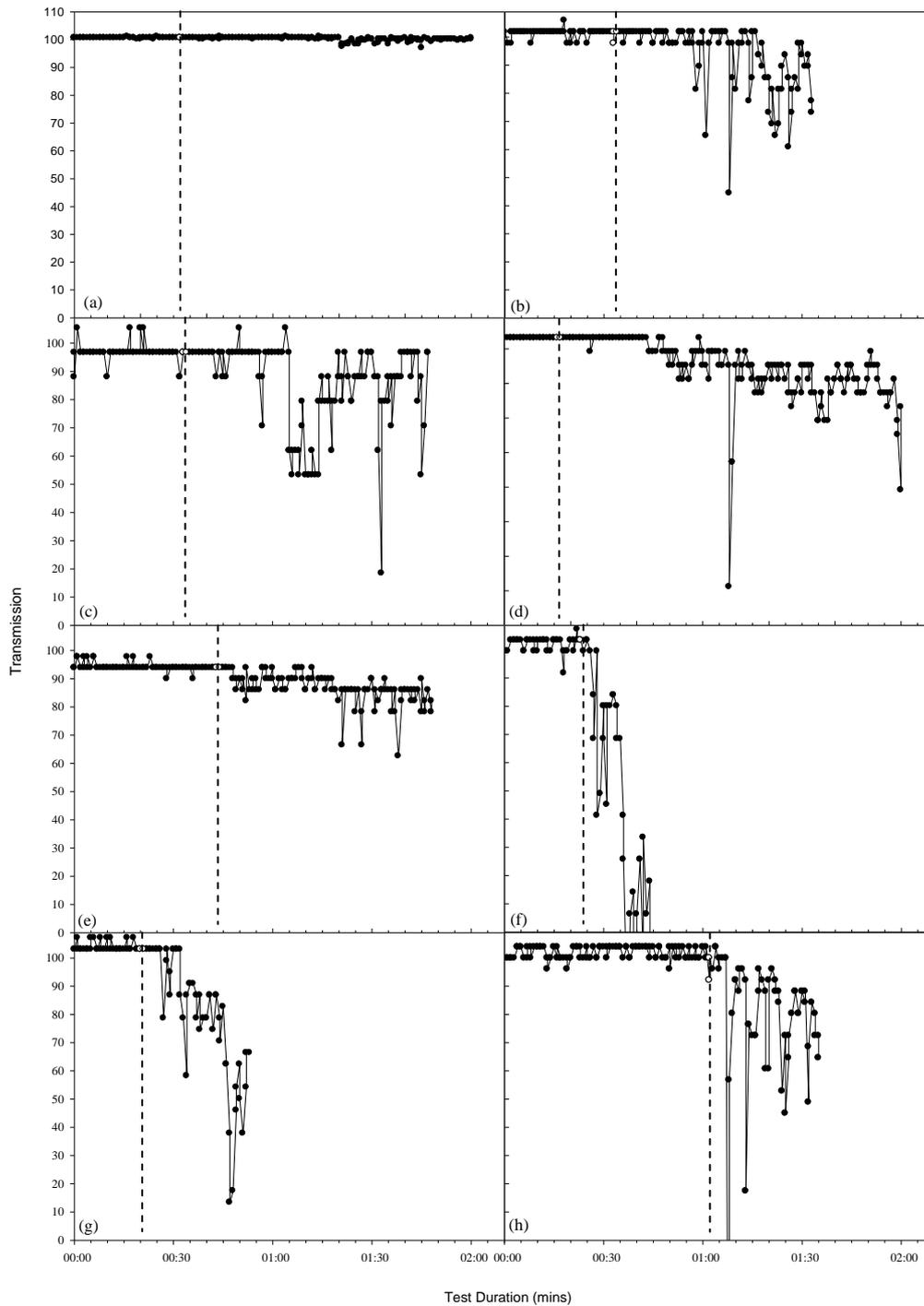


Figure 3.20. The detection levels of suspended sediment by the fibre optic system in different chamber sizes and using variable gains. Each graph shows a single run but is representative of three trials at each setting. Dashed line indicates the passing of critical erosion threshold (a) 80mm chamber, gain 1, (b) 80mm chamber, gain 5 (c) 80mm chamber, gain 10 (d) 50mm chamber, gain 5 (e) 40mm chamber, gain 5 (f) 30mm chamber, gain 5 (g) 40mm chamber, gain 5 (h) 30mm chamber, gain 5.

Fibre optic sensors and chamber size

A gain of 5 gave the best range of transmission results once suspended sediment was detected (Figure 3.20, a-c). However, this setup was not sensitive enough to detect suspended sediment at the critical erosion threshold. Smaller chamber sizes were tested on the assumption that the smaller volume of water contained within them would increase the concentration of the suspended sediment at the critical erosion threshold increasing the level of backscatter (Figure 3.20, d-f). Although smaller chambers do make the initial detection of suspended sediment closer to the critical erosion threshold they are still insufficient. To increase the detection of suspended sediment the sensor needs to be positioned where the highest concentrations will occur, therefore it was lowered to 5mm above the sediment surface (Figure 3.20 g and h). Again this improved the level of detection but the critical erosion threshold remains undetected. The closest any setup came to detecting the critical erosion threshold was the smallest chamber with the sensor positioned 5mm above the sediment surface (Figure 3.20, h). Further reducing the size of the chamber and lowering the sensor may eventually allow the critical erosion threshold to be detected but it was considered that this would interfere with the jet and inhibit the performance of the CSM. The fibre optic system gave low resolution in its output data, the current UV based sensor system gives output data on a scale of roughly 0-100, and a resolution of 0.01. Compared to the fibre optic system which (when set to have the same data output range) has a resolution of roughly 4.1 (Figure 3.21).

3.3.3.2 New method of deploying the chamber

Calculated errors caused by altering the nozzle height

The calculated stagnation pressure on the sediment surface is very dependant on the height from which the jet is fired, with errors increasing exponentially with increasing distance from the correct setting (Table 3.5).

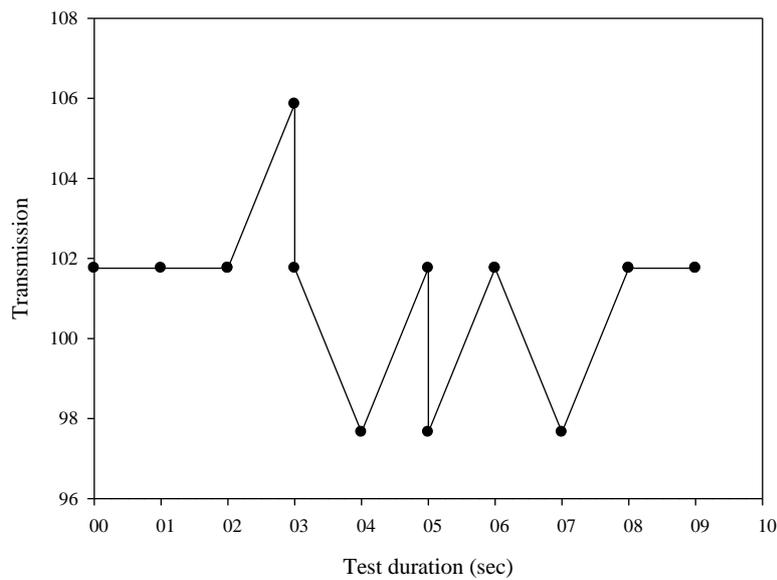


Figure 3.21. Sub sample from figure 3.20 displaying the resolution from the fibre optic system.

In situ analysis

Both methods of deployment proved to allow great precision in deployment of the CSM chamber, with the chamber positioned correctly on all tests, neglecting the need to calculate the potential error. Unsurprisingly, there was no difference between the critical erosion thresholds using either method, although a large site difference did occur (deployment method, $F_1 = 0.64$, $p = 0.430$; site, $F_1 = 282.69$, $p < 0.001$; interaction, $F_1 = 0.03$, $p = 0.874$).

Table 3.5. Calculated stagnation pressures on the sediment surface using a jet volume of 1ml fired from a nozzle positioned at different heights from the sediment surface.

Nozzle Height from sediment surface cm	Calculated Stagnation pressure on the sediment surface Nm⁻²	Difference from calibrated stagnation pressure Nm⁻²	% Error
1.00	311.94	-233.96	-300.00
1.50	138.64	-60.66	-77.78
1.90	86.41	-8.43	-10.80
1.91	85.51	-7.52	-9.65
1.92	84.62	-6.63	-8.51
1.93	83.75	-5.76	-7.39
1.94	82.88	-4.90	-6.28
1.95	82.04	-4.05	-5.19
1.96	81.20	-3.22	-4.12
1.97	80.38	-2.39	-3.07
1.98	79.57	-1.58	-2.03
1.99	78.77	-0.79	-1.01
2.00	77.99	0.00	0.00
2.01	77.21	0.77	0.99
2.02	76.45	1.54	1.97
2.03	75.70	2.29	2.93
2.04	74.96	3.03	3.88
2.05	74.23	3.76	4.82
2.06	73.51	4.48	5.74
2.07	72.80	5.19	6.65
2.08	72.10	5.88	7.54
2.09	71.41	6.57	8.43
2.10	70.74	7.25	9.30
2.15	67.48	10.50	13.47
3.00	34.66	43.33	55.56

3.4 Discussion

3.4.1 Discrepancies in the outputs of CSMs

Differences between machines

All CSM machines have different jet outputs when firing at an equal pressure. These differences are largest between different models of CSM, however different machines of the same model also vary slightly with time, although this is only evident in older machines. Variations in the hose length caused the largest difference in output, possibly as a result of internal friction or drag. This would suggest that changes between the designs of the three CSM models, probably with different internal hose lengths results in more or less drag, causing the differences in output.

Swapping nozzles did not affect the output of the CSM tested while filter wear was also not significant, although the filter type tested was only used on Mark IV CSMs which did not show a change in output over time. Both the Mark III and Mark IV prototype machines differ over time but have an inline filter that was impossible to replace. Combining these factors it is probable to suggest that each CSM has a unique output at each pressure, and that this might change over time. Because of this, results from a CSM should be considered unique to that machine at that time, with comparisons between CSMs or between the same CSM at different times invalid. The impact of this on previous studies where multiple CSMs were used simultaneously and results compared as one is significant, with serious doubts about the validity of any trends in results, especially where there is little variation in stability readings (e.g. Defew *et al.*, 2002; Defew *et al.*, 2003; Tolhurst *et al.*, 2006c). It would still be possible to use the data from each individual CSM if each result could be matched to a machine, however direct comparisons between the CSMs would still be impossible.

A summary of this research has been presented to Sediment Services and new CSMs will have equal internal hose lengths to rectify the differences caused by varying lengths between machines.

Correcting for differences in CSMs

The calibration suggested by Tolhurst *et al.*, (1999) was based on expressing the result based upon the stated firing pressure of the CSM when erosion occurred. This

work has shown that this is an invalid method of measuring the erosion of sediment given it is not consistent with the erosional force of the jet produced by different machines. A new method of quantifying the output of the CSM is suggested by Vardy *et al.* (2007) based upon the jet weights produced by the CSM under test conditions. This method of calibration can be used on all CSMs and allows direct comparisons of results from any machine, although back dating the calibration is impossible without knowing the weights of jets produced at the time of the tests. This method relies on regular measurements of the CSM output, to account for changes with time.

Irregularities in the output of the CSMs

The apparently random occurrence of higher and lower outputs than expected for certain jets was discovered to be a timing error, with higher outputs a result of the jet firing for longer than expected and smaller outputs caused by shorter jets, both with a error of between 0.04 and 0.08 seconds. No change in the firing pressure was found related to these irregular outputs, therefore there is no indication of the irregularities producing stronger or weaker jets than expected. As the timing error is so small and the jets are fired at the correct pressure it is considered that there is no adverse effect on the functioning of the machine. Therefore measurements of erosion that may occur at a pressure that corresponds to a peak or dip can still be considered valid. However, such erroneous measurements of jet weight would have an effect on the calibration method suggested by Vardy *et al* (2007), therefore they need to be removed from the relevant equations.

In consultation with Sediment Services it was subsequently discovered that the timing errors are a result of the programming of the firing mechanism, with an error equal to that discovered within this work. This error has been corrected for all new machines and while existing CSMs will continue to have the fault it should not affect their performance.

3.4.2 Improving the CSM

Changing components of the erosion chamber

In an attempt to improve the CSM changes to the erosion chamber were trailed. The conditions required for a successful trial were that the new component either

improved the accuracy in detecting the critical erosion threshold or allowed the CSM to be manufactured more economically. Both of the new components tested failed in one of these categories. Increasing the diameter of the nozzle would give smaller increases in the force of the jet, probably leading to an increase in accuracy in detecting the critical erosion threshold, however with the current set up the CSM does not produce sufficient pressure to work a larger nozzle and the engineering needed to build a CSM capable of providing such pressures would be impractical. The fibre optic sensor is cheaper than the UV transmission system used currently and would additionally allow the erosion chamber to be constructed more economically. However, the system proved to be less accurate and have considerably less resolution than the current system, not being able to detect the point at which the critical erosion threshold was passed and having large increments in changes in transmission.

Deployment of the CSM

Correctly positioning the erosion chamber, and ensuring the nozzle is positioned correctly within is vitally important, with differences of nearly 10% in the erosional force of the jet on the sediment surface resulting from an error in height of only 1mm. This highlights the importance of correctly deploying the CSM and also ensuring that the nozzle is at the correct height within the chamber. Both methods of deploying the CSM chamber were found to be equally accurate, however since its conception the disk has been used by several experienced CSM operatives who have expressed a preference for it due to the ease of use.

3.5 Conclusions

For the first time since its conception in 1989 there has been a thorough simultaneous evaluation of the construction and operation of the CSM. Given the success of the CSM it was not surprising to find that changing parts of the machine or the deployment method did not improve its operational capabilities. However, the evaluation did discover important differences in the output of individual machines that has severe implications for previous and subsequent work done with multiple machines. With a new calibration method proposed by Vardy *et al.*, (2007) this has been rectified and the CSM should continue to be a successful tool for measuring the critical erosion thresholds of sediment surfaces.

Chapter Four

Intertidal sediment stability from tidal exposure to submersion

Abstract

Intertidal sediment is eroded during submersion when water flow surpasses the critical shear required to resuspend the sediment. Despite this, the *in situ* study of intertidal sediment stability is almost entirely based on measurements taken during exposure during low tide. This is mostly due to the complications of sampling during submersion and the comparative ease of access to exposed sediments. To rectify this inconsistency attempts were made to study sediment stability during the transition from exposed to submerged conditions. *In situ* measurements were taken from two sites within the Eden Estuary under a variety of submersion scenarios while laboratory-based experiments were designed to replicate natural submersion conditions. Both *in situ* and laboratory-based experiments required the adaptation of techniques originally developed for work on exposed sediment. These techniques are considered in reference to their success and application. *In situ* data showed that there was an increase in sediment stability with submersion by a moving incoming tide while results from laboratory-based experiments were inconsistent, probably due to inadequacies in replicating natural submersion. Possible explanations for this are discussed. Two small projects were used as trials for the new laboratory and *in situ* methodologies with relative success.

4.1 Introduction

4.1.1 The study of intertidal sediment stability

The importance of sediment stability relating to coastal erosion and protection of intertidal ecosystems has been highlighted many times (e.g. Widdows & Brinsley, 2002; Watts *et al.*, 2003; Winn *et al.*, 2003; Wood & Widdows, 2003). In studies of intertidal sediment, a suite of measurements are often taken to characterise the sediments and the ecosystem they support. Such measurements usually include grain size, water content, bulk density, organic, chlorophyll *a*, colloidal carbohydrate content and quantification of the macrofaunal community along with a variety of other properties.

Each of these characteristics has been found to have a degree of influence over the overall sediment stability of the systems. A decline in water content increases the critical erosion threshold (Underwood & Paterson, 1993*a*), a pattern which can be observed over an exposure period with stability increasing as the sediment dewateres (Perkins *et al.*, 2003; Tolhurst *et al.*, 2003; Friend *et al.*, 2003*a*).

Proxy measurements of sediment properties are often used in place of actual variables, for example carbohydrate content is used as proxy measurement of EPS and is often correlated with sediment stability (Decho, 2000). Equally, chlorophyll *a* is also used as a proxy measurement for the biomass of microphytobenthic organisms (Underwood & Paterson, 1993*a*). Other variables that often relate to sediment stability include grain size fractions, with non-cohesive sediments becoming more stable with increasing size as a result of the increasing physical force required to erode them (Fig. 1.1). This pattern is reversed in particles below the non-cohesive/cohesive threshold (taken as 63 μ m) where interactions on the sediment particle surface result in cohesiveness and an increase in the force required for erosion.

With so many different influences upon sediment stability in such a heterogeneous environment it is understandable that sediment stability is very difficult to predict. However, with the importance of sediment transport for coastal management and development there is increasing need for a method of measuring large-scale sediment

stability. Some workers have attempted to devise a proxy measurement of sediment stability based on limited measurements of sediment and ecosystem properties (Defew *et al.*, 2003; Friend *et al.*, 2003a). These studies have had limited success, with any significant correlation of sediment properties and stability often stated as site specific. The natural heterogeneity between and within intertidal systems is often given as an explanation for this inconsistency when comparing stability patterns in such areas.

The thorough investigation of sediment stability in intertidal systems has led to a comprehensive, but not complete, understanding of the properties and influences acting upon it. However, nearly all this work is based upon measurements and recordings of sediment exposed during low tide. Continuing the approach of working on exposed sediments will only ever reveal half the story of sediment properties and stability during a tidal cycle.

4.1.2 Submerged sub-tidal sediment stability

Currently, considerably less is known about the properties of subtidal sediments as compared to exposed intertidal sediments, obviously due to the difficulty of working in such an environment. The few studies into the influences upon stability in subtidal sediments have found patterns similar to those found on exposed intertidal sediments. These have included the stabilising influence of EPS, through readings of colloidal carbohydrates (Sutherland *et al.*, 1998; Aspden *et al.*, 2004; Ziervogel & Forster, 2006) and the microphytobenthos (Sutherland *et al.*, 1998; Aspden *et al.*, 2004). Water content is negatively correlated with sediment stability in submerged sediments (Ziervogel & Forster, 2006) as is bulk density (Aspden *et al.*, 2004), and physical disturbance reduces sediment stability (Hauton & Paterson, 2003; Aspden *et al.*, 2004).

As with exposed intertidal sediment, submerged sediment has been studied *in situ* and within the laboratory. *In situ* measurements involve either the deployment of a flume onto the sediment surface (e.g. Sutherland *et al.*, 1998) or measurements of suspended particulate matter referenced to hydrodynamic conditions (e.g. Wang, 2003). As sampling *in situ* is technically and logistically challenging, sediment is often collected from the bed by corers or divers and tested on the surface (e.g. Aspden *et al.*, 2004). It could be argued that by using a core to collect samples in this manner the sampling

can no longer be described as *in situ*, however for this study the definition of *in situ* sampling is taken to mean sediments that are subjected to natural conditions rather than replicated conditions in the laboratory. Equally collected sediment cores are often taken from the field and directly placed in laboratory based flumes (e.g. El Ganaoui *et al.*, 2004; Orvain *et al.*, 2006).

A common occurrence with submerged sediment is the presence of two distinct layers; the main bed made of well-consolidated sediment with a critical erosion threshold higher than that normally experience in the flow conditions of the local environment (Wang, 2003) and above that a layer of less consolidated sediment, termed the fluff layer. The composition of the fluff layer varies with location but mostly comprises of recently deposited sediment (El Ganaoui *et al.*, 2004), organic debris (Jago *et al.*, 2002), and the re-deposited remains from the effects of bioturbation (Orvain *et al.*, 2006). This fluff layer erodes at a much lower stress level than the consolidated layer beneath (Wang, 2003; El Ganaoui *et al.*, 2004; Orvain *et al.*, 2006). Maintaining this fluff layer is important when sampling submerged sediment and sampling methods should include some of the undisturbed overlying water column to prevent the fluff layer being removed or incorporated into the consolidated layer (Schaaff *et al.*, 2006; Spears *et al.*, 2007).

4.1.3 Submerged intertidal sediment stability

The study of sediment stability appears to be based either on exposed sediments in the intertidal zone or submerged sediments in the subtidal, this leaves a link between the two systems unfulfilled, with the erosional characteristics of submerged intertidal sediments largely unstudied. This is a problem that needs to be addressed because while the work on exposed sediment is extensive, it is during submersion that sediment erosion and deposition actually takes place. While it may not be unreasonable to assume that because trends in stability for exposed and submerged sediment are similar, stability on submerged intertidal sediments will be subject to the same influences. However, it would be equally unreasonable to expect that properties of exposed intertidal sediments are not in some way affected by submersion by the incoming tide, and as such predictions of sediment stability based upon measurements of exposed sediments and subsequent models of stability need to be calibrated against the effect of submersion.

In a preliminary study of submerged intertidal sediments Tolhurst *et al.* (2006a) found a reduction in stability, colloidal carbohydrates and water content in intertidal sediment that had been submerged for a 6 hour period and a further change in stability over a tidal cycle in both exposed and submerged sediments. The changes in sediment properties that occur once submerged, highlighted in this study, draws attention to the lack of understanding of several key properties related to sediment stability in submerged intertidal sediments. Additionally, changes in the sediment properties between exposed and submerged sediment means that it may not be valid to use measurements taken on exposed sediment to predict sediment stability once the sediment is submerged.

Many of the properties of sediments that contribute to sediment stability on exposed sediments may be significantly altered by submersion. EPS can act as a stabilising influence on exposed sediments. However, EPS is primarily composed of colloidal carbohydrates that are readily soluble (Taylor *et al.*, 1999; Perkins *et al.*, 2004) and as a result levels have been found to be lower in sediments with high water concentrations (Blanchard *et al.*, 2000). Complete submersion may further lower the EPS concentration within the sediment, reducing its stabilising effect. Equally in intertidal sediments diatoms migrate to the sediment surface over an exposure period, stabilising the sediment surface before returning to deeper sediment prior to submersion (Tolhurst *et al.*, 2003; Consalvey *et al.*, 2004), how and if this pattern continues once the sediment is submerged is unknown. Considerations such as these need to be assessed in order to determine if patterns of sediment properties and stability extend into the submerged period. The final aim of which must be to achieve an understanding of sediment properties over a complete tidal cycle and determine if readings on the exposed sediments can be used to assess submerged sediment.

4.1.4 The submerged intertidal environment

Testing submerged sediment *in situ* is obviously very difficult and potentially hazardous, and because of this, studies examining submerged sediments are often based on replicating submersion within the laboratory. However, for this to be worthwhile accurate replication of the conditions of submersion is required. The few studies that have examined submerged sediments in the laboratory have mostly been based on sample cores from the field transported to the laboratory and placed into

tidal tanks or water filled containers (Tolhurst *et al.*, 2006a). How accurately this replicates tidal submersion by flowing water is debatable. The act of submersion on an estuary or tidal flat is a very dynamic process with an initial “wave” of water moving onto the sediment, while during submersion sediments are subjected to conditions of flow and turbulence from the water column. Measurements of sediment stability are often taken in flumes where flow speed is increased until it surpasses the critical erosion threshold (e.g. Orvain *et al.*, 2006; Schaaff *et al.*, 2006). By their very existence mudflats must not be subjected to flow speeds that exceed the critical erosion threshold on a regular basis but will be continually subjected to slow sub-critical tidal currents. Flow is an important aspect of the intertidal ecosystems as it can change the behaviour of macrofauna species within sediment (Thomsen & Flach, 1997; Ford & Paterson, 2001), alter the nutrient flux between sediment and water (Biles *et al.*, 2003) and bioturbatory activity of several macrofauna species (Biles *et al.*, 2003). However, little is known about the importance of flow and the transition from exposure to submersion in studies of biogenic influences on sediment stabilisation and it is largely ignored for simplicity in many laboratory based studies.

4.1.5 Submerged sediment sampling and analysis techniques

Through the large number of experiments on exposed sediment there has developed an impressive range of techniques, equipment and protocols to test for all aspects of exposed sediment properties (HIMOM, 2005). Devising techniques which produce results from submerged sediments to the accuracy achievable by current techniques on exposed sediments is essential. This will require drawing from the wealth of experience on exposed sediments and selecting, with modifications where necessary, appropriate techniques.

4.1.6 Aims of the Chapter

In this Chapter, the effects of submergence by the natural tide on several physical and biogenic sediment properties are investigated and related to the overall stability of the sediment. *In situ* and laboratory based experiments were performed with the stability results compared in reference to changes in sediment properties. Assessment of the accuracy of laboratory based experiments in replicating submersion was an important consideration. Integral to these experiments was the development and assessment of techniques and methodologies for sampling sediments in the laboratory and *in situ*.

After the initial experiments were performed two supplementary experiments were devised to utilise and assess the methods developed. This supplementary work included a simple laboratory-based experiment and an *in situ* survey, based upon:

- Assessing the effects of submersion on EPS within the sediment and its subsequent effect on sediment stability.
- Surveying submerged sediment properties and stability along a river transect.

4.2. Material and Methods

4.2.1 General approach

The stability and related properties of exposed and submerged intertidal sediments were compared in both laboratory and *in situ* experiments. In the laboratory, the stability and properties of sediment were studied from exposure into submersion under a variety of submersion scenarios and periods. *In situ*, stability and sediment properties were again compared between exposed and submerged sediment using the natural tidal cycle and artificial replications of submersion. Experiments were devised to study the effect of the initial wave of water from the incoming tide as well as prolonged submersion in stationary and sub-critical shear velocity flowing water. All laboratory based experiments used $n = 6$, while logistics limited *in situ* experiments to $n = 5$.

4.2.2 Study sites

As previous studies had found a high level of site specific trends in the factors influencing sediment stability two sample sites within the Eden Estuary were studied (the Papermill and Golf Course sites (See Chapter 2)). Sediment cores were collected from both sites for laboratory analysis between February and April 2006 and *in situ* sampling occurred in July and August of the same year.

4.2.3 Sediment Collection

4.2.3.1 Laboratory-based experiments

Sediment cores were collected at low tide from both sites in the morning, prior to the experiments being performed in laboratory in the afternoon of the same day. On each occasion all cores were taken within an area of about 60cm² that appeared visually homogenous to reduce local heterogeneity. On each sampling occasion cores were taken from the same area of sediment although care was taken not to sample from areas potentially disturbed from previous collection. For analysis the surface of the sediment cores needed to be flush with the top of the core, however, removing undisturbed cores from the sediment which were flush with the sediment surface proved very difficult and many cores were disturbed. To overcome this the plastic cores were pushed to within about 5mm of the sediment surface and removed to the laboratory where additional sediment was added to the bottom of the core, pushing the undisturbed sediment surface to the top of the core (Fig. 4.1).

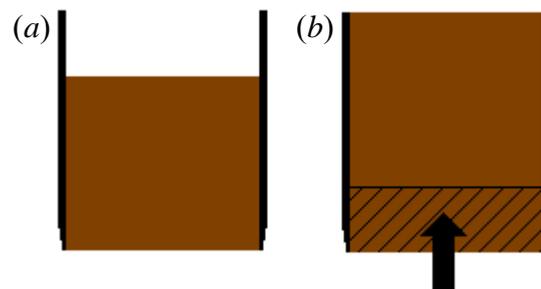


Figure 4.1. (a) Sediment core taken from the intertidal mudflat with sediment surface about 5mm below top of core. (b) Sediment is gently pushed to the top of the core and additional sediment added to the bottom of the core.

4.2.3.2 *In situ* experiments

In situ study of the sediment surface utilised a number of methods of collecting sediment depending upon the requirements. Measurements and sampling was performed directly on the exposed sediment surface when it was exposed during low tide. After initial submersion the sediment was either tested underwater where possible or if an exposed surface was required for sampling, a core of sufficient height to reach the surface was pushed slightly into the sediment and the water carefully extracted through a syringe so not to disturb the sediment surface. In deeper water a sediment core with overlaying water was taken from the substratum before the water

was again siphoned off. Sediment cores were collected by standing in shallow water where possible, while deeper sediments were collected in specialist cores by snorkelling. These cores consisted of a transparent plastic tube (60cm long, 8.5cm diameter), which was pushed 10-15cm into the sediment. The core was then dug out of the sediment and bunged at the top and bottom (while still underwater) to allow an undisturbed core of sediment and overlaying water to be transported to the surface. Once on the surface the top bung was removed and the sediment tested with the overlaying water if possible or where access to the sediment surface was required the bottom bung was pushed up through the core, in turn moving the sediment to the top of the core while the water flowed out of the core.

4.2.4 Sediment submersion

Submersion in the laboratory was replicated in stationary and flowing water. Glass tanks filled with filtered sea water were used for stationary water, into which sediment cores were gently lowered. Flowing water and the initial “wave” of tidal submersion was replicated using an Armfield S6 tilting flume (working section 5m length x 30cm width x 45cm depth; Fig 4.2) set to an angle of 4 degrees to allow the initial water front to flow up the flume in the manner of the incoming tide moving up a tidal flat. A removable section of the base of the flume was modified to allow 6 sediment cores to be placed with the sediment surface flush with the base of the flume. Initial submersion was simulated in stationary water and with flow speeds of 0.12, 0.23 and 0.48ms⁻¹. Prolonged submersion was performed in stationary water in the tanks and in flowing water (set to 0.12ms⁻¹) in the flume for periods of 1, 3 and 5 hours.



Figure 4.2. Armfield S6 tilting flume

In situ experiments mostly utilised the natural incoming tide to submerge sediments. However for the initial submersion experiments two additional submersion treatments were tested in addition to measurements of exposed and tidally submerged sediment. Submersion in still water was replicated by pushing a core (diameter 8.5cm) slightly into the exposed sediment and gently filling it with filtered seawater. A circle of bubble wrap was used to protect the sediment surface while pouring water into the cores. The effect of the incoming wave of incoming tide on exposed sediment was tested by pouring filtered seawater over the exposed sediment before sampling the sediment surface.

4.2.5 Sediment stability

A Mark III CSM was used to determine sediment stability for all experiments, calibrated following Vardy *et al.*, (2007) with results expressed as the stagnation pressure experience on the sediment surface (Nm^{-2}). On exposed sediment, the CSM chamber was placed upon the sediment surface and tests performed normally. The erosion chamber of the CSM is completely waterproof and capable of being used on submerged sediments. This allows a submerged sediment surface to be tested without

the prior removal of overlying water and reducing the disturbance this may have caused. In the laboratory this allowed the CSM chamber to be lowered into either the flume or tank to test the stability of the sediments. *In situ* the CSM was either floated on a small inflatable boat from which the chamber could be lowered onto the sediment surface in shallow conditions, or where the water was too deep, the CSM chamber was lowered into a core containing an undisturbed sediment surface with overlying water. In all cases, great care was taken to prevent the CSM sense head disturbing the sediment surface. This was helped by the removal of the lid of the CSM chamber, allowing water to flow through the chamber as it was pushed through the water, preventing the formation of a pressure wave. The transparent sampling core allowed the chamber to be lowered onto the sediment surface accurately and transmission values were monitored prior to a test to check for suspended sediment in the water caused by disturbance. Fine 1 was selected as the test setting on the CSM due to its high resolution at low erosion pressures expected after initial trials.

4.2.6 Sediment sampling

To sample the sediment surface for later analysis of its properties two recognised techniques from work on exposed sediments were used. In the laboratory, a course coring method based on a small syringe was used (0.8cm diameter; Fig. 4.3). This allowed a core of about 1cm depth to be taken, from which the upper 2mm could be removed by partitioning. The sediment was then quickly wrapped in tin foil and frozen in liquid nitrogen. The course core samples were taken from the same sediment cores used for the CSM. The Contact Core (HIMOM 2005) was used for *in situ* sampling as surface area was not limiting, again the upper 2mm of sediment was collected. In both cases sediment samples were transferred to a -80°C storage freezer before analysis of sediment properties was performed through the protocols detailed in Chapter 2.



Figure 4.3. The course corer, with sediment pushed out of the core and cut to the desired depth.

4.2.7 Statistical analysis

Comparisons of different submersion types and durations was carried out using parametric T-tests and one-way ANOVA with a Tukey's *post hoc* test. A two-way ANOVA compared changes in sediment stability and properties over time between submersion treatments. Determining if sediment stability was related to a sediment property was based upon Spearman rank correlations of stability with the sediment properties combining exposed and submerged sediment measurements.

4.3 Results

4.3.1 Laboratory experiments

4.3.1.1 Initial submersion of exposed sediment cores

Sediment stability

Sediment stability was higher in exposed cores from the Papermill site than those from the Golf Course (d.f. = 4, $T = 3.03$, $p = 0.029$). After initial submersion, sediment from the Golf Course site increased in stability under the two most rapid flow speeds ($F_{4,25} = 3.80$, $p = 0.015$) (Fig. 4.4). Such changes were not found in Papermill sediments where initial submersion caused no change in stability ($F_{4,25} = 1.75$, $p = 0.173$) (Fig. 4.4).

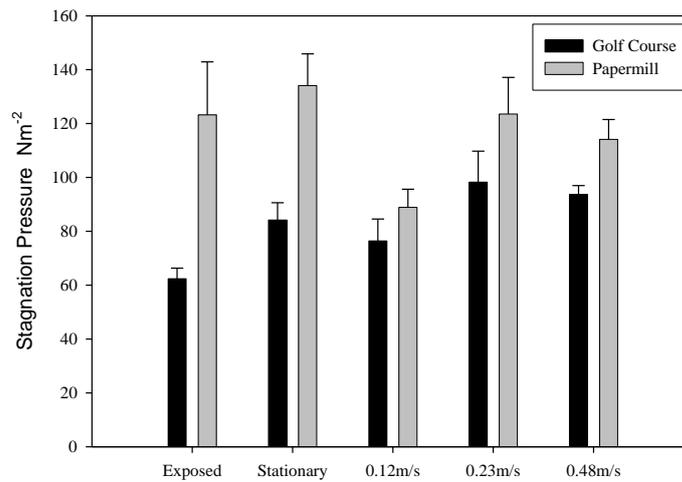


Figure 4.4. Sediment stability after initial submersion in water of different flow speeds.

Sediment properties

Submersion of the Papermill sediments caused no change in any of the measured sediment properties. On Golf Course sediments only colloidal carbohydrate concentration differed between treatments but with no relationship between flow speeds (Fig. 4.5; Table 4.1).

Table 4.1. One-way ANOVA results of comparisons of sediment properties after initial submersion in a laboratory based experiment.

	d.f.	F value	p value	Tukey Differences
Golf Course				
Water Concentration g cm ⁻³	4, 25	0.37	0.829	n/a
Dry Bulk Density g cm ⁻³	4, 25	2.15	0.105	n/a
Organic Concentration mg cm ⁻³	4, 25	1.60	0.206	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	4, 25	3.81	0.015	0.12ms ⁻¹ < 0.23ms ⁻¹
Papermill				
Water Concentration g cm ⁻³	4, 25	0.40	0.806	n/a
Dry Bulk Density g cm ⁻³	4, 25	2.68	0.055	n/a
Organic Concentration mg cm ⁻³	4, 25	0.68	0.612	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	4, 25	1.17	0.348	n/a

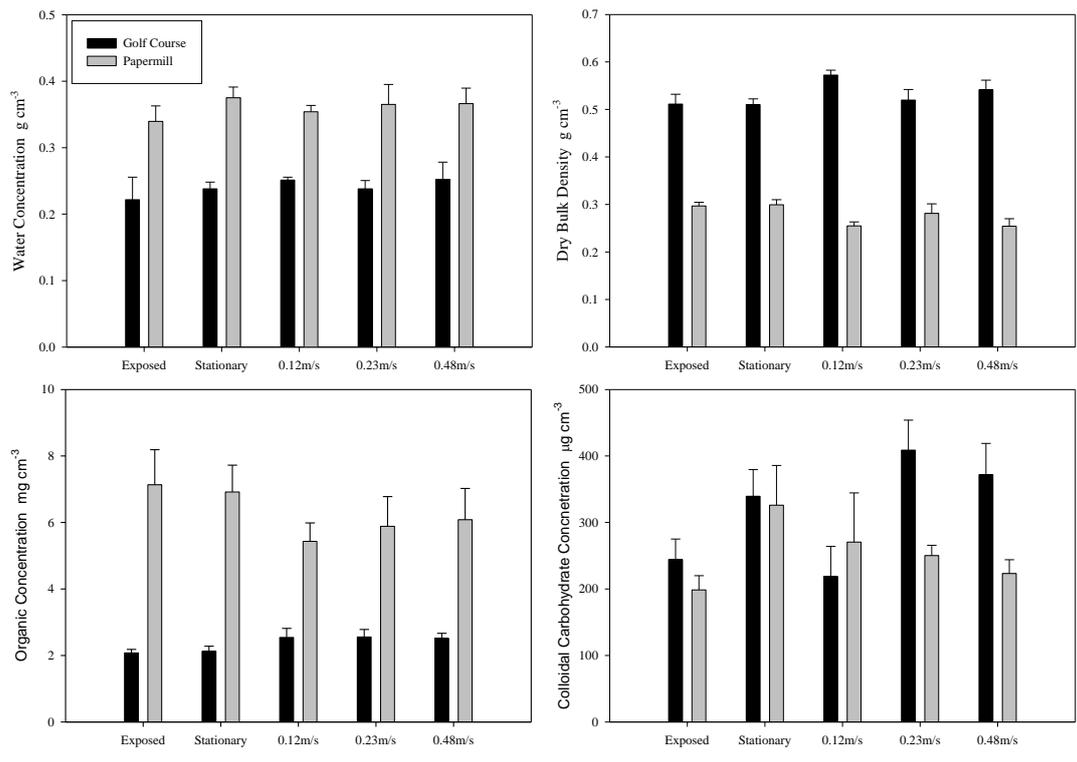


Figure 4.5. Sediment properties after initial submersion in water of different flow speeds.

4.3.1.2 Prolonged submersion of sediment cores

4.3.1.2.1 Submergence of sediment cores in stationary water

Sediment stability

The stability of the exposed cores from the Papermill and Golf Course sites were identical (104Nm^{-2}) although the standard error was higher in the Golf Course sediments (Fig. 4.6). After submersion, sediments from both sites had the same trend of unchanged stability up to three hours and then a drop by the fifth hour (Golf Course site, $F_{3,20} = 7.03$, $p = 0.002$; Papermill site, $F_{3,20} = 6.66$; $p = 0.003$).

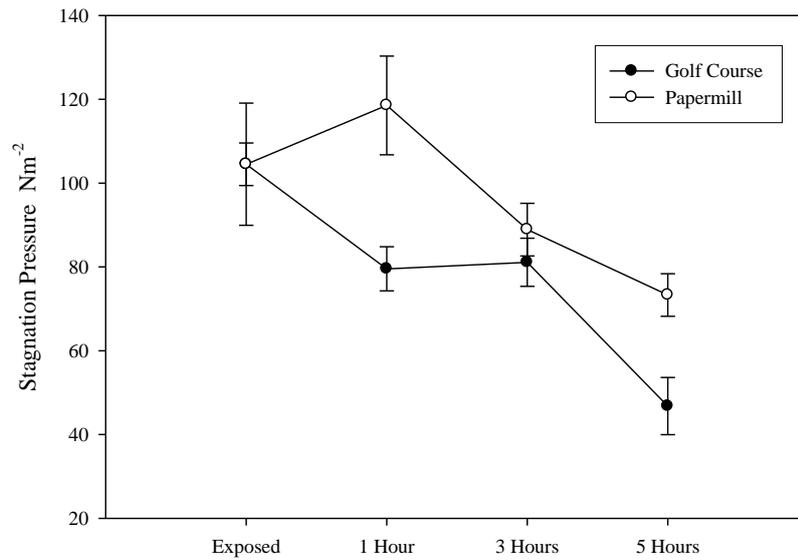


Figure 4.6. Stability of sediments from both sites submerged for different lengths of time in stationary water.

Sediment properties

The water concentration of sediments from both sites increased with longer submersion although decreased after 5 hours in Papermill sediments, while dry bulk density peaked after one and three hours for Papermill and Golf Course sediments respectively. Organic concentration and colloidal carbohydrates remained constant in sediment from both sites throughout submersion (Fig. 4.7; Table 4.2).

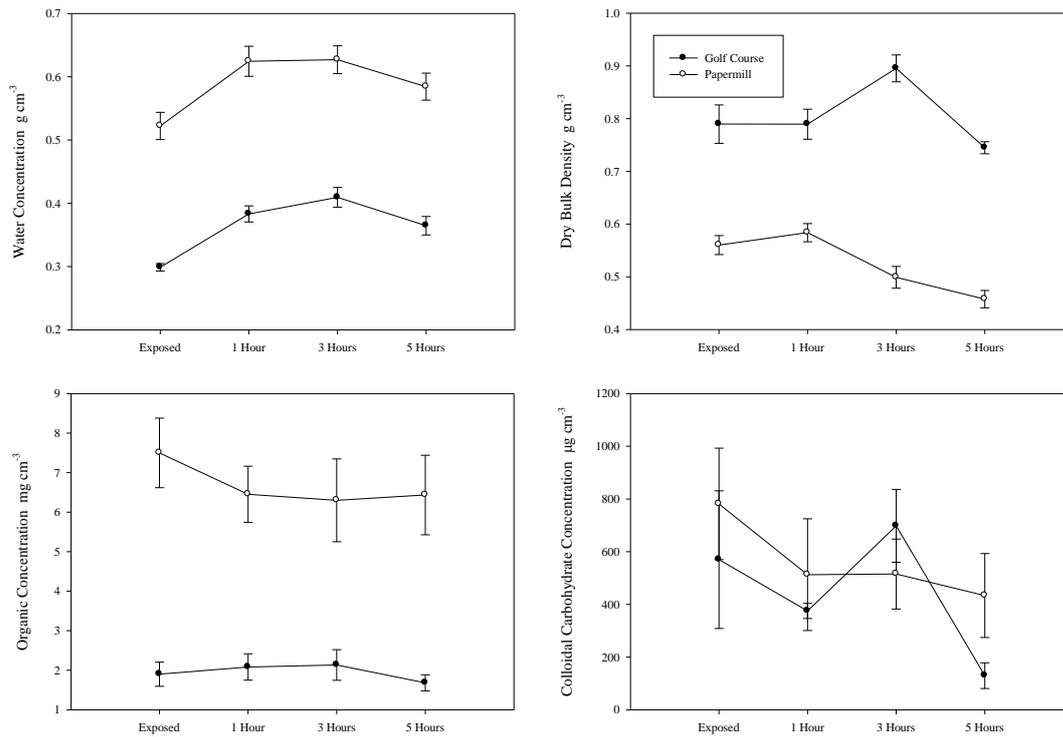


Figure 4.7. Sediment properties of sediment from both sites submerged for different lengths of time in stationary water.

Table 4.2. One-way ANOVA results of comparisons of sediment properties after prolonged submersion in stationary water in a laboratory based experiment.

	d.f.	F value	p value	Tukey Differences
Golf Course				
Water Concentration g cm ⁻³	3, 20	13.69	<0.001	Exposed < 1h, 3h, 5h
Dry Bulk Density g cm ⁻³	3, 20	0.51	0.006	5h < 3h
Organic Concentration mg cm ⁻³	3, 20	0.42	0.739	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 20	2.70	0.565	n/a
Papermill				
Water Concentration g cm ⁻³	3, 20	4.87	0.011	Exposed < 1h, 3h
Dry Bulk Density g cm ⁻³	3, 20	10.00	<0.001	1h < 5h; 3h < Exposed, 3h
Organic Concentration mg cm ⁻³	3, 20	0.37	0.778	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 20	0.70	0.565	n/a

4.3.1.2.2 Submergence of sediments cores in flowing water

Sediment stability

The stability of the Golf Course sediments dropped after one hour of submersion in flowing water to a level that subsequently remained constant with longer submersion ($F_{3, 20} = 50.19, p = 0.008$). Papermill sediments did not change in stability from that of the exposed sediment after any length of submersion ($F_{3, 20} = 1.01, p = 0.408$) (Fig. 4.8).

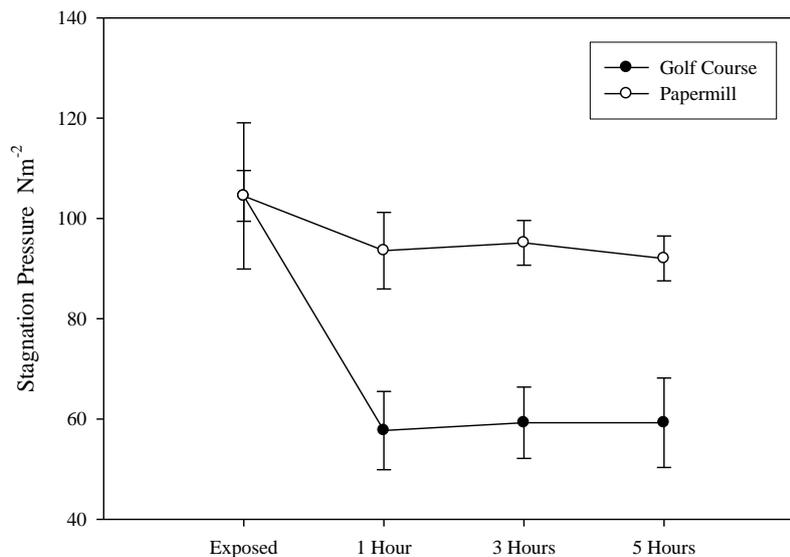


Figure 4.8. Stability of sediments from both sites submerged for different lengths of time in flowing water.

Sediment properties

As with sediments submerged in stationary water, the water concentration of sediments submerged in flowing water increased over time, but levels had dropped by the fifth hour in Papermill sediment and the increase in Golf Course sediments was slow. Dry bulk density again peaked after three hours in Golf course sediments but Papermill sediments were unaffected. No change in organic or colloidal carbohydrate concentration was evident (Fig. 4.9; Table 4.3).

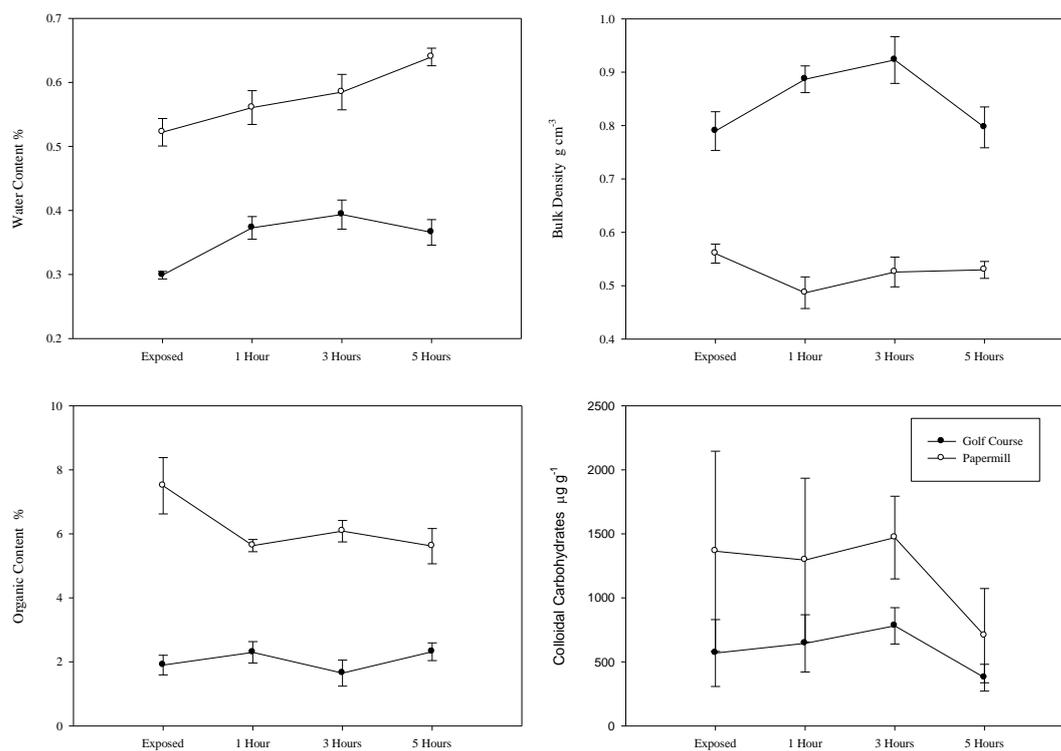


Figure 4.9. Sediment properties of sediment from both sites submerged for different lengths of time in flowing water.

Table 4.3. One-way ANOVA results of comparisons of sediment properties after prolonged submersion in flowing water in a laboratory based experiment.

	d.f.	F value	p value	Tukey Differences
Golf Course				
Water Concentration g cm ⁻³	3, 20	5.33	0.007	Exposed < 1h, 3h
Dry Bulk Density g cm ⁻³	3, 20	3.29	0.046	Exposed, 5h < 3h
Organic Concentration mg cm ⁻³	3, 20	0.94	0.439	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 20	0.77	0.527	n/a
Papermill				
Water Concentration g cm ⁻³	3, 20	4.59	0.013	Exposed < 5h
Dry Bulk Density g cm ⁻³	3, 20	1.62	0.216	n/a
Organic Concentration mg cm ⁻³	3, 20	2.58	0.082	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 20	1.71	0.198	n/a

4.3.1.3 Comparison of sediment submerged in stationary and flowing water

Sediment stability

On both sites there was an overall change in stability over time but flow had no consistent effect. However, on both sites an interaction occurred between time and flow, indicating that over the course of the experiments the stability of the sediment changed differently in flowing and stationary water (Table 4.4). This interaction was not consistent between sites, Golf Course sediment stability levels remained constant in stationary water but dropped after 3 hours in flowing water, whereas stability levels of Papermill sediment were unchanged by submersion in flowing water and dropped over the course of the submersion in stationary water (Fig. 4.10).

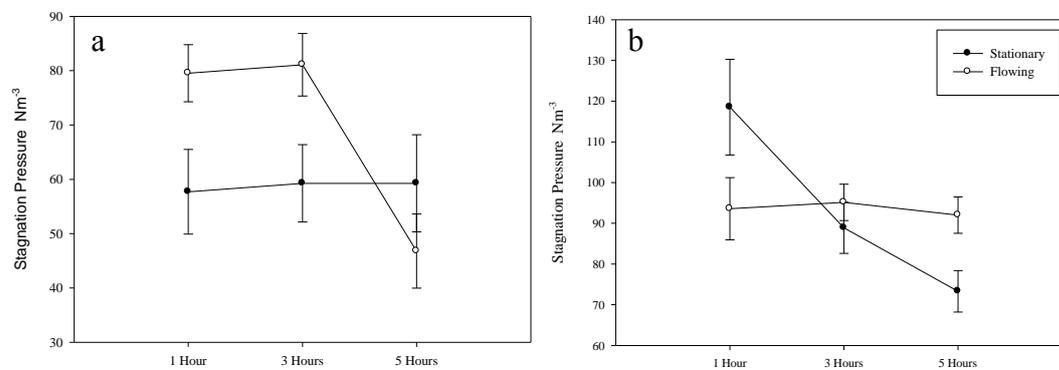


Figure 4.10. Comparison of the stability of sediment submerged in stationary and flowing water from the Golf Course (a) and Papermill (b) sites.

Sediment properties

A consistent change in dry bulk density occurred over time in Golf Course sediments submerged in flowing and stationary water. In Papermill sediments an increase in dry bulk density in flowing water was contrasted with a decrease under stationary conditions. The only other change occurred in colloidal carbohydrate concentration in the Golf Course sediments, this was consistent in both flowing and stationary water (Fig. 4.11 and 12; Table 4.4).

Table 4.4. Two-way ANOVA comparison of stability and properties of sediment submerged in stationary and flowing water in a laboratory based experiment

		Golf Course			Papermill	
		d.f.	F value	p value	F value	p value
Stagnation Pressure Nm^{-3}	Treatment	1, 50	3.26	0.081	0.00	1.000
	Time	1, 50	3.62	0.039	5.52	0.009
	Interaction	4, 50	3.94	0.030	5.03	0.013
Water Concentration g cm^{-3}	Treatment	1, 50	0.34	0.563	0.80	0.378
	Time	1, 50	2.22	0.126	0.38	0.690
	Interaction	4, 50	0.12	0.886	3.83	0.033
Dry Bulk Density g cm^{-3}	Treatment	1, 50	5.60	0.025	0.00	0.979
	Time	1, 50	10.26	0.000	1.79	0.185
	Interaction	4, 50	0.68	0.516	7.88	0.002
Organic Concentration mg cm^{-3}	Treatment	1, 50	0.21	0.654	1.12	0.299
	Time	1, 50	0.42	0.659	0.03	0.968
	Interaction	4, 50	1.46	0.249	0.12	0.890
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	Treatment	1, 50	3.50	0.071	0.11	0.742
	Time	1, 50	6.89	0.003	0.71	0.498
	Interaction	4, 50	0.30	0.744	0.50	0.614

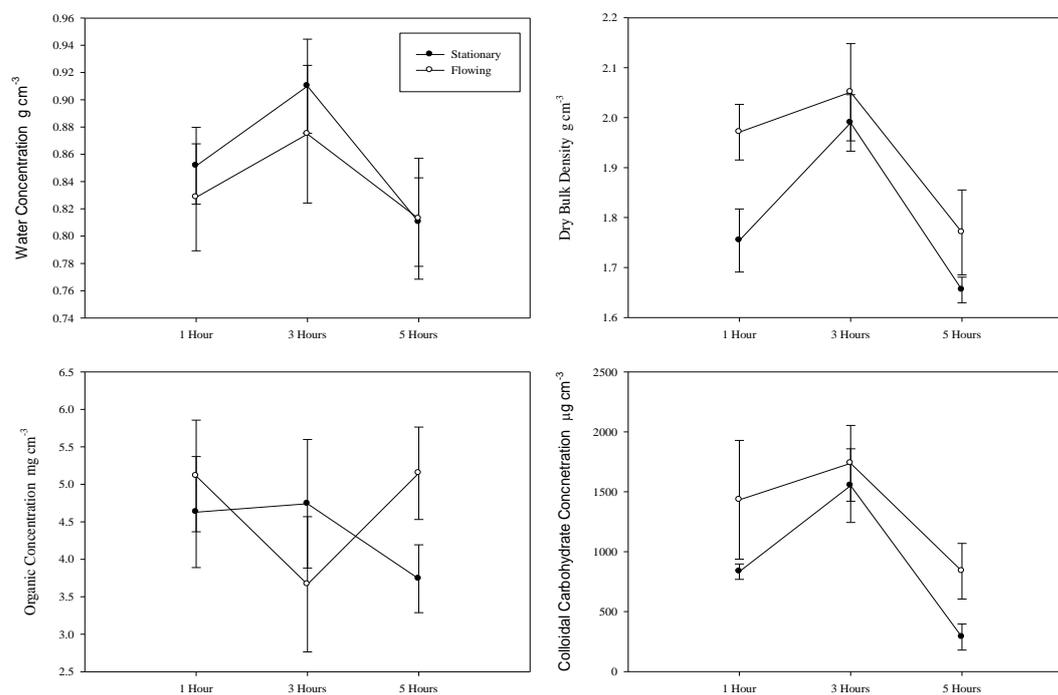


Figure 4.11. Comparison of properties of Golf Course sediments when submerged in stationary and flowing water.

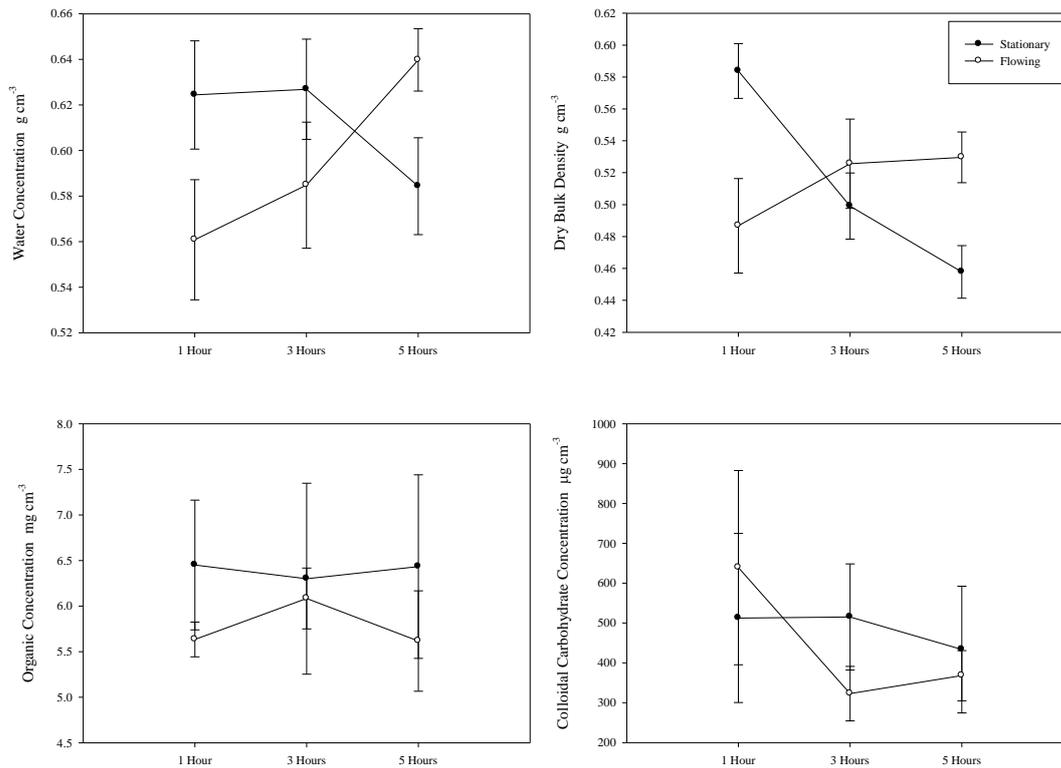


Figure 4.12. Comparison of properties of Papermill sediments when submerged in stationary and flowing water.

4.3.2 *In situ* experiments

4.3.2.1 Initial submersion of exposed sediment

Sediment stability

Exposed sediment from the Golf Course and Papermill sites both had stagnation pressures of about 25Nm^{-2} , both significantly lower than the corresponding values for cores of exposed sediment used in the laboratory-based initial submersion experiments (d.f. = 4, Golf Course site, $T = 6.57$, $p = 0.000$, Papermill site, d.f. = 4, $T = 4.72$, $p = 0.005$). Once submerged by the incoming tide, the stability of Golf Course sediments was higher than the exposed sediment and those submerged in stationary water ($F_{3, 16} = 4.65$, $p = 0.016$). Sediment that was subjected to flowing water did not increase in stability (Fig. 4.13). Sediment from the Papermill site displayed the same pattern of increasing stability with treatments but the increases were not significant ($F_{3, 16} = 0.90$, $p = 0.463$) (Fig. 4.13).

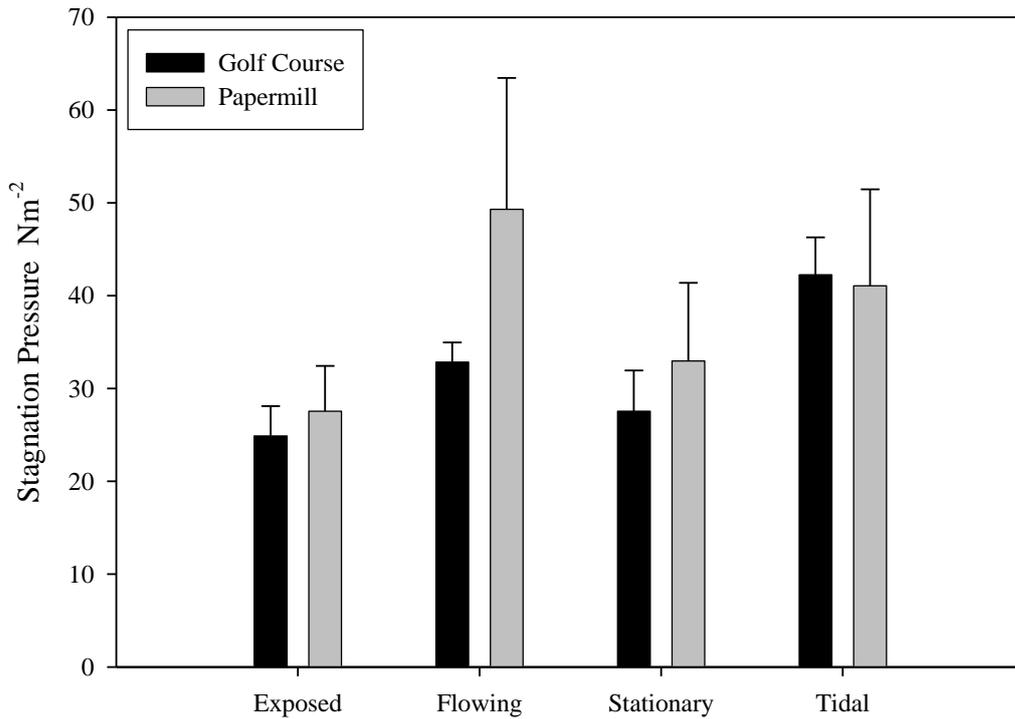


Figure 4.13. Stability of exposed sediment from the Golf Course and Papermill sites when subjected to different submersion treatments; *Exposed* – normal exposed sediment, *Flowing* – exposed sediment after water had been poured over the surface, *Stationary* – exposed sediment submerged in stationary water for 2-3 minutes and *Tidal* – sediment tested after 2-3 minutes of submersion by the on coming tide.

Sediment properties

The sediment properties from the two sites were completely unchanged by any of the treatments (Figs. 4.14; Table 4.5).

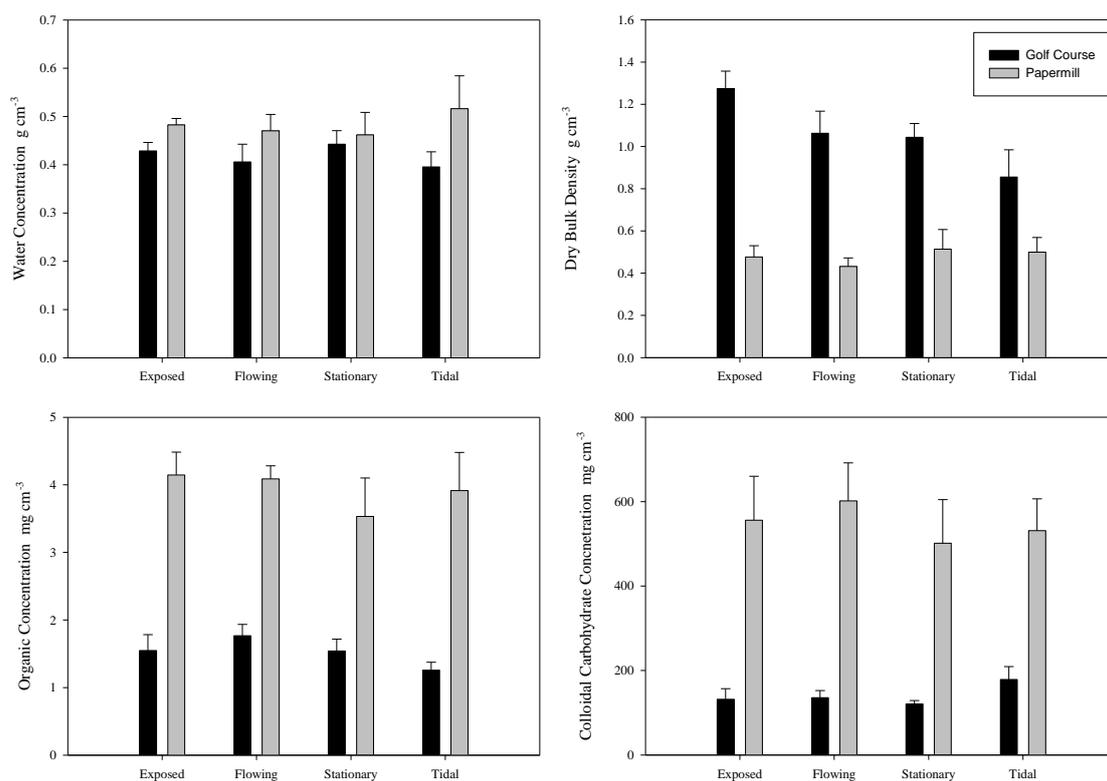


Figure 4.14. Sediment properties of exposed sediments subjected to different submersion treatments (labels detailed in Figure 4.13).

Table 4.5. One-way ANOVA results of comparisons of sediment properties after initial submersion *in situ*.

	d.f.	F value	p value	Tukey Differences
Golf Course				
Water Concentration g cm ⁻³	3, 16	0.53	0.666	n/a
Dry Bulk Density g cm ⁻³	3, 16	3.01	0.061	n/a
Organic Concentration mg cm ⁻³	3, 16	1.36	0.291	n/a
Colloidal Carbohydrate Concentration μg cm ⁻³	3, 16	0.60	0.625	n/a
Papermill				
Water Concentration g cm ⁻³	3, 16	0.28	0.841	n/a
Dry Bulk Density g cm ⁻³	3, 16	0.28	0.836	n/a
Organic Concentration mg cm ⁻³	3, 16	0.39	0.763	n/a
Colloidal Carbohydrate Concentration μg cm ⁻³	3, 16	0.21	0.887	n/a

4.3.2.2 Prolonged submersion of sediment

Sediment stability

Initial stability of exposed sediment from both sites was similar to that from the *in situ* initial submersion experiments. However, again, this was significantly lower than the values obtained from the exposed cores from the respective sites used in the laboratory experiments (Golf Course site, d.f. = 4, $T = 4.25$, $p = 0.004$, Papermill site, d.f. = 4, $T = 10.35$, $p = 0.000$). On both sites there was no change in stability between the two readings taken during exposure. Stability on both sites increased with submersion, before remaining constant after further submersion (Golf Course site, $F_{3, 16} = 21.76$, $p = 0.000$, Papermill site, $F_{3, 16} = 39.79$, $p = 0.000$) (Fig. 4.15).

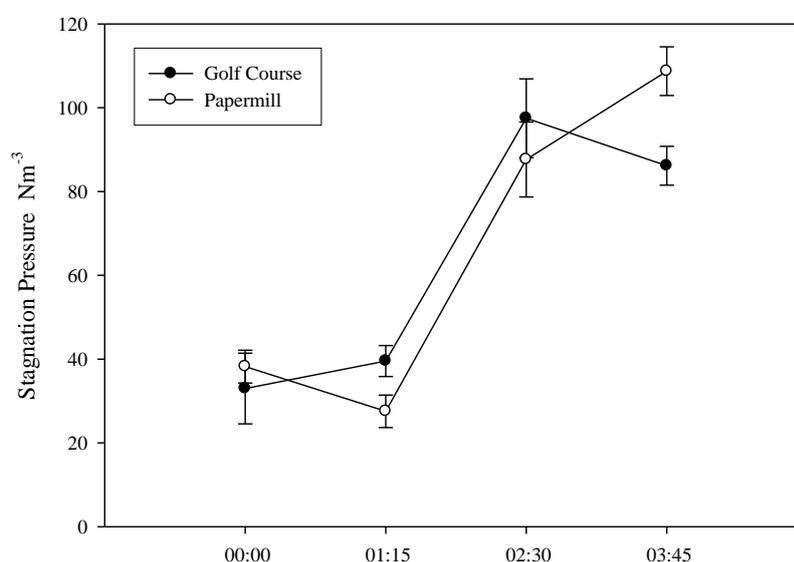


Figure 4.15. Stability of sediment from both sites over the period of submersion by the incoming tide. Sediment was covered by the incoming tide about 1:45 hours after the initial reading.

Sediment properties

Submerged sediment from the Papermill increased in water concentration after prolonged submersion while the dry bulk density only increased in Golf Course sediments between the two submerged sample periods. Colloidal carbohydrate and organic concentrations remained unchanged from either site with submersion (Fig. 4.16; Table 4.6).

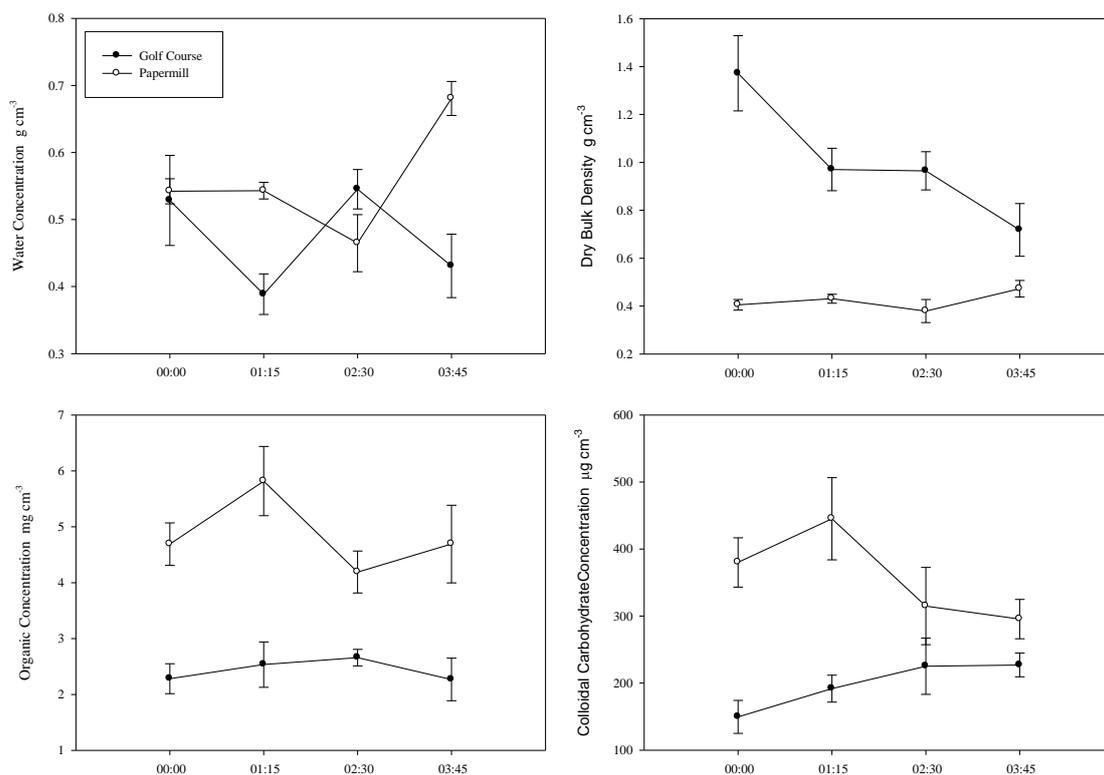


Figure 4.16. Sediment properties of sediment from both sites over a period of submersion by the incoming tide. Sediment was covered by the incoming tide about 1:45 hours after the initial reading.

Table 4.6. One-way ANOVA results of comparisons of sediment properties after prolonged submersion *in situ*

	d.f.	F value	p value	Tukey Differences
Golf Course				
Water Concentration g cm ⁻³	3, 16	2.69	0.081	n/a
Dry Bulk Density g cm ⁻³	3, 16	5.74	0.007	00:00 > 03:45
Organic Concentration mg cm ⁻³	3, 16	0.37	0.779	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 16	0.44	0.729	n/a
Papermill				
Water Concentration g cm ⁻³	3, 16	10.90	<0.001	00:00, 01:15, 02:30 < 03:45
Dry Bulk Density g cm ⁻³	3, 16	1.47	0.260	n/a
Organic Concentration mg cm ⁻³	3, 16	1.65	0.217	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	3, 16	2.49	0.097	n/a

4.3.3 Sediment properties effecting sediment stability

No change in sediment properties correlated with a change in sediment stability within each experiment (Table 4.7). When data from experiments was combined within sites, and then both sites combined there were significant correlations between sediment stability and sediment properties (Table 4.8). Most consistent among these, and also accounting for the largest r_s value was that of colloidal carbohydrate concentration which negatively correlated with sediment stability.

Table 4.7. Results of correlation analysis of sediment stability with sediment properties combining measurements from exposed and submerged sediments.

	Golf Course		Papermill	
	Spearman Rank correlation r_s	p value	Spearman Rank correlation r_s	p value
Laboratory based experiments				
Initial submersion				
Water Concentration g cm^{-3}	0.060	0.753	0.138	0.482
Dry Bulk Density g cm^{-3}	-0.174	0.359	0.201	0.306
Organic Concentration mg cm^{-3}	0.173	0.359	0.316	0.101
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	0.307	0.099	0.204	0.299
Prolonged submersion				
Water Concentration g cm^{-3}	0.045	0.795	0.166	0.332
Dry Bulk Density g cm^{-3}	0.088	0.610	-0.207	0.225
Organic Concentration mg cm^{-3}	0.064	0.710	0.112	0.517
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	-0.035	0.839	0.009	0.959
<i>In situ</i> experiments				
Initial submersion				
Water Concentration g cm^{-3}	-0.144	0.545	-0.292	0.212
Dry Bulk Density g cm^{-3}	-0.237	0.313	-0.087	0.716
Organic Concentration mg cm^{-3}	-0.087	0.716	-0.181	0.446
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	-0.018	0.947	-0.227	0.335
Prolonged submersion				
Water Concentration g cm^{-3}	0.407	0.075	0.257	0.287
Dry Bulk Density g cm^{-3}	-0.271	0.247	-0.073	0.767
Organic Concentration mg cm^{-3}	0.387	0.092	-0.312	0.194
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	0.197	0.405	-0.422	0.072

Table 4.8. Results of correlation analysis of sediment stability with sediment properties from all experiments at each site individually and then all results from both sites combined.

	Spearman Rank correlation r_s	p value
Golf Course		
Water Concentration g cm^{-3}	-0.359	<0.001
Dry Bulk Density g cm^{-3}	-0.473	<0.001
Organic Concentration mg cm^{-3}	0.149	0.121
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	-0.352	<0.001
Papermill		
Water Concentration g cm^{-3}	-0.076	0.434
Dry Bulk Density g cm^{-3}	-0.138	0.154
Organic Concentration mg cm^{-3}	0.291	0.002
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	-0.640	<0.001
Both sites combined		
Water Concentration g cm^{-3}	-0.131	0.053
Dry Bulk Density g cm^{-3}	-0.377	<0.001
Organic Concentration mg cm^{-3}	0.343	<0.001
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	-0.495	<0.001

4.4 Supplementary experiments

4.4.1 The effect of submergence on the stabilising influence of EPS

4.4.1.1 Methods

Artificial EPS was added in the laboratory to sediment cores taken from both sample sites. Cores were taken using the method detailed in section 4.2.3.1 but 5ml of 2.5g l^{-1} Xantham gum was carefully pipetted onto the sediment surface and left overnight to become incorporated into the sediment surface. Pipetting small volumes of Xantham gum solution onto the sediment surface rather than mixing in the solution to homogenised sediment (as performed by Vardy *et al.*, 2007) allowed the natural sediment surface to remain relatively intact. The sediment cores were then submerged in stationary and flowing water for prolonged periods following the procedure detailed in section 4.2.4.

4.4.1.2 Results

Sediment stability

Exposed sediment from both sites with added EPS had stability levels much higher than those encountered in any other experiments and higher than the control sediment, indicating stabilisation by the added EPS. Once submerged in flowing water, stability dropped to levels comparable to the control sediment after only 1 and 3 hours in Golf Course and Papermill sediments respectively (Fig. 4.17a; Table 4.9). Equally Golf Course sediments dropped in stability in stationary water while Papermill sediments only dropped to similar levels after 5 hours (Fig. 4.17b; Table 4.9).

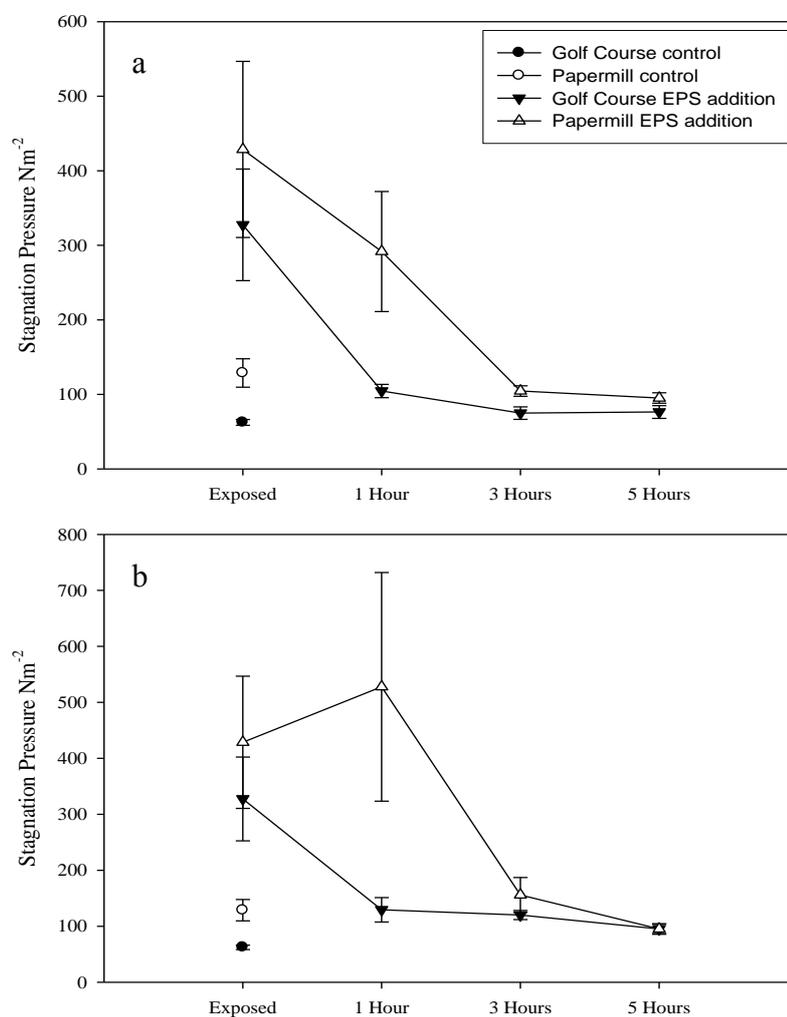


Figure 4.17. Sediment stability from the Golf Course and Papermill sites with added artificial EPS solution after submersion for prolonged periods in flowing (a) and stationary (b) water. Control sediments did not have EPS added.

Table 4.9. Comparisons of sediment stability and properties with the addition of artificial EPS solution after prolonged submersion in flowing water conditions. d.f. for all tests = 4, 25.

	F value	p value	Tukey Differences
Flowing water			
Golf Course			
Stagnation Pressure Nm^{-2}	10.73	<0.001	Control, 1h, 3h, 5h < Exposed
Water Concentration g cm^{-3}	0.49	0.744	n/a
Dry Bulk Density g cm^{-3}	21.33	<0.001	Control, 1h, 3h, 5h < Exposed Control, 5h < 1h
Organic Concentration mg cm^{-3}	2.44	<0.001	1h < 5h
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	1.57	0.213	n/a
Papermill			
Stagnation Pressure Nm^{-2}	4.61	0.007	Control, 3h, 5h < Exposed
Water Concentration g cm^{-3}	0.68	0.610	n/a
Dry Bulk Density g cm^{-3}	16.67	<0.001	1h, 3h, 5h < Exposed, Control
Organic Concentration mg cm^{-3}	2.24	0.098	n/a
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	0.52	0.724	n/a
Stationary water			
Golf Course			
Stagnation Pressure Nm^{-2}	8.69	<0.001	Control, 1h, 3h, 5h < Exposed
Water Concentration g cm^{-3}	1.48	0.237	n/a
Dry Bulk Density g cm^{-3}	13.63	<0.001	Control, 3h 5h < Exposed Control, 5h < 1h
Organic Concentration mg cm^{-3}	0.83	0.520	n/a
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	1.02	0.416	n/a
Papermill			
Stagnation Pressure Nm^{-2}	3.05	0.040	5h < Exposed 3h, 5h, Control < 1h
Water Concentration g cm^{-3}	1.37	0.277	n/a
Dry Bulk Density g cm^{-3}	3.57	0.023	5h < Exposed
Organic Concentration mg cm^{-3}	1.46	0.250	n/a
Colloidal Carbohydrate Concentration $\mu\text{g cm}^{-3}$	1.01	0.425	n/a

Sediment properties

Unexpectedly, the changes in stability were not reflected in changes in organic or carbohydrate concentrations in sediments from either site. Likewise there was no change in water concentration, while there was a general trend of decreasing dry bulk density with submersion time although the magnitude of this change varies between sediments and submersion methods (Figs. 4.18 and 4.19; Table 4.9).

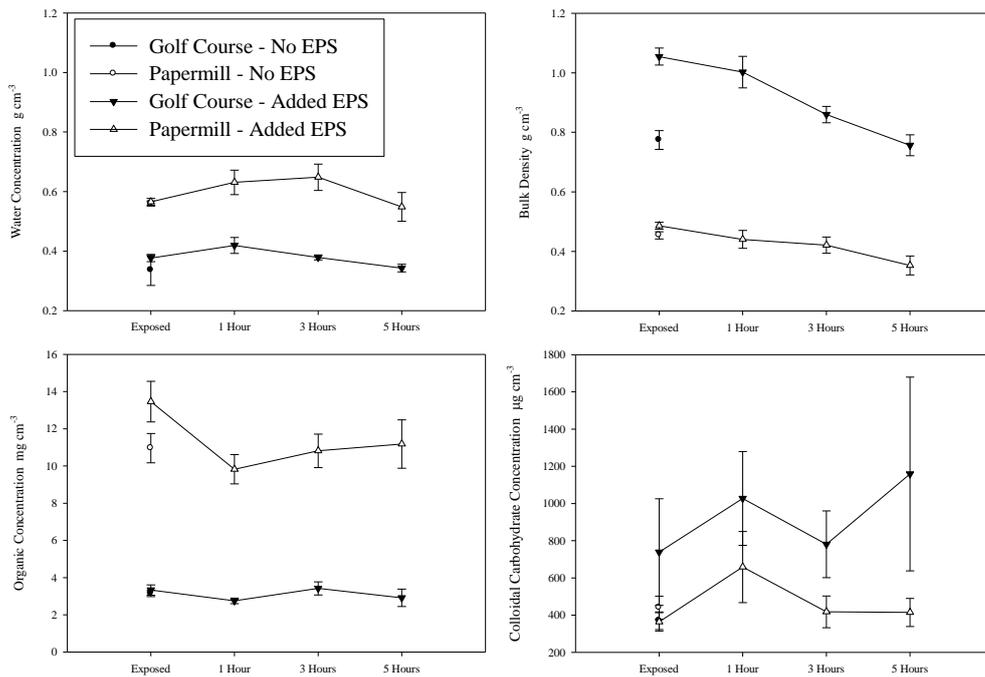


Figure 4.18. Sediment properties from sediment cores taken from both sites with added artificial EPS and subjected to prolonged submersion in flowing water.

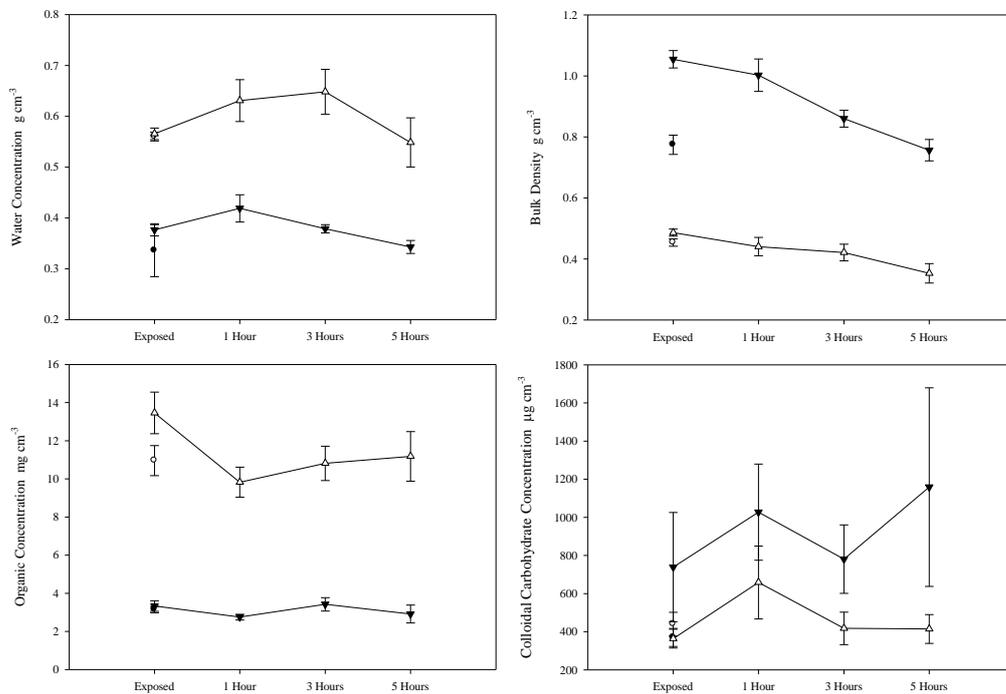


Figure 4.19. Sediment properties from sediment cores taken from both sites with added artificial EPS and subjected to prolonged submersion in stationary water.

4.4.2 Submerged sediment stability along a river transect

4.4.2.1 Methods

Three sites along the River Eden were sampled on the 28th July, 2006. These consisted of a river site, roughly 2 miles from the estuary (56°20'01N; 2°06'46W) and two sites within the estuary, one near the mouth (56°12'45N; 2°49'39W) and the other nearer the head (56°21'52N; 2°53'23W). River samples were taken from an area of sediment submerged in shallow water, with the CSM chamber deployed directly onto the sediment. Contact cores were taken from the sediment after the overlying water had been drained from a core pushed slightly into the sediment. Both estuary sites were sampled while submerged (after several hours of submergence). At the Estuary head sediment cores were taken in standing depth of water, while at the deeper Estuary mouth site cores were collected using snorkelling. CSM measurements and contact cores (n=6) were taken using the protocol detailed in section 4.2.3.1.

4.4.2.2 Results

Stability increased along the river transect from the river to the estuary mouth (Fig. 4.20; Table 4.10) as did water concentration and dry bulk density. Organic concentration remained constant and colloidal carbohydrates were higher in the estuary mouth in comparison to the other two sites (Fig. 4.21; Table 4.10).

Table 4.10. Comparisons of sediment stability and properties from three sites along a river transect. River site (R), Estuary head (EH) and Estuary mouth (EM).

	d.f.	F value	p value	Tukey Differences
Stagnation Pressure Nm ⁻²	2, 15	24.64	<0.001	R < EH < EM
Water Concentration g cm ⁻³	2, 15	15.25	<0.001	R < EH < EM
Dry Bulk Density g cm ⁻³	2, 15	14.31	<0.001	R < EH < EM
Organic Concentration mg cm ⁻³	2, 15	0.29	0.750	n/a
Colloidal Carbohydrate Concentration µg cm ⁻³	2, 15	5.67	0.013	R, EH < EM

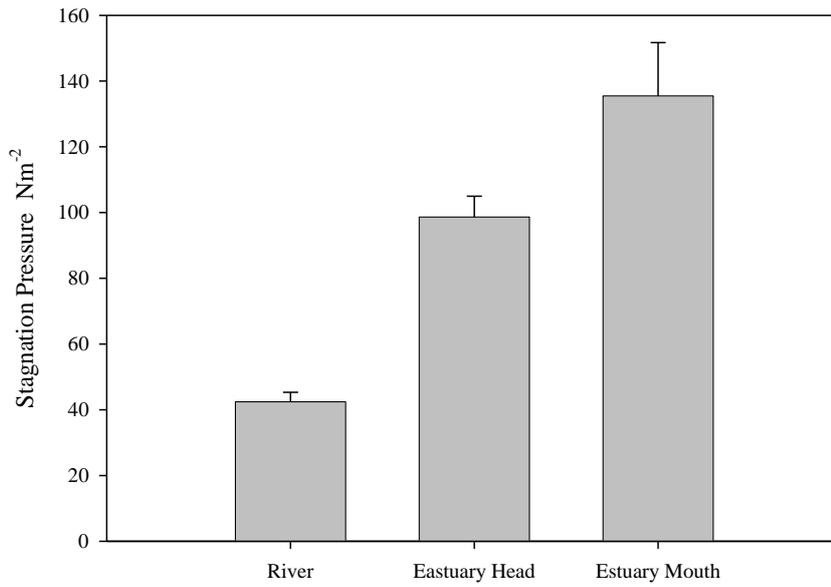


Figure 4.20. Sediment stability from a transect along the Eden river and estuary.

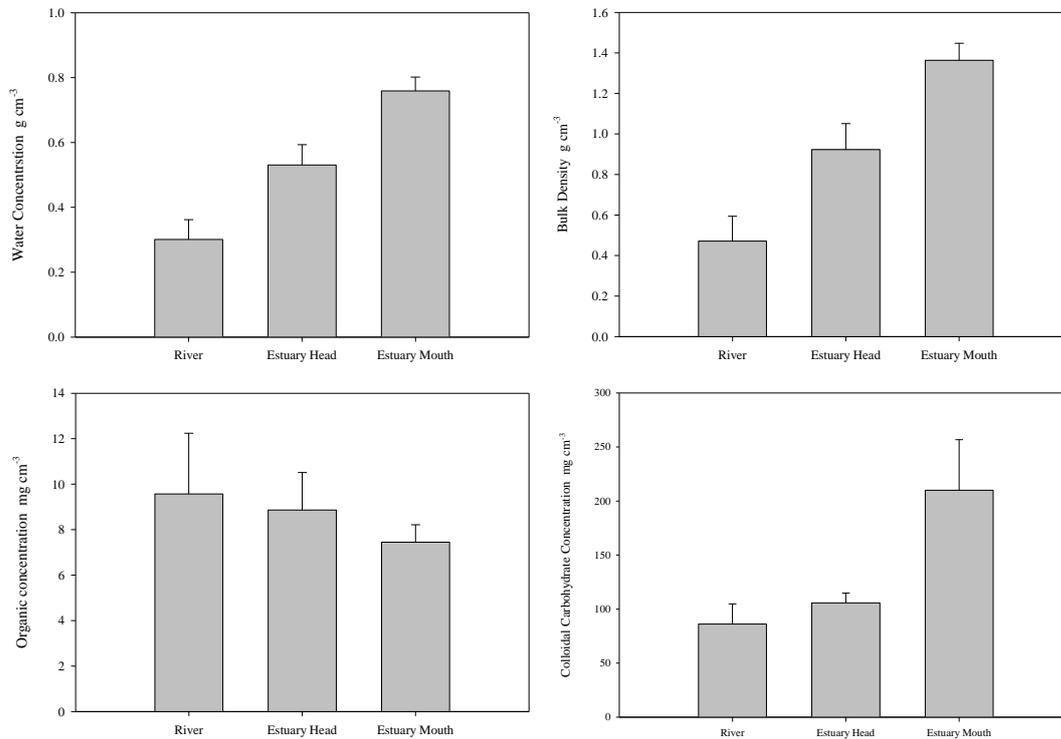


Figure 4.21. Sediment properties from a transect along the Eden river and estuary.

4.5 Discussion

The stability of exposed intertidal sediments is dictated by many factors, many of which interact with, and are subject to, the exposed environment. At the moment of submersion by the incoming tide the environment of the sediment is changed dramatically, with atmospheric conditions replaced by a hydrodynamic regime. Devising experiments to study submerged intertidal sediments and assessing the continuity of sediment properties between these two states is vital for further understanding sediment erosion.

4.5.1 Comparing exposed and submerged sediments

4.5.1.1 Sediment stability

In intertidal systems the initial submersion is by a small tidally driven “wave” of flowing water. From the data presented, this “wave” can cause an increase in sediment stability, with a faster incoming tide, with more energy, more likely to result in an increase in stability. In laboratory-based experiments, submersion in stationary water and the slowest flow speed did not change stability in Golf Course sediments, while faster flows increased stability. Papermill sediments were not affected by submersion. *In situ* results show an increase in stability in Golf Course sediments with submersion and while not significant the pattern was repeated in Papermill sediments. Again submersion in stationary water did not cause a change in stability.

After longer periods of submersion, sediment stability from the laboratory and *in situ* experiments appeared to contradict each other, with *in situ* sediments increasing in stability with submersion, continuing the trend from the initial submersion, but laboratory-based sediments remaining stable or reducing in stability.

The presence of a layer of unconsolidated sediment upon the intertidal exposed sediment surface, similar to the fluff layer found on submerged sediment (Sutherland *et al.*, 1998; Wang *et al.*, 2003; Kuhrts *et al.*, 2006; Schaaff *et al.*, 2006), may be the explanation for the change in stability with submersion and also the apparently contradictory results between laboratory and *in situ* experiments. Such a layer is commonly found in submerged sediments, but its presence has only been suggested in exposed intertidal sediments (Tolhurst *et al.*, in progress). In submerged sediment the

fluff layer has a significantly lower critical erosion threshold than the deeper sediments and can be easily eroded, exposing the more stable sediment beneath. If such a layer were present on exposed sediment then it may possibly be eroded quickly during the initial act of submersion, exposing the deeper more consolidated sediment.

This could explain the increase in stability with submersion found *in situ* as the lower critical erosion threshold on exposed sediment may be attributed to the erosion of this fluff layer. The increased stability once submerged may be because the fluff layer has been removed by the tide and it is the deeper, more consolidated, sediment that is being eroded. The difference in results from exposed and submerged sediments would therefore not be a result of a change in the properties of the sediment, but rather a change in the sediment that is being tested.

The presence of a fluff like layer upon the surface of exposed intertidal sediments could be generated by similar mechanisms to that found in submerged sediment. Deposition of sediment could occur with the receding tide, and bioturbation of sediment will be most evident on the sediment surface resulting in a layer of unconsolidated sediment and organic debris (Orvain *et al.*, 2003). Such a fluff layer may consist of well spaced unconsolidated sediment particles with large inter-particle spaces filled with water or air which may allow it to be eroded at low energy levels.

The contradictory results from the laboratory and *in situ* can be used to support the theory of a fluff layer. The critical erosion thresholds of the exposed sediments from the laboratory and *in situ* differ hugely, with sediments from both sites having critical erosion thresholds of about 104Nm^{-2} and 25Nm^{-2} respectively. The higher stability of the exposed laboratory sediment is possibly because the fluff layer has become consolidated onto the underlying sediment during the time from collection of cores to testing. Due to the experimental design this time gap could have been up to 10 hours for some cores, potentially allowing compaction of the sediment particles or loss of water from the sediment surface through evaporation or drainage into the deeper sediment. This is supported by the similarity in stability of the exposed laboratory cores and the submerged *in situ* sediment from which the fluff layer has been removed.

That such a layer has not been evident in previous studies on intertidal sediments is possibly due to the methods used in detecting erosion. The CSM is capable of very high resolution at low erosion thresholds which makes it ideal for detecting the erosion of the unstable fluff layer. The CSM is also a small and relatively delicate machine to use upon the sediment surface, whereas the use of larger flumes or transportation of sediment cores may result in a disruption to the fluff layer, either allowing it to become more consolidated or inadvertently removing it before it can be tested. When coring submerged sediment it is important to maintain the overlaying water column to prevent disruption to the sediment surface before testing (Hauton & Paterson, 2003; Schaaff *et al.*, 2006; Spears *et al.*, 2007) and it is possible that maintaining the environment in which exposed sediment is sampled is equally vital.

After submersion for the longer time periods that the laboratory experiments allowed, stability of the sediment from the two sites differed, with Papermill sediments remaining unchanged and Golf Course sediments dropping in stability after an hour then remaining constant. However, the validity of these results has to be questioned if a fluff layer has become consolidated into the sediment then the properties of the sediment surface are probably different to that of sediment from which the unconsolidated layer has been removed.

4.5.1.2 Sediment properties

Tolhurst *et al.*, (2005) argued that a given volume of intertidal sediment is made up of six components; non-cohesive mineral grains, cohesive mineral grains, water, gas, biota and other matter. Ignoring the last two components (which is probably incorrect but useful for simplicity) a sediment core can be divided into solid (sediment particles), liquid (water) and gas (air), the contribution of each component to the overall volume of the core can be given as a concentration. When comparing exposed and submerged sediment it is possible that the meanings of these values will change. The dry bulk density (sediment concentration) will also give some indication of the volume of the inter-particle spaces. In exposed sediments this space will be filled with water and air, of which only water can be measured (expressed as water concentration) and is therefore a measure of how much of the inter-particle space is filled with water. However, once submerged it is probable that air is removed from the sediment, and therefore water concentration becomes a direct measurement of the

volume of the inter-particle space. Therefore, while dry bulk density measures the density of the sediment particles in both exposed and submerged sediment, the interpretation of the water concentration measurement is potentially different.

Over all, sediment water concentration tended to increase with submersion although this was not found on longer *in situ* submersion. However, dry bulk density did not consistently change with water concentration, implying that water was replacing air in the inter-particle spaces without causing a change in the density of the sediment particles.

There was no consistent change in organic and colloidal carbohydrate concentrations with submersion. This is not surprising for organic concentration since it is a measure of all organic matter within the sediment, most of which would be expected to remain within the sediment. However, it was expected that the concentration of colloidal carbohydrates would drop with submersion due to their solubility. This did not occur possibly because the colloidal carbohydrates are bound within a mixture of non-soluble components and as such are prevented from dissolving into the water column. However, this does contradict the findings of Blanchard *et al.*, (2000) where carbohydrate levels dropped with increased water levels

If changes in sediment stability are in part due to an unconsolidated fluff layer on exposed sediment it would be expected that dry bulk density would increase and water concentration decrease with its removal. As with measurements of stability this would not be a change in the sediment properties but rather a change in the actual sediment that is being tested. Organic and colloidal carbohydrate levels may also drop if they are present in high proportions within the fluff layer. That this was not found may be a result of the sampling methodology being inadequate to detect the removal of the fluff layer. Sutherland *et al.* (1998) used X-ray tomography in submerged sediment to quantify the bulk density of the fluff layer. They found a low bulk density on the sediment surface which increased with depth until a constant value was obtained in the consolidated sediment. At most the low bulk density extended to a depth of 1.5mm. Both the contact core and course core methods used in this work sampled the upper 2mm of sediment as a whole and to some extent required the instrument to be pushed into the sediment surface slightly. It is quite possible that

this destroyed the fine scale gradient in water concentration and dry bulk density that a fluff layer would display.

4.5.1.3 Sediment properties relating to sediment stability

The changes in stability that occurred between exposed and submerged sediments within each experiment were not found to relate to a change in sediment properties. On a larger scale, decreases in both dry bulk density and colloidal carbohydrates, and increase in organic concentration all correlated with increasing stability. The changes in dry bulk density and colloidal carbohydrates would appear to contradict expectations, as usually increases in the density of the particles and colloidal carbohydrates (as a proxy for EPS) cause increases in the stability of sediment surfaces.

4.5.1.4 Extrapolating from exposed to submerged sediments

There are many changes in sediment stability and related properties between exposed and submerged sediments, and these appear to vary depending upon the nature of the submersion. As erosion will occur when submerged this has important implications for using measurements taken on exposed sediments as an indication of the erosional properties of the sediment. However, this work did not present many definitive answers to these problems, indeed, as with work on exposed sediments it is quite possible that the influences on sediment stability once submerged are as site specific as they are on exposed sediments (Defew *et al.*, 2003; Friend *et al.*, 2003a).

4.5.2 Methodologies

The methods used in sampling exposed and submerged sediment are very specific and transferring a method from the environment in which it was devised into a different one presented problems. Additionally replicating the conditions of flow experienced *in situ* was vital.

4.5.2.1 Measuring sediment stability

The use of a CSM to measure sediment stability in exposed and submerged sediment allowed the stability of submerged sediment to be tested in a new way, bringing many of the advantages the CSM has on exposed sediment into the submerged environment. Rapid replication and high resolution of erosion thresholds allowed high accuracy in

timing and measurements. Additionally, being able to use the CSM in the field and the flume allowed sediment to be subjected to normal sub-critical erosion conditions prior to being tested. Of practical consideration was the addition of the plate to the erosion chamber (section 3.2.3.2), when deploying the chamber directly onto a submerged sediment surface it is very difficult to view the base of the chamber in relation to the sediment surface. Once on the surface the weight of the chamber was rarely supported by the sediment, meaning it was often pushed too deeply into the sediment, potentially reducing the distance from nozzle tip to sediment surface. The increased surface area of the plate allowed the chamber to be supported by the sediment surface and therefore be lowered accurately onto the surface without it accidentally being pushed too far into the sediment. The use of the CSM to measure stability removed the necessity to subject a sample to increasing flow speeds prior to erosion. The CSM was restricted to working in very shallow depths and the use of banded cores to bring undisturbed sediment to the surface was sufficient for the needs of this experiment but would prove inefficient and also depth limiting ($\approx 5\text{m}$ snorkelling and $\approx 30\text{m}$ SCUBA, personal observation) for extensive sampling. This may need to be assessed if more depth was required or the conditions were not suitable for snorkelling or SCUBA. The use of a Jenkins corer was attempted but the highly consolidated nature of intertidal sediment prevented it collecting an undisturbed core. The use of the CSM in the EPS experiment and river transect was highly advantageous. Deploying the erosion chamber in the flume (EPS experiments), directly on to the sediment surface (river samples) and then within sediment cores (estuary sample) allowed direct comparisons of results from three different submerged environments.

4.5.2.2 Sampling exposed and submerged sediment

Two methods of sediment sampling were used, the course core and contact core, both devised for use on exposed sediment and adapted for submerged sampling. Both methods sampled the upper 2mm as a homogenous layer, however with potential for delicate and micro scale changes in sediment properties (Sutherland *et al.*, 1998) both were deemed insufficient for highly accurate measurements of dry bulk density and water concentration. Of the two the contact core required access to the exposed sediment surface so submerged sediment had to be drained of its overlaying water. This probably disturbed the sediment surface beyond an acceptable degree. The

course corer was used in the same fashion on exposed and submerged sediment with the inner core placed onto the sediment surface, again this is probably too destructive to measure very small changes in dry bulk density and water concentration but if it could be modified to sample the overlaying water it may suffice. However, this would not be possible for exposed sediment. Despite this if the nature of the experiment required less accurate measurements such as the organic and colloidal carbohydrate concentration readings taken in the two supplementary experiments then either method of sampling cores is probably sufficient.

4.5.2.3 Replicating submersion in the laboratory

The potential for gradients in dry bulk density and water concentration to occur within the upper millimetres of the sediment (Sutherland *et al.* 1998) makes replicating submersion conditions in the laboratory very important. While simulation of the conditions and hydrodynamics may be possible, great attention needs to be paid to the state of the cores. The surface layer of sediment needs to remain consistent from the estuary to the test chamber. Maintaining this state is highly important as the erosion threshold of the sediment increases dramatically once this sediment layer becomes more consolidated. This must be checked with comparisons of stability between laboratory and the field. In the one experiment in the laboratory where stability was equal to that in the field (initial submersion of Golf Course sediment) the results of submersion matched those found in the estuary. Of equal importance appears to be the need to replicate the hydrodynamic conditions of initial and continued submersion. When stationary water was used instead of flowing water in the laboratory or *in situ* the results differed from those submerged in flowing water. In both the laboratory and field, submersion in stationary water for 2-3 minutes did not replicate the conditions that caused an increase in stability found with flowing water. Equally, prolonged submersion in stationary water caused a continuing decline in sediment stability which was not found in flowing water where stability remained constant after submersion. It appears that without the mechanisms and energy of flowing water the fluff layer is not removed in the initial submersion, causing the stability to remain equal to the exposed sediment. With prolonged submersion in stationary water stability continually decreases, possibly as the sediment becomes more unconsolidated, in flowing water these unconsolidated sediment particles might

be removed but in stationary water they remain on the sediment surface lowering the overall critical erosion threshold.

4.5.3 Supplementary experiments

The effect of submergence on the stabilising influence of EPS

The addition of EPS dramatically increased the stability of exposed sediment, and as expected this stability dropped as the sediments were submerged. To what extent this was caused by the EPS dissolving from the sediment and into the water is unclear as surprisingly measurements of organic and colloidal carbohydrate concentration both remained unchanged throughout the experiment. However, the large increase in initial stability, coupled with the drop while submerged, indicates that this is an area of study that is worth further investigation.

Submerged sediment stability along a river transect

A large change in sediment stability was found between sediments in different parts of the River Eden. Although water concentration and dry bulk density both changed in relation to the stability it is considered unlikely that these are the only variables that will affect sediment properties along such a transect. Grain size remained constant throughout the sample sites (data not shown) while salinity significantly increased between each site towards the sea (data not shown). The implications for this are highly significant with the movement of sediment and related chemicals along a catchment area of great importance to coastal management (Gerbersdorf *et al.*, 2007). Therefore, hopefully the work achieved in measuring underwater sediment properties and stability can be applied.

4.6 Conclusions

- Intertidal sediment stability changes dramatically under submersion with moving water. However, this change is not found when sediment is submerged in stationary water.
- Sediment properties are not affected universally by submersion with a large degree of heterogeneity occurring in the variation of properties.
- The implications for changing sediment properties between exposed and submerged intertidal sediments when relating measurements of exposed

sediments to overall system properties are important although not clarified greatly by this work.

- If there is a fluff layer on exposed sediment then the current methods of detecting it are inadequate and this will require more research.
- If the fluff layer is being removed with the incoming tide then this needs to be quantified as over a tidal flat it will account for a large volume of sediment.
- Methods of sediment sampling devised on exposed sediment are not adequate for sampling submerged sediment.
- Laboratory replication of sediment submersion is possible, however, stationary water is not a suitable substitute to replicate natural flowing conditions.
- The CSM is an ideal machine for measuring stability on exposed and submerged sediment.
- The development of a CSM machine that can be used *in situ* on deeply submerged sediments, removing the dependence on collection of sediment cores, would be highly advantageous.

Chapter Five

The influence of *Arenicola marina* on intertidal sediment stability

Abstract

A holistic ecosystem approach was taken to assess the influence of *Arenicola marina* on intertidal sediment stability. *A. marina* is well known as an ecosystem engineer and the opportunity of an established exclusion experiment on the German island of Sylt was used to compare the sediment stability of ecosystems in the presence and absence of *A. marina*. The experiment comprised of 5 sites on the south shore of a secluded bay on the east side of the island, each with a 20m x 20m exclusion and paired control plot. All sites were sampled in the winter and summer to assess the impact of expected variability in *A. marina* activity with season. No consistent change in the macrofauna community or sediment environment was found between control and exclusion treatments in either season. Likewise there was no overall change in surface or subsurface sediment stability. However, less specific changes in the macrofauna community and sediment environment were found between individual paired control and exclusion plots within each of the 5 sites. The natural heterogeneity and variability within the ecosystem may provide the explanation, with the possibility of ecosystem engineering by *A. marina* having different effects at each site related to the existing conditions. Therefore, it is concluded that despite its perceived importance as an ecosystem engineer *A. marina* did not have a large influence on the sediment ecosystem and, under the current circumstances, did not affect overall sediment stability. However, smaller, more localised effects on the ecosystem were found related to *A. marina*'s exclusion but these did not relate to sediment stability.

5.1 Introduction

5.1.1 Biogenic influences on intertidal sediment stability

Almost every organism present within an intertidal sediment system, from single celled microphytobenthic algae to large macrofauna will affect the characteristics of the sediment through various physical, chemical and biological interactions (de Brouwer *et al.*, 2000; Widdows *et al.*, 2000; Herman *et al.*, 2001; Reise, 2002; Widdows & Brinsley, 2002; Widdows *et al.*, 2004). Such interactions can affect the stability of the sediment, potentially increasing or decreasing the erosion threshold of the surface, leading to the broad classification of species as bio-stabilisers or bio-destabilisers (Widdows & Brinsley, 2002). The influence of many individual species on sediment stability has been studied and quantified using a myriad of different methods (e.g. de Deckere *et al.*, 2000; Andersen *et al.*, 2002). However, the effect of the common and important North East European polychaete worm *Arenicola marina* on surface sediment stability has not been studied in great detail.

5.1.2 *Arenicola marina* – Lifestyle and characteristics

A. marina is considered an important species due in part to its high abundance and biomass (Reise *et al.*, 1994; Riisgard & Banta, 1998) and as its presence significantly alters the conditions of intertidal sediment systems through its lifestyle and feeding behaviour (Cadée, 1976).

A. marina lives in a semi-permanent burrow to depths of 20-40cm which is usually U or J shaped (Fig. 5.1). The burrow can be divided into three sections; the head shaft, which stretches down from the feeding pit on the sediment surface to the feeding pocket; The gallery, where *A. marina* lives, head facing the feeding pocket; and the tail shaft which extends from the gallery to the sediment surface (Riisgard & Banta, 1998; Reise, 2002). *A. marina* feeds by ingesting sediment from the feeding pocket and digesting diatoms and bacteria from within this sediment as it passes through its digestive system (Retraubun *et al.*, 1996b; Riisgard & Banta, 1998; Alyakrinskaya, 2003). Excreted sediment is returned to the sediment surface through the tail shaft forming characteristic faecal casts.

A. marina actively pumps water through its burrow by oscillating its body in a tail to head direction, forcing water to travel in the opposite direction to the sediment. Pumping water through the sediment requires a large energy outlay (Riisgard & Banta, 1998), however it brings several benefits to *A. marina*. Fluidisation of the sediments in the feeding shaft increases the ease of sediment transport into the feeding pocket (Jones & Jago, 1993) and waste material not deposited in the faecal cast may be flushed out of the burrow (Riisgard & Banta, 1998). The supply of fresh oxygenated water to the deeper sediment may also promote the growth of bacteria within the burrow, especially the feeding pocket and head shaft, increasing the amount of food available within the sediment that will be ingested. This is a process termed “gardening” although the level at which this is a deliberate benefit, rather than a by-product of pumping remains a subject of debate (Retraubun *et al.*, 1996b; Riisgard & Banta, 1998).

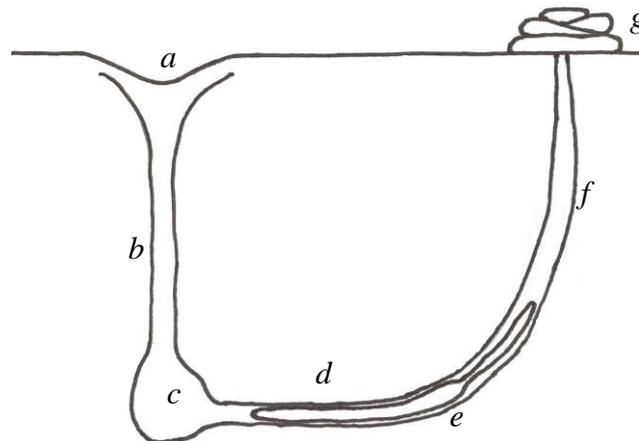


Figure 5.1. Burrow of *A. marina* with feeding pit (*a*), head shaft (*b*), feeding pocket (*c*), gallery (*d*), *A. marina* (*e*), tail shaft (*f*) and faecal cast (*g*).

By feeding in this manner *A. marina* constantly draws particles from the surface into the subsurface sediments, however, while grains of all sizes are drawn into the feeding pocket only finer grains are recycled back to the surface since *A. marina* is selective in the particles it ingests, preferring finer sediments of less than 1mm (Jones & Jago, 1993) but capable of consuming particles of a size up to 2mm (Cadée, 1976; Riisgard & Banta, 1998). This means that while fine particles are recycled to the sediment surface, larger particles, not ingested by *A. marina*, remain incorporated

within the subsurface sediments. When feeding *A. marina* ingest about 1-2ml of sediment per hour. This has been estimated to equate to an annual volume of sediment turn over sufficient to cover a depth of 12cm/year over an intertidal system (Riisgard & Banta, 1998) and 17 and 40cm/year from two intertidal sites with different densities of *A. marina* (Cadée 1976). However, yearly figures like these for a site can be misleading as there is both large variation in digging activity related to shore position and seasonal variation (Retraubun *et al.*, 1996a).

5.1.3 Ecosystem engineering by *A. marina*

A. marina has been termed an “ecosystem engineer” (Riisgard & Banta, 1998) due to the change in the environment and biota that result from its presence (Jones *et al.*, 1994; Jones *et al.*, 1997; Wright & Jones, 2006). The construction of a burrow combined with the pumping of water results in a localised increase in the depth of the oxidic layer (Riisgard & Banta, 1998) and increases oxygen levels, water content and bacterial numbers within the sediment, while there is also a decrease in sulphide concentration (Cadée, 1976; Jones & Jago, 1993; Retraubun *et al.*, 1996b; Volkenborn & Reise, 2006). Less localised effects on the sediment environment include reducing the organic and chlorophyll content of the sediment (Volkenborn & Reise, 2006) and an increase in chemicals sequestered within deeper sediments through drawing sediment particles and water into the deeper sediment (Rasmussen *et al.*, 1998; Petersen *et al.*, 1998; Timmerman *et al.*, 2002).

In addition to changes in the chemical and physical properties of the habitat, the selectivity of *A. marina* in ingesting finer sediments causes an increase in the average grain size and reduction in the cohesive fraction of sediment across an intertidal shore (Cadée, 1976; Volkenborn & Reise, 2006). Larger particles, and in particular shell fragments, which are channelled into the feeding pocket but not ingested accumulate at the depth of the feeding pocket. This can be seen with the common occurrence of a layer of larger particles or a “shell layer” within the sediment across intertidal systems inhabited by *A. marina* (Riisgard & Banta, 1998; Reise, 2002).

This type of ecosystem engineering by *A. marina* is termed allogenic ecosystem engineering, where the organism changes the environment in which it is found through its actions, but the physical body of the organism is not an integral part of the

change to the system (autogenic engineering) (Jones *et al.*, 1994; Jones *et al.*, 1997). The resulting change has consequences for the macrofauna community inhabiting intertidal sediments. *A. marina* is associated with increases in the adult population of the polychaete *Scoloplos armiger* (Volkenborn & Reise, 2006) and decreases the polychaete *Nereis diversicolor*, several bivalve species and the amphipod *Corophium volutator* (Flach, 1992; Flach & De Bruin, 1994; Beukema & Flach, 1995; Flach, 2003; Volkenborn & Reise, 2006). Juvenile populations of the polychaete worms *Nephtys hombergii*, *Heteromastus filiformis*, *S. armiger*, *Capitella capitata* and even *A. marina* itself, and the bivalves *Mya arenaria*, *Cerastoderma edule*, *Macoma balthica*, *Angulus tenuis* are also inhibited (Flach, 1992; Flach & Beukema, 1994; Hardege *et al.*, 1998). On a smaller scale the creation of a burrow with a consistent supply of oxygenated water within the sediment creates a new habitat which is commonly populated by meiofauna species while copepods inhabit the water filled funnel on the sediment surface during tidal exposure (Retraubun *et al.*, 1996b).

5.1.4 Sediment stability as an ecosystem property

Within a coastal ecosystem, the sediment stability is in part dictated by the influences of many different species. The sum of all these influences therefore combines to produce an overall biotic influence that may result in the actual stability being higher or lower than the natural stability of the sediment. Therefore the biogenic influence on the sediment stability should be considered as a property of the ecosystem and can be measured as a level of biogenic (de)stabilisation.

Experimental studies have demonstrated that changes in the macrofauna community within sediment systems through manipulation of diversity (Emmerson *et al.*, 2001; Bolam *et al.*, 2002; Biles *et al.*, 2002; Biles *et al.*, 2003; Solan *et al.*, 2004; Waldusser & Marinelli, 2006), removal of large species (Thrush *et al.*, 2006) and introduction of non-native species (Ruesink *et al.*, 2006) can lead to changes in the properties or functioning of the system. Therefore, a study into the influence of a species on sediment stability should not only account for its direct influence on the sediment but also its indirect role within the ecosystem. An example of this is the effect of birds upon sediment stability; physical disturbance of sediment by feeding birds causes a reduction in its stability (Cadée, 1990), however, Daborn *et al.*, (1993) included migratory birds within an ecosystem analysis of sediment stability. Through changes

in a series of trophic levels the arrival of the migratory birds, a top order predator within the system, caused an increase in microphytobenthic algae, which subsequently increased the sediment stability. This sequence of interactions based on the food web is a trophic cascade. However, the results of this cascade changed the (de)stabilising influence of the ecosystem and it therefore also has to be considered as an ecological cascade (Kitchell & Carpenter, 1993; Wootton, 2002).

Most species within a sediment system may be shown to have an effect on the surface stability, however it is very important to measure the redundancy of this effect. The influence of a single species on a sediment property such as stability may not be unique to that species. If the species was to be removed a change in the macrofauna community may replicate that influence, implying that the original species can be regarded as redundant in relation to that property of the ecosystem (Solan *et al.*, 2004). Alternately, a species may have little direct effect on stability but its presence has a unique influence on other species with which it interacts, so while it may not directly influence stability, the species holds a unique position within the (de)stabilising influence of the system. Therefore the role of a species in sediment stability has to be considered in light of how its influence upon sediment stability relates to its role in the ecosystem and the redundancy of such an influence within the system.

5.1.5 *A. marina*'s effect on sediment stability

Reise (2002) describes the relationships between and within species and the environment in sediment systems as an interaction of “complex habitat mediated interaction webs” and “trophic webs”. The position of *A. marina* as an ecosystem engineer within such a complex system results from its importance in shaping such interactions, therefore it is required to be considered and studied in the same way as a top predator or keystone species. Logically, a study into the affect of *A. marina* on sediment stability cannot be based solely upon its direct interaction with the sediment but as a comparison of the (de)stabilising property of the ecosystem that exists as a result of its engineering and also the parallel ecosystem that would occur in its absence.

An extensive, long-term, exclusion experiment has been established on the German island of Sylt providing an opportunity to examine the influence *A. marina* on sediment stability (Volkenborn & Reise, 2006). An approach based on studying the whole ecosystem in the presence and absence of *A. marina* was used, analysing the macrofauna community and the sediment environment as a whole. From that analysis, changes between ecosystems in some individual species and sediment properties may be identified as more significant than others. Changes in the ecosystem or individual species or properties can then be related to the stability of the sediment.

Within sediment systems the terms “stable” and “stability” can be used to define two different, but potentially interacting properties of sediment particles. This work is concentrated on the stability of the sediment surface, referring to the erosional properties of the sediment with particular respect to the critical erosion threshold. However, “stability” can be and is used in a different way in regard to a measurement of the rate of change in the spatial and temporal dynamics, or redistribution, of sediment particles. Using the second definition, high stability, or a more stable environment, indicates low temporal or spatial movement of sediment particles within a system. Within this study this type stability is of particular importance in reference to bioturbation, with increased bioturbation resulting in a less stable system. This type of stability will therefore be subsequently referred to as “dynamic stability”.

5.1.6 Hypotheses

- H₁: Bioturbation by *A. marina* decreases sediment consolidation.
- H₂: Bioturbation by *A. marina* increases the average grain size and reduces the cohesive grain size fraction of the substratum.
- H₃: Microphytobenthic abundance will increase in the absence of *A. marina*.
- H₄: Ecosystems with *A. marina* will have a different macrofauna community.
- H₅: The bioturbation by *A. marina* and resulting change in ecosystem will result in a lower level of sediment stability.

5.2 Methods

5.2.1 Experiment location and design

A large scale *A. marina* exclusion experiment was established in 2003 in the Königshafen, a secluded bay on the east coast of the northern end of Sylt, Germany (55°02'N; 8°26'E) with average *A. marina* densities of 30 individuals/m² (Reise *et al.*, 1994). The experiment was designed as a paired block design, with 6 sites (3 each at high and low shore) each consisting of an exclusion plot and disturbance control plot (hereafter referred to as the exclusion and control treatment plots) (Fig. 5.2). High shore sites were composed predominantly of fine sand while low shore sites consisted of muddier sediments. Exclusion plots were created by laying a 1mm mesh at a depth of 10cm in 20m by 20m plots to prevent *A. marina* forming its burrows (Fig. 5.3), control plots were created by digging up the sediment but relaying it without the mesh (for full experimental lay out and details on the Königshafen see; Volkenborn & Reise, (2006)). Unfortunately, due to an unexpected absence of adult *A. marina* in the control plot of site 2 on the low shore the entire site was excluded from the experiment and any subsequent analysis. Therefore only 5 replicates of the exclusion and control treatments were used.

Winter and summer sampling was performed in February and August 2005, with each site taking a day to sample fully. Exclusion and control plots were sampled equally with 10 sample locations chosen from each plot, allowing for a 2m edge effect. From each sample location measurements and samples were taken of sediment stability and properties, while the macrofaunal community was sampled and preserved for subsequent identification.

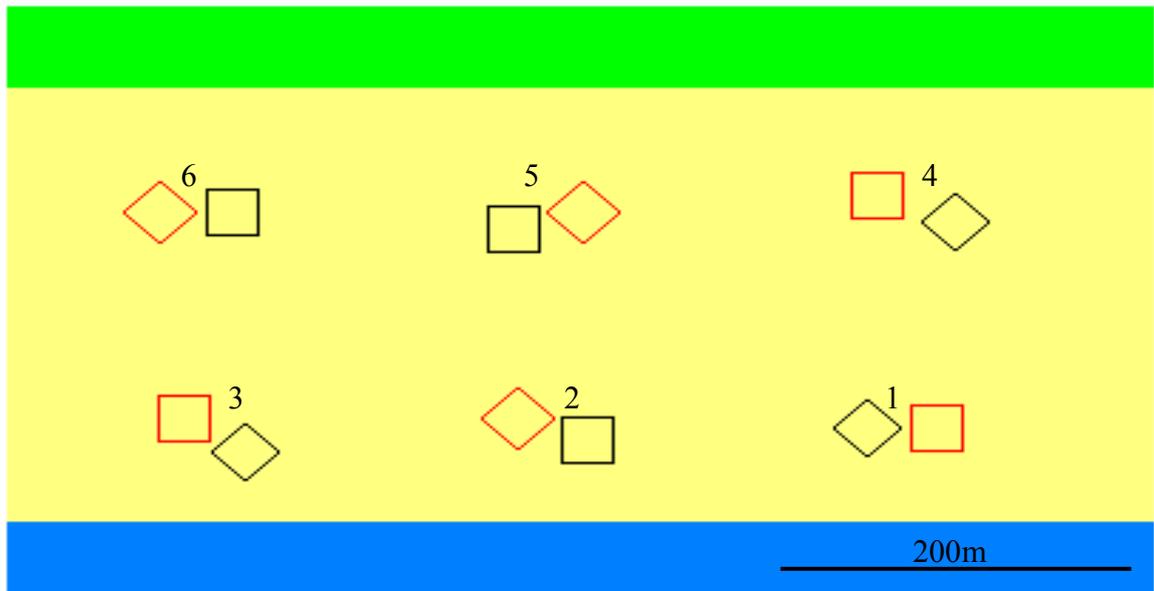


Figure 5.2. Experimental layout of control (black) and exclusion (red) plots within the low shore (1-3) and high shore (4-6) sites.



Figure 5.3. 20m x 20m exclusion plot. Identifiable by the absence of *A. marina* faecal casts on the left.

5.2.2 Measurements of sediment stability

Sediment stability was measured using the Cohesive Strength Meter (CSM) (Paterson, 1989). Fine 1 was selected as the test setting allowing maximum resolution at the low eroding pressures expected at the site after initial trails (see Chapter 3). The critical erosion threshold was deemed to have passed with a 10% drop in transmission indicating sediment suspension (Tolhurst *et al.*, 1999). Results were expressed as surface stagnation pressure (Nm^{-2}) following the calibration method proposed by Vardy *et al.*, (2007).

A shear vane was used at depths of 5 and 10cm to measure the shear strength of the substratum below the surface. Deeper measurements were impossible due to the exclusion mesh at 10cm depth.

5.2.3 Environmental and sediment properties

A contact core (HIMOM, 2003) was used to take a sample of the top 2mm of sediment at each sample location for subsequent analysis of dry bulk density, grain size and water, organic, colloidal carbohydrate and chlorophyll *a* concentrations.

Many of the previous studies into *A. marina* have not analysed the surface sediment with such accuracy, instead using deeper cores (e.g. Jones & Jago, 1993). To assess the stability of surface sediments, measurements have to be based upon sediment properties at the relevant scale and not diluted or contaminated by including deeper sediment with different properties.

All analyse were performed according to the HIMOM (2003) protocols while chlorophyll *a* concentrations were determined using HPLC. Results are expressed as concentrations rather than content (Flemming & Delafontaine, 2000; Perkins *et al.*, 2003; Tolhurst *et al.*, 2005; See Chapter 2 for calculations).

5.2.4 Grain size

6 samples were selected randomly from each site for grain size analysis which was performed using a Coulter LS230 grain size machine with grain size fractions set to 0-63, 64-125, 126-250, 251-500, 501-1000 and 1001-2000 μm . Sediments below 63 μm

are taken as the cohesive fraction. Grain size fractions are labelled in this chapter with the largest diameter in each fraction.

5.2.5 Macrofauna

A. marina density data was supplied by Nils Volkenborn, based upon counts of faecal casts with a 25cm² quadrat (n=10). Adult and juveniles were separated by the size of the cast. *A. marina* numbers are considered separately and not included within the analysis of community structure. The macrofauna community was sampled using a 105mm diameter (86.6cm²) sediment core taken to the depth of the mesh on the exclusion plots (≈10cm) and to a depth of 10cm on the control plots to avoid bias. Cores were sieved through a 500µm sieve and the remains preserved in 10% formalin on the day of sampling for later identification. Macrofauna was dyed with Rose Bengal prior to identification to aid removal from remaining sediment. Identification was performed to species level where possible and numbers were expressed in numbers per core to avoid errors through extrapolation.

5.2.6 Statistical analysis

5.2.6.1 Sample groupings and amalgamation

Measurements of the sediment properties (n=10) or macrofauna community (n=10) from each plot in each season were averaged to give a single value representing the plot within that season. These values were then used for subsequent statistical analysis.

The individual measurements of sediment properties from each plot at each date were used to calculate a coefficient of variation (c.v. value = standard deviation of the samples divided by their average) for each plot as a measure of the heterogeneity of the plot which was then used in subsequent analysis.

5.2.6.2 Univariate analysis methods

Analysis of individual sediment properties, c.v. values and individual species counts were performed through a two-way ANOVA between treatment and season. Prior to analysis all results were tested for assumptions of normality required for parametric tests (Zar, 1998).

5.2.6.3 Multivariate analysis methods

All multivariate analysis was performed using PRIMER 6.0 software package (Clarke & Warwick, 2001).

Due to the dominance of *H. ulvae* in the macrofauna community in the Königshafen, parallel analyses of the community composition were performed with *H. ulvae* included and excluded. Species counts were fourth root transformed when *H. ulvae* was included to reduce its influence while square root transformation was sufficient when *H. ulvae* was not included in the community composition.

Macrofauna communities from each plot and season were compared using the Bray-Curtis similarity index (*S*) with the results used to construct an n-MDS ordination showing treatment and season. A two-way ANOSIM was used to determine if the exclusion of *A. marina*, or the different seasons resulted in an overall change in the macrofauna community, while additional one way ANOSIMs were performed between treatments within the winter and summer communities.

SIMPER analysis was performed on the community to identify how different the communities from each treatment were and which, if any, species accounted for the majority of the variation. *H. ulvae* was excluded from this analysis to prevent its dominance hiding the influence of other species. The abundances of species identified as contributing to the variation between treatments was then analysed through univariate methods to determine if their abundance significantly differed between treatments or seasons, *H. ulvae* was also included in this analysis due to its numerical dominance.

In addition to examining the overall change in macrofauna community, a subsequent analysis was used to determine if the exclusion of *A. marina* caused more localised and less universal changes in the macrofauna community. Using the 10 samples taken from each plot, an ANOSIM was performed to compare the macrofauna communities between each of the paired exclusion and control plots. The *R* values from these tests were averaged within season and placed into a one tailed T test to determine if they were different from 0, therefore indicating if there were changes in the macrofauna community between paired treatments. This analysis does not demonstrate a

consistent change in macrofauna community across the entire experiment as the R value from each of the paired treatments is specific to that site

The sediment environment was analysed using similar procedures to the macrofauna community, with all measurements normalised to allow direct comparisons, no transformation was required. The sediment environment from each plot on each date was compared and quantified through measurement of the euclidean distance between each. A PCA and ANOSIM analysis were used to determine if any change in sediment environment occurred between treatments or dates.

The macrofaunal community was correlated with sediment properties, experimental treatments, season and location on shore using the BIOENV procedure. This would determine which variables and factors best explained any variation in the macrofaunal community.

5.3 Results

5.3.1 *A. marina* exclusion

The implanted mesh prevented the colonisation of the sediment by adult *A. marina*. Unsurprisingly this gave a significant treatment effect but there was no additional change in *A. marina* density between seasons. Juvenile *A. marina* were not excluded by the mesh with no differences in density between treatments although densities were higher in the summer than the winter (Table 5.1).

5.3.2 Macrofauna

5.3.2.1 Species abundance and distribution

The macrofauna community of the Königshafen was dominated by the gastropod *Hydrobia ulvae*, with average numbers of nearly 1000 per core in the winter increasing to nearly 1500 per core in the summer (equating to 11000 and 17500 individuals m⁻², respectively). *H. ulvae* accounted for 90% of all macrofauna sampled in both the summer and winter. In the winter, the remaining 10% was comprised of oligochaete species (4%) and nematodes (4%) with the final 2% mostly polychaete worms and bivalves. During the summer, oligochaetes again comprised about 4% of

sampled individuals while bivalves numbers increased to 3% and nematodes reduced to just over 1%. Polychaete worms again comprised most of the remaining macrofauna. Numbers of individual organisms were significantly higher in summer than winter samples (Table 5.2).

Table 5.1 ANOVA comparisons of mean densities of adult and juvenile *A. marina* m² for winter and summer samples. Mean densities are given with standard errors. Significant differences in bold.

	Average	s.e.		d.f.	F Value	p value
Adults						
Winter / Control	3.22	0.68	Treatment	1, 16	30.52	<0.001
Winter / Exclusion	0.00	0.00	Season	1, 16	0.04	0.845
Summer /Control	3.46	1.00	Interaction	1, 16	0.04	0.845
Summer / Exclusion	0.00	0.00				
Juvenile						
Winter / Control	0.90	0.28	Treatment	1, 16	1.79	0.200
Winter / Exclusion	0.42	0.12	Season	1, 16	9.54	0.007
Summer /Control	1.38	0.29	Interaction	1, 16	0.77	0.394
Summer / Exclusion	1.28	0.10				

Table 5.2 Comparison between seasons of individual organism abundances from each macrofauna core sample. T- test (d.f. = 159), Total species counts analysed with and without the numerically dominant gastropod *H. ulvae*. Mean abundances are given with standard errors. Significant differences in bold.

	Average no. organisms per macrofauna core sample	s.e.	T value	p value
All Species				
Winter	1176.2	81.81	3.97	<0.001
Summer	1552.2	47.57		
All species except <i>H. ulvae</i>				
Winter	89.5	5.45	5.78	<0.001
Summer	137.7	6.13		

5.3.2.2 Comparisons of communities between treatments

Community composition was not affected by treatment across the whole experiment, however, there was a large change between seasons (Fig. 5.4; Table 5.3), possibly due to the increase in the numbers of individual organisms (Table 5.2). No difference in community was found between treatments within each season, either with or without *H. ulvae* included in the community composition (Table 5.3).

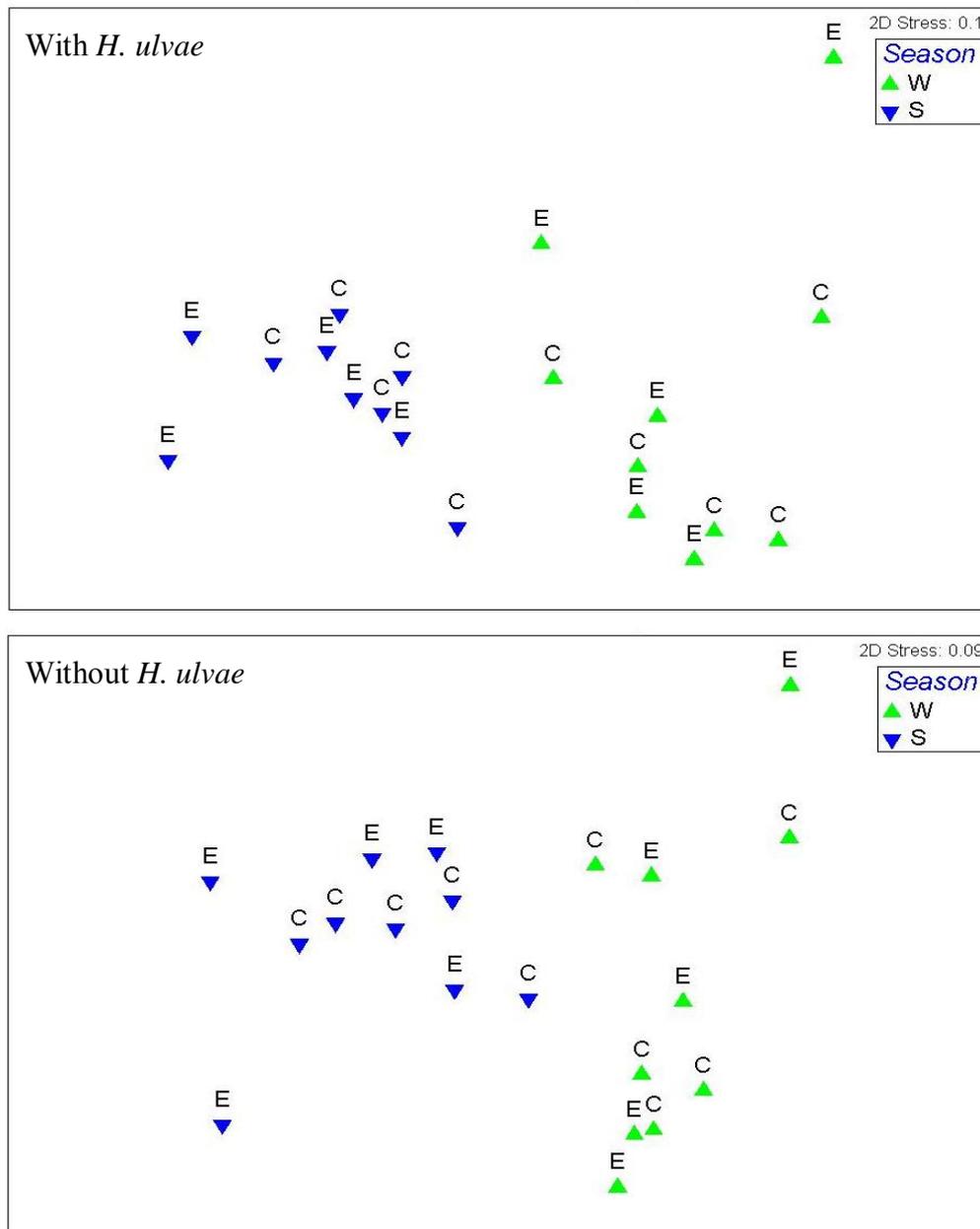


Figure 5.4 n-MDS Ordinations of the macrofauna community from each plot, with and without *H. ulvae* from winter and summer samples labelled with control (C) or exclusion (E) treatments.

Table 5.3. Comparisons of macrofauna communities. A two-way ANOSIM of plot communities compared between season and treatment and one way ANOSIMs comparing treatments within each season. Communities analysed with and without the presence of the numerically dominant gastropod *H. ulvae*. Significant differences in bold.

	With <i>H. ulvae</i>		Without <i>H. ulvae</i>	
	R value	p value	R value	p value
All samples				
Treatment	-0.068	0.789	-0.056	0.699
Season	0.720	<0.001	0.680	0.002
Winter samples				
Treatment	-0.084	0.770	-0.108	0.754
Summer Samples				
Treatment	-0.052	0.571	-0.004	0.429

5.3.2.3 Community changes within plots

Individual ANOSIMs were performed between the paired exclusion and control plots using the 10 macrofauna samples from each. Averaged R values from all 5 of the paired plots within each season were not large but were significantly higher than zero (Table 5.4). This indicates that the macrofauna community in the exclusion plots was different to that in the paired control plots. Differences occurred in both seasons and with and without *H. ulvae* included into the macrofauna community analysis (Table 5.4). Averaged R values were higher in the analysis with *H. ulvae* removed, indicating that the changes that occurred were a result of other, less dominant, species. However, the difference found in the macrofauna communities between each of the paired treatment plots was not necessarily the same over the whole experiment, rather that the exclusion of *A. marina* does change the community within each of the paired plots.

Table 5.4 Mean average R values from ANOSIM comparisons of the macrofauna community from paired control and exclusion treatments within each plot and season. Averaged R values are tested for difference from 0 (One tailed T-test). Significant differences in bold.

	Average R value	s.e.	T Value	p value
With <i>H. ulvae</i>				
Winter	0.166	0.044	3.75	0.020
Summer	0.155	0.017	9.28	0.001
Without <i>H. ulvae</i>				
Winter	0.276	0.099	2.80	0.049
Summer	0.367	0.131	2.81	0.049

5.3.2.4 Species accountable for the variation in macrofauna communities between treatments

H. ulvae was not considered in this analysis and was separately analysed due to the high abundance. The similarity between the remaining macrofauna communities within each treatment and season was high (<70%), while the dissimilarity between the communities in the two treatments within each season was low (Table 5.5). Species that accounted for the majority of the dissimilarity in communities between exclusion and control plots in both seasons were Nematode spp., Oligiochaete spp. and *Pygospio elegans*, while differences in the abundances of *Tubificoides benedii* and *Scolopsis armiger* were also large in the winter and *Cerastoderma edule* and *Mya arenaria* in the summer.

5.3.2.5 Treatment comparisons of individual species abundances

In addition to the numerically dominant *H. ulvae*, species identified by the SIMPER analysis were analysed separately from the community analysis. There was no change in the abundances of any of these species between control and exclusion treatments, with only *C. edule* increasing in abundance between the winter and summer seasons (Table 5.6).

Table 5.5 SIMPER comparisons of the macrofauna communities (with *H. ulvae* excluded) within and between the control and exclusion treatments in each season. Species identified as contributing the majority of the dissimilarity between communities in each treatment are shown with their relative contribution to the dissimilarity.

	Winter		Summer	
Similarity %				
Control	76.19		81.34	
Exclusion	70.04		73.90	
Dissimilarity %				
Control	25.00		21.81	
Exclusion				
Species and % Contribution	Nematode <i>spp.</i>	28.61	<i>C. edule</i>	15.91
	<i>T. benedii</i>	13.65	Nematode <i>spp.</i>	13.59
	<i>S. armiger</i>	11.39	<i>P. elegans</i>	11.95
	Oligiochaete <i>spp.</i>	10.29	<i>M. arenarea</i>	11.87
	<i>P. elegans</i>	9.00	Oligiochaete <i>spp.</i>	7.18

5.3.3 Sediment properties

5.3.3.1 Overall sediment environment

There was no change in the overall sediment environment between exclusion and control treatments, however there was a small change between seasons (Two Way ANOSIM, Treatment $R = 0.128$, $p = 0.920$; Season $R = 0.214$, $p = 0.035$; Fig 5.5). Equally, when the sediment environment was compared between treatments within each season there was again no difference (Winter, $R = -0.140$, $p = 0.857$; Summer, $R = -0.096$, $p = 0.698$).

5.3.3.2 Sediment environment changes within paired treatments

Individual ANOSIM comparisons of the sediment environment between each of the 5 paired exclusion and control treatments within season showed that the exclusion of *A. marina* did result in a different sediment environment. Differences occurred in both seasons (Table 5.7). As with the similar results from the macrofauna community, the significance of the R value indicates that the sediment environment in each exclusion plot was different from its associated control plot, but the difference may vary between each of the paired treatments across the experiment.

Table 5.6 Mean average abundance and standard errors of the numerically dominant *H. ulvae* and additional species identified through SIMPER analysis as contributing to the majority of dissimilarity between control and exclusion treatments. Abundances compared between treatment and season through a two-way ANOVA. Significant differences in bold.

	Average	s.e.		d.f.	F Value	p value
<i>Cerastoderma edule</i>						
Winter / Control	1.10	0.64	Treatment	1, 16	0.13	0.728
Winter / Exclusion	1.88	1.64	Season	1, 16	17.94	<0.001
Summer / Control	34.72	23.93	Interaction	1, 16	0.21	0.650
Summer / Exclusion	28.88	21.16				
<i>Hydrobia ulvae</i>						
Winter / Control	948.94	363.60	Treatment	1, 16	1.09	0.311
Winter / Exclusion	1224.32	809.93	Season	1, 16	2.11	0.166
Summer / Control	1316.12	371.66	Interaction	1, 16	0.03	0.864
Summer / Exclusion	1512.94	306.90				
<i>Mya arenaria</i>						
Winter / Control	0.3	0.51	Treatment	1, 16	0.85	0.371
Winter / Exclusion	0.84	1.66	Season	1, 16	1.93	0.183
Summer / Control	3.04	5.63	Interaction	1, 16	0.65	0.433
Summer / Exclusion	11.1	20.06				
<i>Nematode spp.</i>						
Winter / Control	35.86	24.57	Treatment	1, 16	0.62	0.441
Winter / Exclusion	46.86	39.02	Season	1, 16	2.31	0.148
Summer / Control	19.14	6.74	Interaction	1, 16	0.02	0.901
Summer / Exclusion	27.10	26.61				
<i>Oligochaete spp.</i>						
Winter / Control	26.58	11.47	Treatment	1, 16	0.47	0.501
Winter / Exclusion	22.14	9.38	Season	1, 16	4.23	0.056
Summer / Control	35.38	6.74	Interaction	1, 16	0.05	0.819
Summer / Exclusion	33.18	14.14				
<i>Pygospio elegans</i>						
Winter / Control	1.42	1.07	Treatment	1, 16	3.00	0.103
Winter / Exclusion	3.80	4.09	Season	1, 16	1.65	0.217
Summer / Control	2.32	2.06	Interaction	1, 16	1.03	0.325
Summer / Exclusion	11.44	14.09				
<i>Scolopsis armiger</i>						
Winter / Control	4.10	3.04	Treatment	1, 16	0.01	0.934
Winter / Exclusion	4.00	4.51	Season	1, 16	0.02	0.893
Summer / Control	3.66	1.86	Interaction	1, 16	0.02	0.883
Summer / Exclusion	4.02	3.75				
<i>Tubificoides benedii</i>						
Winter / Control	16.64	9.15	Treatment	1, 16	1.90	0.188
Winter / Exclusion	11.22	6.61	Season	1, 16	1.82	0.196
Summer / Control	19.06	4.64	Interaction	1, 16	0.26	0.619
Summer / Exclusion	16.56	4.08				

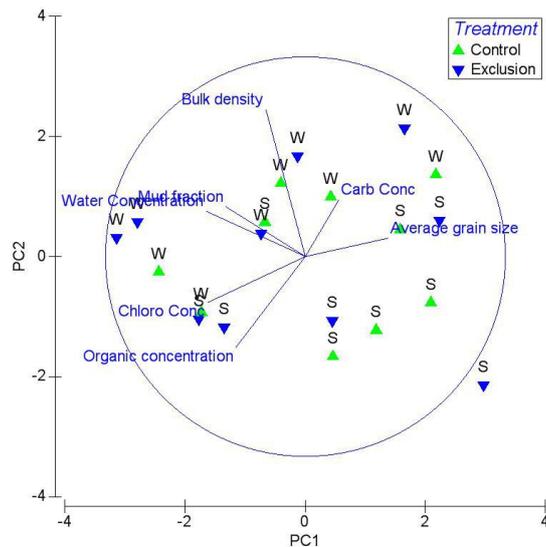


Figure 5.5 PCA of the sediment environment from all plots in both winter (W) and summer (S) seasons. Samples are distributed in relation to measurements of water, organic, carbohydrate and chlorophyll *a* concentrations, dry bulk density, average grain size and cohesive grain size fraction with variable lines within the PCA indicating increasing value of the stated variable.

Table 5.7 Mean average R values of ANOSIM comparisons of the sediment environment from control and exclusion treatments within each plot and season. Averaged R values were tested for difference from 0 (One tailed T-test). Significant differences in bold.

	Average R value	s.e.	T Value	<i>p</i> value
Winter	0.184	0.054	3.44	0.026
Summer	0.241	0.085	2.85	0.046

5.3.3.3 Individual sediment properties

There was no change in any of the measured sediment properties between exclusion and control treatments (Table 5.8; Figs. 5.6-7). Seasonal differences did occur in water concentration, dry bulk density and cohesive grain size fraction, all being higher in the winter than the summer.

5.3.3.4 Sediment environment heterogeneity

The coefficient of variation (c.v. values) was calculated for each of the sediment properties as a measurement of the heterogeneity of the plots. There was no change between *A. marina* exclusion and control plots in any of the sediment properties. Equally differences between seasons were not found in any property except organic concentration which was more varied in the winter than the summer (Table 5.9).

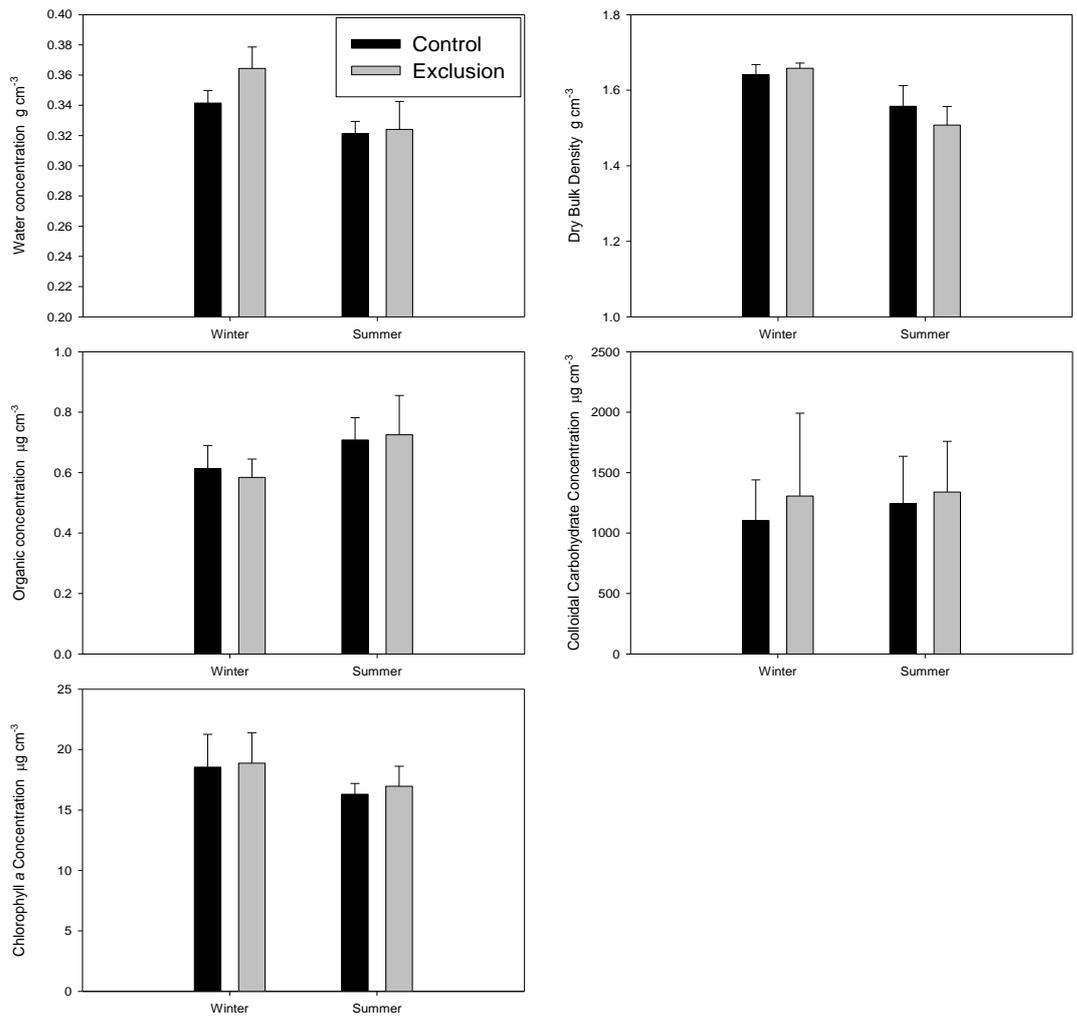


Figure 5.6 Properties of sediments from control and *A. marina* exclusion plots in the winter and summer.

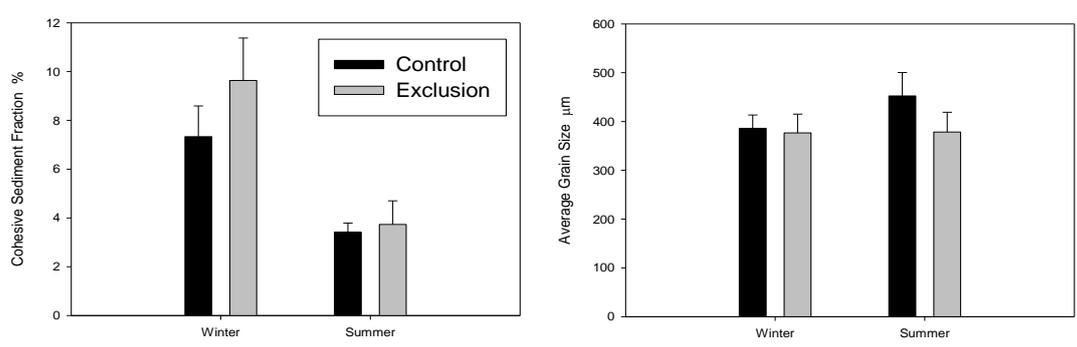


Figure 5.7 Cohesive grain size fraction and average grain size of sediments from control and *A. marina* exclusion plots in the winter and summer.

Table 5.8 Two-way ANOVA comparison of individual sediment properties between treatments and seasons. Significant differences in bold.

	d.f.	F Value	p value
Water Concentration			
Treatment	1, 16	0.97	0.339
Season	1, 16	5.42	0.033
Interaction	1, 16	0.60	0.451
Dry Bulk Density			
Treatment	1, 16	0.17	0.683
Season	1, 16	8.68	0.009
Interaction	1, 16	0.70	0.417
Organic Concentration			
Treatment	1, 16	0.00	0.949
Season	1, 16	1.75	0.205
Interaction	1, 16	0.07	0.801
Carbohydrate Concentration			
Treatment	1, 16	0.10	0.759
Season	1, 16	0.03	0.857
Interaction	1, 16	0.01	0.910
Chlorophyll <i>a</i> concentration			
Treatment	1, 16	0.06	0.815
Season	1, 16	1.01	0.331
Interaction	1, 16	0.01	0.940
Cohesive grain size fraction			
Treatment	1, 16	1.21	0.289
Season	1, 16	16.98	0.001
Interaction	1, 16	0.70	0.417
Average grain size			
Treatment	1, 16	1.13	0.305
Season	1, 16	0.75	0.399
Interaction	1, 16	0.69	0.421

Table 5.9 Heterogeneity of sediment properties, measured through coefficient of variation (c.v.) values from each plot. Mean average and standard error values given and compared between seasons and treatments through a two way ANOVA. Significant values in bold.

	Average	s.e.		d.f.	F Value	P value
Water Concentration						
Winter / Control	0.073	0.014	Treatment	1, 16	0.11	0.741
Winter / Exclusion	0.086	0.015	Season	1, 16	0.25	0.626
Summer /Control	0.091	0.027	Interaction	1, 16	0.07	0.789
Summer / Exclusion	0.089	0.026				
Dry Bulk Density						
Winter / Control	0.057	0.014	Treatment	1, 16	0.11	0.746
Winter / Exclusion	0.056	0.012	Season	1, 16	4.17	0.058
Summer /Control	0.081	0.010	Interaction	1, 16	0.16	0.698
Summer / Exclusion	0.101	0.026				
Organic Concentration						
Winter / Control	0.403	0.043	Treatment	1, 16	1.30	0.271
Winter / Exclusion	0.254	0.041	Season	1, 16	4.97	0.040
Summer /Control	0.194	0.033	Interaction	1, 16	2.93	0.106
Summer / Exclusion	0.229	0.079				
Carbohydrate Concentration						
Winter / Control	0.778	0.208	Treatment	1, 16	0.00	0.969
Winter / Exclusion	0.762	0.320	Season	1, 16	2.56	0.129
Summer /Control	0.506	0.172	Interaction	1, 16	0.00	0.971
Summer / Exclusion	0.340	0.096				
Chlorophyll a Concentration						
Winter / Control	0.476	0.056	Treatment	1, 16	1.41	0.252
Winter / Exclusion	0.328	0.043	Season	1, 16	0.97	0.338
Summer /Control	0.393	0.031	Interaction	1, 16	4.43	0.051
Summer / Exclusion	0.500	0.030				
Cohesive grain size fraction						
Winter / Control	0.390	0.117	Treatment	1, 16	1.49	0.240
Winter / Exclusion	0.235	0.044	Season	1, 16	1.25	0.280
Summer /Control	0.393	0.061	Interaction	1, 16	0.78	0.389
Summer / Exclusion	0.500	0.050				
Average grain size						
Winter / Control	0.147	0.036	Treatment	1, 16	0.02	0.878
Winter / Exclusion	0.120	0.014	Season	1, 16	8.11	0.012
Summer /Control	0.061	0.011	Interaction	1, 16	2.15	0.162
Summer / Exclusion	0.088	0.012				

5.3.4 Relating macrofauna community composition to sediment properties, treatment and site location

The change in season was identified as explaining the majority of variation in the macrofauna community with and without *H. ulvae* included in the analysis. Organic concentration was also included in comparisons without *H. ulvae*. When analysed within each season shore height was a major factor in both winter and summer, while position across the shore and water concentration were included in winter but not the summer (Table 5.10).

Table 5.10 BIOENV correlations the macrofauna community with sediment properties, plot location on the shore and treatment.

	Correlation	No. of variables	Variables
With <i>H. ulvae</i>			
Overall	0.62	1	Season
Winter	0.73	3	Water concentration, Shore height, Shore position
Summer	0.75	2	Chlorophyll <i>a</i> concentration, Shore height
Without <i>H. ulvae</i>			
Overall	0.62	3	Season, Organic concentration
Winter	0.80	4	Water concentration, Organic concentration, Shore height, Shore position
Summer	0.73	2	Chlorophyll <i>a</i> concentration, Shore height

5.3.5 Sediment stability

5.3.5.1 Sediment surface stability

A. marina exclusion or season did not have an effect on the surface sediment stability, with stability remaining unchanged across the whole experiment (Table 5.11; Fig. 5.8).

Table 5.11 Two-way ANOVA of surface sediment stability as measured by the Cohesive Strength Meter (CSM), comparisons between *A. marina* exclusion and control treatments from both winter and summer sampling. Significant differences in bold.

	d.f.	F value	<i>p</i> value
Treatment	1, 16	0.27	0.613
Season	1, 16	0.00	0.991
Interaction	1, 16	0.00	0.983

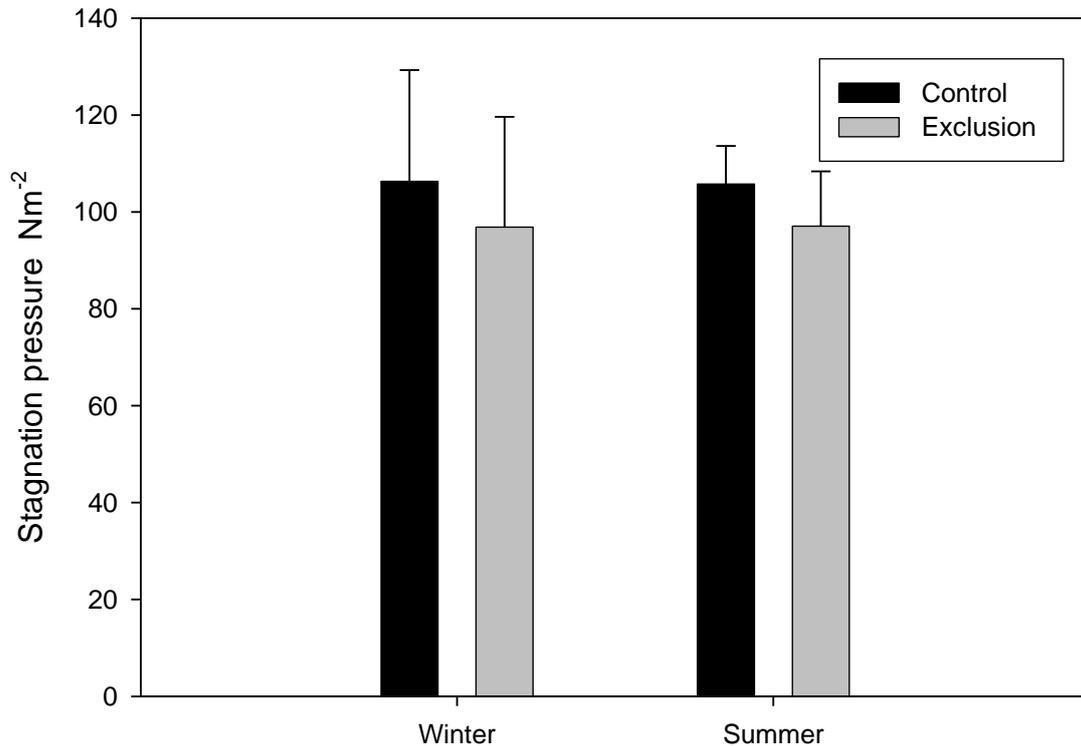


Figure 5.8 Surface sediment stability measured by the CSM from exclusion and control plots from both winter and summer sampling.

5.3.5.2 Sub surface sediment shear strength

Subsurface sediment shear strength was unaffected by *A. marina* exclusion at either of the measured depths. Higher shear strength at 5cm depth during the summer resulted in a season effect but this did not continue into deeper sediments (Table 5.12; Fig. 5.9).

5.3.5.3 Treatment effect on the heterogeneity of sediment stability

There was no difference between *A. marina* exclusion and control plots in the heterogeneity of surface or sub-surface sediment stability. Surface sediment stability was more varied in the winter than the summer, although this pattern did not continue in to the deeper sediments (Table 5.12).

Table 5.11 Two way ANOVA of sub surface sediment shear strength, comparisons made between treatment and season at two different depths of sediment. Significant differences in bold.

	d.f.	F value	p value
5cm Depth			
Treatment	1, 16	0.06	0.805
Season	1, 16	25.54	<0.001
Interaction	1, 16	0.64	0.435
10cm Depth			
Treatment	1, 16	1.00	0.333
Season	1, 16	1.40	0.253
Interaction	1, 16	1.22	0.285

Table 5.12 Coefficient of variation for surface sediment stability and sub surface shear strength with ANOVA comparisons for treatment and season. Significant differences in bold.

	Average	s.e.		d.f.	F Value	p value
Surface stability						
Winter / Control	0.385	0.102	Treatment	1, 16	0.74	0.402
Winter / Exclusion	0.462	0.091	Season	1, 16	9.29	0.008
Summer /Control	0.192	0.028	Interaction	1, 16	0.05	0.825
Summer / Exclusion	0.221	0.030				
5cm deep sediment shear strength						
Winter / Control	0.254	0.100	Treatment	1, 16	0.14	0.710
Winter / Exclusion	0.170	0.037	Season	1, 16	4.27	0.055
Summer /Control	0.078	0.014	Interaction	1, 16	1.40	0.254
Summer / Exclusion	0.121	0.015				
10cm deep sediment shear strength						
Winter / Control	0.100	0.015	Treatment	1, 16	3.36	0.086
Winter / Exclusion	0.105	0.011	Season	1, 16	0.12	0.737
Summer /Control	0.075	0.010	Interaction	1, 16	2.43	0.138
Summer / Exclusion	0.121	0.017				

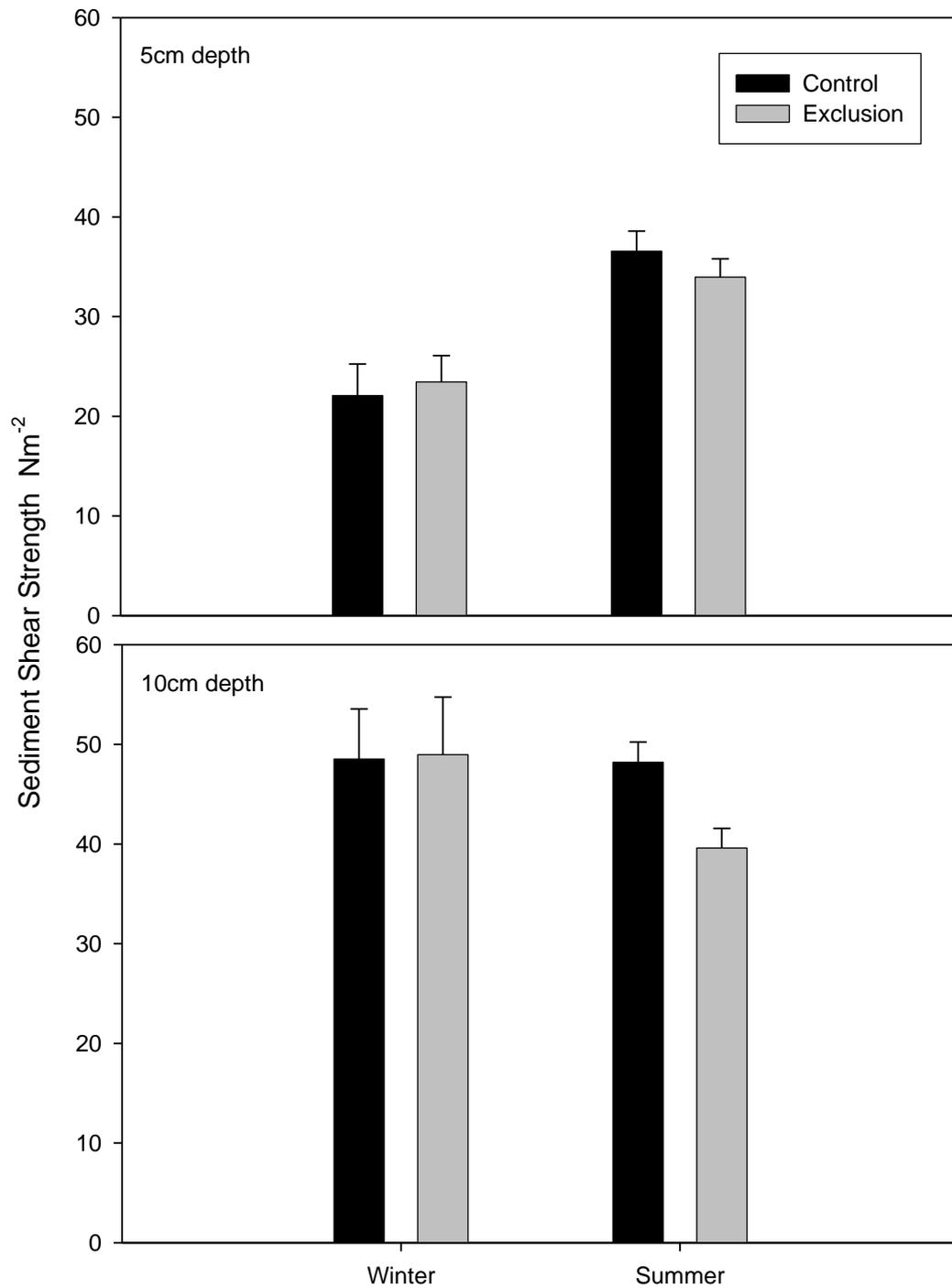


Figure 5.9 Sub surface sediment shear strength measured through a shear vane from two depths on control and exclusion plots and over winter and summer sampling periods.

5.4 Discussion

5.4.1 Assessment of hypotheses

- H₁: Bioturbation by *A. marina* decreases sediment consolidation.
- H₂: Bioturbation by *A. marina* increases the average grain size and reduces the cohesive grain size fraction of the substratum.
- H₃: Microphytobenthic abundance will increase in the absence of *A. marina*.
- H₄: Ecosystems with *A. marina* will have a different macrofauna community.
- H₅: The bioturbation by *A. marina* and resulting change in ecosystem will result in a lower level of sediment stability.

H₁: There was no change in either the dry bulk density or water concentration of the sediment. Therefore there was no indication that *A. marina* decreased the consolidation of the sediment and H₁ was rejected.

H₂: There was no difference in the average grain size or the cohesive grain size fraction between control and exclusion treatments and therefore H₂ has to be rejected.

H₃: There was no increase in the chlorophyll *a* concentration of the sediment in the *A. marina* exclusion plots. Therefore there is no indication that *A. marina* reduced the MPB abundance and H₃ was rejected.

H₄: There was no consistent change in the macrofauna community as a whole or in the abundance of specific species between control and exclusion treatments across the entire experiment. However, there were more localised changes between paired treatment plots within each site which were not consistent between sites. This implies that the removal of *A. marina* did have an effect on the macrofauna community but that it varied between sites. Therefore H₄ can be accepted with the proviso that the change was not consistent across the experiment.

H₅: Surface and subsurface sediment stability was unchanged by the exclusion of *A. marina*. Therefore there was no indication that the presence of *A. marina* in the ecosystem decreased the stability of the sediment and H₅ was rejected.

5.4.2 Exclusion of *A. marina* from the intertidal system

The complete exclusion of adult *A. marina* from the plots allowed the treatments to be regarded as two parallel ecosystems, in the presence and absence of ecosystem engineering by *A. marina*. Despite a reported preference for sediment with low adult *A. marina* populations (Hardege *et al.*, 1998) juvenile *A. marina* were equally abundant on both the exclusion and control plots. It is probable that these smaller worms can establish a shallow burrow in the sediment above the exclusion mesh. Due to their small size, the volume of sediment turnover caused by these worms is considered unlikely to have a large effect on the ecosystem in comparison to the adult worms.

5.4.3 Effect of *A. marina* on the ecosystem

5.4.3.1 The macrofauna community

There was no consistent change in the macrofauna community across the five sites within either the winter or summer seasons. Equally, no individual species consistently increased or decreased in abundance between control and exclusion plots. However, when the communities of paired treatments within each site were compared there was a difference between control and exclusion plots. This indicates that the exclusion of *A. marina* did result in different macrofauna communities but that the change was not consistent across the entire experiment. This is probably a result of natural variation in the macrofauna community in the Königshafen, with different communities at each of the 5 sites affected differently by the exclusion of *A. marina*.

Although not found in this study previous work based on the same experimental plots found increases in the population of *N. diversicolor* and decreases in *S. armiger* numbers with the exclusion of *A. marina* (Volkenborn & Reise, 2006). While the ecosystem engineering activity of *A. marina* has been associated with reductions in the juvenile populations of several bivalve species and adult *P. elegans* (Flach, 1992; Flach, 2003). The lack of any such change in this experiment was unexpected. Even given the variation between sites it was expected that the influence of *A. marina* would be evident across several plots, however this was not found and changes in community that did occur were smaller and more specific to the nature of the original community.

5.4.3.2 The sediment environment

As with the macrofauna community there was no consistent difference between individual sediment properties or the sediment environment between the control and exclusion plots across the experiment. However, changes between paired treatments within each plot were found. The suggested explanation for this is similar to that for the macrofaunal community, that given the natural heterogeneity within the area covered by the experiment each plot had a different initial sediment environment. The removal of *A. marina* from different sediment environments therefore produced different changes within each plot, rather than a more universal change across the entire experiment. The lack of an overall change in sediment environment or specific sediment properties was unexpected as it contradicts previous research into the impact of *A. marina* on the sediment environment. This includes elevated water within the substratum (Cadée, 1976; Jones & Jago, 1993) and a larger average grain size and a reduction in the cohesive grain size fraction (Cadée, 1976; Volkenborn & Reise, 2006). However, these properties are usually measured using a deeper scale than the upper 2mm used in this study and includes the larger sediments not ingested by *A. marina* and sequestered into the substratum. It might have been expected that by continually depositing fine sediments on to the surface, the grain size of surface sediments would be finer in the presence of *A. marina*. However, this was not found, implying that the cycling of the sediment causes finer sediment to be removed from the system altogether, probably through hydrodynamic forces (Cadée, 2001), rather than remaining in the upper surface layer.

As with other sediment properties, microphytobenthic abundance was unaffected by the exclusion of *A. marina*. An increase in MPB with the exclusion of *A. marina* was expected as a result of the removal of its bioturbation creating a more dynamically stable sediment surface on which MPB could grow. Equally the release from the grazing pressure from *A. marina* would allow MPB abundance to increase. As no change was measured in chlorophyll *a* these expectations have been disproved. It is possible that grazing pressure from *A. marina* was negligible and as a result the MPB did not benefit from its exclusion. However, this is considered unlikely given the dense populations of *A. marina* within the site and the large volume of material ingested by *A. marina*. Instead it is possible that any increase in the MPB that may

have resulted from the release of *A. marina*'s grazing pressure would have been consumed by other species.

5.4.3.3 The overall influence of *A. marina* in changing a sediment ecosystem

The lack of consistent changes over the entire experiment in either the macrofauna community or the sediment environment implies the activities of *A. marina* cannot be considered as a major influence in shaping the ecosystem of the Königshafen. Instead it appears that changes in the season and position on the shore within the area studied are larger influences upon macrofauna composition than the ecosystem engineering of *A. marina*. Although shore position and seasonal variation are commonly associated with changes in macrofaunal community the perceived status of *A. marina* as an ecosystem engineer would be expected to have a larger influence on the macrofaunal community within the confines of such temporal and spatial variation.

5.4.3 The influence of *A. marina* on sediment stability

5.4.3.1 The direct influence of *A. marina* on sediment stability

There was no change in the surface or sub surface sediment stability between ecosystems with or without *A. marina*. The activity of *A. marina* was expected to create a more dynamically stable and consolidated sediment. Equally without the cycling of finer sediments by *A. marina* it was expected that there would be an increase in the cohesive grain size fraction of the substratum. Both of these consequences of *A. marina*'s activity would lead to an increase in the surface and subsurface sediment stability. However, the lack of change in dry bulk density, water concentration, average grain size and cohesive grain size fraction indicates that none of the expected changes occurred and likewise there was no change in the stability of the sediment.

The nature of the area studied may have had a bearing upon the lack of change in sediment stability. The Königshafen is not an estuary but a small enclosed bay with minimal fresh water input. Equally the island of Sylt itself is predominantly comprised of sandy particles and it is possible that there is only a very small potential for fine cohesive particles to enter the ecosystem within the Königshafen. If this is true then the results from this experiment may not be representative of other systems where riverine input or greater exposure to the open sea may give a greater supply of

finer sediment particles. If this was to happen then a larger input in finer particles may either decrease or increase the sediment stability depending upon if the particles were larger or smaller than the non cohesive / cohesive size boundary respectively.

5.4.3.2 The influence of *A. marina* on the biogenic (de)stabilisation of the sediment

There was no difference in surface and subsurface sediment stability between the ecosystems with and without *A. marina*. Therefore it can be stated that within this experiment the (de)stabilising influence of the two ecosystems was the same.

No other macrofauna species within the Königshafen has the potential to replicate the bioturbation of *A. marina* in terms of the volume and depth of sediment it turns over. Equally this bioturbation has been found to influence the abundance of other species within the sediment. Therefore, the sediment system in the absence of *A. marina* was expected to have different properties driven by the influences of the other species affecting the sediment in different ways. However, the lack of a consistent change in the macrofauna community between treatments was problematic, with different changes in assemblages in each site possibly having a different overall influence on the stability of the sediment. It is still feasible that the exclusion of *A. marina* would cause a consistent change over the 5 sites through the removal of its direct influence on sediment stability, even if the remaining community had no effect. However, no difference in stability was found so it can be stated that within this experiment the macrofauna community that occurred in the presence of *A. marina* had the same influence on sediment stabilisation as the community that existed with its exclusion

These results contradict previous research which associated decreasing surface stability with increasing *A. marina* numbers (Defew *et al.*, 2002). However, that conclusion was based upon a survey, where *A. marina* abundances were at natural levels and low abundances would have been due to the unsuitable nature of the habitat. Therefore in that study a correlation actually occurred between stability and *A. marina* numbers, rather than *A. marina* reducing stability as stated. This experiment differed because conditions at all sites were suitable for *A. marina* colonisation but colonisation was prevented.

Previous work has shown no change in sediment stability with changing macrofauna community and diversity (Bolam *et al.*, 2002). However within that study *A. marina* was not considered and all communities always had a representative species present from five identified functional groups. Therefore if each of the identified functional groups had an influence on sediment stability, regardless of the actual species that filled that role, the lack of change in stability in that study could be associated with no change in the functional diversity and hence the (de)stabilising influence of the community. Had a functional group been removed this situation might have been different.

There may be parallel situation with the current experiment, where the importance of *A. marina* as an ecosystem engineer meant its removal could be considered as a removal of an entire functional group. However, in respect to its influence on sediment stability it appears that *A. marina* does not hold such a unique position under the experimental conditions. This contradicts the perceived importance of the species as a unique keystone species and ecosystem engineer (Wright & Jones *et al.*, 2006).

5.4.5 Importance of an ecosystem approach in the study of sediment stability

The holistic approach to studying surface sediment stability adopted in this experiment is essential when aiming to quantify the role of a single species within an intertidal ecosystem. Biotic changes to the habitat within an intertidal sediment system are a result many different influences interacting with the sediment (Reise, 2002), where the presence and actions of a single species can affect the influences of many other species (Waldusser & Marinelli, 2006), of which an ecological cascade is but one example (Kitchell & Carpenter, 1993; Wootton, 2002). In this experiment, the removal of *A. marina* had no effect on the (de)stabilising influence of the system. However, while many other species have been found to directly influence sediment stability their contribution to the (de)stabilising influence of each ecosystem in which they are found remains unclear.

An example which was apparent from this experiment was the role of *H. ulvae*. The high numbers of *H. ulvae* found may account for the low levels of microphytobenthic organisms, reducing their input to biogenic stabilisation to a minimum (Austen *et al.*, 1999; Decho, 2000). Additionally, *H. ulvae* directly reduces stability through

interactions with the sediment (Andersen *et al.*, 2002; Andersen *et al.*, 2005; Lumborg *et al.*, 2006; Orvian *et al.*, 2006). From this knowledge, it is possible to quantify how *H. ulvae* affects sediment stability, but to what extent these influences are specific to *H. ulvae* is unknown and in its absence would changes in the ecosystem compensate for them? Therefore assessing the influence of *H. ulvae*, or any other species, in sediment stability requires a holistic ecosystem approach to determine if it fulfils a unique role or if in its absence one or several species replicate its influence. This approach would incorporate the many different species that stabilise or destabilise intertidal sediments (Reise, 2002; Widdows & Brinsley, 2002) and may lead to a better understanding of sediment (de)stabilisation as an ecosystem property.

Chapter Six

The impact of bait digging on the intertidal sediment ecosystem

Abstract

The biotic and abiotic impact of bait digging for the large polychaete worm *Arenicola marina* was studied on the Eden Estuary, Scotland. Although similar studies have been performed before, most have focused on the impact on target and non-target macrofauna species. This study undertook an ecosystem approach, simultaneously considering the impact on the macrofauna community and on a range of sediment properties. The stability of the sediment was regarded as an important property of the ecosystem that is influenced by sediment properties and the behaviour of a variety of species that may be affected by the disturbance. Experimental plots were established in spring 2007 and bait digging was simulated either once or twice over the next two months. Samples were taken for 3 months after the first digging event. Few consistent changes in the sediment environment or macrofauna community were found as a result of the digging disturbance, although the high level of natural heterogeneity in the control plots may have masked any effect. Surface sediment stability was unaffected, while subsurface sediment shear strength was reduced in shallow sediment (5cm depth) but not deeper. The area chosen for the experiment comprised muddy sand and it is suggested that such largely non-cohesive systems can absorb the disturbance of bait digging with minimal consequences. The density of *A. marina* adjacent to the dug plots was studied to observe potential local recolonisation or dilution of the population. No effect was found in these areas and minimal recolonisation by *A. marina* of the dug areas occurred indicating that recolonisation occurs over a longer period than the time frame of this experiment. Therefore bait digging over a wide area of the intertidal is likely to have lasting impact on *A. marina* populations but with little further effect on the sediment environment.

6.1 Introduction

6.1.1 Tayport Sands bait digging impact assessment

Scottish Natural Heritage (SNH) is concerned about the effects of bait diggers within the Tayport Sands area of the Tay Estuary. This study was designed to assess the impact of bait digging upon target and non-target macrofauna species, as well as the overall sediment environment, including the impact on erosional properties of the sediment. As the Tay Estuary is a Special Area of Conservation (SAC) and Special Protection Area (SPA) the possible implications of bait digging for the important conservation species that inhabit or frequent the area is also important. This study comprised two elements, a survey of target and non-target species within the area and a study into the impact of bait digging, comprising a literature review and *in situ* experiment. This information will be given to SNH to help decide upon an acceptable level of bait digging within the area. The experimental section of this work is covered within this chapter.

6.1.2 Bait digging history

The harvesting and exploitation of species living within intertidal sediment systems by humans has occurred for centuries around the world (Ferns *et al.*, 2000). Although the target species change depending upon locality and geographic region, the general focus is towards large species of bivalves and polychaete worms found buried in the upper sediment. In North West Europe these target species are most commonly the common cockle (*Cerastoderma edule*) and the lugworm (*Arenicola marina*), respectively.

6.1.3 Bait digging methods

The process, technique and scale of bait digging operations varies between and within areas, ranging from a single person using manual tools, to large commercial operations with specialised equipment (Kaiser *et al.*, 2001).

At its most basic bait digging is performed by an individual person using a rake (for bivalves in the upper 5-10cm of the sediment) or spade (for deeper, ≈30-40cm, polychaete worms) to dig up the sediment and remove individual organisms. This is

mostly for recreational purposes (specifically bait for fishing), and therefore its scale is limited, with an individual only taking the number of animals required.

These techniques are also used by organised groups of professional operatives. While the mechanisms are the same the scale is considerably larger, with the aim of the workers to extract as many animals as possible to maximise profit. Such groups may move across an entire area removing as many animals as possible and may return to a specific area several times a year.

Progressing from manual bait digging, some commercial operations use large trawls to dredge the sediment and remove target animals. Although this is also called bait digging, the scale of the operation leads to the term “bait harvesting” (Kaiser *et al.*, 2001) being more appropriate. These operations are mostly based on extracting bivalve species for selling as food (Hall & Harding, 1997) although large polychaetes are also sometimes targeted (Beukema, 1995). There are two main methods used by these organisations to collect the bivalves. A tractor pulled trawl removes the upper layer of sediment and channels it into a large rotating cylinder within the trailer. The cylinder has holes large enough to allow the sediment and small non-target species to fall through where they are deposited back on to the sediment surface. Larger cockles are retained in the base of the cylinder and collected. This method is performed during low tide and the tractor allows for the trawl to be operated very accurately allowing total coverage of an area, although this is not often done, with gaps between trawls often left undug. The second method uses a boat to pull a hydraulic suction sledge across the sediment surface. The sledge pumps water into the sediment surface, effectively fluidising the upper sediment. The sediment is then dredged up into filters in which the cockles are contained while all remaining species and sediment are allowed to pass back to the sediment surface. As this obviously has to be performed during high tide, combined with the inherent lack of manoeuvrability in using a boat, the method produces a random trawl that does not allow for 100% coverage (Hall & Harding, 1997).

6.1.4 Effect on macrofauna species

6.1.4.1 Target species

In any assessment of bait digging the results are very much dependant upon the scale of the original operation. A population of *A. marina* within a mudflat is probably able to accommodate a small level of recreational bait digging by a few individuals (Blake, 1979). However, if digging levels are increased then the populations of target species will be affected. By their nature, the targeted species are often climax species with the largest bodies, slowest growth rates and longest life cycles. This means that in areas where digging has occurred for several years the populations of target species are often low and of a smaller size than in adjacent undug areas (Beukema, 1995; Griffiths *et al.*, 2006). Due to the slow growth and life cycles of target species the populations can then take several years to recover after bait digging is stopped (Beukema, 1995).

Commercial digging and dredging only removes individuals of a necessary size, usually indicating adult status. Although this leaves small and juvenile individuals in the sediment, the process of dredging (especially using the tractor trawl) is highly energetic and damages a large proportion of non-target individuals and subsequent mortality is high (Kaiser *et al.*, 2001).

6.1.4.2 Non target macrofauna species

Digging disturbance nearly always alters the macrofauna community, with reductions in non-target species number and abundance. Recovery of the community to pre-digging levels appears to take between 3 to 6 months, however, the extent and rate of recovery is highly variable and often dependant upon the method and scale of the digging.

Bait digging

Recreational digging with a spade for larger worms is often done by digging a small hole and depositing the sediment in a mound next to the hole. The recolonisation of these features differs; often the pit will have a low number of inhabitants and the populations recover slowly while the mound will show a short-term increase in some motile species from a combination of the species within the deposited sediment and those originally there, before an evening out of the numbers between the mound and

pit (McLusky *et al.*, 1983; Griffiths *et al.*, 2006). Infilling, where the dug sediment is replaced in the pit, hastens the return to the original species distribution (McLusky *et al.*, 1983). Less motile and surface dwelling species are usually more adversely affected by this method. Burying these species within the dug up sediment often places them a depth at which they cannot return to the surface, possibly in anoxic sediment with high sulphur concentrations, and subsequent mortality is high (Jackson & James, 1979; Brown & Wilson, 1997). Raking the sediment for cockles does not disturb the sediment to the same depth as digging for worms, however changes in the macrofauna community are similar (Cowie *et al.*, 2000).

Bait Harvesting

The short-term consequences of larger scale digging and trawling are similar to those of small scale operations, although on a greater scale. However, the recovery of the community after such dredging often takes longer. This is a consequence of the larger area being dug resulting in a larger affect and a higher initial level of mortality due to reburial and the intensely physical action of being placed though the dredging machine (Beukema, 1995; Hall & Harding, 1997; Ferns *et al.*, 2000). As with target species, if bait digging is a continuous process occurring throughout and over several years then the non-target macrofauna community will not recover to pre-dig levels (Brown & Wilson, 1997; Cowie *et al.*, 2000; de Boer & Prims, 2002; Griffiths *et al.*, 2006).

6.1.5 Sediment conditions

There has been considerable work on the consequence of bait digging on the macrofauna community as a whole, including both target and non-target species. However, the effect on the actual sediment environment or the sediment ecosystem as a whole receives very little consideration, indeed Hall & Harding (1997) list three areas of concern related to bait digging without mentioning the impact on the sediment environment.

The sediment ecosystem is highly complex (Reise, 2002) and only assessing the affect of bait digging on the macrofauna (e.g. Brown & Wilson, 1997; Ferns *et al.*, 2000; Griffiths *et al.*, 2006) may fail to identify impacts on the ecosystem as a whole. These impacts may have potentially important environmental and economic consequences

for the system with many sediment properties such as stability and productivity influencing species abundance and behaviour. Both surface and subsurface sediment stability may be affected by the disturbance of bait digging, potentially leading to a loss of sediment from the overall system or a reduction in ability of the sediment system to sequester nutrients and pollutants (Gerbersdorf *et al.*, 2007). The productivity of an intertidal system that supports the ecosystem is largely related to the abundance of the microphytobenthos that inhabit the very upper sediments. Disturbing the sediment surface may disrupt the microphytobenthic community and lead to a loss in productivity, with potentially serious implications for organisms higher in the food chain.

Some studies of disturbance on intertidal flats have been more comprehensive in measuring sediment properties (e.g. Dornie *et al.*, 2003a, b) but these studies did not replicate the nature of the bait digging disturbance. Although the methods of disturbance are sometimes similar, the nature of bait digging disturbance is unique in that it selectively removes some species from the ecosystem and as such the results from the disturbance experiments should not be used to represent the effect of bait digging on the sediment ecosystem. Therefore the current study will replicate the physical disturbance and biological consequences of bait digging and assess the impact on the sediment ecosystem as a whole, focusing upon sediment stability as a property of the ecosystem of particular environmental importance.

6.1.6 Effect on adjacent sediments

Experiments into bait digging or sediment disturbance are often based on comparing dug or disturbed plots with control areas. Such plots or sites are usually separated by a set “buffer” distance to avoid any effect of the disturbance treatment on adjacent plots or the control area (Thrush *et al.*, 1996; Cowie *et al.*, 2000) Setting this “buffer” distance implies that the effect of digging or disturbance may not be contained within the actual treated area but may spread into the adjacent sediment. The cause of these more widespread consequences are unclear but could include contamination of the adjacent sediment with disturbed sediment, or a change in the macrofauna community based on horizontal migration of animals to recolonise the disturbed sediment. While this buffer between plots is obviously important in the context of such experiments the principles behind the assumption of its importance have not been studied. This

has particular importance for work on bait digging, where the disturbed or dug area may account for a large proportion of the whole system. If recolonisation by macrofauna is largely based on horizontal migration of individual organisms then the population of a species across the entire system may decrease after extensive bait digging operations, having effectively been diluted by the bait digging. Additionally, if adjacent sediments are acting as sources for larval or juvenile recolonisation then their relative health needs to be considered. Therefore, in considering an acceptable level of bait digging for a system, the consequences of digging needs to be assessed on both the dug and undug areas of sediment.

6.1.7 Main hypotheses for the influence of bait digging on sediment habitats:

- H₁: The disturbance of bait digging will reduce the both the surface sediment stability and the subsurface shear strength.
- H₂: Disturbance from bait digging will reduce the compaction of the sediment, decreasing dry bulk density and increasing water concentration.
- H₃: Surface productivity by MPB will be reduced by bait digging activity.
- H₄: The effect of the disturbance from bait digging on the sediment properties will not be long lasting. With measured sediment properties including water concentration, dry bulk density, organic and carbohydrate concentrations will return to levels found in the control plots within three months.
- H₅: Bait digging activity will decrease the abundance and biodiversity of the macrofaunal assemblages.
- H₆: Bait digging will lower the density of *A. marina* within the dug areas but will not lead to a reduction in the abundance in adjacent sediments.

6.2 Methods

6.2.1 Location

Although the original proposal was to study the impact of bait diggers on Tayport Sands the experimental section of the work was performed on the nearby Eden Estuary (see Chapter 2). This was for two reasons related to the commercial bait digging that occurs on Tayport Sands. Initially a non-impacted area was required for

the experiment, given the limited knowledge of the bait digging operations in Tayport Sands it was not possible to identify such an area that would also represent a habitat where bait digging would occur. In addition to this, the risk of having the experiment dug up by the commercial bait diggers during their activity was considered too great when a similar habitat without this inherent risk was available.

6.2.2 Experimental design

Nine plots measuring 5m by 5m were established on the 28th March 2007, three of each for control, single dig and multiple dig treatments (Labelled C, S and M respectively). Plots were positioned randomly in an area of high *A. marina* density on the upper intertidal zone of the south shore of the Eden Estuary. All plots were sampled on the 29th March for pre-disturbance measurements (referred to as the pre-dig date), single and multiple dig plots were then dug up two days later, and multiple dig plots again two and a half months later on the 11th June. Digging the plots involved using spades to turn over the sediment to a depth of about 30cm and *A. marina* were removed by hand. A reasonable effort was placed into this, to replicate an efficient bait digging operation, but this did not guarantee that all *A. marina* were removed. Collected *A. marina* were relocated >200m from the experiment. Measurements were subsequently taken on the 4th and 25th April and 21st June, providing data for 1 week, 1 month and 3 months after the initial digging disturbance.

6.2.3 Sampling and measurements

6.2.3.1 Sediment properties

Contact cores (n=5) were taken from each plot on each date, from which the water, organic, carbohydrate, and chlorophyll *a* concentrations were subsequently obtained (see chapter 2).

6.2.3.2 Macrofauna

105mm diameter (86.6cm²) sediment cores (n=3) were taken randomly from within each plot on all four sample dates. These were sieved through a 500µm sieve with the remaining macrofauna preserved in 10% formaldehyde and dyed with Rose Bengal to aid identification. Species were identified to as high level as possible. Measurements of species number, individual number and diversity (measured through the Shannon-Weaver diversity index, H') were obtained from each sample.

6.2.3.3 *A. marina* counts

A. marina casts were taken as a non-destructive proxy measurement of *A. marina* density using a 0.25m² quadrat positioned randomly inside each plot (n=5). Counts were also taken 0.5, 1.5 and 3.0m from the edge of each plot to determine if there was any change in population in the vicinity of the disturbance (n=5).

6.2.3.4 Sediment stability

Sediment stability was measured with the CSM set to Sand 7 after initial tests of stability. The chamber was positioned randomly within each plot (n=5) and the critical erosion threshold measured as a 10% drop in transmission (Tolhurst *et al.*, 1999) and expressed as stagnation pressure (Vardy *et al.*, 2007).

Subsurface sediment shear strength was measured by the Shear Vane at depths of 5 and 15cm (n=5), although due to equipment failure no measurements were taken on the 3 month sample date.

6.2.4 Statistical analysis

6.2.4.1 Sample groupings and amalgamation

Measurements of the sediment properties (n=5) or macrofauna community (n=3) from each plot on each date were averaged to give a single value representing the plot. These values were then used for subsequent statistical analysis. Additionally, the individual measurements of sediment properties from each plot at each date were used to calculate a coefficient of variation (c.v. value = average for the samples divided by their standard deviation) for each plot as a measure of the heterogeneity of the plot which was then used in subsequent analysis.

6.2.4.2 Univariate analysis methods

Analysis of individual sediment properties, c.v. values, macrofauna community properties and individual species counts were performed through a Two-way ANOVA between treatment and date, with an effect of digging expected to be shown through a significant interaction between the two. Prior to analysis all results were tested for assumptions of normality required for parametric tests (Zar, 1998)

6.2.4.3 Multivariate analysis methods

All multivariate analysis was performed using PRIMER 6.0 software package (Clarke & Warwick, 2001). Prior to analysis all macrofauna abundances were square root transformed to prevent the analysis being biased towards dominant species. The macrofauna communities from each plot and date were compared using the Bray-Curtis similarity index (S) with the results used to construct an n-MDS ordination showing treatment and date. A two-way ANOSIM was used to compare differences between treatment and date for all samples while additional one way ANOSIMs were performed between treatments on each sample date. SIMPER analysis was performed to identify how different the communities from each treatment were on each date and which, if any, species accounted for the majority of the variation. The abundances of these species were then analysed through univariate methods to determine if their abundance significantly differed between treatments.

The sediment environment was analysed using a similar procedure, with all measurements normalised to allow direct comparisons, no transformation was required. The sediment environment from each plot on each date was compared and quantified through measurement of the euclidean distance between each. A PCA and ANOSIM analysis were used to determine if any change in sediment environment occurred between treatments or dates.

6.3 Results

6.3.1 Sediment stability

6.3.1.1 Surface sediment stability

There was no change in surface sediment stability with treatment, although there was a change over the experiment with time (Table 6.1; Fig. 6.1). This may be due to the drop in stability after the pre dig samples but as this occurs on all three treatments it can not be associated with the digging disturbance.

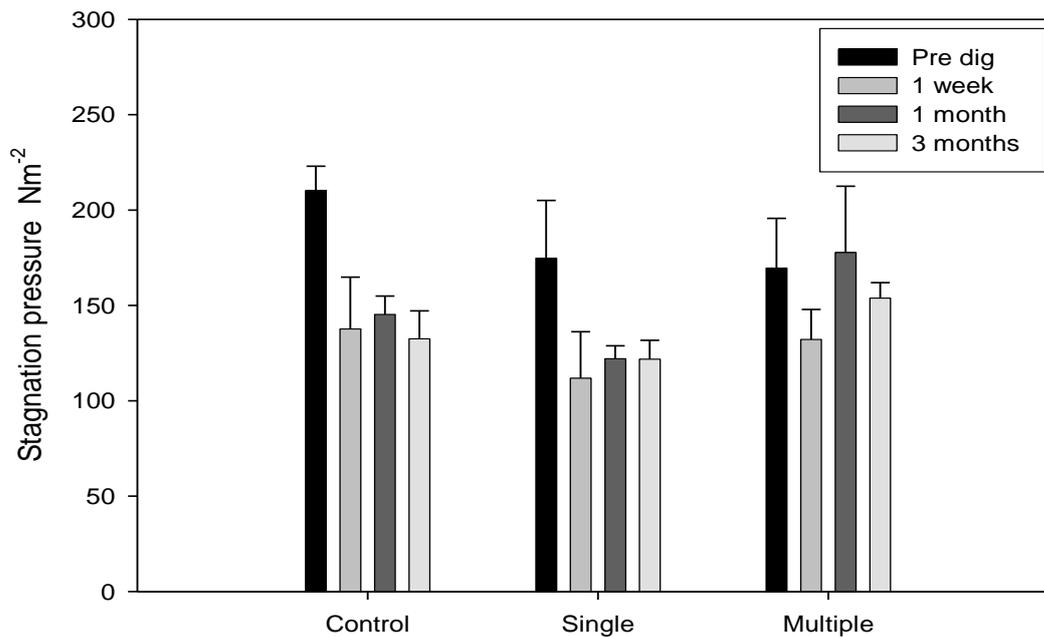


Figure 6.1 Surface sediment stability from the 3 different digging treatments (control, single and multiple digs) over the 3 months of the experiment. Bait digging occurred after the Pre dig samples on both single and multiple dig plots, while bait digging occurred again after 2 months on the multiple dig plots.

Table 6.1 Two-Way ANOVA between treatment and date on measurements of surface sediment stability.

	d.f.	F value	<i>p</i> value
Treatment	2, 24	1.94	0.165
Date	3, 24	4.57	0.011
Interaction	6, 24	0.71	0.643

6.3.1.2 Subsurface sediment shear strength

There was a change in sub surface sediment shear strength between treatments and dates at a 5cm depth in addition to an interaction effect (Table 6.2; Fig. 6.2). There was a drop in shear strength in both of the dug treatments after the digging disturbance while shear strength in control plots slightly increased over time. Deeper sediments (15cm depth) had higher shear strength than the shallow sediments but did not differ in relation to treatment or date.

6.3.2 Sediment properties

6.3.2.1 Individual sediment properties

Across all sediment properties the only changes that occurred were a treatment effect in dry bulk density and a change in carbohydrate concentration between dates (Table 6.3; Figs 6.3 to 6.7). Neither of these changes corresponded to the digging disturbances. The changes in dry bulk density between treatments probably occurred because of higher values on the control plots throughout the experiment, including the pre dig dates (Fig. 6.4) and the date effect in carbohydrate concentration was probably due to the low values found on the pre dig date (Fig 6.6).

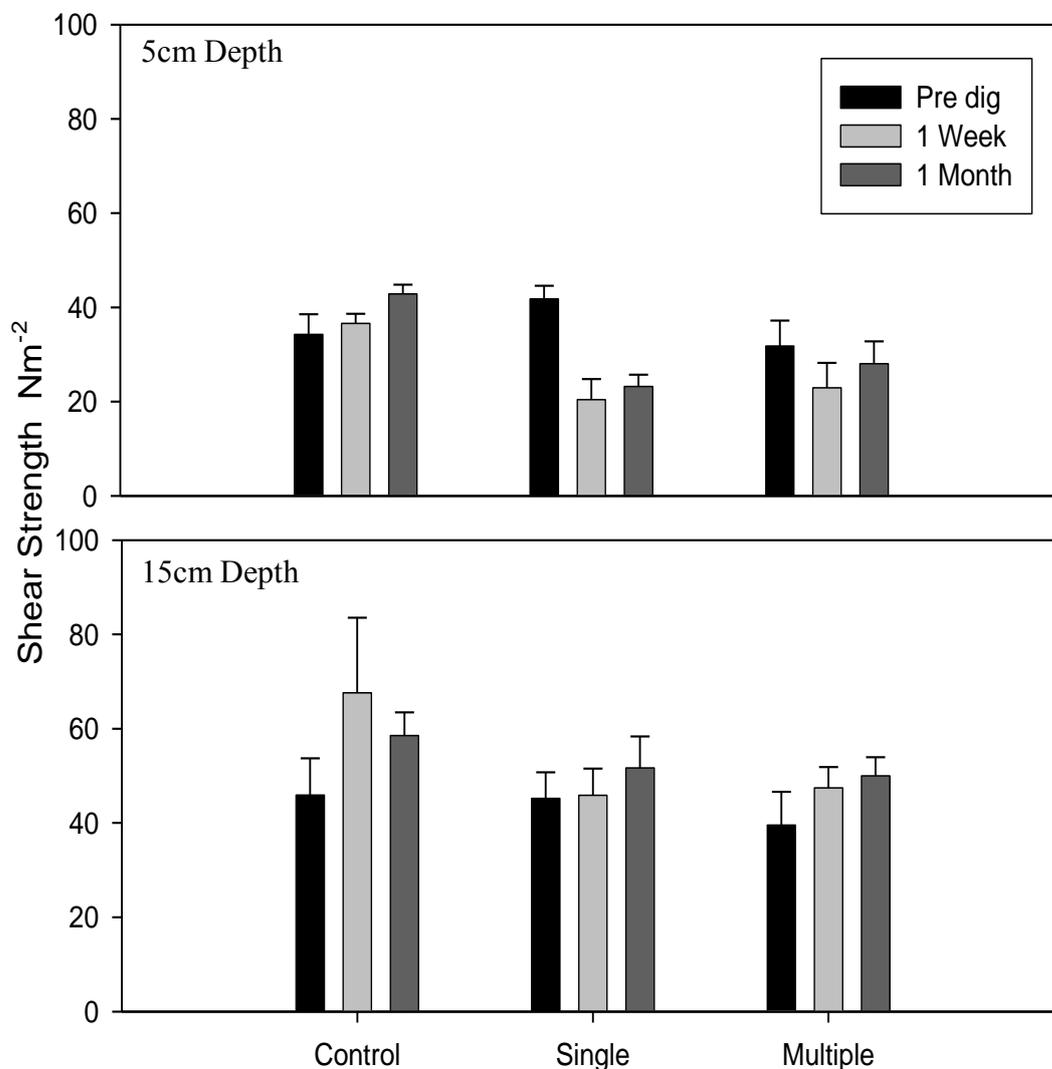


Figure 6.2 Sub surface sediment shear strength of the 3 treatments from the first three sample dates (Pre dig, 1 week and 1 month). 3 Month samples could not be taken due to equipment failure.

Table 6.2 Two-Way ANOVAs from two depths of sediment shear strength. Comparisons made between digging treatment and sample date.

	d.f.	F value	p value
5cm Depth			
Treatment	2, 24	6.35	0.008
Date	3, 24	4.21	0.032
Interaction	6, 24	3.55	0.026
15cm Depth			
Treatment	2, 24	2.00	0.165
Date	3, 24	1.68	0.214
Interaction	6, 24	0.52	0.722

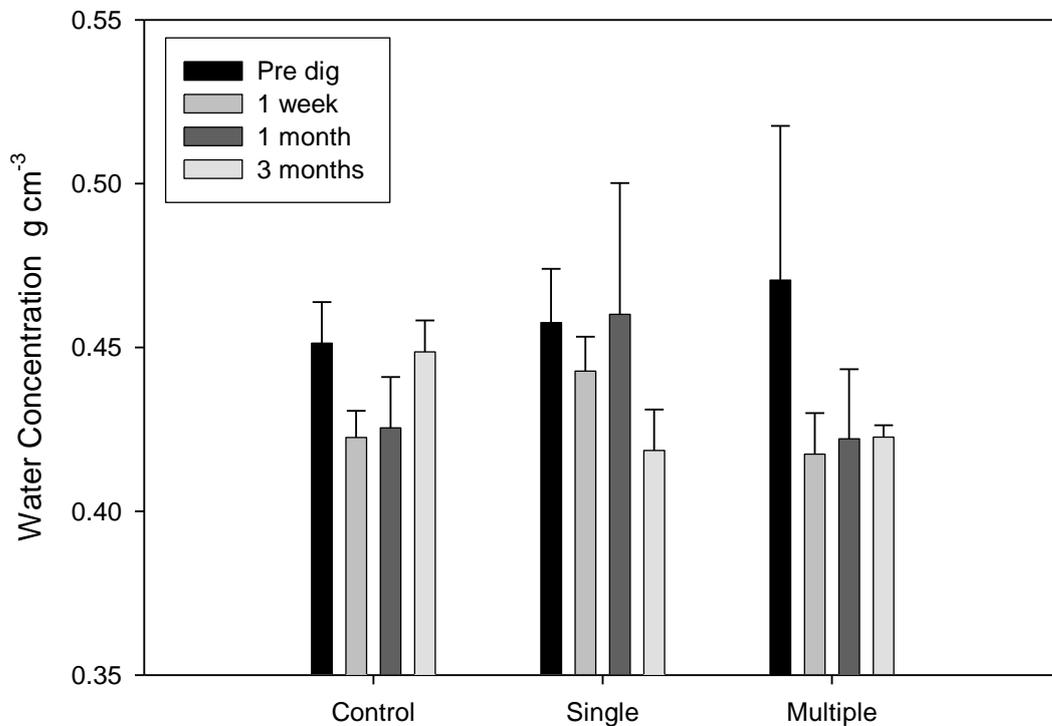


Figure 6.3 Water concentration of surface sediments taken from experimental plots subjected to three different digging disturbances. Bait digging occurred after the pre dig sample date on the single and multiple plots with a second bait digging event after 2 months on the multiple plots.

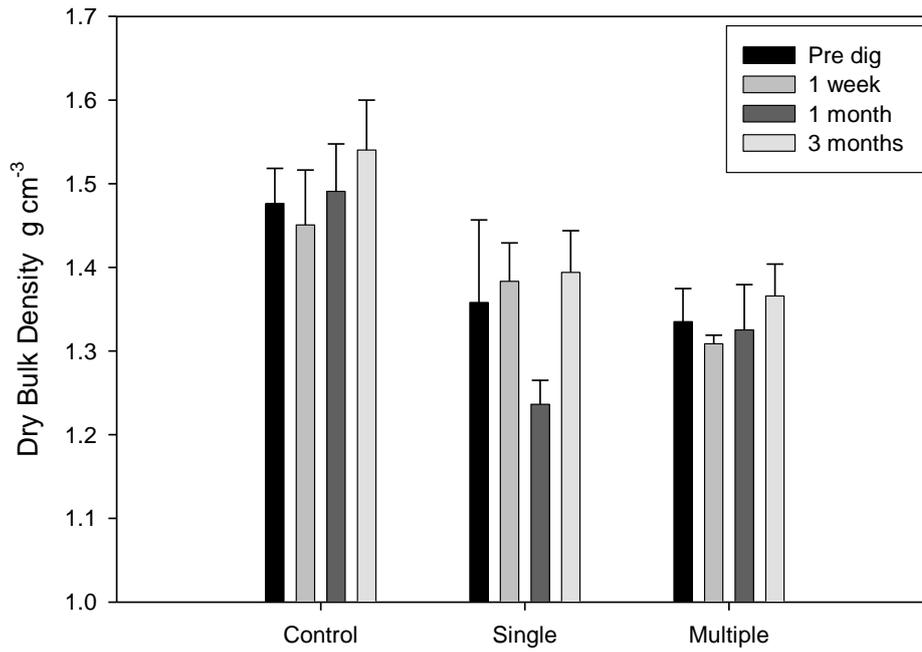


Figure 6.4 Dry bulk density of surface sediments taken from experimental plots subjected to three different digging disturbances.

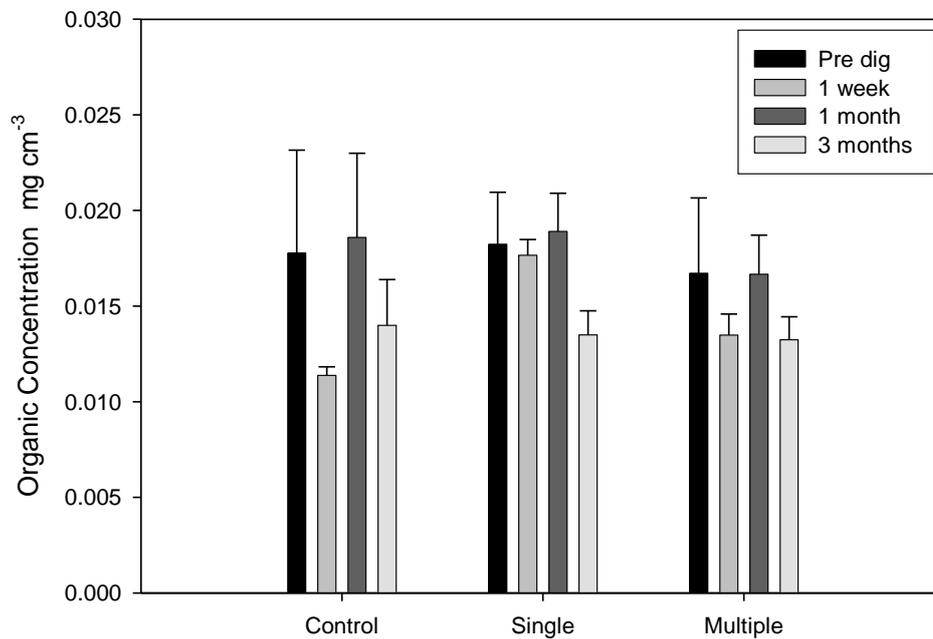


Figure 6.5 Organic concentration of surface sediments taken from experimental plots subjected to three different digging disturbances.

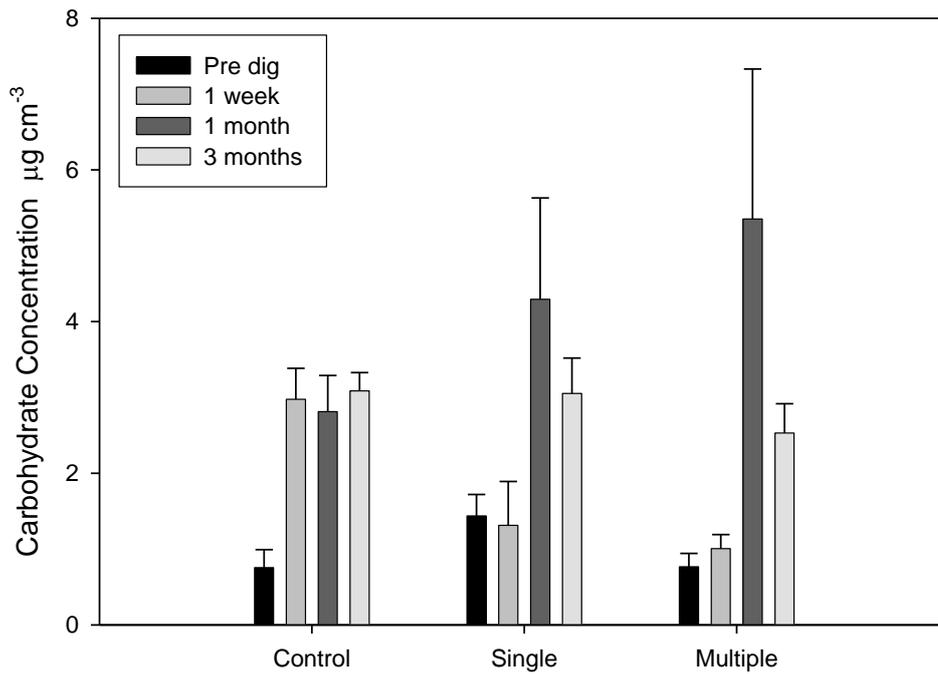


Figure 6.6 Colloidal carbohydrate concentration of surface sediments taken from experimental plots subjected to three different digging disturbances.

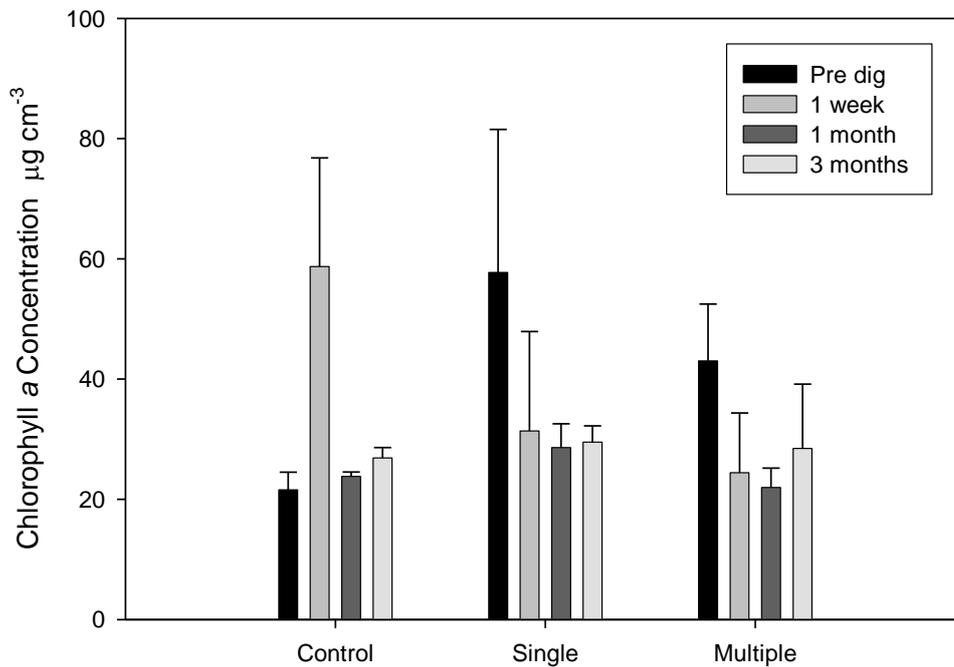


Figure 6.7 Chlorophyll a concentration of surface sediments taken from experimental plots subjected to three different digging disturbances.

Table 6.3 Two-Way ANOVAs of several sediment properties. Comparisons made between disturbance treatment and sample date.

	d.f.	F value	<i>p</i> value
Water Conc.			
Treatment	2, 24	0.30	0.742
Date	3, 24	1.42	0.262
Interaction	6, 24	0.61	0.719
Dry Bulk Density			
Treatment	2, 24	10.80	<0.001
Date	3, 24	1.23	0.320
Interaction	6, 24	0.67	0.677
Organic conc.			
Treatment	2, 24	0.62	0.544
Date	3, 24	2.11	0.126
Interaction	6, 24	0.34	0.907
Carbohydrate conc.			
Treatment	2, 24	0.03	0.972
Date	3, 24	9.71	<0.001
Interaction	6, 24	1.69	0.166
Chlorophyll <i>a</i> conc.			
Treatment	2, 24	0.21	0.814
Date	3, 24	2.03	0.137
Interaction	6, 24	0.95	0.479

6.3.2.2 Overall sediment environment

There was no overall change in the sediment environment over the experiment or in any comparison of treatments within dates, with the exception of control and multiple dug plots in the 1 week samples (Table 6.4; Fig. 6.8). Equally there was no change in the sediment environment between sample dates.

Table 6.4 Comparisons of the sediment environments using one and two way ANOSIM (p values given in brackets). All measurements from the whole of the experiment were placed into a two way ANOSIM between treatment and date. The sediment environment from the three treatments were compared within each sample date using a one way ANOSIM (highest possible level of significance equals 0.1 for one way ANOSIMs in each date).

	Entire experiment	Pre dig	1 Week	1 Month	3 Months
Global R					
Treatment	0.047 (0.315)	-0.111 (0.732)	0.317 (0.075)	0.045 (0.364)	-0.062 (0.629)
Date	0.11 (0.051)				
Pairwise tests					
CvS	0.194 (0.095)	0.074 (0.5)	0.296 (0.2)	0.259 (0.3)	0.148 (0.4)
CvM	0.176 (0.099)	-0.074 (0.7)	0.593 (0.1)	-0.074 (0.8)	0.259 (0.2)
SvM	-0.185 (0.883)	-0.333 (1.0)	0.148 (0.4)	-0.074 (0.5)	-0.481 (1.0)

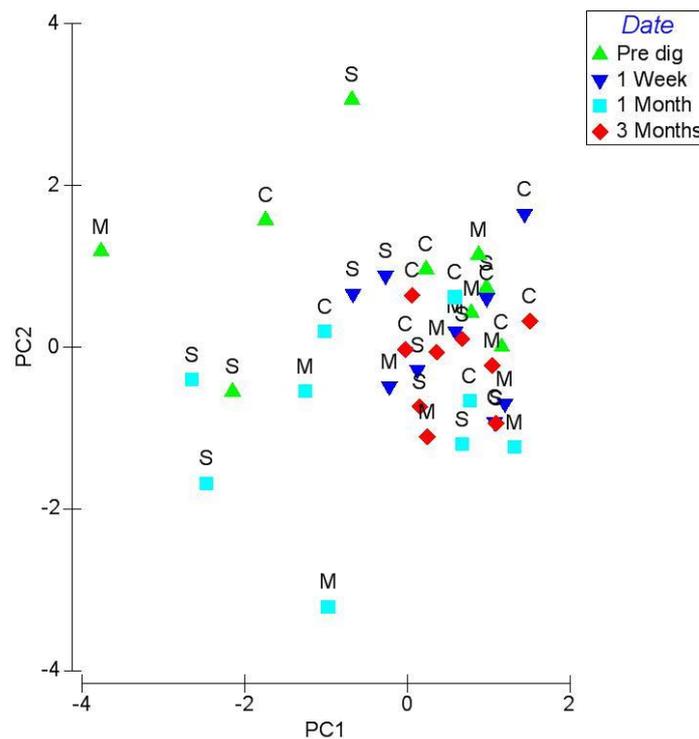


Figure 6.8 PCA of overall sediment environment with plots identified by treatment and sample date. Letters identify treatments; Control plots (C), Single dig plots (S) and multiple dug plots (M).

6.3.2.3 Heterogeneity of the sediment properties between treatments

The only change in the heterogeneity of the plots occurred in the surface sediment stability between dates, with the variation in stability increasing from pre dig samples to 1 month and then declining again by the 3 month samples. This occurred on all treatments and cannot be associated with the digging disturbance (Table 6.5).

6.3.4 Macrofauna community

6.3.4.1 Macrofauna community properties

Bait digging treatment or sample date had no effect on the number of species within the plots, however the number of organisms did change with sample date (Table 6.6; Fig. 6.9). This is probably due to the low numbers found on all treatments on the 3 months sample date, although there is a pattern of lower numbers after 1 week on the dug plots, however this is not identified as either a treatment or interaction effect. There was an interaction between treatment and sample date for the diversity of the communities but this does not relate to the digging disturbances.

Table 6.6 Comparisons of the numbers of species, individual organisms and community diversity on each plot over the duration of the experiment. Two way ANOVA between treatment and sample date.

	d.f.	F value	p value
Number of species			
Treatment	2, 24	3.25	0.056
Date	3, 24	1.96	0.146
Interaction	6, 24	1.44	0.243
Number of organisms			
Treatment	2, 24	3.15	0.061
Date	3, 24	3.51	0.031
Interaction	6, 24	0.29	0.933
Diversity (H')			
Treatment	2, 24	0.55	0.584
Date	3, 24	2.07	0.130
Interaction	6, 24	2.57	0.046

Table 6.5 Average coefficient of variation (c.v.) values with standard errors (se) and ANOVAs between treatment and date for sediment stability and sediment properties.

	Pre dig	1 Week	1 Month	3 Months		d.f.	F value	p value	
Stagnation pressure									
Control									
Average	0.17	0.27	0.38	0.29	Treatment	2, 24	0.60	0.555	
Se	0.0022	0.032	0.044	0.042		Date	3, 24	13.32	<0.001
						Interaction	6, 24	1.41	0.252
Single									
Average	0.16	0.22	0.04	0.24					
Se	0.014	0.038	0.045	0.042					
Multiple									
Average	0.14	0.18	0.50	0.24					
Se	0.011	0.065	0.115	0.028					
Water conc.									
Control									
Average	0.04	0.07	0.09	0.12	Treatment	2, 24	0.56	0.576	
Se	0.001	0.018	0.044	0.030		Date	3, 24	2.01	0.140
						Interaction	6, 24	1.13	0.374
Single									
Average	0.08	0.07	0.13	0.10					
Se	0.017	0.021	0.030	0.030					
Multiple									
Average	0.12	0.06	0.14	0.07					
Se	0.032	0.012	0.052	0.002					
Dry Bulk Density									
Control									
Average	0.05	0.08	0.08	0.11	Treatment	2, 24	1.67	0.210	
se	0.009	0.006	0.005	0.017		Date	3, 24	1.98	0.144
						Interaction	6, 24	1.34	0.277
Single									
Average	0.09	0.07	0.14	0.14					
se	0.043	0.019	0.001	0.017					
Multiple									
Average	0.01	0.08	0.11	0.07					
se	0.029	0.023	0.011	0.022					

Table 6.5 cont.

	Pre dig	1 Week	1 Month	3 Months		d.f	F value	<i>p</i> value	
Organic conc.									
Control									
Average	0.54	0.14	0.48	0.40	Treatment	2, 24	2.53	0.101	
se	0.272	0.044	0.284	0.199		Date	3, 24	0.73	0.545
						Interaction	6, 24	0.75	0.618
Single									
Average	0.13	0.14	0.23	0.13					
se	0.013	0.024	0.026	0.017					
Multiple									
Average	0.43	0.32	0.18	0.24					
se	0.196	0.082	0.021	0.026					
Carbohydrate conc.									
Control									
Average	0.78	1.05	0.59	0.39	Treatment	2, 24	0.87	0.432	
se	0.309	0.248	0.121	0.101		Date	3, 24	1.89	0.158
						Interaction	6, 24	1.25	0.316
Single									
Average	0.90	0.66	0.65	0.61					
se	0.334	0.107	0.041	0.052					
Multiple									
Average	0.60	1.13	0.84	0.84					
se	0.168	0.225	0.057	0.066					
Chlorophyll <i>a</i> conc.									
Control									
Average	0.24	1.04	0.13	0.36	Treatment	2, 24	0.21	0.814	
se	0.10	0.41	0.04	0.08		Date	3, 24	2.03	0.137
						Interaction	6, 24	0.95	0.479
Single									
Average	0.87	0.68	0.30	0.31					
se	0.38	0.44	0.07	0.10					
Multiple									
Average	0.71	0.55	0.37	0.58					
se	0.34	0.21	0.03	0.35					

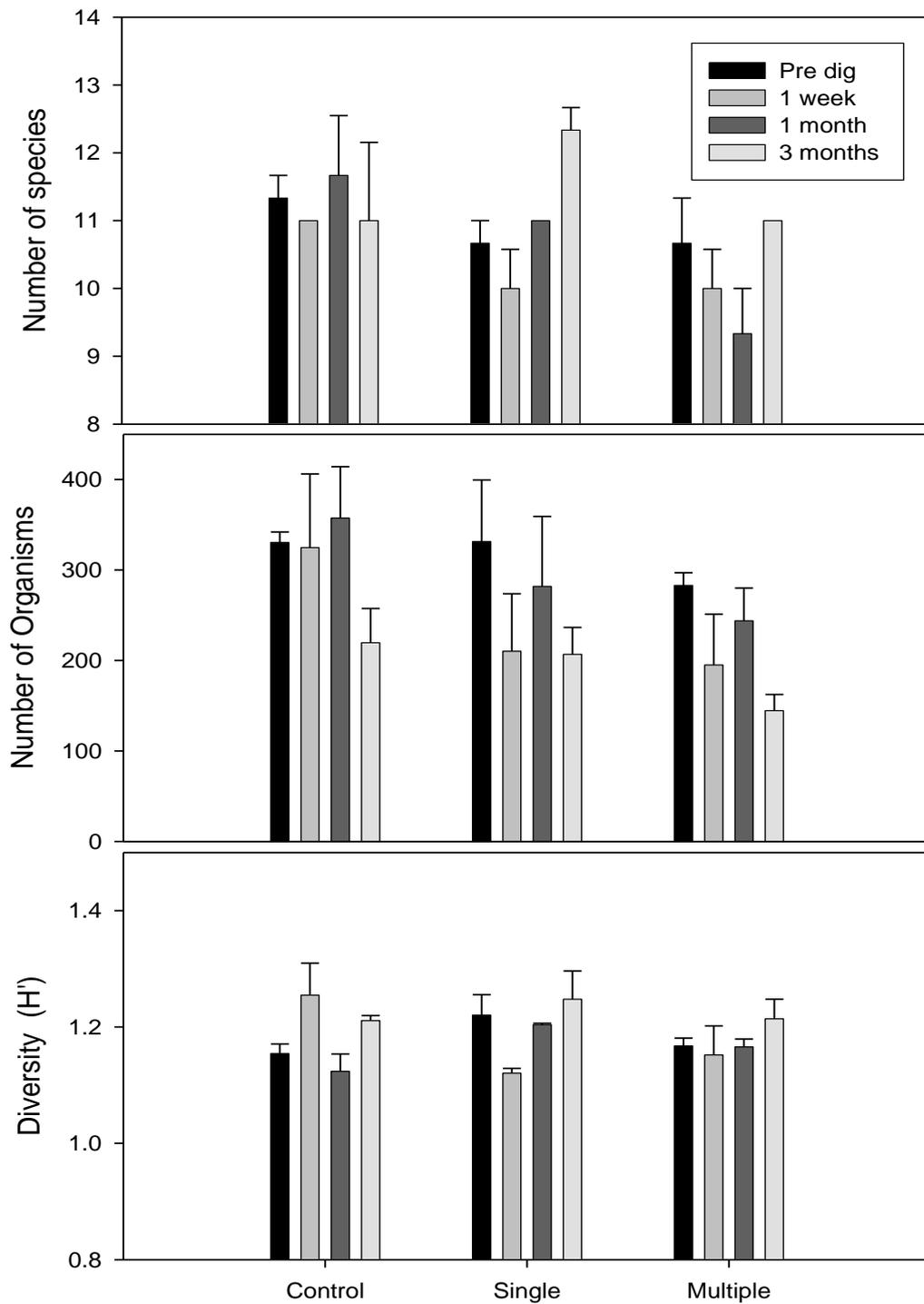


Figure 6.9 Number of species, individual organisms and community diversity (measured through the Shannon Weaver diversity index (H')) from the three treatments over the 3 month duration of the experiment. Bait digging occurred after the pre dig sample date on the single and multiple plots and again after 2 months on the multiple plots.

6.3.4.2 Macrofauna community composition

There was a small change in community composition between the communities in the control and multiple dig plots over the whole experiment, however, this is not as large as the differences that occur between sample dates regardless of treatment (Table 6.7; Fig 6.10). When analysed within each sample date there was a difference between the communities in the control and multiple dig plots before any disturbance had occurred and then a difference between the multiple dig plots and other treatments after the second digging disturbance on these plots.

Table 6.7 Comparison of macrofauna community composition using one and two way ANOSIM (p values given in brackets). All measurements from the duration of the experiment were placed into a two way ANOSIM between treatment and date. The community composition from the three treatments was compared within each sample date using a one way ANOSIM (highest possible level of significance equals 0.1 for one way ANOSIMs at each date).

	Entire experiment	Pre dig	1 Week	1 Month	3 Months
Global R					
Treatment	0.115 (0.126)	0.292 (0.036)	-0.111 (0.68.2)	0.07 (0.350)	0.21 (0.125)
Date	0.215 (0.006)				
Pairwise tests					
CvS	0.037 (0.322)	0.111 (0.4)	0.000 (0.7)	0.148 (0.3)	-0.111 (0.8)
CvM	0.269 (0.024)	0.63 (0.1)	-0.148 (0.9)	0.296 (0.3)	0.296 (0.1)
SvM	0.093 (0.259)	0.185 (0.2)	-0.111 (0.7)	-0.222 (0.8)	0.519 (0.1)

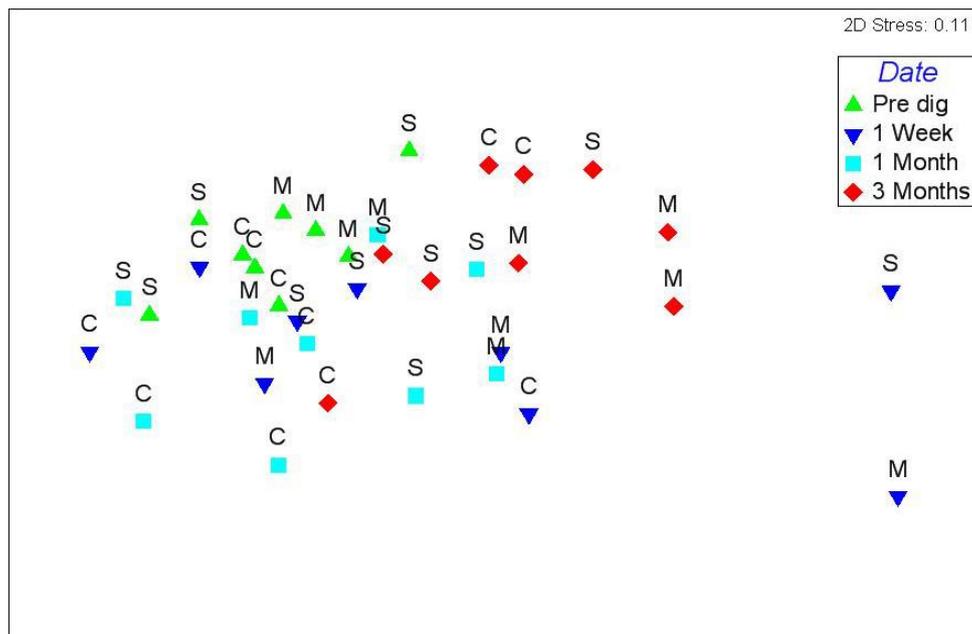


Figure 6.11 n-MDS Ordination of the macrofauna community composition from each plot on all sample dates. Letters identify treatments; control plots (C), single dig plots (S) and multiple dug plots (M).

6.3.4.3 Individual species analysis

SIMPER analysis between community composition identified four species and groups, Nematode *spp.*, *Tubificoides benedii*, Oligiochaete *spp.* and Oweniidae *spp.* that contributed to the majority of the dissimilarity between treatments, however, the overall dissimilarity between communities in different treatments was not very large for any comparison (Table 6.8). Of the four species identified as contributing most of the differences between communities, only *T. benedii* showed a change in abundance between dates with a drop in abundance for each sample date, a pattern that was found in all treatments so could not be attributed to the digging disturbance (Table 6.9).

Table 6.8 Average dissimilarity of macrofauna communities between treatments on each date obtained through SIMPER analysis of community composition.

	Pre-dig	1 Week	1 Month	3 Months
C v S	12.01	23.24	16.56	13.71
C v M	9.7	22.61	15.79	16.22
S v M	12.15	21.77	15.29	15.07

Table 6.9 Two-way ANOVA between treatment and date on the abundances of four species identified through SIMPER analysis as contributing to the majority of dissimilarity between the three digging treatments.

	Pre dig	1 Week	1 Month	3 Months		d.f.	F value	p value	
Nematode spp.									
Control									
Average	98.11	118.44	184.22	71.00	Treatment	2, 24	2.99	0.051	
se	6.45	36.18	46.03	30.06		Date	3, 24	3.65	0.041
						Interaction	6, 24	0.56	0.759
Single									
Average	93.44	65.89	107.78	59.44					
se	39.94	27.21	33.43	19.89					
Multiple									
Average	67.11	69.44	84.56	47.22					
se	2.35	23.94	13.95	4.77					
<i>T. benedii</i>									
Control									
Average	187.00	153.33	133.44	112.78	Treatment	2, 24	5.78	0.004	
se	16.98	42.34	14.56	5.39		Date	3, 24	1.28	0.297
						Interaction	6, 24	0.24	0.958
Single									
Average	184.22	120	132.44	112.78					
se	27.94	31.59	37.98	7.77					
Multiple									
Average	173.11	101.22	126.89	76.78					
se	12.31	25.89	23.87	12.73					
Oligiochaete spp.									
Control									
Average	24.44	27.22	19.78	19.67	Treatment	2, 24	2.33	0.100	
se	3.44	9.59	2.51	3.76		Date	3, 24	0.53	0.596
						Interaction	6, 24	0.71	0.644
Single									
Average	25.11	9.33	23.22	14.89					
se	5.81	1.35	4.16	2.12					
Multiple									
Average	21.00	8.78	15.78	9.44					
se	5.00	2.66	3.78	2.61					

Table 6.9 cont.

	Pre dig	1 Week	1 Month	3 Months		d.f.	F value	p value
Oweniidae								
<i>spp.</i>								
Control								
Average	3.33	5.67	4.22	0.33	Treatment	2, 24	2.33	0.100
se	0.69	2.67	0.99	0.19	Date	3, 24	0.53	0.596
					Interaction	6, 24	0.71	0.644
Single								
Average	7.11	2.22	5.22	0.89				
se	2.72	1.60	3.93	0.22				
Multiple								
Average	2.22	3.44	3.44	0.67				
se	0.48	3.28	1.44	0.33				

6.3.5 *A. marina* density

6.3.5.1 *A. marina* density inside the plots

Inside the plots there were changes in *A. marina* density with date and an interaction effect. However, the change in density with date was not due to a reduction in *A. marina* numbers in treated plots after bait digging, but rather a failure of numbers to increase in these plots after the disturbance in relation to the increase found in the control plots. The different pattern in changing *A. marina* abundances between treatments probably accounts for the interaction effect (Table 6.10; Fig. 6.11).

Table 6.10 ANOVA comparison of treatment and date on *A. marina* density inside the treatment plots.

	d.f.	F value	p value
Date	3, 24	18.05	<0.001
Treatment	2, 24	1.34	0.286
Interaction	6, 24	2.75	0.035

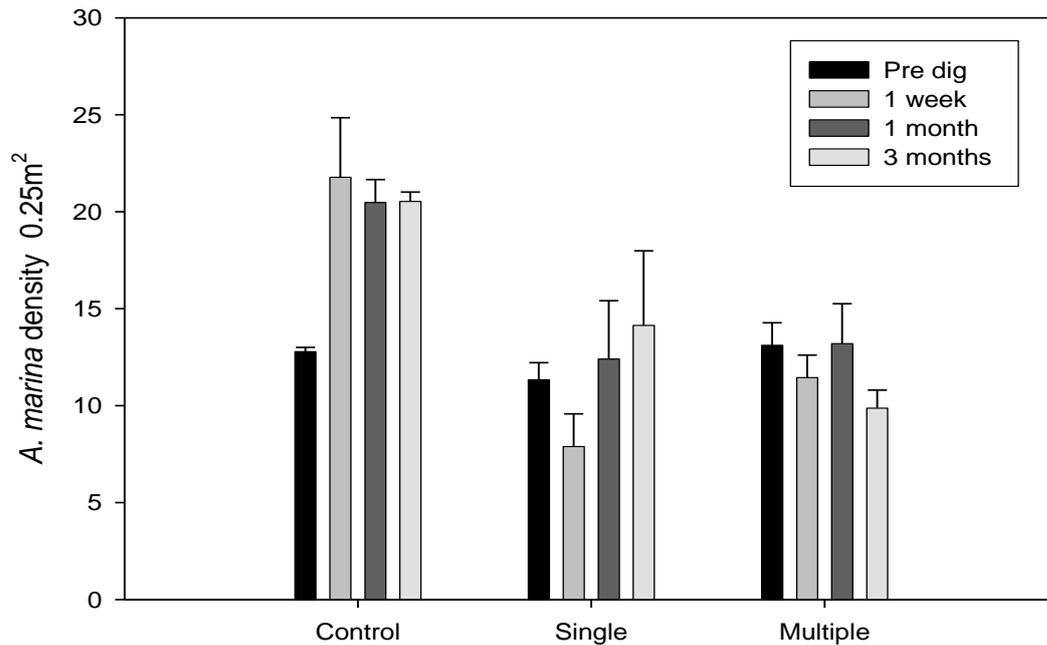


Fig 6.11 *A. marina* cast density inside the treatment plots over the 3 month duration of the experiment. Digging occurred after the pre dig sample date on the single and multiple plots with a second digging disturbance after 2 months on the multiple plots.

Table 6.12 Changes in *A. marina* density in the adjacent area to the edge of the plots compared through a two-way ANOVA between date and distance from the edge.

	d.f.	F value	p value
Control			
Date	3, 32	7.79	<0.001
Distance	3, 32	0.79	0.509
Interaction	9, 32	0.24	0.985
Single			
Date	3, 24	9.64	<0.001
Distance	3, 24	0.55	0.584
Interaction	6, 24	0.14	0.989
Multiple			
Date	3, 24	223.55	<0.001
Distance	3, 24	0.01	0.911
Interaction	6, 24	0.49	0.810

6.3.5.2 *A. marina* density outside the plots

Outside the plots there was a change in the density of *A. marina* with date on all treatments, but no distance or interaction effect (Table 6.12; Fig. 6.12). The change with date is most probably due to the low density of *A. marina* found at all distances on the pre dig sample date.

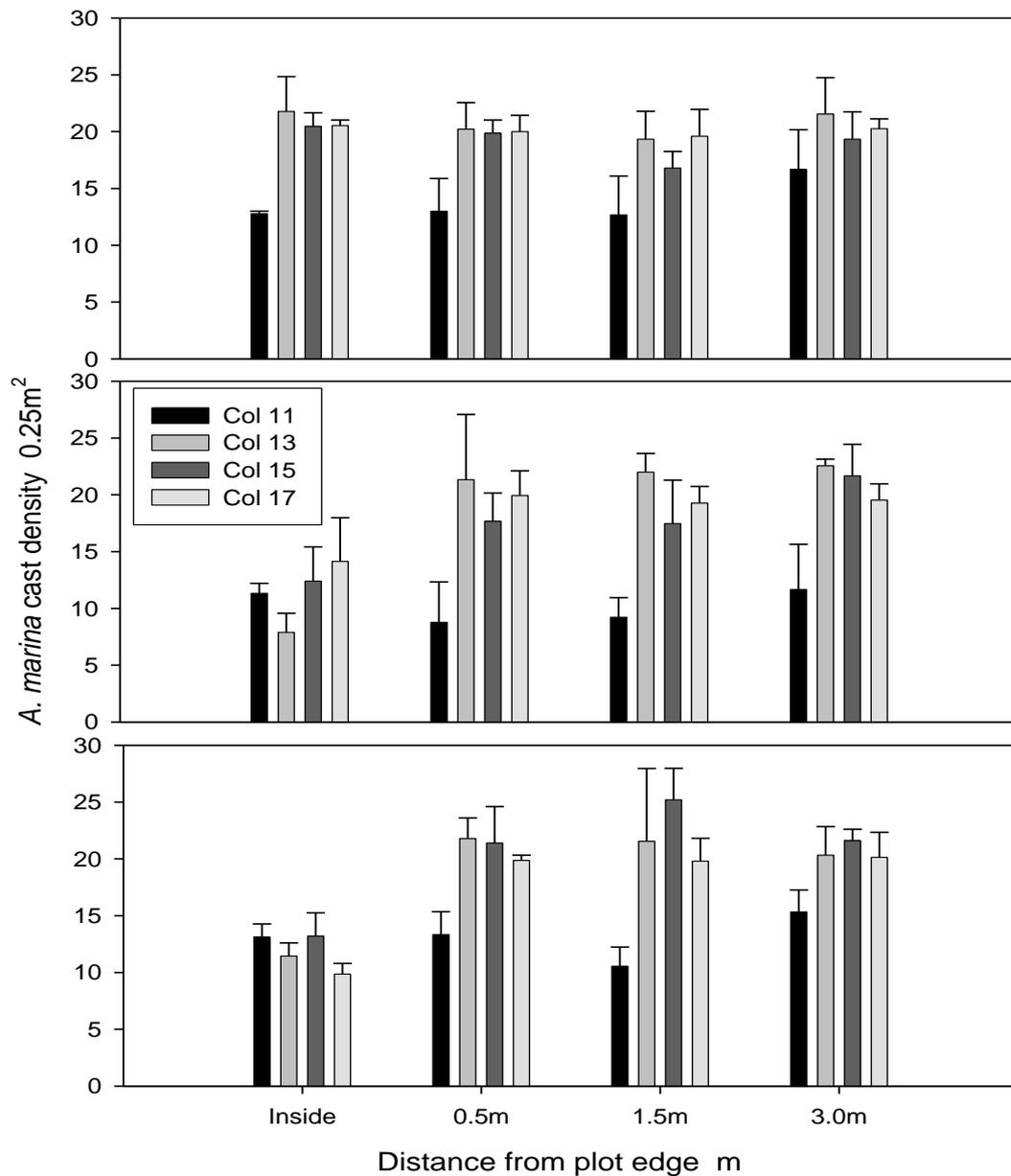


Fig. 6.12 *A. marina* density at three distances from the edge of the treatment plots.

6.4 Discussion

6.4.1 Assessment of hypotheses

- H₁: The disturbance of bait digging will reduce the both the surface sediment stability and the subsurface shear strength.
- H₂: Disturbance from bait digging will reduce the compaction of the sediment, decreasing dry bulk density and increasing water concentration.
- H₃: Surface productivity by MPB will be reduced by bait digging activity.
- H₄: The effect of the disturbance from bait digging on the sediment properties will not be long lasting. With measured sediment properties including water concentration, dry bulk density, organic and carbohydrate concentrations will return to levels found in the control plots within three months.
- H₅: Bait digging activity will decrease the abundance and biodiversity of the macrofaunal assemblages.
- H₆: Bait digging will lower the density of *A. marina* within the dug areas but will not lead to a reduction in the abundance in adjacent sediments.

H₁; sediment surface stability was unaffected by the bait digging disturbance as was the shear strength of the deeper (15cm) sediments, while there was a drop in shear strength in shallower (5cm) sediments. Therefore the majority of H₁ is rejected, except for the shallow sediments.

H₂; There was no change in the compaction of the sediment, measured through water concentration and sediment dry bulk density. H₂ is therefore rejected.

H₃; MPB abundance, measured through chlorophyll *a* concentration, was constant throughout the experiment and therefore productivity of the sediment ecosystem appears to be unaffected by the disturbance of bait digging. H₃ is therefore rejected.

H₄; The predicted time frame for recovery of 3 months was deemed irrelevant by the lack of any measurement of disturbance on the sediment environment. Or, recovery occurred within the first week between disturbance and the first sampling date and therefore recovery could be very quick although this can not be confirmed by this experiment. Subsequently H₄ can be rejected.

H₅; There was no change in the macrofauna community and as a result H₅ must be rejected.

H₆; The predicted change in *A. marina* density both inside and outside the dug plots did occur, with a reduction inside and no change in density in the adjacent sediment. Therefore H₆ is accepted.

6.4.2 Effects on sediment stability

Bait digging for *A. marina* did not result in a change in surface stability. This was unexpected and contradicted the expected impact (H₁), since disturbance, both biotic and abiotic is commonly associated with a reduction in sediment stability (Underwood & Paterson, 1993; Thrush *et al.*, 1996; Black & Paterson, 1999; Cadée, 2001; Reise, 2002; Widdows & Brinsley, 2002; Black *et al.*, 2002). However, it is possible that the unexpectedly large change in stability in the control plots between pre dig and 1 week sampling may be masking a small change in stability in the single and multiple plots after the digging disturbance. It is also possible that the recovery from the impact was very rapid, within the single week between the first measurements. This possibility cannot be excluded given the sampling methodology and should be considered in future research.

The impact of bait digging on the sub surface sediment was to reduce its shear strength at a depth of 5cm but this effect was not continued into deeper sediment. Sediment shear strength is closely related to bed compaction and it is likely that after being dug the upper sediments did not return to their previous levels of compaction while the lower sediments did. It would have been very interesting to see how these results changed after 3 months but this was not possible.

6.4.3 Impact on the sediment surface environment

Neither individual sediment properties nor the overall sediment environment was changed by the digging. Equally no change in the heterogeneity of the sample plots occurred after disturbance. However, there was a high level of variation within and between treatments and plots. It was expected that the disturbance caused by bait digging would reduce the compaction of the sediment particles and increase the water concentration (H₂), however this was not found, with the dry bulk density and water

concentration remaining unchanged. It is possible that this is a result of sampling the surface sediments which are subjected to constant disturbance through direct exposure to tidal submergence. In deeper sediments the sediment particles may have become less compacted by the digging disturbance and did not return to levels equal to the control plots as quickly as the surface sediments, this may account for the reduction in the shear strength found at 5cm depth, although without measurements of water concentration or dry bulk density of sub-surface sediments this can only be suggested.

Different results have been found with respect to sediment properties and disturbance in other studies, with either an impact (Kaiser *et al.*, 2001) or no impact recorded (Dernie *et al.*, 2003a, b). Although the nature of the disturbance and the original sediment environment varied between these experiments, possibly accounting for this discrepancy.

The lack of a measured impact in this study may indicate that the area chosen for the experiment may have a faster rate of recovery than predicted and in comparison to other systems and recovery occurred within the first week after disturbance and therefore was not measured. However, this would be a very fast recovery, much faster than the predicted period of three months (H₄), and it is considered more likely that there was only minimal if any impact on the sediment environment from bait digging. Grain size measurements were not taken throughout this experiment, although no change in grain size had been found in previous studies related to bait digging (Kaiser *et al.*, 2001). The area studied in this experiment comprised of muddy sand that is expected to have relatively rapid recovery rates given its usual level of exposure to disturbance (Schratzberger & Warwick, 1998; Schoeman *et al.*, 2000; Ferns *et al.*, 2000). Not accommodating different sediment environments could be considered a failing in the experimental design. However, the experiment was designed to examine the effects of bait digging on *A. marina* and an area which represented their normal habitat choice was selected. Indeed the habitat chosen was probably towards the lower limit of tolerable sediment size in which *A. marina* is found (Riisgard & Banta, 1998) and as such the result probably represent the slowest recovery rate in an *A. marina* dominated environment, and therefore the lack of impact in the sediment environment is probably true for most bait digging disturbance of this nature.

6.4.4 The impact on macrofauna species

The lack of change in macrofauna community composition after the bait digging disturbance was unexpected (H_5) given the results of previous research (Beukema, 1995; Brown & Wilson, 1997; Ferns *et al.*, 2000; Kaiser *et al.*, 2001; Dernie *et al.*, 2003a, b; Zajac & Whitlatch, 2003). Community properties were equally unaffected by the bait digging disturbance, although there appeared to be a general drop in number of individual organisms after the first dig. Again, as with sediment properties, this may be a result of heterogeneity between plots and within the controlled measurements on different dates. The results contradict previous findings of lower diversity and individual organism numbers after digging disturbance (Hall & Harding, 1997; Griffiths *et al.*, 2006). The lack of change was also true of specific species, either those identified as contributing to the variation in the samples or species previously identified as particularly susceptible to bait digging disturbance such as bivalves (Jackson & James, 1979; Beukema, 1995) and polychaetes (Brown & Wilson, 1997; Hinchey *et al.*, 2006).

The lack of change in the macrofauna community that occurred may be a consequence of the habitat chosen for the experiment. Overall sediment grain size will affect the rate of recovery, with sandier sediment environments having a quicker recovery than muddy environments (Schoeman *et al.*, 2000; Ferns *et al.*, 2000). This is probably a result of the increased complexity of the original community within muddy sediment and the inherent level of resistance to disturbance in sandier sediment communities, both consequences of a higher exposure to disturbance events in natural conditions (Schratzberger & Warwick, 1998). As stated previously, the sediment in the experimental plots was predominantly muddy sand, and as such the impact would be expected to be relatively low and recovery fast.

Given that there was no change in the macrofauna community it is probable that the digging disturbance was not fatal for the majority of the individual organisms. It is, however, possible that the plots were quickly recolonised after the first digging event and before the sampling a week later, however the speed of this recovery would imply very rapid changes in community were possible and would contradict the small change in community that occurred over the subsequent three months. Such rapid recolonisation of disturbed sediment is possible in very small areas through migration (Zajac *et al.*, 1998). However, this is usually in areas less than 1m^2 and larger areas

require longer for recovery (Zajac *et al.*, 1998; Zajac & Whitlatch, 2003). In larger plots recovery is achieved through a combination of migration and larval recruitment, the importance of the latter increasing with increasing size of the disturbed area. Relating this, and other experiments on bait digging, to the scale of a commercial bait digging operation is complicated (Ellis *et al.*, 2000) as it is difficult to replicate disturbance on that scale. Within the experimental plots in this study recovery was most likely be through migration, although over a longer time period than that studied. However, a commercial bait digging operation will disturb a larger area and larval recruitment will probably be more important.

6.4.5 Ecosystem assessment

Overall, bait digging for *A. marina* had very little effect on the sediment ecosystem under the present conditions. Although there were small changes in some factors, these did not present themselves as a general trend from which an obvious impact could be observed. Given this lack of change it is not surprising that the stability of the sediment surface was not altered by the digging. The indication that there was a change in subsurface sediment shear strength is interesting and worthy of further study as this may have important consequences on the longer term nature of the sediment system. However, with the limited number of measurements taken of sediment shear strength and the lack of any measurement of additional subsurface sediment properties it is impossible to draw many conclusions.

It is important to note that the digging disturbance was not the only impact on the plots. The removal of a sizable proportion of the *A. marina* population should also be considered an impact, especially given its recognition as an ecosystem engineer (Riisgard & Banta, 1998 (see Chapter 5). This is an approach not taken in some previous studies of bait digging where the sediment was disturbed but the target species not removed (e.g. Brown & Wilson, 1997) or it is not stated if they are removed or not (Jackson & James, 1997; Griffiths *et al.*, 2006). In such studies the process of recovery and recolonisation will undoubtedly be influenced by the continued presence of the target species in abundances that do not replicate those that would be found after a normal bait digging process (Turner *et al.*, 1997). From an ecosystem point of view, this is an important consideration and one that might be used

to question the validity of the results from these studies when assessing the impact of bait digging, especially where measurements of species number and biomass are used.

The removal of a large number of *A. marina* was expected to have consequences on the sediment environment, especially given the volume of sediment it can turnover in such densities as found in this experiment. On this assumption it is possible that the change in subsurface sediment strength would have been different had the existing population been maintained. However, it is equally possible that the remaining numbers of *A. marina* were sufficiently high to compensate for the reduction in its population. Complete removal of *A. marina* can lead to changes in the sediment ecosystem (see Chapter 4). This may highlight a difference between complete removal of *A. marina* and the actions of bait digging where a number of individuals remain. It may be that there is a threshold level in the population of *A. marina* below which they cannot to exert a dominant influence on the sediment environment, below this they could no longer be termed ecosystem engineers. If this is so, then it would appear that the bait digging in this experiment did not lower the *A. marina* population below this threshold value. The ecosystem engineering of *A. marina* has been found to inhibit the activity of several other species, including larval settlement (Flach, 1992; Flach & De Bruin, 1994; Flach & Beukema, 1994; Beukema & Flach, 1995; Retraubun *et al.*, 1996b; Hardege *et al.*, 1998; Flach, 2003; Volkenborn & Reise, 2006), therefore its abundance within a recovering community, especially one where larval recruitment is important has to be considered. The protocols for many studies on disturbance do not accommodate this (Thrush *et al.*, 1996; Beukema *et al.*, 1999; Schoeman *et al.*, 2000) and as such their results may not be applicable to the specific disturbance caused by bait digging. This is similar in principle to the results found by Beukema *et al.*, (1999) where populations of some species in defaunated plots increased above that in the surrounding sediment, a consequence of the release of competitive inhibition from other species that had yet to recover.

6.4.6 The impact on *A. marina* abundance

6.4.6.1 *A. marina* density inside the plots

The removal method used to capture *A. marina* was not intended to be one hundred percent efficient, rather to represent a viable amount of effort per return (McLusky *et al.*, 1983). This was intended to replicate the efforts of a normal bait digging

operation. This caused an unexpected pattern of change in *A. marina* density within the plots; rather than a large drop in numbers there appeared to be a failure for numbers to increase after the digging. This may have two explanations, firstly that the digging disturbance prevented development or colonisation of new *A. marina* which appeared to happen in the control plots from the pre-dig sampling onwards. However, given the gap of only one week between the first two measurements it is considered unlikely that many new individuals would have colonised the plots. Rather it is possible, probably related to environmental conditions, that the cast counts from the pre-dig measurements were universally low on all plots. If this was so then the expected numbers for the pre-dig date are probably more similar to those found on the control plots on all other days. This would also make a drop in abundance on the dug plots after bait digging more pronounced and agree with the predicted drop in density (H_6). Equally the lack of change in density of *A. marina* in the surrounding sediments would contradict a rapid migration of *A. marina* in to the dug plots. If the *A. marina* populations did increase within the plots between pre-dig and 1 week samples then this increase may be expected to be mirrored in the area just outside the plots, or alternatively the density of *A. marina* in the surrounding plots may drop as *A. marina* migrate from this area into the plots.

6.4.6.2 *A. marina* movement from adjacent sediment

There was very little, if any, movement of *A. marina* from adjacent sediment to the plots from which they had been removed. This implies that horizontal migration by *A. marina* into an available habitat is slow, certainly slower than that of the time scale of this experiment. The possibility that this would have been different in a different season cannot be excluded but *A. marina* tends to be most motile during the same period of time covered by this experiment (Retraubun *et al.*, 1996a). It is possible that after the digging the sediment environment was not suitable for recolonisation, however, given the lack of a large change in the measured sediment properties this is considered unlikely. It is also possible that horizontal migration is performed over longer distances than those measured so there was no drop in the local density, but again this is not supported as there is very little evidence of *A. marina* abundance within the plots increasing over time.

The implications for this to bait digging practices are important as it implies that a system can not quickly compensate for a small scale reduction in *A. marina* abundance and equally if a larger area is dug then the resulting recolonisation will probably depend more on larval recruitment than horizontal migration.

Macrofaunal recolonisation is highly dependant upon the area of disturbance (Thrush *et al.*, 1996, Zajac *et al.*, 1998), with larger areas taking longer to recolonise. Relating this to target species recolonisation is difficult as most experimental disturbance plots are relatively small in comparison to a commercial bait digging operation. Equally it is difficult to quantify the area covered by commercial bait diggers, both temporally and spatially. However, given that it is difficult to experimentally replicate the impact of commercial bait diggers extrapolations from smaller scaled experiment must be used. The lack of recolonisation of the dug plots and the lack of change in the adjacent populations indicates that on the scale of this disturbance *A. marina* do not migrate into available sediment over a short time scale. Horizontal migration of adults through tidal currents does happen with *A. marina* although this is a slow process (Beukema *et al.*, 1999). Therefore, once an area of this size has been impacted by bait diggers the *A. marina* population is unlikely to recover quickly through horizontal migration.

Normally, if recolonisation of large fauna (target species) is dependant upon larval recruitment, then abundances may take a year to recover and biomass will take several years (Beukema *et al.*, 1999). However, in the case of *A. marina* this may be different as juvenile *A. marina* tend to inhabit different areas to adults on an intertidal area and then migrate into suitable adult areas when they are a suitable stage of development (Flach, 1992; Hardege *et al.*, 1998). Therefore *A. marina* recolonisation does not rely on larval recruitment as such, rather recruitment of large juveniles or young adults from other areas, and as such the lag in the recovery of biomass may not be so pronounced.

6.4.7 Impact of bait digging on Tayport Sands

It would appear that bait digging in this sort of habitat has minimal impact on the non-target macrofaunal community or sediment environment although the target species will obviously be affected. However, based on the scale of this experiment this would

require considerably more research to be able to state this with confidence. Of particular interest is the consistency in the sediment environment and the subsequent lack of impact on the sediment stability. In terms of conservation this has to be considered a good thing as a reduction in stability could result in a loss of sediment that could become substantial given the volume of commercial bait digging activity in the area. Although predicted to be lowered (H_3), the lack of impact on the chlorophyll *a* concentration must also be considered as positive as this can be taken as a proxy measurement of productivity and therefore a reduction caused by disturbance could potentially have severe implications for the food web that they support.

6.4.8 Important conservation species

Of particular relevance to Tayport Sands is the impact of bait digging on the conversationally important species which include common and grey seals and many bird species.

Seals

In general seal diets are based upon fish and larger animals that they catch in deeper waters and not species (target and non target) that are affected by bait digging. Haul out sites such as the Tayport Sands are used primarily for resting and rearing young seals and not feeding. While it is possible that reductions in the macrofauna community caused by bait digging may have implications for the food web that supports the prey fish of seals, this is considered unlikely to be a significant effect and has not been studied. Of more importance to the seal population would be the immediate disturbance caused by the bait diggers to the seals during the act of digging. Disturbance by humans can distress seals at haul out sites (Stevens & Boness, 2003; Cassini *et al.*, 2004) although occasional disturbance is unlikely to result in long-term effects the seals (Engelhard *et al.*, 2002) and as such bait digging should not be very detrimental to seal populations.

Birds

Intertidal sediment systems are highly important for both migratory and permanent resident species of birds. The diet of these birds is mostly based upon the macrofauna community within the sediment, either specifically on species collected by bait diggers (e.g. the oystercatcher (*Haematopus ostralegus*) and cockles) or more

opportunistic species that feed within the macrofauna (e.g. the redshank (*Tringa tetanus*), curlews (*Numenius arquata*) and gulls). As the food source of these birds may be directly affected by bait digging the consequences are more immediate than those of marine mammals. Indeed, the very action of bait digging provides a very short term benefit to birds as the disturbed sediment contains numerous dead and disturbed animals on the surface unable to rapidly burrow for protection (Ferns *et al.*, 2000). However the short term benefits of bait digging are far outweighed by the subsequent months of lower species number and abundance offering a lower volume of food for the bird population (Norris *et al.*, 1998; Ferns *et al.*, 2000).

6.5 Conclusions

The results of this study indicate that the combined effects of bait digging disturbance and removal of a proportion of the ecosystem engineering *A. marina* on the upper southern shore of the Eden Estuary does not have a large impact the sediment environment or macrofauna community. These minimal changes in the ecosystem do not result in a change in the surface sediment stability. Despite these findings it is difficult to draw many conclusions based upon the impact of bait digging on a larger scale and therefore extending these results to the scale of Tayport Sands has to be done with caution. It is suggested that a larger project established over a longer time period would be beneficial to this field of research.

Chapter Seven

General Discussion

7.1 The CSM

In conjunction with the work performed by Vardy *et al.*, (2007) the operation of the CSM has been heavily revised (Chapter 3). Although the work performed did not result in any substantial advancements in the design of the CSM, it was in these attempts that the problems with the original operation procedures were identified and rectified. Of particular importance was the discovery of differences in firing efficiency between machines and the resulting method of calibration that accounts for these differences and allows different machines to be compared. This was a considerable improvement in methodology which allows the validation of future CSM studies.

A major consequence of this work was the disproving of the calibration method proposed by Tolhurst *et al.* (1999) where the vertical jet pressure was converted into horizontal shear stress, and the subsequent lack of a workable equivalent based upon the new calibration method. Expressing the critical erosion threshold of sediments as a property of the vertical jet (e.g. firing pressure or stagnation pressure) is often considered a failing of the CSM, given that it does not replicate natural erosion conditions (Widdows *et al.*, 2007). Instead the critical erosion threshold of sediments is commonly given as horizontal shear stress values, often derived from flume based experiments where horizontal flow is increased until sediment is eroded (e.g. Widdows *et al.*, 1998, Widdows *et al.*, 2000b; Bale *et al.*, 2006). However, the assumption that this is a better replication of natural conditions is arguable. Horizontal flow in estuaries is predominantly a product of tidal and river currents which are rarely fast and do not regularly surpass the critical erosion threshold (Bell *et al.*, 1997; Le Hir *et al.*, 2000; Janssen-Stelder, 2000). Indeed, the prevailing hydrodynamic conditions of tidal and river flow in a depositional sediment system

must not regularly surpass the critical erosion threshold as the system would be completely eroded. The majority of sediment is instead eroded by extreme events, usually storms, which produce wind driven waves and currents. The type of flow produced by these waves is highly turbulent, consisting of vortexes and eddies. Although the vertical jet of the CSM does not replicate these conditions, it is important to consider these processes before dismissing the validity of the CSM because it does not produce horizontal laminar flow.

The CSM is a unique and highly useful machine in the study of sediment stability, and in particular the biogenic influences that effect it. The capabilities of the CSM for rapid test replication, ease of use and accuracy in deployment give it a specialised position in the study of intertidal sediment stability. With the detailed study into its operation completed the CSM should remain a vital tool in the study of sediment stability.

7.2 Sediment stability as an ecosystem function/service

As stated by Reise (2002), the ecosystem of intertidal sediment systems is comprised of a mixture of “complex habitat mediated interaction webs”. Indeed, with the combination of a highly mobile and unstable substratum and the tidal change between marine and terrestrial conditions, with their related stresses, there can be few other environments that have such inherent complexity. This means that any process occurring within the ecosystem is unlikely to be occurring in isolation. Instead it will most likely be influencing, and influenced by, numerous other biotic and abiotic processes. This can be seen in some of the studies of primary production (Forster *et al.*, 2006), nutrient flux (Biles *et al.*, 2002; Biles *et al.*, 2003) and bioturbation (Emmerson *et al.*, 2001; Solan *et al.*, 2004) in sediment systems where changes in the species composition change the resulting process. These studies have been based on the principles of ecosystem functioning and services (Chapin *et al.*, 1997). An ecosystem service being effectively an ecosystem function of which the results have tangible benefits for humans.

Much of this research has been based on the supposition that ecosystem function is closely related to changes in the biodiversity (Emmerson *et al.*, 2001; Solan *et al.*, 2004), and in particular the loss in biodiversity that may occur as a result species extinction (local, regional and global scales) through climate change and habitat destruction (Balvanera *et al.*, 2006).

The overall stability of the intertidal sediment is a combination of numerous interacting biotic and abiotic factors. The product of which can be described as the (de)stabilising influence of the system on the sediment stability when compared to the level of stability that would exist based on purely physical properties. This will then have an influence on the erosion and transport of sediment within the system. An ecosystem function is a process within an ecosystem that involves the transport, transfer or metabolism of materials (Chapin *et al.*, 1997). While the term materials is principally aimed at chemicals and nutrients in processes of metabolism it could equally be applied to the movement of sediment in the same fashion. When considered like this the (de)stabilising influence of the ecosystem on overall sediment stability could be taken as an ecosystem function. Indeed, in the context of coastal erosion, sediment deposition and pollution sequestering it could possibly be considered an ecosystem service.

Sediment stability as an ecosystem function is explored in Chapters 5 and 6. In Chapter 5 the effect of removing the ecosystem engineering polychaete worm *A. marina* is minimal, with small localised changes in the sediment environment and macrofauna community, which do not lead to a change in the sediment stability. While in Chapter 6 the disturbance to biotic and abiotic variables caused by bait digging on the sediment ecosystem was also found to be minimal, resulting in no impact on sediment stability. Within these studies a change in either a biotic or abiotic influence could be considered in isolation, however within intertidal sediments such changes rarely occur in isolation, and consequently the combination of the biotic and abiotic influences is required to be considered together as a change in the ecosystem.

7.3 An ecosystem approach to sediment stability

The study of sediment stability in intertidal ecosystems has been approached in a multidisciplinary fashion by many authors, relating a plethora of biotic and abiotic factors to an overall product of sediment stability. Some studies have focused on large scale surveys of intertidal systems with attempts made to explain correlations between changes in the ecosystem and the sediment stability (e.g. Christie *et al.*, 1999; Paterson *et al.*, 2000; Defew *et al.*, 2002; Defew *et al.*, 2003; Friend *et al.*, 2003a) (Table 1.1). However, invariably in these studies the inherent complexities of the influences into stability prevent their isolation or quantification. Alternately, studies have been experimentally based and have manipulated one or more variables, usually a single species, and observed the subsequent change in stability (e.g. Andersen *et al.*, 2002; Orvian, 2006; Orvian *et al.*, 2006; Ciutat *et al.*, 2007) (Table 1.1). This approach has identified the methods in which many species interact with the sediment and the resulting consequence on stability. However, within such experiments the ecosystem is grossly simplified, and the experiment is usually based on changing the abundance of a single species in mesocosms without considerations of the impact this may have on other species, and consequently what additional effect this may have on sediment stability. This means that the two types of study are ultimately testing different things, on one hand the ecosystem is being surveyed, while on the other the component parts of the ecosystem are being manipulated. To progress the understanding of species influences on sediment stability the two approaches need to be combined, using large scale *in situ* surveys based on experimental manipulations similar to that in Chapter 5. Obviously, this is a very difficult proposition with large scale *in situ* experiments complicated and expensive. However, by manipulating the ecosystem it is possible that the true importance of individual species or sediment properties will become more apparent in a situation which is more likely to yield valid interpretations.

This approach will allow the study of sediment stability to be applied to realistic situations, such as the approach taken in Chapter 6 where the effect on the ecosystem was related to measurements of the sediment stability. That no change in surface stability was found is an indication that there may have been other dominant influences on sediment stability that were not effected by the disturbance.

Unfortunately, such an approach is difficult to establish, especially in an experimental manipulation rather than a survey, and results are often complicated. For example in both Chapters 5 and 6 results from different replicates of the same treatment are sometimes contradictory. Additionally the large variation in control measurements in both experiments made isolating a cause and effect related to the treatments highly difficult. Overcoming this problem would probably require more replicates of each treatment, however, the limiting factor in such experiments is usually the logistics and expense, and adding more replicates is often not feasible.

An ecosystem approach to sediment stability is therefore a complex undertaking but is one that could be simplified into laboratory based experiments with a sufficient understanding of the original system. Given that intertidal sediment macrofaunal systems are already associated with studies of ecosystem functioning with changing diversity due to their low diversity (Biles *et al.*, 2002; Biles *et al.*, 2003; Kaiser *et al.*, 2005) this should be possible. Using macrofaunal functional groups could simplify the systems (Bolam *et al.*, 2002), allowing laboratory mesocosms to be used to assess changes in the stability related to controlled changes in biotic or abiotic factors (e.g. Biles *et al.*, 2002; Biles *et al.*, 2003). As it is unlikely that a single variable will ever be sufficient to understand and predict sediment stability the need to develop such studies becomes more important as the threat to intertidal sediment systems from climate change and human development increases.

7.4 Working underwater

Taking the study of sediment stability into the submerged period of the tidal cycle encountered many problems (Chapter 4), with results that suggested several significant effects without much success in their explanation. In particular, the unexpected increase in stability that occurs with submersion. Despite this, the importance of understanding stability during submersion means the research should be continued. With no established method of measuring submerged sediment properties *in situ* a large amount of the planned research to be taken from the field into the laboratory. This was performed with a certain level of success but can only be seen as an initial step. In particular the importance of accurately replicating *in situ*

conditions, especially flow and submersion duration, was highlighted. This will allow more comprehensive and accurate laboratory based studies of sediment stability, with the results being more applicable to the natural environment.

In both the laboratory and *in situ* the methods devised to collect and sample sediment and measure stability will hopefully be useful for future studies, especially considering the importance of progressing stability studies from the exposed sediment into the submerged zone.

7.5 What determines sediment stability?

The research and application of understanding sediment stability is based on scales and interpretation of one scale to another. An area of study where this has been performed extensively is in the effect of MPB and EPS in stabilising sediment surfaces. The understanding of how this stabilisation occurs starts on the level of chemical composition, with the chemical properties of EPS. This is developed further to the microstructure of sediment, with electron microscopy used to capture images of single cells, EPS and individual particles bound together (Fig. 7.1). A combination of these levels can be used to understand sediment stabilisation over a biofilm, which can then be extrapolated into larger areas and possibly whole system scales. Despite this accuracy in studying EPS, other areas of research in intertidal sediment stability have not been studied as comprehensively. The composition of sediment, relating to the mix of particles, water, air and additional components (Tolhurst *et al.*, 2005) has not been taken to the microscopic level in intertidal sediments in the same fashion. Although the practice of taking large cores and homogenising the upper 1 or 2 centimetres of sediment is no longer well accepted as a measurement to represent the sediment surface it is probable that sampling the upper 2mm is still at an insufficient resolution. It is suggested that within the distribution and spacing of individual sediment particles and the composition of the related space between particles are probably the main factors that determine sediment stability. If so then this adds further importance to the expression of sediment properties as concentrations which can be compared to the sediment concentration (dry bulk density) rather than content which includes elements of dry bulk density in the calculation. Despite the successful

microscale study of MPB and EPS, research into the composition of sediment on the same scale would be very difficult. The methods of sampling used in this work used two types of core (contact and coarse) which, to some extent, probably both compact the surface sediment on a microscale, preventing very accurate measurements of dry bulk density and water concentrations. The cryolander core (Wiltshire *et al.*, 1997) was designed to overcome this problem, however, on the very small scale it is likely that freezing of a sample will distort sediment particles through the formation of ice crystals. Existing work from freshwater systems has used a gamma ray densitometer to obtain the density of sediment particles on very fine scales, with the results correlating well with changing stability (Lick & McNeil, 2001; McNeil & Lick, 2004; Gerbersdorf *et al.*, 2007) and this may be an area in which further research in both exposed and submerged intertidal sediments could progress.



Figure 7.1. Low Temperature Electron Microscope images of sediment, diatoms and EPS.

7.6 Conclusions

The CSM

- The CSM remains a vital tool for studying intertidal sediment stability, but needs to be calibrated using the procedure devised by Vardy *et al.*, (2007).
- The operation of the CSM must be undertaken with great care, with potential errors easily made if the chamber is deployed incorrectly.

Sediment stability as an ecosystem function

- The stability of intertidal sediment is a hugely complex product of numerous interacting biotic and abiotic factors.
- Understanding the interactions of these factors is as important as identifying the factors in isolation.
- Studying a species in isolation is unlikely to result in an accurate estimation of its influence on the (de)stabilising influence of the ecosystem.

Sediment stability

- Expressing sediment properties as a concentration allowed the matrix of sediment, water and air to be quantified more accurately than the use of content. However, this did not identify a variable that explained changes in sediment stability.
- Studying concentrations on a microscale may allow a more accurate explanation of changing stability.
- Progressing research into sediment stability into the submerged period of the tidal cycle is increasingly important.

References

- Allen, J.L.R. (1985). Principles of physical sedimentology. Cambridge University Press.
- Alyakrinskaya, I.O. (2003). Some ecological features of the Lugworm *Arenicola marina* L., (Annelida, Polychaeta) and its morphological and biochemical adaptations to burrowing. *Biol. Bull.* **30**(4): 411-418.
- Amos, C.L., Bergamasco, A., Umgiesser, G., Cappucci, S., Cloutier, D., DeNat, L., Flindt, M., Bonardi, M. & Cristante, S. (2004). The stability of tidal flats in Venice Lagoon – the results of *in-situ* measurements using two benthic, annular flumes. *J. Marine Syst.* **51**: 211-241.
- Andersen, T.J. (2001). Seasonal variation in erodability of two temperate, microtidal mudflats. *Estuar. Coast. Shelf Sci.*, **53**; 1-12.
- Andersen, T.J. & Pejrup, M., (2001). Suspended sediment transport on a temperate, microtidal mudflat, the Danish Wadden Sea. *Mar. Geol.* **173**: 69-85.
- Andersen, T.J. & Pejrup, M. (2002). Biological mediation of the settling velocity of bed material eroded from an intertidal mudflat, the Danish Wadden Sea. *Estuar. Coast Shelf Sci.* **54**: 737-745.
- Andersen, T.J., Jensen, K.T., Lund-Hansen, L.C., Mourtisen, K.N. & Pejrup, M. (2002). Enhanced erodability of fine-grained marine sediments by *Hydrobia ulvae*. *J. Sea. Res.* **48**: 51-58.
- Andersen, T.J., Lund-Hansen, L.C., Pejrup, M., Jensen, K.T. & Mourtisen, K.N. (2005) Biologically induced differences in erodability and aggregation of subtidal

- and intertidal sediments: a possible cause for seasonal changes in sediment deposition. *J. Marine Syst.* **55**: 123-138.
- Aspden, R.J., Vardy, S., Perkins, R.G., Davidson, I. R., Bates, R. & Paterson, D.M. (2004). The effects of clam fishing on the properties of surface sediment in the lagoon of Venice, Italy. *Hydrol. Earth Syst. Sci.* **8**(2): 160-169.
- Austen, I., Andersen, T.J. & Edelvang, K. (1999). The influence of benthic diatoms and invertebrates on the erodability of an intertidal mudflat, the Danish Wadden Sea. *Estuar. Coast Shelf Sci.* **49**: 99-111.
- Bale, A.J., Widdows, J., Harris, C.B. & Stephens, J.A. (2006). Measurements of the critical erosion threshold of surface sediments along the Tamar estuary using a mini-annular flume. *Cont. Shelf Res.* **26**: 1206-1216.
- Bell, R.G., Hume, T.M., Dolphin, T.J., Green, M.O. & Walters, R.A. (1997). Characterisation of physical environmental factors on an intertidal sandflat, Manukau Harbour, New Zealand. *J. Exp. Mar. Biol. Ecol.* **216**: 11-31.
- Bertness, M.D. (1999). The ecology of Atlantic shorelines. Sinauer Associates, Inc.
- Beukema, J.J. (1995). Long-term effects of mechanical harvesting of lugworms *Arenicola marina* on the zoobenthic community of a tidal flat in the Wadden Sea. *Neth. J. Sea Res.* **33**(2): 219-227.
- Beukema, J.J. & Flach, E.C. (1995). Factors controlling the upper and lower limits of the intertidal distribution of two *Corophium* species in the Wadden Sea. *Mar. Ecol. Prog. Ser.* **125**: 117-126.
- Beukema, J.J., Flach, E.C., Dekker, R. & Starink, M. (1999). A long-term study of the recovery of the macrozoobenthos on large defaunated plots on a tidal flat in the Wadden Sea. *J. Sea Res.* **42**: 235-254.

- Biles, C.L., Paterson, D.M., Ford, R.B., Solan, M. & Raffaelli, D.G. (2002). Bioturbation, ecosystem functioning and community structure. *Hydrol. Earth Syst. Sci.* **6**(6): 999-1005.
- Biles, C.L., Solan, M., Isaksson, I., Paterson, D.M., Emes, C. & Raffaelli, D.G. (2003). Flow modifies the effect of biodiversity on ecosystem functioning: an *in situ* study of estuarine sediments. *J. Exp. Mar. Biol. Ecol.* **285-286**: 165-177.
- Black, K.S., Tolhurst, T.J., Paterson, D.M. & Hagerthey, S.E. (2002a) Working with natural cohesive sediments. *J. Hydrol. Eng.* **128**: 2-8.
- Black, I., Black, K.S. & Tolhurst, T.J. (2002b). The Mk4 60psi cohesive strength meter (CSM), operating manual. *Sediment Services*.
- Blake, R. W. (1979). Exploitation of a natural population of *Arenicola marina* (L.) from the North-east coast of England. *J. Appl. Ecol.* **16**: 663-670.
- Blanchard, G.F., Paterson, D.M., Stal, L.J., Richard, P., Galois, R., Huet, J., Kelly, J., Honeywill, C., de Brouwer, J., Dyer, K., Christie, M. & Seguignes, M. (2000). The effect of geomorphological structures on the potential biostabilisation by microphytobenthos and intertidal flats. *Cont. Shelf Res.* **20**: 1243-1256.
- Blavanera, P., Pfisterer, A.B., Buchmann, N., He, J., Nakashizuka, T., Raffaelli, D. & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecol. Lett.* **9**: 1-11.
- Bolam, S.G. & Fernandes, T.F. (2003). Dense aggregations of *Pygospio elegans* (Claparède): effect on macrofaunal community structure and sediments. *J. Sea Res.* **49**: 171-185.
- Bolam, S.G., Fernandes, T.F. & Huxham, M. (2002). Diversity, biomass, and ecosystem processes in the marine benthos. *Ecol. Monogr.* **72**(4): 599-615.

- Brown, B. & Wilson, W.H. (1997). The role of commercial digging of mudflats as an agent for change of infaunal intertidal populations. *J. Exp. Mar. Biol. Ecol.* **218**: 49-61.
- Brown, A.C. & McLachlan, A. (2002). Sandy shore ecosystems and the threats facing them: some predictions for the year 2025. *Environ. Conserv.* **29**: 62-77.
- Brown, E., Colling, A., Park, D., Phillips, J., Rothery, D. & Wright, J. (1999). Waves, tides and shallow water processes (2nd edition). The Open University.
- Cadée, G.C. (1976). Sediment reworking by *Arenicola marina* on tidal flats in the Dutch Wadden Sea. *Nether. J. Sea Res.* **10**(4): 440-460.
- Cadée, G.C. (1900). Feeding traces and bioturbation by birds on a tidal flat, Dutch Wadden Sea. *Ichnos.* **1**: 23-30.
- Cadée, G.C. (2001). Sediment dynamics by bioturbation animals. In; ecological studies, Ecological comparisons of sedimentary shores. Vol. 151, p127-148.
- Carpenter, S.R. & Kitchell, J.F. (1993). The trophic cascade in lakes. Cambridge Studies in Ecology. Cambridge University Press. pp 385.
- Cassini, M.H. & Szteren, D. (2004). Fence effects on the behavioural responses of South American fur seals to tourist approaches. *J. Ethol.* **22**: 127-133.
- Chapin, F.S., Walker, B.H., Hobbs, R.J., Hooper, D.U., Lawton, J.H., Sala, O.E. & Tilman, D. (1997). Biotic control over the functioning of ecosystems. *Science.* **277**: 500-504.
- Christie, M.C., Dyer, K.R. & Turner, P. (1999). Sediment flux and bed level measurements from a macro tidal mudflat. *Estuar. Coast Shelf Sci.* **49**: 667-688.
- Christie, M.C., Dyer, K.R., Blanchard, G., Cramp, A., Mitchener, H.J. & Paterson, D.M. (2000). Temporal and spatial distributions of moisture and organic contents across a macro-tidal mudflat. *Cont. Shelf Res.* **20**: 1219-1241.

- Ciutat, A., Widdows, J. & Pope, N.D. (2007). Effect of *Cerastoderma edule* density on near-bed hydrodynamics and stability of cohesive muddy sediments. *J. Exp. Mar. Biol. Ecol.* **346**: 114-126.
- Clarke, K. R. & Ainsworth, M. (1993). A method of linking multivariate community structure to environmental variables. *Mar. Ecol. Prog. Ser.* **92**: 205-219.
- Clarke, R.K. & Warwick, R.M. (2001). Changes in marine communities; an approach to statistical analysis and interpretation (2nd edition). Plymouth Marine Laboratory, Plymouth.
- Consalvey, M., Paterson, D.M. & Underwood, G.J.C. (2004). The ups and downs of life in a benthic biofilm: migration of benthic diatoms. *Diatom Res.* **19**(2): 181-202.
- Consalvey, M., Perkins, R.G., Paterson, D.M. & Underwood, G.J.C. (2005) PAM Fluorescence: a beginners guide of benthic diatomist. *Diatom Res.* **20**(1): 1-22.
- Cowie, P.R., Widdicombe, S. & Austen, M.C. (2000). Effects of physical disturbance on an estuarine intertidal community: field and mesocosm results compared. *Mar. Biol.* **136**: 485-495.
- Cundy, A.B., Croudace, I.W., Cearreta, A. & Irabien, M.J. (2003). Reconstructing historical trends in metal input in heavily-disturbed, contaminated estuaries: studies from Bilbao, Southampton Water and Sicily. *Appl. Geochem.* **18**: 311-325.
- Daborn, G.R., Amos, C.L., Brylinsky, M., Christian, H., Drapeau, G., Faas, R.W., Grant, J., Long, B., Paterson, D.M., Perillo, G.M.E. & Piccolo, M.C. (1993). An ecological cascade effect – migratory birds affect stability of intertidal sediments. *Limnol. Oceanogr.* **38**: 225-231.
- Dade, W.B., Davis, J.D., Nichols, P.D., Nowell, A.R.M., Thistle, D., Trexler, M.B. & White, D.C. (1990). Effects of exopolymer adhesion on the entrainment of sand. *Geomicrobiol. J.* **8**: 1-16.

- Davidson, N.C. & Buck, A.L. (1997). An inventory of UK estuaries. Vol 1. Introduction and methodology. Joint Nature Conservation Committee. Vol. 1 of 7.
- De Boer, W.F. & Prins, H.H.T. (2002). The community structure of a tropical intertidal mudflat under human exploitation. *J. Mar. Sci.* **59**: 1237-1247.
- de Brouwer, J.F.C., Bjelic, S., de Deckere, E.M.G.T. & Stal, L.J. (2000). Interplay between biology and sedimentology in a mudflat (Biezelingse Ham, Westerschelde, The Netherlands). *Cont. Shelf Res.* **20**: 1159-1177.
- de Brouwer, J.F.C., de Deckere, E.M.G.T. & Stal, L.J. (2003). Distribution of extracellular carbohydrates in three intertidal mudflats in Western Europe. *Estuar. Coast Shelf Sci.* **56**: 313-324.
- de Brouwer, J.F.C., Wolfstein, K., Ruddy, G.K., Jones, T.E.R. & Stal, L.J. (2005). Biogenic stabilisation of intertidal sediments: The importance of extracellular polymeric substances produced by benthic diatoms. *Microb. Ecol.* **49**: 501-512
- de Deckere, E.M.G.T., van de Koppel, J. & Heip, C.H.R. (2000). The influence of *Corophium volutator* on resuspension. *Hydrobiologia.* **426**: 37-42.
- de Deckere, E.M.G.T., Tolhurst, T.J. & de Brouwer, F.C.J. (2001). Destabilisation of cohesive intertidal sediments by infauna. *Estuar. Coast Shelf Sci.* **53**: 665-669.
- Decho, A.W. (2000). Microbial biofilms in intertidal systems: an overview. *Cont. Shelf Res.* **20**: 1257-1273.
- Defew, E.C., Tolhurst, T.J. & Paterson, D.M. (2002). Site-specific features influence sediment stability of intertidal flats. *Hydrol. Earth Syst. Sci.* **6**(6): 971-981.
- Defew, E.C., Tolhurst, T.J., Paterson, D.M. & Hagerthey, S.E. (2003). Can the stability of intertidal sediments be predicted from proxy measurements? An *in situ* investigation. *Coastal Zone Topics.* **5**: 61-70.

- Dernie, K.M., Kaiser, M.J. & Warwick, R.M. (2003a). Recovery rates of benthic communities following physical disturbance. *J. Anim. Ecol.* **72**: 1043-1056.
- Dernie, K.M., Kaiser, M.J., Richardson, E.A. & Warwick, R.M. (2003a). Recovery of soft sediment communities and habitats following physical disturbance. *J. Exp. Mar. Biol. Ecol.* **285**: 415-434.
- Dyer, K.R. (1973) Estuaries - a physical introduction. Wiley-Interscience.
- Dytham, C. (1999). Choosing and using statistics, a biologist's guide. Blackwell Publishing
- Edgar, P.J., Hursthouse, A.S., Matthews, J.E. & Davies, I.M. (2003). An investigation of geochemical factors controlling the distribution of PCBs in intertidal sediments at a contamination hot spot, the Clyde Estuary, UK. *Appl. Geochem.* **18**: 327-338.
- El Ganaoui, O., Schaaff, E., Boyer, P., Amielh, M., Anseimet, F. & Grenz, C. (2004). The deposition and erosion of cohesive sediments determined by a multi-class model. *Estuar. Coast Shelf Sci.* **60**: 457-475.
- Ellis, J.I., Norkko, A. & Thrush, S.F. (2000). Broad-scale disturbance of intertidal and shallow sublittoral soft-sediment habitats; effects on the benthic macrofauna. *J. Aquat. Ecosys. Stress Rec.* **7**: 57-74.
- Emmerson, M.C., Solan, M., Emes, C., Paterson, D.M. & Raffaelli, D. (2001). Consistent patterns and the idiosyncratic effects of biodiversity in marine ecosystems. *Nature.* **411**: 73-77.
- Fernandes, S. Sobral, P & Costa, M.H. (2006). *Nereis diversicolor* effect on the stability of cohesive intertidal sediments. *Aquat. Ecol.* **40**: 567-579.
- Flach, E.C. (1992). Disturbance of benthic infauna by sediment-reworking activities of the lugworm *Arenicola marina*. *Neth. J. Sea Res.* **30**: 81-89.

- Flach, E.C. (2003). The separate and combined effects of epibenthic predation and presence on macro-infauna on the recruitment success of bivalves in shallow soft-bottom areas on the Swedish west coast. *J. Sea Res.* **49**: 59-67.
- Flach, E.C. & Beukema, J.J. (1994). Density-governing mechanisms in populations of the lugworm *Arenicola marina* on tidal flats. *Mar. Ecol. Prog. Ser.* **115**: 139-149.
- Flach, E.C. & de Bruin, W. (1994). Does the activity of cockles, *Cerastoderma edule* (L.) and lugworms, *Arenicola marina* L., make *Corophium volutator* Pallas more venerable to epibenthic predators: a case of interaction modification? *J. Exp. Mar. Biol. Ecol.* **182**: 265-285.
- Flemming, B.W. & Delafontaine, M.T. (2000). Mass physical properties of muddy intertidal sediment: some applications, misapplications and non-application. *Cont. Shelf Res.* **20**: 1179-1197.
- Ford, R.B. & Paterson, D.M. (2001). Behaviour of *Corophium volutator* in still versus flowing water. *Estuar. Coast Shelf Sci.* **52**: 357-362.
- Forster, R.M., Créach, V., Sabbe, K., Vyverman, W. & Stal, L.J. (2006). Biodiversity-ecosystem function relationship in microphytobenthic diatoms of the Westerschelde Estuary. *Mar. Ecol. Prog. Ser.* **311**: 191-201.
- Friend, P.L., Ciavola, P., Cappucci, S. & Santos, R. (2003a). Bio-dependant bed parameters as a proxy tool for sediment stability in mixed intertidal habitats. *Cont. Shelf Res.* **23**: 1899-1979.
- Friend, P.L., Collins, M.B. & Holligan, P.M. (2003b). Day-Night variation of intertidal flat sediment properties in relation to sediment stability. *Estuar. Coast Shelf Sci.* **58**: 663-675.
- Gerbersdorf, S.U., Jancke, T. & Westrich, B. (2007). Sediment properties for assessing the erosion risk of contaminated riverine sites. *J. Soil Sed.* **7**(1): 25-35.

- Gerdol, V. & Hughes, R.G. (1994). Effect of *Corophium volutator* on the abundance of benthic diatoms, bacteria and sediment stability in two estuaries in southeastern England. *Mar. Ecol. Prog. Ser.* **114**: 109-115.
- Gibbs, R.J., Matthews, M.D. & Ink, D.A. (1971). The relationship between sphere size and settling velocity. *J. Sediment. Petrol.* **41**: 7-18.
- Gleizon, P., Punt, A.G. & Lyons, M.G. (2003). Modeling hydrodynamics and sediment flux within a macrotidal estuary: problems and solutions. *Sci. Total Environ.* **314**: 589-597.
- Graf, G. & Rosenberg, R. (1997). Biosuspension and biodeposition: a review. *J. Marine Syst.* **11**: 269-278.
- Grant, J. & Daborn, G. (1994). The effects of bioturbation on sediment transport on an intertidal mudflat. *Neth. J. sea Res.* **32**(1): 63-72.
- Hagerthey, S.E., Defew, E.C. & Paterson, D.M. (2002). Influence of *Corophium volutator* and *Hydrobia ulvae* on intertidal diatom assemblages under different nutrient and temperature regimes. *Mar. Ecol. Prog. Ser.* **245**: 47-59.
- Hall, S.J. & Harding, M.J.C. (1997). Physical disturbance and marine benthic communities: the effects of mechanical harvesting of cockles on non-target benthic infauna. (1997). *J. Appl. Ecol.* **34**: 497-517.
- Hardege, J.D., Bentley, M.G & Snape, L. (1996). Sediment selection by juvenile *Arenicola marina*. *Mar. Ecol. Prog. Ser.* **166**: 187-195.
- Hauton, C. & Paterson, D.M. (2003). A novel shear vane used to determine the evolution of hydraulic dredge tracks in sub-tidal marine sediments. *Estuar. Coast Shelf Sci.* **57**: 1151-1158.

- Herman, P.M.J., Middelburg, J.J. & Heip, C.H.R. (2001). Benthic community structure and sediment processes in an intertidal flat: results from the ECOFLAT project. *Cont. Shelf Res.* **21**: 2255-2071.
- HIMOM. (2005). Hierarchical Monitoring Methods. European commission fifth framework programme. Contract: EVK3-CT-2001-00052.
- Hinchey, E.K., Schaffner, L.C., Hoar, C.C., Vogt, B.W. & Batte, L.P. (2006). Responses of estuarine benthic invertebrates to sediment burial: the importance of mobility and adaptation. *Hydrobiologia.* **556**: 85-98.
- Holland, A.F., Zingmark, R.G. & Dean, J.M. (1974). Quantitative evidence concerning the stabilisation of sediments by marine benthic diatoms. *Mar. Biol.* **27**: 191-196.
- Honeywill, C. (2001). *In situ* analysis of the biomass and distribution of microphytobenthos. University of St Andrews. Ph.D. thesis.
- Hoskins, D.L., Stancyk, S.E. & Decho, A.W. (2003). Utilisation of algal and bacterial extracellular polymeric secretions (EPS) by the deposit-feeding brittlestar *Amphipholis gracillima* (Echinodermata). *Mar. Ecol. Prog. Ser.* **247**: 93-101
- Houwing, E.J. (1999). Determination of the critical erosion threshold of cohesive sediments on intertidal mudflats along the Dutch Wadden Sea coast. *Estuar. Coast Shelf Sci.* **49**: 545-555.
- Hughes, R.G. & Paramor, O.A.L. (2004). On the loss of saltmarshes in south-east England and methods for their restoration. *J. Appl. Ecol.* **41**: 440-448.
- Huxham, M., Gilpin, L., Mocogni, M. & Harper, S. (2006). Microalgae, macrofauna and sediment stability: an experimental test of a reciprocal relationship. *Mar. Ecol. Prog. Ser.* **310**: 55-63.

- IPCC (2007). Summary for policy makers. In: Climate Change 2007: The physical science basis. Contribution of Working Group 1 to the fourth Assessment Report of the Intergovernmental Panel on Climate Change [Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M. & Miller, H.L. (eds.)]. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.
- Jackson, M.J. & James, R. (1979). The influence of bait digging on cockle, *Cerastoderma edule*, populations in North Norfolk. *J. Appl Ecol.* **16**: 671-679.
- Jago, C.F., Jones, S.E., Latter, R.J., McCandliss, Hearn, M.R. & Howarth, M.J. (2002). Resuspension of benthic fluff by tidal currents in deep stratified waters, northern North Sea. *J. Sea Res.* **48**: 259-269.
- Janssen-Stelder, B. (2000). The effect of different hydrodynamic conditions on the morphodynamics of a tidal mudflat in the Dutch Wadden Sea. *Cont. Shelf Res.* **20**: 1461-1478.
- Jie, H. Zhinan, Z. Zishan, Y. & Widdows, J. (2001). Differences in the benthic-pelagic particle flux (biodeposition and sediment erosion) at intertidal sites with and without clam (*Ruditapes philippinarum*) cultivation in Eastern China. *J. Exp. Mar. Biol. Ecol.* **261**: 245-261.
- Jickells, T.D. & Rae, J.E. (1997). Biogeochemistry of intertidal sediments. Cambridge environmental chemistry series 9. Cambridge university Press.
- Jones, S.E. & Jago, C.F. (1993). *In situ* assessment of modification of sediment properties by burrowing invertebrates. *Mar. Biol.* **115**: 133-142.
- Jones, C.G., Lawton, J.H. & Shachak, M. (1994). Organisms as ecosystem engineers. *Oikos.* **69**(3): 373-386.
- Jones, C.G., Lawton, J.H. & Shachak, M. (1997). Positive and negative effects of organisms as physical ecosystem engineers. *Ecology.* **78**(7): 1946-1957.

- Jonsson, P.R., van Duren, L.A., Amielh, M., Asmus, R., Aspden, R.J., Oliver, F., Pope, N., Precht, E., Sauriau, P.G & Schaaff, E. (2006). Making water flow: a comparison of the hydrodynamic characteristics of 12 different benthic biological flumes. *Aquat. Ecol.* **40**(4): 409-438.
- Kaiser, M.J., Broad, G. & Hall, S.J. (2001). Disturbance of intertidal soft-sediment benthic communities by cockle hand raking. *J. Sea Res.* **45**: 119-130.
- Kaiser, M.J., Attrill, M.J., Jennings, S., Thomas, D.N., Barnes, D.K.A., Brierley, A.S., Polunin, N.V.C., Raffaelli, D.G. & Williams, P.J.L. (2005). *Marine ecology: processes, systems, and impacts*. Oxford University Press.
- Krasnow, L.D. & Taghon, G.L. (1997). Rate of tube building and sediment particle size selection during tube construction by the tanaid crustacean, *Leptochelia dubia*. *Estuaries*. **20**: 534-546.
- Kuhrts, C. Seifert, T. & Fennel, W. (2006). Modeling transport of fluff layer material in the Baltic Sea. *Hydrobiologia*. **554**: 25-30.
- Lansard, B. Grenz, C. Charmasson, S., Schaaff, E. & Pinazo, C. (2006). Potential plutonium remobilisation linked to marine sediment resuspension: first estimates based in flume experiments. *J. Sea Res.* **55**: 74-85.
- Le Hir, P., Roberts, W., Cazaillet, O., Christie, M., Bassoullet, P. & Bacher, C. (2000). Characterisation of intertidal flat hydrodynamics. *Cont. Shelf Res.* **20**: 1433-1459.
- Lee, M. (2001). Coastal defence and the Habitats Directive: predictions of habitat change in England and Wales. *The Geographic J.* **167**(1): 39-56.
- Lelieveld, S.D., Pilditch, C.A. & Green, M.O. (2003). Variation in sediment stability and relation to indicators of microbial abundances in the Okura Estuary, New Zealand. *Estuar. Coast Shelf Sci.* **57**: 123-136.

- Leopold, M.F. (2002). Eiders *Somateria mollissima* scavenging behind a lugworm boat. *J. sea Res.* **47**: 75-82.
- Levington, J. & Kelaher, B. (2004). Opposing organizing forces of deposit-feeding marine communities. *J. Exp. Mar. Biol. Ecol.* **300**: 65-82.
- Lick, W. & McNeil, J. (2001). Effects of sediment bulk properties on erosion rates. *Sci. Total Environ.* **266**: 41-48.
- Lucas, C.H., Widdows, J. & Wall, L. (2003). Relating spatial and temporal variability in sediment chlorophyll a and carbohydrate distribution with erodability of a tidal flat. *Estuaries.* **26**: 885-893.
- Lumborg, U. & Windelin, A. (2003). Hydrography and cohesive sediment modeling: application to the Rømø Dyb tidal area. *J. Marine Syst.* **38**: 287-303.
- Lumberg, U. Andersen, T.J. & Pejrup, M. (2006). The effect of *Hydrobia ulvae* and microphytobenthos on cohesive sediment dynamics on an intertidal mudflat described by means of numerical modeling. *Estuar. Coast Shelf Sci.* **68**: 208-220.
- MacIntyre, H.L., Geilder, G.J. & Miller, D.C. (1996). Microphytobenthos; the ecological role of the “secret garden” of unvegetated, shallow water marine habitats, 1. Distribution, abundance and primary production. *Estuaries.* **19**: 186-201.
- Martin, D., Bertasi, F., Colangelo, M.A., de Vries, M., Frost, M., Hawkins, S.J., Macpherson, E., Moschella, P.S., Satta, M.P., Thompson, R.C. & Ceccherelli, V.U. (2005). Ecological impact of coastal defence structures on sediment and mobile fauna: Evaluating and forecasting consequences of unavoidable modifications of native habitats. *Coast. Eng.* **52**: 1027-1051.

- Mason, C.F., Underwood, G.J.C., Baker, N.R., Davey, P.A., Davidson, I., Hanlon, A., Long, S.P., Oxborough, K., Paterson, D.M. & Watson, A. (2003). The role of herbicides in the erosion of salt marshes in eastern England. *Environ. Pollut.* **122**: 41-49.
- McLusky, D.S., Anderson, F.E. & Wolfe-Murphy, S. (1983). Distribution and population recovery of *Arenicola marina* and other benthic fauna after bait digging. *Mar. Ecol. Prog. Ser.* **11**: 173-179.
- McNeil, J. & Lick, W. (2004). Erosion rates and bulk properties of sediments from the Kalamazoo River. *J. Great Lakes Res.* **30**(3): 407-418.
- Minoura, K. & Osaka, Y. (1992). Sediments and sedimentary processes in Mutsu Bay, Japan – Pelletization as the most important mode in depositing argillaceous sediments. *Mar. Geol.* **103**: 487-502.
- Morgan R.P.C. (1995). Soil erosion and conservation – second edition. Longman Group Limited.
- Mitchell, S.B., Couperthwaite, J.S, West, J.R. & Lawler, D.M. (2003). Measuring sediment exchange rates on an intertidal bank at Blacktoft, Humber Estuary, UK. *Sci. Total Environ.* **314-316**: 535-549.
- Norris, K., Bannister, R.C.A. & Walker, P.W. (2000). Changes in the Oystercatcher *Haematopus ostralegus* wintering in the Burry Inlet in relation to the biomass of cockles *Cerastoderma edule* and its commercial exploitation. *J. Appl. Ecol.* **1**:75-85.
- Nybakken, J.W. (1997) Marine Biology: An ecological approach, 4th edition. Addison-Wesley Educational Publishers Inc.
- O'Brien D.J., Whitehouse R.J.S. & Cramp A. (2000). The cyclic development of a macrotidal mudflat on varying timescales. *Cont. Shelf Res.* **20**: 1593-1619.

- Ólafsson, J.S. & Paterson, D.M. (2004). Alteration of biogenic structure and physical properties by tube-building larvae in cohesive sediments. *Aquat. Ecol.* **48**: 219-229.
- Orvain, F. (2006). A model of sediment transport under the influence of surface bioturbation: generalization to the facultative suspension-feeder *Scrobicularia plana*. *Mar. Ecol. Prog. Ser.* **286**: 43-56.
- Orvain, F., Le Hir, P. & Sauriau, P.G. (2003). A model of fluff layer erosion and subsequent bed erosion in the presence of the bioturbator, *Hydrobia ulvae*. *J. Mar. Res.* **61**: 823-851.
- Orvain, F., Sauriau, P.G., Bacher, C. & Prineau, M. (2006). The influence of sediment cohesiveness on bioturbation due to *Hydrobia ulvae* on the initial erosion of intertidal sediments: A study combining flume and model approaches. *J. Sea Res.* **55**: 54-73.
- Paramor, O.A.L. & Hughes, R.G. (2005). Effects of the invertebrate infauna on early saltmarsh plant colonization of managed realignment areas in south-east England. *Mar. Ecol. Prog. Ser.* **303**: 61-71.
- Paterson, D.M. (1989). Short term changes in the erodability of intertidal cohesive sediments related to the migratory behaviour of epipelagic diatoms. *Limnol. Ocean.* **34**: 223-234.
- Paterson, D.M. & Black, K.S. (1999). Water flow, sediment dynamics and benthic biology. *Advances in ecological research: Estuaries* (Ed. D.B. Nedwell and D.G. Raffaelli) pp. 155-194. Academic Press, London.
- Paterson, D.M., Tolhurst, T.J., Kelly, J.A., Honeywell, C., de Deckere, E.M.G.T., Huet, V., Shayler, S.A., Black, K.S., de Brouwer, J. & Davidson, I. (2000). Variations in sediment properties, Skeffling mudflat, Humber Estuary, UK. *Cont. Shelf Res.* **20**: 1373-1396.

- Perkins, R.G., Honeywell, C., Consalvey, M., Austin, H.A., Tolhurst, T.J. & Paterson, D.M. (2003). Changes in microphytobenthic chlorophyll *a* and EPS resulting from sediment compaction due to de-watering: opposing patterns in concentration and content. *Cont. Shelf Res.* **23**: 575-586.
- Perkins, R.G., Paterson, D.M., Sun, H., Watson, J. & Player, M.A. (2004). Extracellular polymeric substances: quantification and use in erosion experiments. *Cont. Shelf Res.* **24**: 1623-1635.
- Pearson, T.H. (2001). Functional group ecology in soft-sediment marine benthos: The role of bioturbation. *Oceanogr. Mar. Biol.* **39**: 233-267.
- Pearson, T.H. & Rosenberg, R. (1978). Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol.* **16**: 229-311.
- Petersen, K., Kristensen, E. & Bjerregard, P. (1998). Influences of bioturbating animals on flux of cadmium into estuarine sediments. *Mar. Environ. Res.* **45**: 403-415.
- Porter, E.T., Cornwell, J.C. & Sanford, L.P. (2004). Effect of oysters *Crassostrea virginica* and bottom shear velocity on benthic-pelagic coupling and estuarine water quality. *Mar. Ecol. Prog. Ser.* **271**: 61-75.
- Rasmussen, A.D., Banta, G.T. & Andersen, O. (1998). Effects of bioturbation by the lugworm *Arenicola marina* on cadmium uptake and distribution in sandy sediments. *Mar. Ecol. Prog. Ser.* **164**: 179-188.
- Reading, C.J., Garbutt, R.A., Watts, C.W., Saunders, J.E. & Wolters, M. (2004). Managed realignment at Tollesbury, Annual report, 2003-2004. Report for NERC, prepared by Centre for Hydrology and Ecology.

- Reise, K., Herre, E. & Sturm, M. (1994). Biomass and abundance of macrofauna in intertidal sediment of the Königshafen in the northern Wadden Sea. *Helgolander Meeresun.* **48**: 201-215.
- Reise, K. (2002). Sediment mediated species interactions in coastal waters. *J. Sea Res.* **48**: 127-141.
- Retraubun, A.S.W., Dawson, M. & Evans, S.M. (1996a). Spatial and temporal factors affecting sediment turnover by the lugworm *Arenicola marina* (L.). *J. Exp. Mar. Biol. Ecol.* **201**: 23-35.
- Retraubun, A.S.W., Dawson, M. & Evans, S.M. (1996b). The role of the burrow funnel in feeding processes in the lugworm *Arenicola marina* (L.). *J. Exp. Mar. Biol. Ecol.* **202**: 107-118.
- Riethmüller, R., Heineke, M. Kühl, H. & Keuker-Rüdiger, R. (2000). Chlorophyll *a* concentration as an index of sediment surface stabilisation by microphytobenthos? *Cont. Shelf Res.* **20**: 1351-1372.
- Riisgård, H.U. & Banta, G.T. (1998). Irrigation and deposit feeding by the lugworm *Arenicola marina*, characteristics and secondary effects on the environment. A review of current knowledge. *Vie Milieu.* **48**(4): 243-257.
- Roast, S.D., Widdows, J., Pope, N. & Jones, M.B. (2004). Sediment-biota interactions: mysid feeding activity enhances water turbidity and sediment erodability. *Mar. Ecol. Prog. Ser.* **281**: 145-154.
- Romano, C., Widdows, J., Brinsley, M.D. & Stall, F.J. (2003). Impact of *Enteromorpha intestinalis* mats on near-bed currents and sediment dynamics: flume studies. *Mar. Ecol. Prog. Ser.* **256**: 63-74.
- Ryu, S.O. (2003). Seasonal variation in sedimentary processes in a semi-enclosed bay: Hampyong bay, Korea. *Estuar. Coast Shelf Sci.* **56**: 481-492.

- Ruesink, J.L., Feist, B.E., Harvey, C.J., Hong, J.S., Trimble, A.C. & Wisehart, L.M. (2006). Changes in productivity associated with four introduced species: ecosystem transformation of a “pristine” environment. *Mar. Ecol. Prog. Ser.* **311**: 203-215.
- Schaaff, E., Grenz, C., Pinazo, C. & Lansard, B. (2006). Field and laboratory measurements of sediment erodability: A comparison. *J. Sea Res.* **55**: 30-42.
- Schoeman, D.S., McLachlan, A & Dugan, J.E. (2000). Lessons from a disturbance experiment in the intertidal zone of an exposed sandy beach. *Estuar. Coast Shelf Sci.* **50**: 869-884.
- Schratzberger, M. & Warwick, R.M. (1998). Effects of physical disturbance on nematode communities in sand and mud: a microcosm experiment. *Mar. Biol.* **130**: 643-650.
- Sgro, L., Mistri, M. & Widdows, J. (2005). Impact of the infaunal Manila clam, *Ruditapes philippinarum*, on sediment stability. *Hydrobiologia.* **550**: 175-182.
- Smith, D., Hughes, R.G. & Cox, E.J. (1996). Predation of epiphytic diatoms by the amphipod *Corophium volutator* and the polychaete *Nereis diversicolor*. *Mar. Ecol. Prog. Ser.* **145**: 53-61.
- Smith, D.J. & Underwood, G.J.C. (1998) Exopolymer production by intertidal epipellic diatoms. *Limnol. Oceanogr.* **43**: 1578-1591.
- Smith, G.M., Thomson, A.G., Möller, I. & Kromkamp, J.C. (2004). Using hyperspectral imaging for the assessment of mudflat surface stability. *J. Coast. Res.* **20**: 1165-1175.
- Solan, M., Cardinale, B.J., Downing, A.L., Engelhardt, K.A.M., Ruesink, J.L. & Srivastava, D.S. (2004). Extinction and ecosystem function in the marine benthos. *Science.* **306**: 1177-1180

- Soulsby, R. (1997). Dynamics of marine sands. Thomas Telford Publications.
- Spears, B.M., Funnell, J., Saunders, J.E. & Paterson, D.M. (2007). On the boundaries: stability measurements across aquatic ecosystems. In Westrich, B., & U. Föstner (Eds) Sediment dynamics and pollutant mobility in rivers: an interdisciplinary approach. Springer, Germany.
- Spencer, K.L. (2002). Spatial variability of metals in the inter-tidal sediments of the Medway Estuary, Kent, UK. *Mar. Poll. Bull.* **44**: 933-944.
- Staats, N., de Deckere, E., Kornman, B., van der Lee, W., Termaat, R., Terwindt, J. & de Winder, B. (2001). Observations on suspended particulate matter (SPM) and microalgae in the Dollard Estuary, The Netherlands: Importance of late winter ice cover of the intertidal flats. *Estuar. Coast Shelf Sci.* **53**: 297-306.
- Stevens, M.A. & Boness, D.J. (2003). Influences of habitat features and human disturbance on use of breeding sites by a declining population of southern fur seals (*Arctocephalus australis*). *J. Zool.* **260**: 145-152.
- Sutherland, T.F., Amos, C.L. & Grant, J. (1998). The effect of buoyant biofilms on the erodability of sublittoral sediments of a temperate microtidal estuary. *Limnol. Oceanogr.* **43**(2): 225-235.
- Taylor, I.S. & Paterson, D.M. (1998). Microspatial variation in carbohydrate concentrations with depth in the upper millimetres of intertidal sediments. *Estuar. Coast Shelf Sci.* **48**: 359-370.
- Taylor, I.S., Paterson, D.M. & Mehlert, A. (1999). The quantitative variability and monosaccharide composition of sediment carbohydrates associated with intertidal diatom assemblages. *Biochemistry.* **45**: 303-327.
- Thomsen, L. & Flach, E. (1997). Mesocosm observations of fluxes of particulate matter within the benthic boundary layer. *J. Sea Res.* **37**: 67-79.

- Thrush S.F., Whitlatch, R.B., Pridmore, R.D., Hewitt, J.E., Cummings, V.J. & Wilkinson, M.R. (1996). Scale dependant recolonisation: the role of sediment stability in a dynamic sandflat habitat. *Ecology*. **77**(8): 2472-2487.
- Thrush, S.F., Hewitt, J.E., Gibbs, M., Lundquist, C & Norkko, A. (2006). Functional role of large organisms in intertidal communities: Community effects and ecosystem function. *Ecosystems*. **9**: 1029-1040.
- Timmerman, K., Christensen, J.H. & Banta, G.T. (2002). Modeling of advective solute transport in sandy sediments inhabited by the lugworm *Arenicola marina*. *J. Mar. Res.* **60**: 151-169.
- Tolhurst, T.J., Black, K.S., Shayler, S.A., Mather, S., Black, I., Baker, I. & Paterson, D.M. (1999). Measuring the *in situ* erosion shear stress of intertidal sediments with the Cohesive Strength Meter (CSM). *Estuar. Coast Shelf Sci.* **49**: 281-294.
- Tolhurst, T.J., Black, K.S., Paterson, D.M., Mitchener, H.J., Termaat, G.R. & Shayler, S.A. (2000a). A comparison and measurement standardisation of four *in situ* devices for determining the erosion shear stress of intertidal sediments. *Cont. Shelf Res.* **20**: 1379-1418
- Tolhurst, T.J., Riethmüller, R. & Paterson, D.M. (2000b). *In situ* laboratory analysis of sediment stability from intertidal mudflats. *Cont. Shelf Res.* **20**: 1317-1334.
- Tolhurst, T.J., Jesus, B., Brotas, V. & Paterson, D.M. (2003). Diatom migration and sediment armouring – an example from the Tagus Estuary, Portugal. *Hydrobiologia*. **503**: 183-193.
- Tolhurst, T.J., Underwood, A.J., Perkins, R.G. & Chapman, M.G. (2005). Content verses concentration: Effects of units on measuring the biogeochemical properties of soft sediments. *Estuar. Coast Shelf Sci.* **63**: 665-673.

- Tolhurst, T.J., Defew, E.C., Perkins, R.G., Sharples, A. & Paterson, D.M. (2006a). The effects of tidally-driven temporal variation on measuring intertidal cohesive sediment erosion threshold. *Aquat. Ecol.* **40**: 521-531.
- Tolhurst, T.J., Friend, P.L., Watts, C., Wakefield, R., Black, K.S. & Paterson, D.M. (2006b). The effects of rain on the erosion threshold of intertidal cohesive sediments. *Aquat. Ecol.* **40**: 533-541.
- Tolhurst, T.J., Defew, E.C., de Brouwer, J.F.C., Wolfstein, K., Stal, L.J. & Paterson, D.M. (2006c). Small-scale temporal and spatial variability in the erosion thresholds and properties of cohesive intertidal sediments. *Cont. Shelf Res.* **26**: 351-362.
- Tolhurst, T.J., Watts, C.W., Vardy, C.W., Saunders, J.E., Consalvey, M.C. & Paterson, D.M. (In progress). The effects of rain on the erosion threshold and biogeochemical properties of intertidal sediments.
- Townend, I. (2002). Marine science for strategic planning and management: the requirement for estuaries. *Mar. Policy.* **26**(3): 209-219.
- Townend I. & Whitehead, P. (2003). A preliminary net sediment budget for the Humber Estuary. *Sci. Total Environ.* **314-316**: 755-767.
- Turner, S.J., Grant, J., Pridmore, R.D., Hewitt, J.E., Wilkinson, M.R., Hume, T.M. & Morrisey, D.J. (1997). Bedload and water-column transport and colonization processes by post-settlement benthic macrofauna: Does infaunal density matter? *J. Exp. Mar. Biol. Ecol.* **216**: 51-75.
- Uncles, R.J. (2003). From catchment to coastal zone: examples of the application of models to some long term problems. *Sci. Total Environ.* **314-316**: 567-588.
- Underwood, G.J.C. & Paterson, D.M. (1993a). Recovery of intertidal benthic diatoms after biocide treatment and associated sediment dynamics. *J. Mar. Biol. Ass.* **73**: 25-43.

- Underwood, G.J.C. & Paterson, D.M. (1993*b*). Seasonal changes in diatom biomass, sediment stability and biogenic stabilisation in the Severn Estuary. *J. Mar. Biol. Ass.* **73**: 871-887.
- Underwood, G.J.C. & Smith, D.J. (1998). Predicting epipellic diatom exopolymer concentrations in intertidal sediments from sediment chlorophyll *a*. *Microb Ecol.* **35**: 116-125.
- United Nation Population Division. World Population Prospects: The 2006 Revision. <http://www.un.org/esa/population/publications/wpp2006/wpp2006.htm>
- Vardy, S., Saunders, J.E., Tolhurst, T.J., Davies, P.A. & Paterson, D.M. (2007). Calibration of the high-pressure cohesive strength meter (CSM). *Cont. Shelf Res.* **27**(8): 1190-1199.
- Volkenborn, N & Reise, K. (2006). Lugworm exclusion experiment: Responses by deposit feeding worms to biogenic habitat transformations. *J. Exp. Mar. Biol. Ecol.* **330**: 169-179.
- Waldbusser, G.G. & Marinelli, R.L. (2006). Macrofaunal modification of porewater advection: role of species function, species interaction, and kinetics. *Mar. Ecol. Prog. Ser.* **311**: 217-231.
- Wang, Y.H. (2003). The intertidal erosion rate of cohesive sediment: a case study from Long Island Sound. *Estuar. Coast Shelf Sci.* **56**: 891-896.
- Watts, C.W., Tolhurst, T.J., Black, K.S. & Whitmore, A.P. (2003). *In situ* measurements of shear stress and geotechnical shear strength of the intertidal sediments of the experimental managed realignment scheme at Tollesbury, Essex, UK. *Estuar. Coast Shelf Sci.* **58**: 611-620.
- Whitehouse, R., Soulsby, R., Roberts, W. & Mitchener, H. (2000). Dynamics of estuarine muds. Thomas Telford Publications.

- Widdows, J., Brinsley, M.D., Bowley, N & Barrett, C. (1998). A benthic annular flume for *in situ* measurement of suspension feeding/biodeposition rates and erosion potential of intertidal cohesive sediments. *Estuar. Coast Shelf Sci.* **48**: 27-38.
- Widdows, J., Brown, S., Brinsley, M.D., Salkeld, P.N. & Elliot, M. (2000a). Temporal changes in intertidal sediment erodability: influences of biological and climatic factors. *Cont. Shelf Res.* **20**: 1275-1289.
- Widdows, J., Brinsley, M.D., Salkeld, P.N. & Lucas, C.H. (2000b). Influence of biota on spatial and temporal variation in sediment erodability and material flux on a tidal flat (Westerschelde, The Netherlands). *Mar. Ecol. Prog. Ser.* **194**; 23-37.
- Widdows, J. & Brinsley, M.D. (2002). Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. *J. Sea Res.* **48**: 143-156.
- Widdows, J., Blauw, A., Heip, C.H.R., Herman, P.M.J., Lucas, C.H., Middelburg, J.J., Schmidt, M.D., Brinsley, M.D., Twisk, F. & Verbeek, H. (2004). Role of physical and biological processes in sediment dynamics of a tidal flat in Westerschelde Estuary, SW Netherlands. *Mar. Ecol. Prog. Ser.* **274**: 41-56.
- Widdows, J., Brinsley, M.D., Pope, N.D., Staff, F.J., Bolam, S.G. & Somerfield, P.J. (2006). Changes in biota and sediment erodability following the placement of fine dredged material on upper intertidal shore and estuaries. *Mar. Ecol. Prog. Ser.* **319**: 27-41.
- Widdows, J., Friend, P.L., Bale, A.J., Brinsley, M.D., Pope, N.D. & Thompson, C.E.L. (2007). Inter-comparison between five devices for determining erodability of intertidal sediments. *Cont. shelf. Res.* **27**: 1174-1189.
- Willows, R.I., Widdows, J. & Wood, R.G. (1998). Influences of an infaunal bivalve on the erosion of an intertidal cohesive sediment; A flume and modeling study. *Limnol. Oceanogr.* **43**(6): 1332-1343.

- Wiltshire, K.H., Blackburn, J. & Pateson, D.M. (1997). The cryolander: A new method for fine-scale *in situ* sampling of intertidal surface sediments. *J. Sed. Res.* **67**: 977-981.
- Winn, P.J.S., Young, R.M. & Edwards, A.M. C. (2003). Planning for the rising tides: the Humber Estuary Shoreline Management Plan. *Sci. Total Environ.* **314**: 13-30.
- Wood, R. & Widdows, J. (2003). Modelling intertidal sediment transport for nutrient change and climate change scenarios. *Sci. Total Environ.* **314-316**: 637-649.
- Wright, J.P. & Jones, C.G. (2006). The concept of organisms as ecosystem engineers ten years on: Progress, limitations and challenges. *Bioscience.* **56**(3): 203-209.
- Wootton, J. T. (2002). Indirect effects in complex ecosystems: recent progress and future challenges. *J. Sea Res.* **48**: 157-172.
- Yallop, M.L., de Winder, B., Paterson, D.M. & Stal, L.J. (1994). Comparative structure, primary production and biogenic stabilisation of cohesive and non-cohesive marine sediments inhabited by microphytobenthos. *Estuar. Coast Shelf Sci.* **39**: 565-582.
- Yallop, M.L., Paterson, D.M. & Wellsbury, P. (2000). Interrelationships between rates of microbial production, exopolymer production, microbial biomass, and sediment stability in biofilms of intertidal sediments. *Microb. Ecol.* **39**: 116-127.
- Ysebaert, T., Fettweis, M., Meire, P. & Sas, M. (2005). Benthic variability in intertidal soft-sediments in the mesohaline part of the Schelde Estuary. *Hydrobiologia.* **540**: 197-216.
- Zajac, R.N., Whitlatch, R.B & Thrush, S.F. (1998). Recolonization and succession in soft-sediment infaunal communities: the spatial scale of controlling factors. *Hydrobiologia.* **375**: 227-240.

Zajac, R.N. & Whitlatch, R.B. (2003). Community and population-level responses to disturbance in a sandy community. *J. Exp. Mar. Biol. Ecol.* **294**: 101-125.

Ziervogen, K. & Forster, S. (2006). Do benthic diatoms influence erosion thresholds of coastal subtidal sediments? *J. Sea Res.* **55**: 43-53.

Zipperle, A. & Reise, K. (2005). Freshwater springs on intertidal sand flats cause a switch in dominance among polychaete worms. *J. Sea Res.* **54**: 143-150.

Appendix 1

Calibration of the high pressure Cohesive Strength Meter

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Calibration of the high pressure Cohesive Strength Meter.

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Abstract

Coastal erosion is an immense economic and social problem that has been receiving increased attention in recent years. A number of devices have been developed to measure and quantify sediment stability, from direct measurement devices such as large laboratory and field flumes to proxy measures such as shear vanes and fall cone penetrometers. The Cohesive Strength Meter (CSM) erosion device was developed to measure temporal and spatial variation in the erosion threshold of muddy intertidal sediments directly and *in situ*. Technological developments have enabled considerable improvements to be made to the original design over the last 15 years.

This paper describes modifications to the CSM system that extend the range of eroding pressures the device can generate, to enable measurements to be made on very stable and consolidated sediments, such as salt marshes. A recalibration of the

modified device found inconsistencies in the currently used calibration. Therefore, a new method of equating the CSM jet pressure to the stagnation pressure on the surface sediment is put forward. The application of the device under laboratory conditions using the new calibration on muddy sediment is also presented. The following calibration equations were generated for the individual CSM models: CSM Mark IV_{hp} $y = 22.652 x$; CSM Mark IV_p $y = 8.5282 x$ and for the CSM Mark III $y = 15.844 x$ where y = stagnation pressure at sediment surface (Nm^{-2}) and x = jet exit pressure (kPa).

Keywords: Cohesive Strength Meter; Intertidal Sediment Stability; Erosion Threshold; Calibration.

1. Introduction

Coastal areas are coming under increased hydrodynamic forcing due to global climate change leading to sea level rise and increased storm frequency. Sea level rise will expose new coastal areas to wave and tidal action, these new areas will not be in equilibrium with these forces and thus undergo erosion. Public expenditure on defending the coastline in Europe in 2001 amounted to € 3.2 billion, and this is expected to rise (Doody et al., 2004). This has led to an urgent need to accurately measure and quantify erosion of coastal areas in order to develop predictive models of the dynamics of sediment. Many devices exist that can measure erosion thresholds on unvegetated mudflats, but most of these devices cannot generate enough stress to erode consolidated or armoured sediments such as those stabilised by natural exopolymers (Tolhurst et al., 2002), nor can they be deployed on vegetated habitats

such as mangroves or saltmarshes. There is a requirement for a device that can be used to measure the erosion threshold of such sediments.

The Cohesive Strength Meter (CSM), was originally designed and described by Paterson (1989) and has become quite widely used to determine the relative stability of many estuarine and intertidal sites (e.g. Tolhurst et al., 2000; de Deckere et al. 2001; Defew et al., 2002; Friend et al., 2003; Tolhurst et al., 2003; Watts et al., 2003). It is a compact device used to measure the erosion threshold of exposed sediments, and the system employs a vertical jet of water that is pulsed at the sediment with gradually increasing force. Paterson devised it as a way of measuring relatively small-scale spatial and temporal variation in sediment stability. There are many advantages of using the CSM to measure sediment stability over the more traditional *in situ* devices: it is portable and easy to carry, it is simple and quick to set-up, and measurement time is rapid (~5 min). In addition the CSM footprint is relatively small and so it provides high spatial resolution. Habitats such as saltmarshes and mangrove forests have vegetation cover that prevents the deployment of many erosion devices (Paterson and Black, 2000; Friend et al., 2003). The CSM device, however, can easily be positioned between plants on the sediment. Originally, the device was intended as a method of producing ordinal measurements, with the critical erosion point expressed in terms of the exit velocity of the jet. However, sediment stability is most often described in terms of a critical erosion threshold (τ_0). Tolhurst et al. (1999) identified a need to equate the output of the CSM to the equivalent horizontal shear stress experienced by the sediment, since the critical erosion threshold is one of the parameters used to model sediment dynamics (e.g. Willis and Crookshank, 1997; Whitehouse et al., 2000). They devised a method of equating the exit jet pressure of

the CSM to the critical thresholds for the suspension of sands, producing a conversion equation that is now part of the standard operating procedures of the CSM.

The CSM has been most commonly used on intertidal flats, but more recently has been used for investigations into the erosion thresholds of saltmarsh sediment. The erosion thresholds found on saltmarshes often exceed those found in intertidal flats. For example, Friend et al. (2003) conducted an extensive study using the CSM (Mark III) of the stability of intertidal areas in the Ria Formosa tidal lagoon, Portugal, and found that in some of the saltmarsh areas, detection of the erosion point was beyond the limits of the CSM. Thus, a new High Pressure CSM (CSM Mark IV_{hp}) has been developed with a maximum jet exit pressure double that of the previous devices (up to 413.7 kPa), to allow detection of the erosion thresholds of these highly stable systems. This paper describes an attempt to calibrate the CSM Mark IV_{hp} in the manner described by Tolhurst et al. (1999), and puts forward a new way of describing the force exerted by the jet on the bed. Test data from implementation of the CSMs on muddy sediments in the laboratory is also presented.

2. Materials and methods

2.1. The Cohesive Strength Meter Mark IV prototype and Mark IV high pressure

The basic design of the CSM is unchanged from that described in Tolhurst et al. (1999), although small changes were made to the jet nozzle and length of tubing. The eroding water jet pulse is driven by air pressure supplied by a diving cylinder and the force, length and timing between pulses is controlled by an onboard microprocessor. The sediment suspended into the test chamber is recorded as a change in light

transmission across the chamber and this data is logged for each jet pulse. The CSM can store many different routines for the erosion run protocols, which are used depending on the nature of the substratum (e.g. sand or mud, stable or unstable sediment). For example the pressure steps are small with relatively long intervals between pulses for sediment of low resistance, whereas for sediment of high resistance, the change in pressure is large and the intervals relatively short. The modifications that led from the CSM Mark III to the CSM Mark IV Prototype (CSM Mark IV_p) involved creating a more compact system, and the CSM Mark IV_{hp} was further modified to include the use of components suited to operating under higher pressure. For both of these types of CSM, the reservoir providing water for the jet has been built into the device, and for the CSM Mark IV_{hp} it has been increased in size to allow for the larger water consumption at higher jet pressures and longer test durations.

The effect of constructional differences on the eroding pressure generated by the CSM were investigated by using three different lengths of hose to connect the CSM Mark IV_{hp} to its nozzle. Hose lengths of 0.7 metres, 2.4 metres (original length) and 3.1 metres were tested on both Sand 9 and Fine 1 settings (described in section 2.3). Additionally, two Mark IV_{hp} CSM's borrowed from Sediment Services and Silsoe Research Institute were tested using this method. Pressure readings were taken with a Digitron 2022P Manometer at a range of pressures, comparing the stated firing pressure of the CSM with the actual output pressure.

The duration of the jet pulses was timed by recording the jets for a full test on a 25 frame per second digital camera. Frames where the jet was fired were counted to give a jet duration accurate to 0.04 seconds (n=3). Two test settings were used, Sand 3 and 9 with jet durations of 0.3 and 1 second, respectively. Both these tests start at a

jet exit pressure starting of 3.45 kPa, incrementing at 3.45 kPa per test, up to a jet exit pressure of 34.47 kPa. Thereafter incremental increases are 6.895 kPa up to 413.7 kPa.

2.2. Equations for the suspension of sand

The original derivation of the Shields Criterion for the suspension of sands by Bagnold (1966) uses the solution of equation 2 to formulate equation 1.

$$\theta \geq \frac{CV^2}{gD} \quad (1)$$

where θ = the Shield's criterion for the suspension of sand particles, C = a constant (0.19), V = the settling velocity for a grain of diameter D (ms^{-1}), g = the acceleration due to gravity (9.81 ms^{-2}), and D = the grain diameter (m)

$$\theta = \frac{\tau_0}{(\rho_s - \rho_w)gD} \quad (2)$$

where ρ_s = density of the sediment (kgm^{-3}), and ρ_w = density of the fluid (kgm^{-3})

The Tolhurst et al. (1999) calibration used both these equations to calculate τ_0 and hence resulted in a circular argument. Bagnold (1966) defines the critical threshold of the suspension of sands as being:

$$\tau_0 = \frac{\rho_w V^2}{1.25^2} \quad (3)$$

A modification by McCave (1971) leads to:

$$\tau_0 = \frac{\rho_w V^2}{1.85^2} \quad (4)$$

Thus, equation 4 should have been used for the original calibration. A further modification from the Tolhurst et al. (1999) calibration was the use of the settling velocity formula described by Soulsby (1998).

2.3 Garnet calibration

A suitable answer to the question of how to calibrate the new high pressure CSM appeared to be the use of a denser material than the quartz sand used in the previous calibration, but of a consistent shape and size range to the sand. Garnet sand was used as it has the same size range and shape as sand grains (Figure 1), but with a density of 4200 kgm^{-3} compared to 2600 kgm^{-3} for quartz. Therefore, a higher CSM jet pressure should have been required to move the garnet grains into suspension than quartz grains of an equivalent size. The garnet sand calibration was undertaken on the Mark III, Mark IV_p and Mark IV_{hp} CSM's following the previously outlined method of Tolhurst et al. (1999), with settling velocities calculated according to Soulsby (1998).

Garnet sand with a density of 4200 kgm^{-3} was obtained from Power Garnet™ and sieved using Endecotte brass sieves, with mesh sizes of 150, 212, 250, 300, 425, 500, 600, 710, 850, 1000, 1180, 1400, 1700 and 2000 microns. Sieving was

undertaken with the aim of producing a series of fine and coarse grain sizes for calibration. The median grain size of each sieved fraction was determined using a Coulter Laser Particle Sizer. Garnet sand of each sieve fraction was placed in an 8 cm diameter dish, with a depth of 4 cm. The surface was levelled and then placed in a larger container so that the garnet sand was submerged. The CSM head was then placed on the garnet and the larger container was filled with water so the CSM chamber was filled. For median grain sizes of less than 600 microns, CSM test ‘Fine 1’ was used. This test fires the jet for 1 s with the jet exit pressure starting at 0.6895 kPa, incrementing at 0.6895 kPa per test, up to a jet exit pressure of 16.548 kPa. Thereafter incremental increases are 2.07 kPa up to 41.37 kPa, followed by increments of 13.79 kPa up to 207 kPa (413.7 kPa for the CSM Mark IV_{hp}). For median grain sizes of greater than 600 microns, CSM test ‘Sand 9’ was used. This test fires the jet for 1 s with the jet exit pressure starting at 3.45 kPa, incrementing at 3.45 kPa per test, up to a jet exit pressure of 34.47 kPa. Thereafter incremental increases are 6.895 kPa up to 207 kPa (413.7 kPa for the CSM Mark IV_{hp}). For both tests data was logged at every 0.1 s for 3 s. Transmission data from 0.3 s to 1.3 s was averaged and plotted against the jet exit pressure. A drop in light transmission of 10% was taken to indicate that an erosion event had occurred. Following the method of Tolhurst et al. (1999), the jet exit pressure at which suspension occurred for each median grain size was recorded. A theoretical critical suspension value for each median grain size (section 2.2) was then plotted against the jet exit pressure. This process was applied to the Mark III, Mark IV_p and Mark IV_{hp} CSMs. Exit jet pressures were confirmed using a Digitron 2022P Manometer for each CSM.

The use of a clear chamber made to the direct specifications of the usual CSM chamber showed that the detection point of the CSM is about 1 cm above the

sediment surface. Suspension of sand grains in the CSM is therefore defined as the point when grains are suspended 1cm or more above the bed.

2.4. Calibration Using Q (flux)

With the development of the high pressure CSM, with a maximum jet pressure of around 420 kPa, a new calibration was needed to determine the relationship between the CSM jet pressure and the stress exerted on the sediment surface since this new pressure range fell outside of the existing calibration. Preliminary tests showed that there were significant differences in the erosion thresholds for a particular median grain size between the different CSM designs. It was therefore decided to develop a calibration that used the actual flux of the jet and to express this as the stagnation pressure against the test surface, rather than the pressure setting of the CSM, to make the new calibration applicable to any CSM no matter what its design. To calculate the flux (m^3s^{-1}) the volume of water released at each jet exit pressure was measured. Each jet was fired into a dried, weighed container and then reweighed ($n=5$). The Sand 9 programme was used to give the full range of pressures (3.45 kPa to 207 kPa (414 kPa CSM Mark IV_{hp})), and Fine 1 was used to determine the volumes released at lower jet exit pressures (0.7 kPa to 29 kPa).

The calculations for a turbulent jet impinging on surface are as follows: by continuity

$$Q = U_0 \left(\frac{\pi d^2}{4} \right) \quad (5)$$

where Q is the volume flux (m^3/s), d is the orifice diameter (m), and U_0 is the jet velocity at the source (ms^{-1}).

For a fully developed turbulent jet, the jet velocity u_j at a vertical distance z from the source has been determined experimentally (Fischer et al., 1979) as:

$$u_j = 7.0 \left(\frac{M_0^{\frac{1}{2}}}{(z - z_0)} \right) \quad (6)$$

where $u_j(z)$ is the jet velocity at a vertical distance z from the source (ms^{-1}), z is the vertical distance (m) from the source, $z = z_0$ is the virtual origin of the jet, and M_0 is the source momentum flux of the jet (m^4s^{-2}), whereby

$$M_0 = U_0 Q \quad (7)$$

In the above, $z = z_0$ represents the end of the zone of flow establishment (ZFE) and the start of the zone of established flow (ZEF) where jet profiles are self-similar.

According to the experimental data summarised by (Fischer et al (1979), for a fully-developed turbulent flow in an uncontained fluid,

$$z_0 \sim 10 \ell_Q \quad (8)$$

where

$$\ell_Q = (\pi/4)^{1/2} d \quad (9)$$

for a round turbulent jet.

Thus from 5,6 and 7, the jet velocity in the self-similar region ($z > z_0$) is given by

$$u_j = 7.0 \left(\frac{4}{\pi} \right)^{\frac{1}{2}} \frac{Q}{d(z - z_0)} \quad (10)$$

The stagnation pressure on the test surface below the jet orifice is given by

$$P = \frac{1}{2} \rho_w (u_j)^2 \quad (11)$$

Thus, from (10) and (11)

$$P = \frac{1}{2} \frac{\rho_w (7.0)^2 4Q^2}{d^2 (z - z_0)^2 \pi} \quad \text{for } z > z_0 \quad (12)$$

Since z_0 is typically less than the total depth of the working volume, its effect on the expression 12 is not expected to be significant, so long as the jet is fully turbulent.

2.5. Testing the calibration on cohesive sediment.

To test the equations, a model system was created by sieving 2 kg of mud collected from the Eden Estuary, Scotland through a Endecotte brass sieve, with 500 μm mesh size, and homogenised. The mud samples consisted of an average of 60%

silt/clay (< 63 μm), 20% very fine sand (63 μm – 125 μm), 12 % fine sand (125 μm – 500 μm), and 1 % sand (> 500 μm) (Wentworth 1922). Grain size was determined using a Beckman Coulter [LS 230 Particle Size Analyser](#). To another 2 kg of mud, 20 g of xanthan gum (a bacterial polymer) dissolved in 500 ml of water was added and mixed thoroughly. The mud/Xanthan gum mixture was left for two hours before tests were run. For the homogenised mud, CSM test Fine was used, and for the mud/xanthan gum mixture Sand 9 was used (n=5). The non-parametric Kruskal-Wallis Test (Zar 1984) was used to determine whether the application of the equations normalised the data obtained from the different CSMs. Results were considered to be significantly different at $p < 0.05$.

3. Results and Discussion

3.1. Garnet Calibration

When the new calibration was undertaken, the theoretical suspension threshold (Nm^{-2}) for each median garnet grain size was calculated and, as expected, was found to exceed the theoretical suspension threshold (Nm^{-2}) of that calculated for corresponding median quartz grain sizes (Figure 2). The curves obtained by the use of garnet to calibrate the three CSMs (as described in section 2.3) were of the same shape as those determined by Tolhurst et al. (1999) using quartz sand (Figure 3). However, it was immediately obvious that the jet exit pressures (kPa) at which the garnet eroded were much lower than expected, and that the differences between the quartz and lowest garnet eroding pressure vs grain size curves and the different garnet curves for different versions of CSMs were of the same order. Furthermore, when the

eroding pressure was plotted against the critical suspension threshold for the three CSMs (calculated using the formulas described in Tolhurst et al., 1999) and the quartz calibration, there was a wide variation between the curves (Figure 4). For the calibration to hold, the garnet data from all CSMs and the quartz data obtained from the CSM Mark III should all have fallen on roughly the same line, with higher eroding pressures for the larger garnet grains. The discrepancy was partially explained by the problems in the calculations described in section 2.2. However, even with the recalculation according to equation 4, the curves did not drop into line (Figure 5), although the differences were much reduced. Of greatest concern was the difference between the quartz and garnet curves taken from the CSM Mark III, which indicated that there was some inconsistency somewhere in the original calibration.

The jet exit pressures were confirmed using a Digitron 2022 P manometer, and corresponded exactly with those displayed on each CSM, indicating some other factor needed to be taken into account to explain the discrepancy between the different calibration results.

3.2 Effects of constructional differences on jet

One possible cause of the inconsistency in results between the different CSM devices was constructional differences. The most obvious constructional difference between the three types of CSMs was different in pressure hose lengths, and for the CSM Mark IV_p a slightly larger diameter of tubing. The three CSMs (Mark IV_{hp}) tested were all supplied with different hose lengths by the manufacturer, and additionally were tested with shorter (0.7 m) and longer (3.1 m) hose lengths. For each hose length on a CSM, different fluxes were observed (Figure 6 a-c), and for the

same hose length on different CSMs, a variation in flux was observed (Figure 6d), indicating that not all variation between CSM's was due to different hose lengths. Thus, at least some of the differences in the suspension calibration can be attributed to different fluxes between devices.

Anomalous jet weights (Figure 6) were noticed during these tests, and an investigation into the pressure and firing time of each jet was conducted. A linear relationship was found between stated firing pressure and actual firing pressure, hence there was no indication of increases or decreases in output related to the anomalous flux readings.

On the Sand 9 (section 2.1) test the majority of jets had durations of 1 second as programmed by the test setting. However, several jets had shorter or longer durations with errors between 0.04 and 0.08 seconds detected. These timing differences corresponded to the jets that produced unexpected flux values, with longer duration jets resulting in higher flux values and shorter durations resulting in lower flux values (Figure 7a). The error is more apparent on Sand 3 (section 2.1) where errors of 0.06 seconds are more significant to a programmed jet duration of only 0.3 seconds (Figure 7b). The variations that result from errors in the jet duration from 0.04 to 0.08 seconds on Sand 9 are unlikely to influence the erosion measurement. However, the manufacturer is aware of this problem and new CSM's will not have the inherent timing errors.

3.3 Pressure on surface (Nm^{-2})

Converting the mass of water produced at each pressure increment into pressure on the sediment bed (as described in section 2.2) shows that there is a near linear relationship between jet pressure and pressure on the bed (Figure 8). However, constructional differences between the different CSM devices results in different relationships between jet pressure and pressure on the bed. (Note that the differences in the relationships between the jet and bed pressure are manifested as differences in slope. If the discrepancy had been due solely to the neglect of the term z_0 in equation 12, the linear relationships of Figure 8 would have been displaced with respect to each other but would have retained the same slope. The form of the graph confirms that the constructional details are responsible for the differences in slope between the different devices). The anomalous readings (described in section 3.2) were removed from the data set for the purpose of the calibration.

The relationship between jet exit pressure and stagnation pressure at the surface of the sediment is described by the following relationships.

For the CSM Mark IV_{hp}:

$$y = 22.652 x \quad (13)$$

and for the CSM Mark IV_p

$$y = 8.5282 x \quad (14)$$

and for the CSM Mark III

$$y = 15.844 x \quad (15)$$

where y = stagnation pressure at sediment surface (Nm^{-2}), and x = jet exit pressure (kPa).

3.4 Testing on cohesive sediment in the laboratory.

To test if the flux equations worked in a real system, the three CSMs were deployed on a homogenised bed of mud to which 10 g of Xanthan gum was added to each kg of mud. Erosion threshold was expressed for each CSM as the jet exit pressure (kPa) (Figure 9a) and as stagnation pressure at the surface (Nm^{-2}) (Figure 9b) taken from equations 13-15. There was a significant difference between the three types of CSM for the erosion threshold expressed as jet exit pressure ($p < 0.05$) for both the homogenised mud and the mud/EPS mixture. When the results were converted to stagnation pressure at the surface, there was no significant difference between the three CSM's for either the homogenised mud mixture or the mud/EPS mixture ($p < 0.05$). The standard deviation between readings remained high for the mud/EPS mixture and there remain some differences among the different devices, but these differences are no longer significant. Thus, this calibration is successful in eliminating significant differences between different devices.

A weakness in using this method for calibrating the CSM in terms of modelling is that a relationship between the CSM results and critical suspension thresholds cannot

be established at present. The relative stability between one area of sediment and another is all that can be stated. However, results from different CSM models can be confidently expressed, and results between different research groups will be comparable. Additionally, as the method is fast, a CSM can be calibrated monthly as part of a maintenance program. Further work needs to be undertaken to understand the flow within the CSM chamber before a satisfactory relationship between the CSM pressures and critical suspension thresholds can be developed.

4. Conclusions

The benefits of expressing the jet exit pressure as stagnation pressure at the surface are:

- 1) It will give a measure of the initial impact of the jet on the surface of the cohesive sediment bed
- 2) Each laboratory will easily be able to calibrate their own CSM on a regular basis allowing a more accurate comparison of results between labs
- 3) It is much less time consuming than the calibration of Tolhurst *et al* (1999).
- 4) It will give a comparative measure of the force at the higher end of the CSM scale.
- 5) Published results using older CSM's are relative and instrument specific. Comparisons between instruments should be treated with caution.

Acknowledgments

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References:

Bagnold, R.A., 1966, An approach to the sediment transport problem from general physics. *U.S. Geological Survey Professional Paper* 422-I.

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

Defew, E.C., Tolhurst, T.J., Paterson, D.M., 2002. Site-specific features influence sediment stability of intertidal flats. *Hydrology And Earth System Sciences* 6 (6), 971-981.

Doody, P., Ferreira, M., Lonbardo, S., Luclus, I., Misdorp, R., Niesing, H., Salman, A., Smailegange, M., (Eds.), 2004. Coastal erosion in Europe, Sediment and Space for Sustainability, Results from the EuroSION Study, online, brochure, 1.19 MB, <http://www.euroSION.org/project/eventnews.htm>.

Fisher, H. B., List, E.J., Koh, R.C.Y., Imberger, J., and Brooks, N. H., 1979. Mixing in Inland and Coastal Waters. Academic Press, New York, USA.

Friend, P.L., Ciavola, P., Cappucci, S., Santos, R., 2003. Bio-dependent bed parameters as a proxy tool for sediment stability in mixed habitat intertidal areas. *Continental Shelf Research* 23 (17-19), 1899-1917.

McCave, I.N., 1971. Sand waves in the North Sea off the coast of Holland. *Marine Geology*. (10), 199-225.

Paterson, D.M., 1989. Short-term changes in the erodibility of intertidal cohesive sediments related to the migratory behaviour of epipelagic diatoms. *Limnology and Oceanography* (34) 223-234.

Paterson, D.M., Black, K.S., 2000. Temporal variability in the critical erosion threshold of saltmarsh and upper intertidal sediments. In: Sherwood, B.R., Gardiner, B.G., and Harris, T. (Eds.), *British Saltmarshes*. Forrest Text Cardigan for The Linnean Society of London, London, pp 51-63.

Soulsby, R., 1998. *Dynamics of Marine Sands*, Thomas Telford Publications, pp131-137.

Tolhurst, T. J., Black, K. S., Shayler, S. A., Mather, S., Black, I., Baker, K., Paterson, D. M., 1999. Measuring the in situ erosion shear stress of intertidal sediments with the Cohesive Strength Meter (CSM). *Estuarine Coastal and Shelf Science* (49), 281-294.

Tolhurst, T. J., Black, K. S., Paterson, D. M., Mitchener, H., Termaat, R., Shayler, S. A., 2000. A comparison and measurement standardisation of four in situ devices for determining the erosion shear stress of intertidal sediments. *Continental Shelf Research* (20), 1397-1418.

Tolhurst, T. J., Gust, G., Paterson, D. M., 2002. The influence of an extracellular polymeric substance (EPS) on cohesive sediment stability. In: Winterwerp, J.C., Kranenburg, C., (Eds.), *Fine Sediment Dynamics in the Marine Environment*, *Proceedings in Marine Science* (5), 409-425.

Tolhurst, T.J., Jesus, B., Brotas, V., Paterson D. M., 2003. Diatom migration and sediment armouring - an example from the Tagus Estuary, Portugal. *Hydrobiologia* 503, 183-193.

Watts, C.W., Tolhurst, T.J., Black, K.S., Whitmore, A.P., 2003. *In situ* measurements of erosion shear stress and geotechnical shear strength of the intertidal sediments of the experimental managed realignment scheme at Tollesbury, Essex, UK. *Estuarine Coastal And Shelf Science* 58 (3), 611-620.

Whitehouse, R., Soulsby, R., Roberts, W., Mitchener, H., 2000. Dynamics of Estuarine Muds. Thomas Telford Publishing, London.

Willis, D.H., Crookshank, N.L., 1997. Modelling multiphase sediment transport in estuaries. In: Burt, N., Parker, R. and Watts, J. (Eds.), *Cohesive Sediments*. Wiley, Chichester, UK, pp.383-394.

Zar, J.H., 1984 *Biostatistical Analysis*, Prentice-Hall International, Engelwood Cliffs, New Jersey, USA, p176.

Figure Captions

Fig. 1. Garnet grains used for Cohesive Strength Meter calibration. Mesh size – 450-825 μm . Magnification (x25).

Fig. 2. Theoretical critical suspension threshold for a variety of grain sizes for garnet (●) and quartz (▼).

Fig. 3. Mean eroding pressure (\pm s.e.) at different grain sizes and densities. CSM Mark IV_p using garnet (○), CSM Mark IV_{hp} using garnet (▲), CSM Mark III using garnet (■), CSM Mark III using quartz (◇).

Fig. 4. Critical suspension threshold calculated using the method of Tolhurst et al., 1999 for different erosion thresholds measured using various CSM devices. CSM Mark IV_p using garnet (○), CSM Mark IV_{hp} using garnet (▲), CSM Mark III using garnet (■), CSM Mark III using quartz (◇).

Fig. 5. Critical suspension threshold calculated using the method of Bagnold (1966) for different erosion thresholds measured using various CSM devices. CSM Mark IV_p using garnet (○), CSM Mark IV_{hp} using garnet (▲), CSM Mark III using garnet (■), CSM Mark III using quartz (◇).

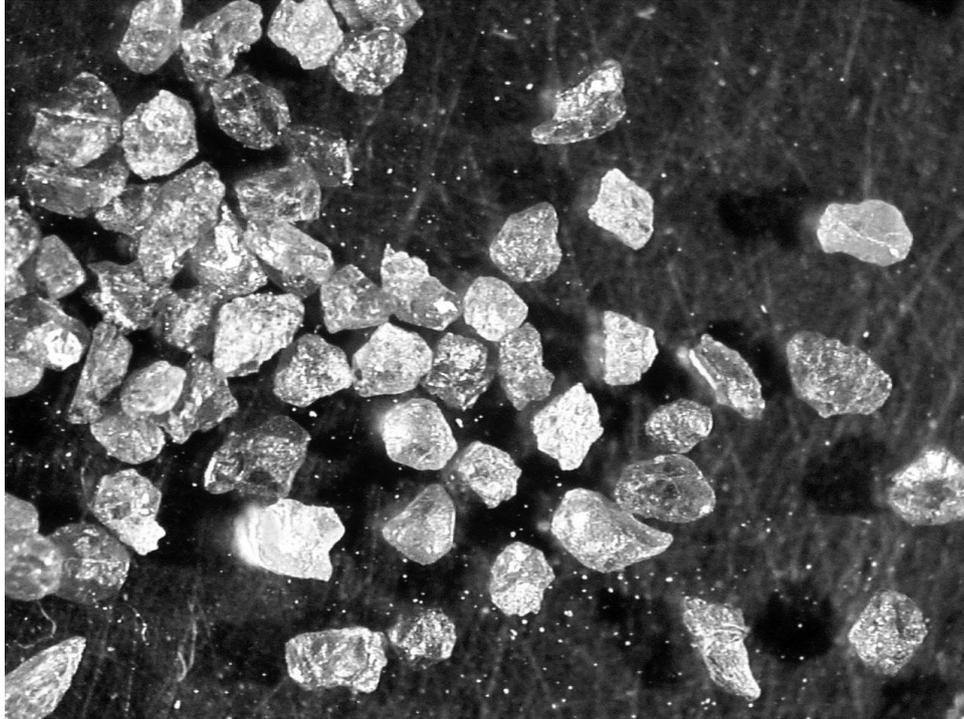
Fig. 6. Effects of hose length on different CSM Mark IV_{hp} devices. Supplied by Sediment Services (ss), The University of St Andrews (sa) and Silsoe Research Institute (sl). (a) CSM Mark IV_{hp} ss (▲) 0.7 m, (○) 1.2 m, (■) 3.1 m. (b) CSM Mark IV_{hp} sa (▲) 0.7 m, (○) 2.4 m, (■) 3.1 m. (c) CSM Mark IV_{hp} sl (▲) 0.7 m, (○) 1.7 m, (■) 3.1 m. (d) Hose length 0.7 m, CSM Mark IV_{hp} (▲) ss, (○) sa, (■) sl (mean ± s.e., n=5).

Fig. 7. Q flux (■) and jet duration (▲) (a) test sand 9 (b) test sand 3 (mean± s.e., n=5).

Fig. 8. Stagnant pressure on the surface for exit jet pressure (▲) CSM Mark IV_{hp}, (●) CSM Mark III, (■) CSM Mark IV_p (mean ± s.e., n=5).

Fig. 9. Application of new calibration equations to homogenised mud bed and a mud/EPS mixture (mean ± s.e., n=5). (a) results expressed as jet exit pressure (kPa) and (b) results expressed as stagnant pressure on the bed (Nm⁻²).

Figure 1 Vardy *et al.*



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Figure 2 Vardy *et al.*

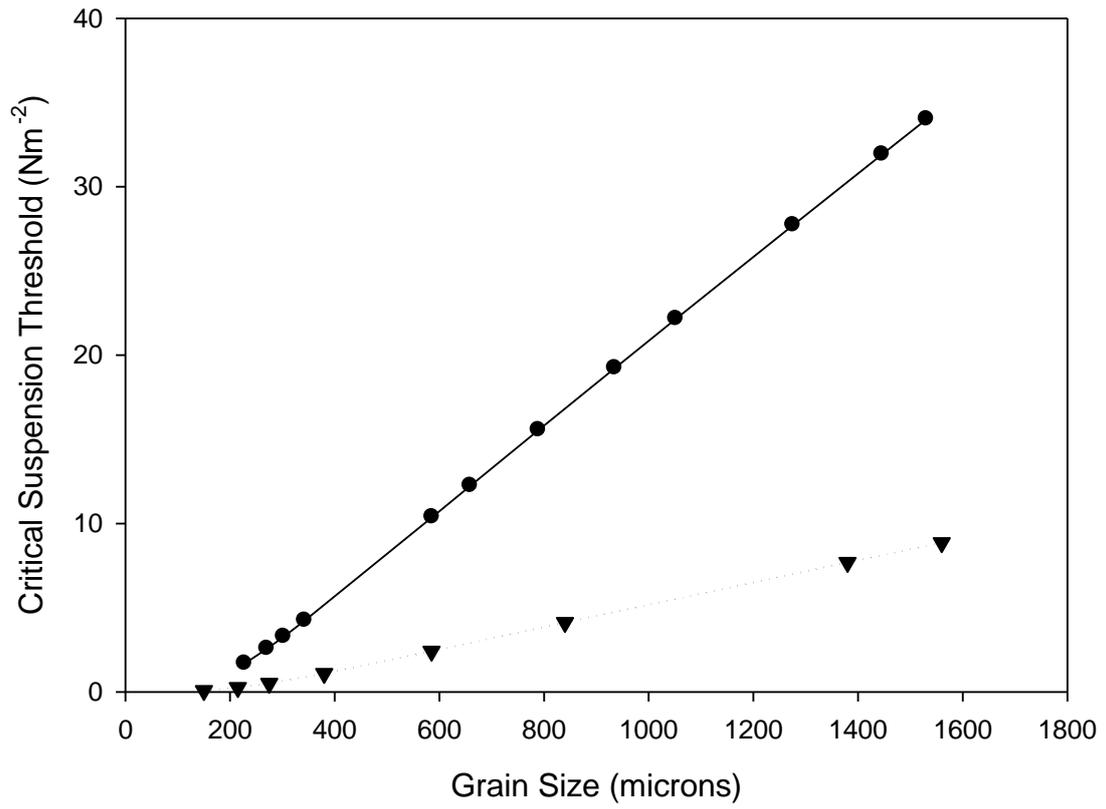


Figure 3 Vardy *et al.*

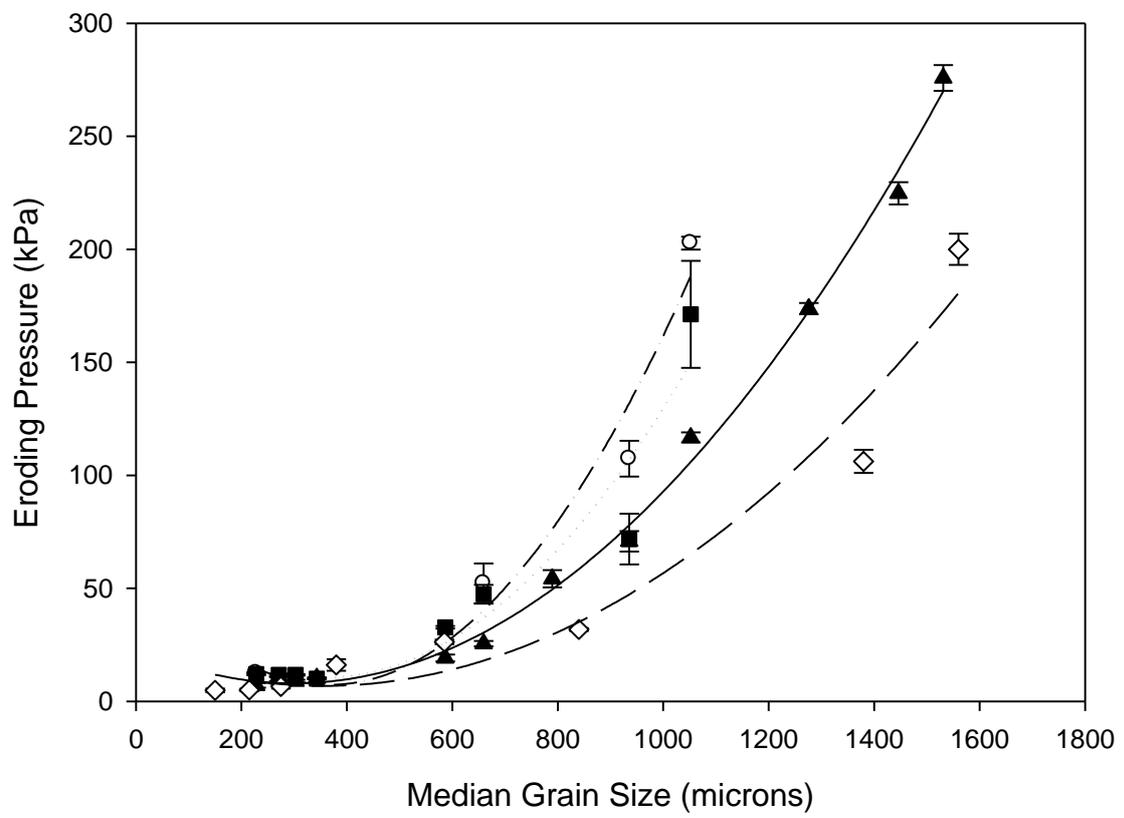


Figure 4 Vardy *et al.*

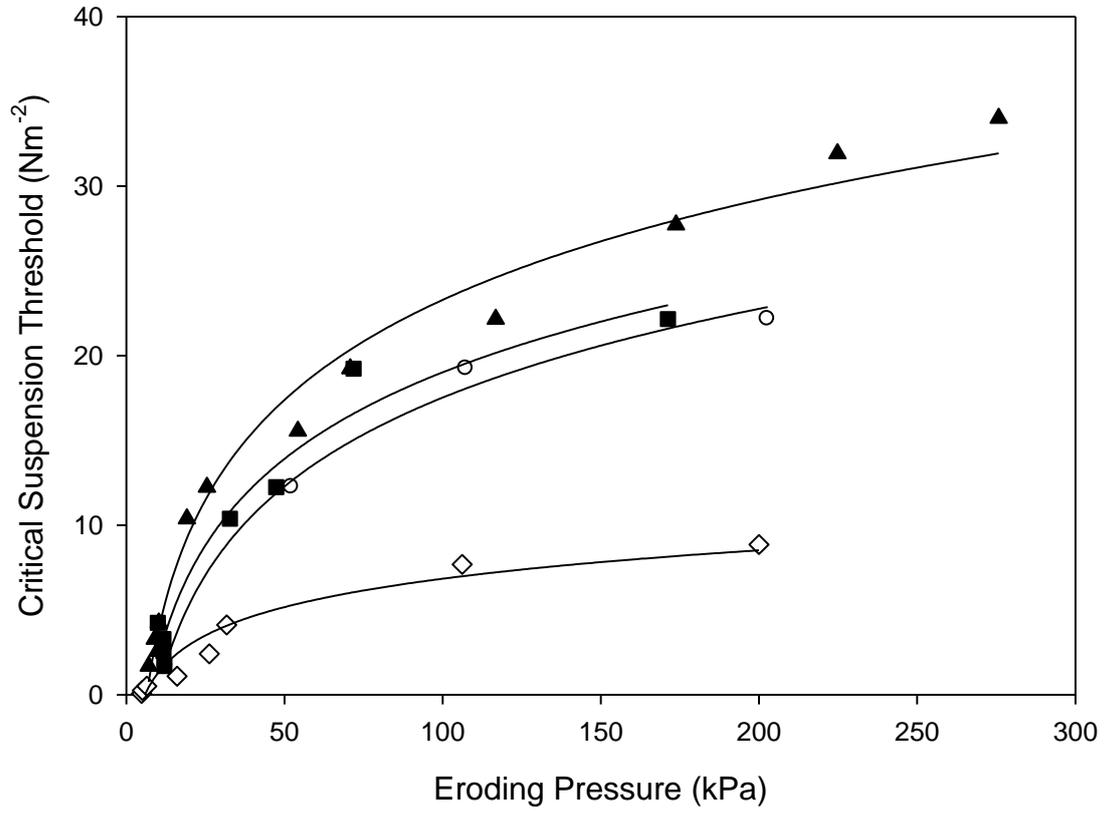


Figure 5 Vardy *et al.*

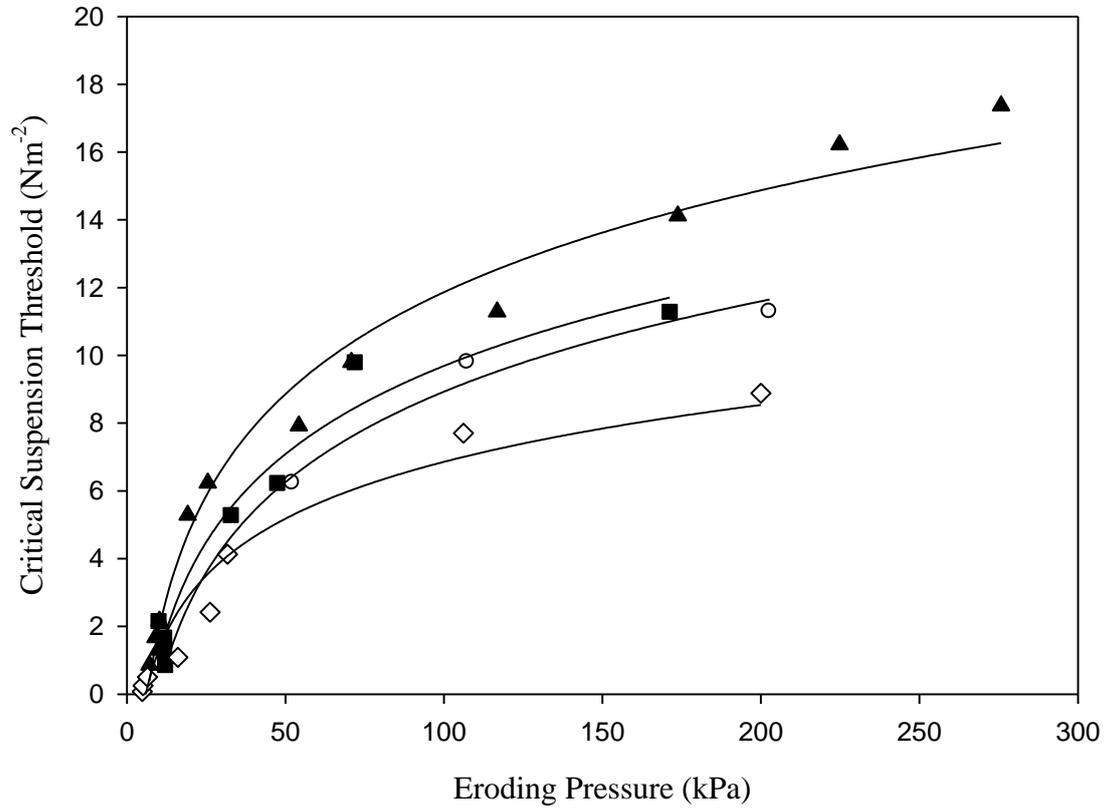
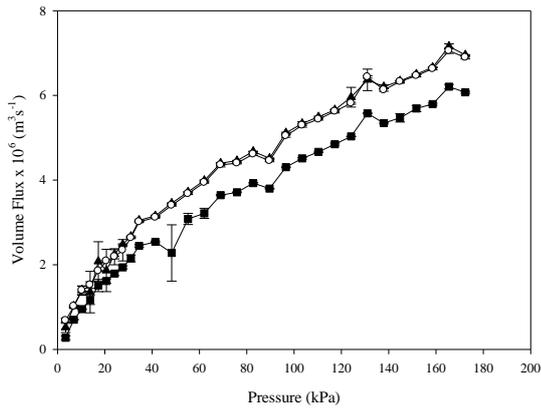
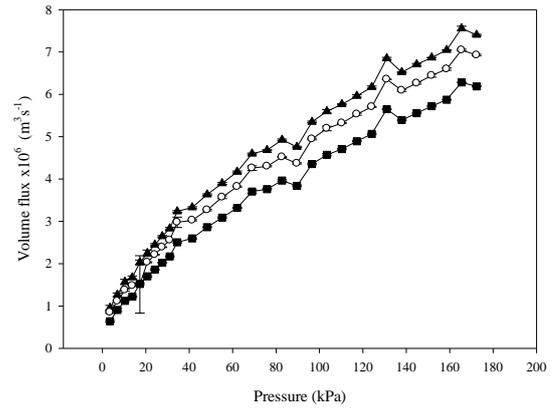


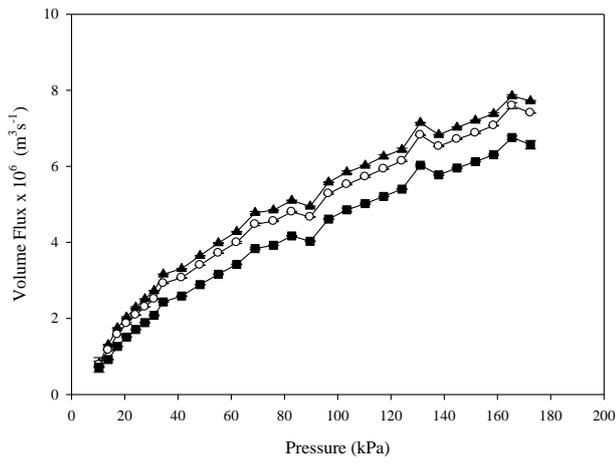
Figure 6 Vardy *et al.*



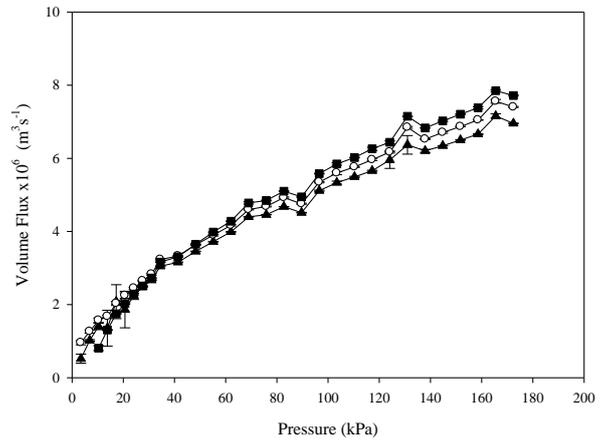
(a)



(b)

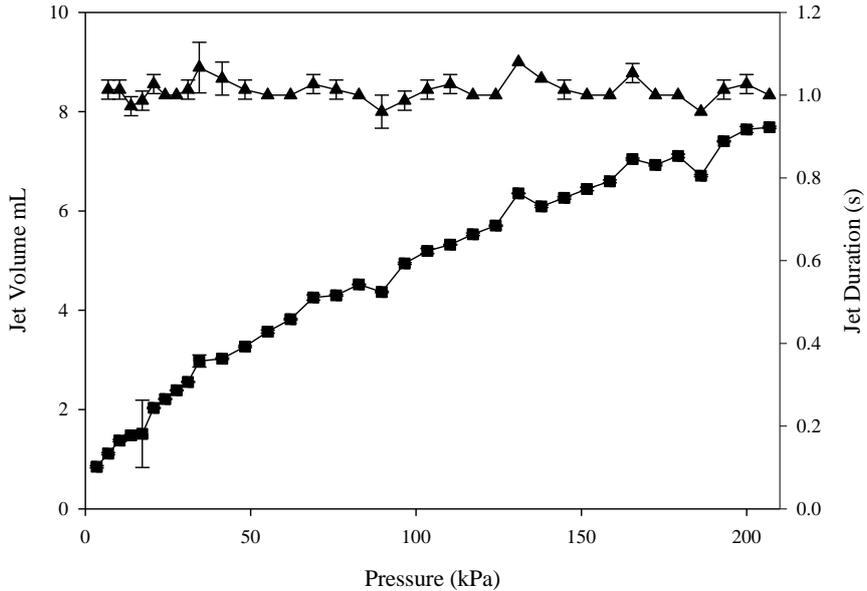


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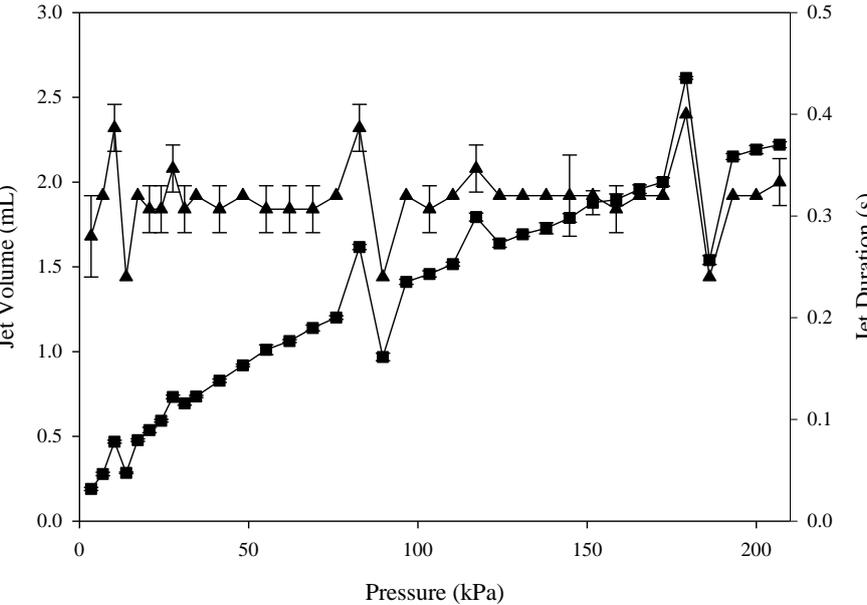


(d)

Figure 7 Vardy *et al.*



(a)



(b)

Figure 8 Vardy *et al.*

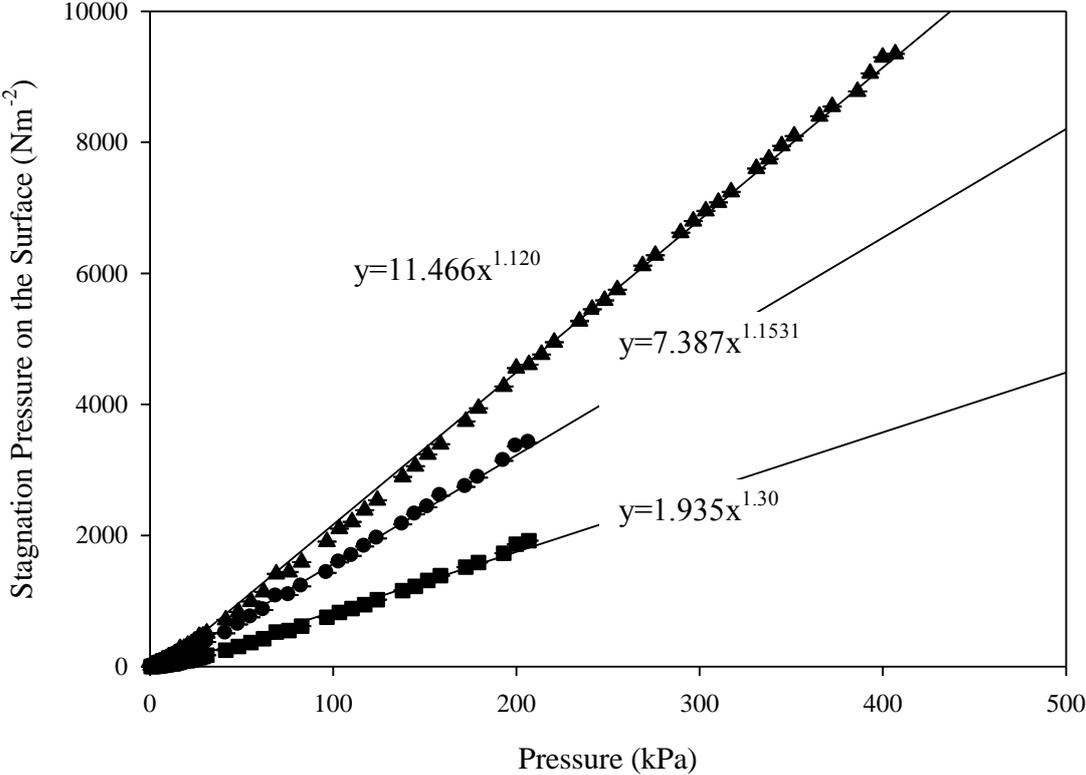
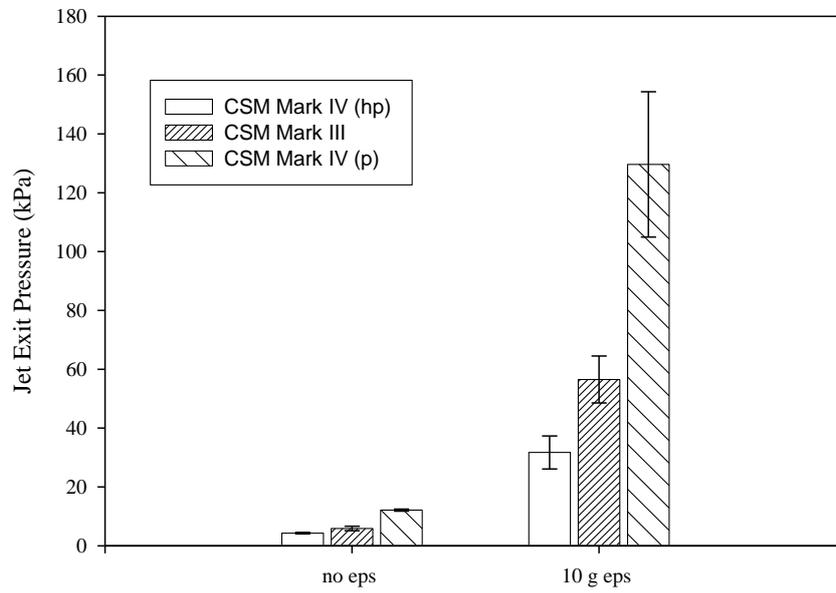
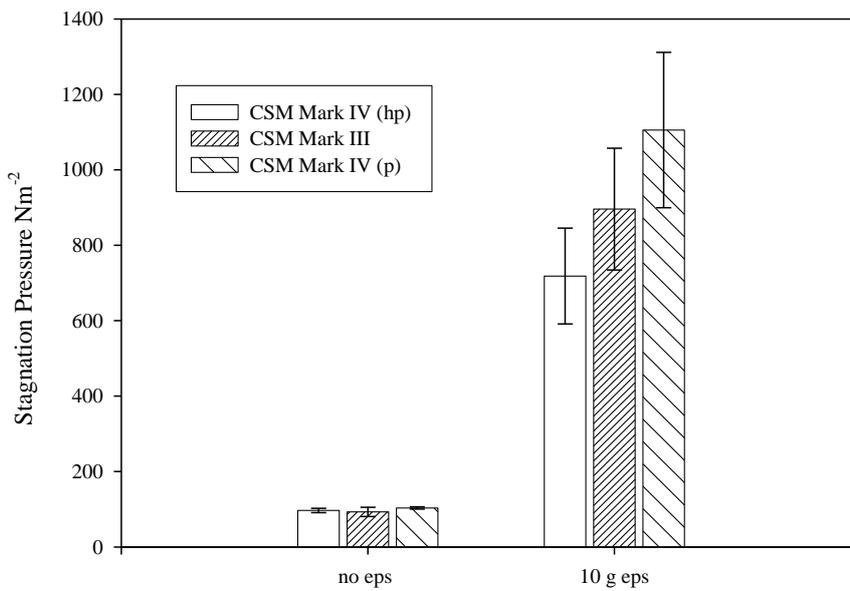


Figure 9 Vardy *et al.*



(a)



(b)

