University of Southampton

ANIMAL-SEDIMENT INTERACTIONS: MACROFAUNA COMMUNITY STRUCTURE AND SEDIMENTS OF AN INTERTIDAL MUDFLAT, SOUTHAMPTON WATER, UK

A thesis submitted for the degree of Doctor of Philosophy

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ABSTRACT

FACULTY OF SCIENCE

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ANIMAL-SEDIMENT INTERACTIONS: MACROFAUNA COMMUNITY STRUCTURE AND SEDIMENTS OF AN INTERTIDAL MUDFLAT, SOUHAMPTON WATER, UK

by Michelle Dawn Lindsay

Interdisciplinary research, incorporating macrobenthic ecology and sediment dynamics, was undertaken in order to investigate the effects of recharge on the lower shore of an intertidal mudflat at Hythe, Hampshire, UK. A field programme was carried out to examine the impact of a trial recharge experiment (planned and carried out by Associated British Ports) on the macrofauna community and associated sediments, and to collect baseline information on the site. The fieldwork included biological sampling, in situ physico-chemical measurements of key environmental parameters, measurements to identify changes in bed elevation indicative of accretion or erosion at the site, and the collection of sediment cores for analysis of sediment properties and structure.

Design and development of a 1000 l capacity, recirculatory experimental laboratory or microcosm system incorporating simulation of tidal exposure/inundation was undertaken in order to provide a facility for conducting controlled, manipulative laboratory experiments on sediments and live fauna from Hythe. A series of five controlled smothering experiments were carried out in the new system to assess the impact of burial (as may occur during recharge) with native sediments on individual species and natural mixed species assemblages from the site. Effects of smothering were analysed in terms of survival and observed vertical migratory ability, and 'tolerance thresholds' were identified for individual species. The influence of the macrofauna community at Hythe on sediment erodibility and bed stability was investigated in controlled laboratory erosion experiments using two instruments; EROMES and the CSM (Cohesive Strength Meter). Erosion thresholds (τ_c) and erosion rates (E) were calculated for each core, regression analysis was utilised to identify faunal effects, and to perform a comparison of instruments. Computer Tomography (CT) was used to obtain high-resolution data and images to produce fine-scale sediment profiles of wet bulk density, and to investigate/identify effects of the recharge and modification of sediment structure and properties by macrofauna.

Both the field (and the CT) studies indicated that no significant deposition of material occurred at the site as a result of recharge and the benthic community was unaffected. This is in agreement with the results of surveys carried out by ABP which concluded that the deposited material was transported away from the site by the tidal currents (ABP, 2001). Results obtained from the smothering experiments indicated that effects were species specific, and were dependent upon animal functional morphology. Tolerance thresholds for species ranged from less than 2 cm of burial (e.g. H. ulvae) to >50 cm (N. diversicolor). No clear relationship was identified between τ_c and macrofauna density from the erosion experiments. Results suggest that erosion rate and gradient as a relative measure of internal friction coefficient (ϕ) may be better parameters for future investigation, as causative relationships were implied. Experimental effects arising from instrument differences, laboratory effects and treatment effects were identified and addressed. CT proved to be a highly suitable technique for the investigation of sediment structure and fine-scale bulk density distribution. Several distinct layers of sediment, including a collapse zone (sensu Droppo & Amos, 2001) and self-weight consolidation, were identified from the bulk density data. Faunal modification of sediment structure by bioturbation, and individual burrows were also identified. No long-term, or seasonal trends of erosion or deposition were revealed by the CT images and data, and earlier observations were supported in that no clear signs of recharge were evident.

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DECLARATION

I hereby declare that the work included in this thesis has not previously been accepted in substance for any degree and is not being concurrently submitted in candidature for any degree.

This thesis is the result of my own investigations, except where otherwise stated. Other sources are acknowledged by explicit references. A bibliography giving full details of all sources is included.

I hereby give consent for my thesis, if accepted, to be available for photocopying and for inter-library loan, and for the title and summary to be made available to outside organisations.

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For my grandparents, Dorothy & James Scott, and Bill Parker

you are sorely missed and thought of often,
with love and gratitude for the happiest of memories

List of abbreviations and symbols

τ_o	bed shear stress (Pa)
d	biological dominance, Berger-Parker Index of Dominance
H'	biological diversity, Shannon-Weiner Diversity Index
D	biological diversity, Simpson's Index
BOD	biological oxygen demand
B_1	maximum bioturbation zone
B_2	limit of bioturbation (infauna)
ρ_b	bulk density (for fine-grained sediments) (kgm^{-3})
BD_w	wet bulk density (kgm^{-3})
CSM	Cohesive Strength Meter
c	collapse zone
cb	consolidated bed
CT	computer tomography
R^2	correlation coefficient
τ_c	critical threshold for erosion (Pa)
df	degrees of freedom
d	deposition (layer of)
DO	dissolved oxygen content (mgl^{-1})
x	eroding pressure (Pa)
R	EROMES propeller speed (rpm)
E	erosion rate ($mgm^{-2}s^{-1}$)
E_m	mean erosion rate ($mgm^{-2}s^{-1}$)
J	eveness (after Pielou)
EPS	extra-cellular polymer
f	fluff layer
HU	Hounsfield Unit
ITI	Infaunal Trophic Index
I	intensity of attenuated X-ray beam
I_o	initial intensity of X-ray incident beam
ϕ	internal friction coefficient
LD50	lethal dose fifty
LD90	lethal dose ninety
LD10	lethal dose ten
tx	% light transmission
μ	linear attenuation coefficient (of X-rays through sample)
μ_s	linear attenuation coefficient of sample
μ_w	linear attenuation coefficient of pure water
M_1	mass of filter (g)
M_2	mass of filter and sample (g)
MLW	mean low water
MLWN	mean low water neap
MLWS	mean low water spring
MDS	Multi-Dimensional Scaling
n	number of individuals in sample
OBS	optical backscatter sensor
ppm	parts per million
psu	practical salinity unit

PCA	Principal Component Analysis
Eh	redox potential
ROI	region of interest
α	sediment surface area (m^2)
σ_{n-1}	standard deviation
SSC	suspended sediment concentration (mgl^{-1}, gl^{-1})
T	time (sec)
M_t	total water-saturated mass (g)
V	volume (m^{-3})
V_t	volume of water-saturated sample (m^{-3})

1 Introduction

1.1 Research perspective

Estuaries are recognised as highly productive ecosystems and are of great importance to fisheries, plants and bird populations (McLusky, 1981). Many estuaries are fringed by broad intertidal flats, which act as natural sinks for fluvial sediments, organic matter and nutrients (McLusky, 1981). However, as many of the world's greatest cities are located on estuarine shores, these environments are subjected to intense exploitation and pressures from human activity resulting in degradation of the natural environment, including chemical pollution and organic enrichment from industrial, domestic and agricultural sources and shipping. Many estuaries have also been subject to morphological and physical modifications from port construction and development, coastal protection and flood defence structures and leisure activities such as the construction of marinas. One of the most significant activities in industrialised and urbanised estuaries is the dredging of navigation channels, which is carried out periodically to counteract sediment accumulation and allow the safe passage of vessels. Impacts of these activities in estuaries has resulted in the disruption and modification of natural sediment transport processes, deposition of material by engineered structures and modified flow patterns, and the contamination of sediments by hydrocarbons, metals and other chemicals (McLusky, 1981).

Estuarine organisms are believed to have evolved resilience to natural environmental fluxes and stress, as they are naturally subjected to significant fluctuations in salinity, temperature, nutrient supply and sediment regimes. Estuarine communities are thus generally considered to be relatively low in biological diversity consisting of organisms able to tolerate this fluctuation of environmental conditions (Gray, 1981; McClusky, 1981). Intertidal mudflats fringing estuaries are highly productive habitats that provide a home to numerous invertebrates, including polychaete worms and molluscs, which build burrows and tubes in the sediment. These macrofauna communities live largely within the sediment, some moving to the surface (e.g. *Hydrobia ulvae*), or extending body parts to the surface to feed, in a three-dimensional habitat, linked to the water column by fluxes of water, sediments, nutrients and also contaminants.

However, anthropogenic organic enrichment (e.g. sewage effluents, artificial fertilizers etc.) and chemical contamination of the water column and sediments inflict further stress upon the biota present both directly through toxicity effects, and indirectly through hypoxia. These factors (and/or physical disturbance caused through activities such as dredging, mineral extraction and some methods of commercial fishing) cause displacement, increased mortality, modified distribution, and reduced fitness for foraging, avoidance of predation, competitive and reproductive ability of the biota. This commonly results in habitat degradation and reduced biodiversity and species richness (McLusky, 1981).

This research focuses on the interactions between the macrofauna community structure and the sedimentary environment they inhabit, the principle components of which are shown in Figure 1.1. The aims of the study and details of scientific objectives relating to each component are presented below (sections 1.2.1 & 1.2.2). The field work was undertaken at Hythe, Hampshire (Figure 1.2), to assess the impacts of recharge on the benthic macrofauna community. Field sampling was also utilised to obtain background information on the infaunal community, and to provide a basis for detailed laboratory experiments on animal-sediment interactions. The experiments were intended to form a sub-set of details at temporal and spatial scales that could not be undertaken in the field due to the severely restricted access to the mudflat. These experiments focused on the impacts of episodic deposition on the community structure and individual populations as would be expected to occur, for example, during recharge. Recharge is the planned and deliberate deposition of large quantities of dredge spoil, either from routine dredging of navigation channels, or capital dredging (the one-off dredging of large quantities of sediments during port construction). The material is transported to, and discharged at, a chosen site where it is intended to remain – effectively 'capping' the existing sediment; see below and Chapter 2 for further details. Experiments were also run to investigate the impacts of the community structure on bed stability (chapter 5). In addition, a CT Scanner (Wellington & Vinegar, 1987) was used to analyse sediment cores from the field site to investigate the effects of the infaunal community on sediment structure and properties (chapter 6).

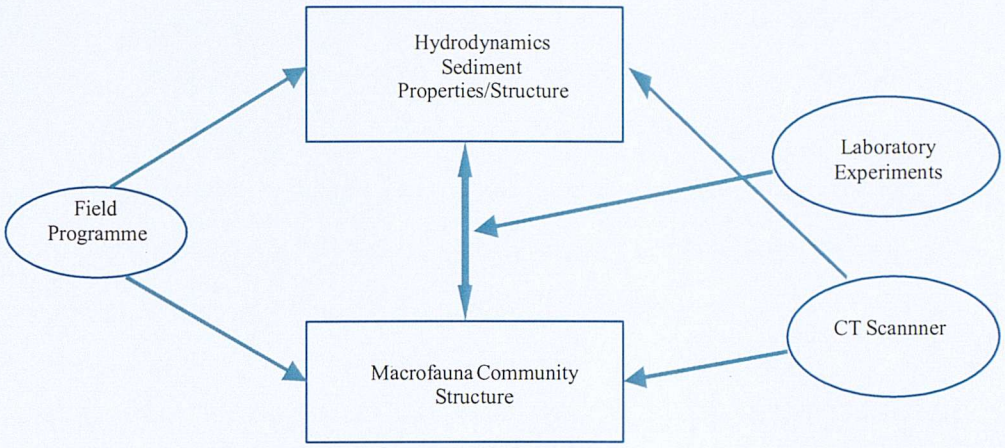


Figure 1.1: Flow diagram outlining principal components of the research programme

Permission was obtained by ABPMER (Associated British Ports Marine Environmental Research) to carry out a trial recharge on the intertidal mudflat at Hythe, on the western shore of Southampton water on October 11th 2000. The recharge involved use of hoppers to discharge 1000 m^3 of dredged material taken from the navigation channel, which was intended to 'cap' the mudflat. One of the intentions of this study was to capitalise on this activity by monitoring the response of the tidal flat community as well as its behaviour leading up to the recharge.

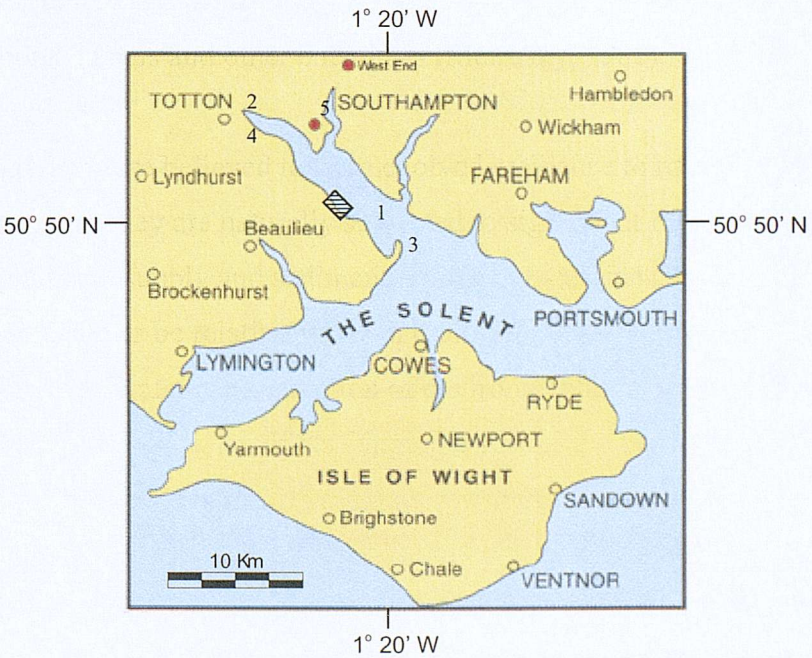


Figure 1.2 Map showing location of Southampton Water & Hythe, Hampshire. Hythe is shown by the rectangle on the western side of the estuary. 1 = Southampton Water, 2 = Redbridge Viaduct, 3 = Calshot Spit, 4 = River Test, 5 = River Itchen.

The town of Hythe is situated on the western shore of Southampton Water between the area known as Dibden Bay, and Fawley (indicated by the rectangle in Figure 1.2). The field site at Hythe is located on an area of intertidal mudflats, approximately 200 ha in total, bordered by saltmarsh to the west, Southampton Water to the east, Hythe Town to the north and Fawley to the south. For a local map and full details of sampling location see Figure 2.1 and the site description in chapter 2.

Understanding the processes underlying bed stability and the erosional processes of intertidal mudflats is of great significance (and economic importance) to the authorities and personnel involved in coastal zone and port management, to coastal protection and the implications of the effects of climate change, and to habitat and species conservation. It is also of interest both to ecologists in understanding both the role of biological communities in modifying their environment, and to sedimentologists and physical oceanographers as a major parameter essential to the construction of physical and numerical models for sediment transport and estuarine processes.

With such complex arrays of interactions, it has been recognised by biologists that simple ecological measures alone are inadequate to explain the cause of a species' distribution, spatial and temporal heterogeneity or community dynamics of macrofauna in intertidal soft sediments. Multidisciplinary research, incorporating both sedimentological and ecological methods, is thus required to improve understanding of these environments and their dynamics, improve the accuracy of existing mathematical models for sediment transport and habitat management and obtain a clearer understanding of the processes determining community structure.

An extensive literature search indicated that the effects of recharge on macrofauna vary greatly between studies (Drucker, 1995; Johnston, 1981; Levings *et al.*, 1985; Lockwood, 1985; Maurer *et al.*, 1986; Maurer *et al.*, 1985; McLusky *et al.*, 1989; Newell *et al.*, 1998; Ray *et al.*, 1995; Rees *et al.*, 1992; Roberts *et al.*, 1998; Rowlatt *et al.*, 1986; Rowlatt & Limpenny, 1987; SOAEFD, 1996; Valdes-Cogliano *et al.*, 1988; van Dolah, 1979; Wildish & Thomas, 1985). This finding suggests that site specificity is of major importance, and that accurate predictions of impacts of future operations at different sites cannot be made with confidence without detailed prior, site-specific investigations. The nature of previous studies, in terms of duration of sampling period

and time lapse between data collection, methodology, and analysis and interpretation of data, varies significantly. It is, therefore, difficult to make comparisons between results for different sites, and these factors probably account for some of the variation in conclusions reached.

1.2 Research framework and literature review

Prior to the 1990's, the traditional approach to research into intertidal mudflats has tended to divide the biological and physical processes. This has led to the biological processes being largely ignored by oceanographers studying sediment dynamics, and vice versa. As knowledge of these complex environments has expanded, it has been recognised that the physical processes studied by sedimentologists and physical oceanographers are significantly affected and modified by biological processes (Paterson, 1989). Intertidal mudflats are shaped by a combination of abiotic and biotic factors and interactions, which in turn are controlled directly or indirectly by the hydrodynamic regime. Overall, sediment supply and removal, and the type of sediment present (i.e., clay, silt etc.) are determined by the site hydrodynamics in terms of current and wave processes (Mehta, 1989). Also, supply of nutrients (and food resources) and planktonic larvae from the water column are at least partially controlled by hydrodynamic flows, which in turn affect feeding, productivity and recruitment of the biota (Snelgrove & Butman, 1994). Therefore, it has long been established that physical factors (in addition to chemical factors such as toxicity) to some extent determine and control the assemblages of organisms present. However, community structure, species richness and biodiversity are also affected by biotic interactions between organisms such as inter- and intraspecific competition for food and resources, predation and population dynamics of prey (Wilson, 1991; Woodin, 1974) further adding to the complexity of these systems. This leads to a requirement for studies that attempt to bridge the traditional divisions between disciplines. Therefore a review of a range of topics is presented here, including sediment dynamics of intertidal mudflats (and cohesive sediments); the background ecological theory and research into mudflat communities; animal – sediment interactions and relationships, and the effects of biological communities on intertidal cohesive sediments; the effects of dredge spoil disposal on biological communities – including environmental impact type case studies, laboratory studies investigating the effects on individual species; impacts on sedimentology and environmental or physical/chemical conditions, and finally the

results of previous research conducted in Southampton Water of relevance to this study.

1.2.1 Intertidal mudflats

Intertidal flats are an area of the shore periodically exposed and covered by the tides, i.e. they incorporate the area between extreme high water spring tide level (EHWSTL) and extreme low water spring tide level (ELWSTL). Intertidal flats can be broad expanses of sediment, either predominantly mud (clays and silts), sand or a mixture of the two, sometimes several kilometers wide "characterised by a relatively shallow gradient in elevation (hence the term 'flats'), and often intersected by creeks or channels" (Dyer, 1998). Intertidal mudflats are composed of a mixture of mud and sand, with the mud content (silt and/or clays) sufficiently high for the sediment to be cohesive in nature. They are bordered by lower lying sandflats and the subtidal zone (main water body) at the lower margin, and are often fringed by a vegetated area such as a saltmarsh at the upper margin (Evans, 1965). Occurring along estuaries, deltas, lagoons or open coastlines, intertidal mudflats fringe thousands of kilometres of coastlines worldwide forming a protective barrier against inundation (Dyer, 1998). Although intertidal mudflats are depositional environments and are often intuitively thought of as being relatively sheltered, they are still very much exposed to the influence of currents and wave action (see below for a discussion of mudflat evolution or progradation). Detailed definitions and classifications of tidal flats are given by Amos (1995) and Dyer (1998).

Research on mudflats has been limited in extent and nature due to difficulty of access onto such soft substrates and the fact that direct observation tends to destroy the features being investigated (Dyer, 1998). However, a growing volume of published literature on intertidal mudflats provides information about various aspects of these environments including their morphology and typology, sediment properties and mudflat processes such as progradation, erosion and bed stability. Published literature is also available on instrumentation and methodology such as flumes and erosion threshold measuring devices (e.g. Black & Paterson, 1987; Nowell & Jumars, 1987; Schunemann & Kuhl, 1991; Tolhurst *et al.*, 1999). Many of the earlier research papers focusing on individual processes originate from laboratory studies. These were often carried out using abiotic 'model' sediments such as potters' clay and tended to ignore

the role of biology entirely. It is now generally accepted that within mudflats there is a close coupling between the hydrosphere, geosphere, atmosphere and biosphere, and that biological communities and organisms play an important role in shaping and modifying the sediment in which they live. For example, multidisciplinary research on the role of microphytobenthic communities in stabilizing the sediment surface has shown that, in some cases, these organisms may increase the measured resistance of surface sediments to erosion by up to several orders of magnitude (Paterson, 1979, 1994). These realisations have, more recently, led to the development of large scale collaborative projects on intertidal mudflats, funded by the EU, which are based around multidisciplinary field campaigns and *in situ* data collection such as INTRMUD and LISP(UK) (Black, 1996).

1.2.2 Mudflat processes

Since the 1950's, studies on the formation and progradation of mudflats (e.g Evans, 1965; Evans & Collins 1975; Van Straaten, 1957) have addressed questions on the mechanisms determining the creation and maintenance of mudflats in terms of sediment supply and transport, deposition and accretion, and the role of features such as creeks. From studies carried out on the extensive mixed tidal flats of the Wash (North Sea), Evans (1965, 1975) proposed vertical accretion dominates on the marshes and upper flats, and lateral progradation dominates on the lower flats. Much debate took place, for example, Kestner (1975, *in* Amos, 1995) contested theories of continuous progradation. He suggested that growth was inhibited at the lower margins by the low water channel, and that it was only in the presence of a sedimentation *umbra* caused by intervention such as marsh reclamation or channel entrainment through engineered structures that accretion occurred. This was disputed by Amos (1995), who pointed out that this conclusion must be flawed to some extent or tidal flats would never have developed to begin with, although there is evidence to support Kestner's view in that progradation has been shown to be dominated by marsh reclamation. The conclusion reached by Amos (1995) was that intertidal mudflats prograde and are maintained by a balance between sediment supply and 'accommodation space'. This occurs through the process of settling and scour lag – i.e. "The product of a dynamic balance between landward transport due to tidal asymmetry and seaward dispersion due to the resulting suspended sediment concentration gradient" (Amos, 1995). The impacts of engineered structures behind tidal flats such as

causeways and seawalls were also investigated (e.g. Amos & Mosher, 1985; Evans & Collins, 1975). It was discovered that engineered structures caused profound effects such as the disappearance of 'upper flats' (Dyer, 1998), or the rapid development of entire mudflats as occurred on the seaward side of a solid-fill causeway at the mouth of the Avon River in the Minas Basin (Amos & Mosher, 1985).

It is now recognised that the processes shaping and 'controlling' these dynamic and sensitive environments are extraordinarily complex, and that in order to fully understand them, a comprehensive knowledge and understanding of the underlying processes is required. These include processes traditionally studied by physical oceanographers and sedimentologists (the transport of cohesive and non-cohesive sediments, deposition, consolidation, exposure, erosion, bed stabilisation, re-suspension, the impacts of wave action, currents and combined wave-current activity), as well as the biological processes involved (including biostabilization, bioturbation, and biodeposition), traditionally the domain of biologists and ecologists. In addition feeding and foraging behaviour by animals at the upper end of the food chain such as fish (during inundation) and birds (during exposure) have been found to influence tidal flat stablility (e.g. Daborn *et al.*, 1993; Evans *et al.*, 1999). As one of the foci of this thesis, the processes of erosion and stability of intertidal mudflats and the biological processes involved are reviewed below in more detail.

1.2.3 Erosion and bed stability

The processes and factors influencing bed stability and erosional processes are numerous, extremely complex and incompletely understood (Amos, 1995; Jumars & Mehta, 1989; Nowell, 1984; Nowell *et al.*, 1981; Paterson, 1997). They can be broadly split into four categories: 1) External physical processes (and factors) such as tidal exposure, inundation, desiccation through exposure to wind and sun, rainfall and freshwater influxes (Amos *et al.*, 1988; Anderson, 1983; Daborn *et al.*, 1993; Paterson, 1989; Paterson & Underwood, 1990; Paterson *et al.*, 1990); 2) hydrodynamic processes, including wave and current interactions (e.g. Mehta, 1989); 3) sedimentological factors such as grain size and related cohesive or adhesive forces, organic content, bulk density (water content) and the temperature and salinity of pore waters (e.g. Dyer, 1986), and 4) biological processes such as the production of EPS (extra-polymeric substances) and filamentous structures from microphytobenthic

organisms (diatoms, filamentous algae, cyanobacteria) living on the sediment surface (Daborn *et al.*, 1993; Paterson, 1989a,b; Paterson, 1994; Paterson *et al.*, 1999b), the production of mucus and reinforced tube walls (biostabilisation) by benthic infauna (bivalves, polychaetes, amphipod crustaceans, meiofauna) and bioturbation through the feeding and burrowing of infauna (Daborn *et al.*, 1993; Davey & Partridge, 1998; Grant & Daborn, 1994; Jones & Jago, 1993; Meadows *et al.*, 1988; Meadows & Tait, 1989; Widdows *et al.*, 1988; Willows *et al.*, 1998).

Bed stability, as a focus of such studies, is usually measured as the resistance of the sediment (surface) to erosion or erodibility (e.g. Amos, 1995; Black & Paterson, 1997). Erosion occurs when the fluid forces (lift and drag) exceed the combined effects of gravity and frictional and cohesive forces between the grains in the sediment (Collins, 1989). The eroded (sedimentary) material is removed in the form of flocs or grains from the bed to be transported as bed load or suspended load. Two parameters commonly measured in studies investigating bed stability are erosion threshold (or the critical threshold for erosion, τ_c) and erosion rate (E) (Amos, 1985; Black & Paterson, 1997).

Numerous (and often incompatible) definitions of erosion threshold have arisen, for example: Ten or more inorganic grains move at the same time (Heinzelmann & Wallisch, 1991); both organic material and inorganic grains move (Madsen *et al.* 1993); resuspended sediment leads to a substantial reduction (e.g. 10%) in light transmission (Paterson, 1989). Inconsistencies and subjectivity also arise from the variety of methods and instrumentation used to measure erodibility (Sutherland *et al.*, 1998). For example, use of the Cohesive Strength Meter (CSM), a portable, *in situ* erosion device (see section 1.2.4 for a review of instrumentation), facilitates erosion threshold being defined as reduction of light transmission below 90 % (after Paterson, 1989).

Erosion rate can simply be defined as the quantity of sediment erosion taking place over unit time. Many of the instruments mentioned below, in particular annular flumes, can be used to obtain measurements from which erosion rates and erosion thresholds can be calculated. The former is often considered to be the more sensitive parameter

(e.g. Sutherland *et al.*, 1998). However, erosion rate has been found to vary as a complex function of the applied bottom shear stress and cannot be represented by a single coefficient; it is related to the excess bed shear stress rather than the bed shear stress *per se* (Amos & Mosher, 1985). Studies that quote results as erosion rates from these measurements are derived from time-averaged values. Units and methods for the measurement and calculation of both these parameters are discussed in Chapter 5.

Mehta & Parthenaides (1982) proposed two types of bed erosion; surface erosion (Type I) and bulk erosion (Type II). Type I erosion is a self-limiting process which decays with time as deeper sediments with a higher yield strength become exposed. It has been equated with either the breakdown of primary floc bonds in the surface zone under hydrodynamically turbulent smooth flows (Type Ia), or the erosion of secondary bonds by drag and pressure ejection under turbulent rough flows (Amos *et al.*, 1992). In contrast, Type II erosion is chronic erosion which remains constant with time (as long as the shear stress exerted on the bed remains greater than the sediment shear strength). Type II erosion was defined by Amos *et al.* (1992) as the failure of the horizontally laminated microfabric of the consolidated bed.

The resistance of muds to erosion is highly complex, and cannot be readily predicted from physical properties. By contrast the relationship between grain size and threshold of erosion velocity for non-cohesive sediments can be readily calculated using a standard method such as the Shield's Curve (Dyer, 1986; Widdows *et al.*, 1998). Research into these processes has been carried out through both laboratory studies and field campaigns to obtain *in situ* data. Laboratory studies for obtaining measures of 'erodibility' are carried out on re-formed (and often abiotic) beds, using either natural sediment or a substitute, such as potters clay, or on sediment samples transported from the natural environment. The advantages of laboratory studies are that detailed investigations of physical and hydrodynamic processes are possible through closely controlled manipulation of environmental parameters. They also allow investigation at timescales not possible in intertidal environments where access is often severely restricted. The disadvantages of laboratory studies are that re-formed and abiotic beds vary greatly from natural mudflats and react quite differently so that results cannot be extrapolated to natural conditions. Such studies are nonetheless, of great use in increasing understanding of individual processes and hydrodynamic interactions which

are too complex to observe in the field (e.g. Amos & Mosher, 1985). Sampling and transport inevitably disturbs and disrupts the sedimentary and biological processes (Black & Paterson, 1997; Grant *et al.*, 1990; Nowell & Jumars, 1997), although techniques have been developed to minimize this disturbance, and some success has been reported (Tolhurst *et al.*, 2000). It is generally agreed that some disturbance is unavoidable and great care must be taken when extrapolating results from such investigations to *in situ* conditions. However, this type of study is of great value as it represents a kind of interim between field studies in terms of both the level of control and access it provides. Laboratory flume or microcosm/mesocosm experiments are undoubtedly of use in the investigation of biological processes, invertebrate behaviour and interaction with sediment processes (Meadows & Tait, 1989; Sutherland *et al.*, 1998; Willows *et al.*, 1998; Widdows *et al.*, 1998). A review of some of the results obtained from this type of study is given below (section 1.2.5).

Given the above problems with laboratory investigations, field studies are desirable to investigate stability and properties of natural sediments. The main limitations are likely to be access and cost. Large instruments like the Sea Carousel (Amos, 1992) can only be deployed by ship and are generally not suitable for use in shallow intertidal areas. This led to the development of smaller portable instruments such as the Mini Flume, the Plymouth annular flume and the CSM (Cohesive Strength Meter) (Black & Paterson, 1997). However, access can still be a major limitation as the lower intertidal shore may only be exposed for a few hours a month. There are a growing number of sedimentological field studies investigating sediment erodibility (Amos *et al.*, 1992; Black, 1996; Cappucci *et al.*, 2000; Day, 2000; Friend *et al.*, 1999; Tolhurst *et al.*, 1999; Tolhurst *et al.*, 2000). Field studies of the role of biological processes in determining sediment stability and erosion can be divided into those involving diatoms and other micro algae (e.g. Paterson, 1989; Paterson *et al.*, 1990; Tolhurst *et al.*, 1999), and those focusing on the macrobenthos or infauna (macrofauna). The latter include a couple of isolated *in situ* studies involving the application of biocides (Faas *et al.*, 1993; Grant & Daborn, 1994), a portable *in situ* flume (Widdows *et al.*, 1998; Widdows *et al.*, 2000), and a field study using geophysical techniques (Jones & Jago, 1993). Sediment-animal interactions and the influences of infauna on erosion and bed stability are discussed more fully in section 1.2.5 below.

1.2.4 Sediment – animal interactions

Most previous multidisciplinary studies, incorporating both the sediment dynamics and the biology, have largely focused on the microphytobenthos and stabilization of the top layer of sediment (Black, 1989; Paterson, 1989 a,b; Paterson *et al.*, 1990; Paterson, 1994; Paterson & Black,1999 a,b). Studies investigating the impacts of macrofauna on soft intertidal sediments have focused on processes such as bioturbation, biodeposition and biostabilization. Organisms are traditionally placed into functional groupings based on their feeding habits and lifestyle, i.e. deposit versus suspension feeders, scavengers versus predators, burrowers versus tube builders etc. However, it is now recognised that many of the common intertidal species exhibit what is termed as plasticity in feeding modes and can switch between deposit and suspension feeding (Jumars & Nowell, 1984; Snelgrove & Butman, 1994). It has also been found that the impact of these organisms on their environment varies according to population density (Jumars & Nowell, 1984).

The organisms present in soft sediments, including intertidal mudflats, affect and modify their environment, in terms of both sediment structure and, to some extent, the flow regime at the bed surface (Jones & Jago, 1993; Nowell & Jumars, 1984). Sediments are reworked and redistributed by burrowing organisms, releasing trapped nutrients and causing oxygenation of buried material (Watling, 1991). Surface sediments are disturbed by surficial deposit feeders in a process known as bioturbation which has been found to decrease sediment stability (e.g Widdows *et al.*, 1998; Willows *et al.*, 1998). Biodeposition occurs through the production of faecal pellets (Widdows *et al.*, 1998). Extra-cellular polymers (EPS) in the form of mucus produced by macrofauna during tube and burrow construction, feeding and movement are known to increase the inherent shear strength of the sediment and therefore, the erosion threshold through biostabilization (Meadows & Tait, 1989). The presence of macrofauna tubes and burrows, constructed by species such as the amphipod crustacean *Corophium volutator*, have also been found to alter flow structure at the sediment surface, and at certain densities will reduce the shear stress exerted on the bed through a skimming effect (Grant & Daborn, 1994). Grazers such as *Hydrobia* may indirectly reduce erosion threshold by the removal of the diatom biofilms, which stabilize the sediment surface (Grant & Daborn, 1994). These interactions are complex, and it has been found that a single species of macrofauna may act to either

increase or decrease bed stability (Snelgrove & Butman, 1994). For example, *Corophium* burrows may decrease shear stress and thus erosion, however, feeding by this animal may also have a negative effect on bed stability through removal of diatom cover, and disturbance of surface sediment (Grant & Daborn, 1994). *Nereis diversicolor,* a common estuarine polychaete (Annelida), has been found to switch between predation on other fauna, scavenging at the sediment surface, deposit feeding and suspension feeding depending upon the type and amount of food resources available (Meadows & Tait, 1989; Scaps, 2002). Thus, several effects are possible including bioturbation, biodeposition, and the removal and/or suppression of both bioturbating and biostabilizing organisms through predation, competition and disturbance.

1.2.5 Effects of dredge spoil disposal on biological communities

The literature on the effects of dredge spoil disposal on benthic communities and their environments was reviewed (Drucker, 1995; Johnston, 1981; Levings *et al.*, 1985; Lockwood, 1985; Maurer *et al.*, 1986; Maurer *et al.*, 1985; McLusky *et al.*, 1989; Newell *et al.*, 1998; Ray *et al.*, 1995; Rees *et al.*, 1992; Roberts *et al.*, 1998; Rowlatt *et al.*, 1986; Rowlatt & Limpenny, 1987; SOAEFD, 1996; Valdes-Cogliano *et al.*, 1988; van Dolah, 1979; Wildish & Thomas, 1985). Most previous studies were focused on the sublittoral zone. Sampling sites were located in open coastal waters where environmental and biological conditions vary significantly from those found in the relatively sheltered environment of an intertidal, estuarine mudflat. These studies showed no consistent trends, indicating that the overall variation is greater than can be accounted for by differences in methods alone. This suggests that the effects of recharge on infaunal assemblages may only be predicted in a site-specific manner, if at all.

The disposal of dredged material into the oceans in recent decades is considered to be the largest mass input of waste substances into the system (Collins, 1989). 70 – 75 % is believed to be deposited on wetlands and nearshore areas. The scale of these operations alone highlights a need for better understanding and capacity for prediction of environmental impacts if significant potential environmental damage is to be prevented. The shorelines of estuaries are highly dynamic environments subject to wave action and tidal currents which can resuspend and transport deposited

sedimentary material, therefore a fundamental consideration for recharge is retaining the material at the chosen disposal site. In order to understand and predict the movement of fine dredged material during dredging operations and following disposal in a coastal environment, it is essential to consider fundamental principles of sediment transport under currents, waves and combined flows and to characterize the hydrodynamic regime at the site. This is no small undertaking, involving considerable cost, time and expertise (Collins, 1989).

Intertidal recharge offers an alternative to expensive offshore disposal, which leads to future problems associated with the removal of large amounts of sediment from the system (Ray *et al.*, 1995). In the case of capital dredging operations, millions of tonnes of sediment may be removed as a one-off event (e.g. during port construction/engineering). Recharge may then become particularly attractive if the environmental loss of, for example, wetland or mudflats used by feeding bird populations, may be potentially ameliorated or compensated for by the restoration or creation of suitable habitat further along shore through the 'capping' of polluted sediments with less polluted material (Ray *et al.*, 1995).

The impacts of recharge and/or dredge disposal activity on individual benthic species have received relatively little attention. Experimental studies [Brenchley (1981), Chandrasekara & Frid (1998), Chang & Levings (1978), Maurer *et al.* (1980, 1981, 1982, 1985, 1986), Nichols *et al.* (1978) and Turk & Risk (1981)] focus on just a few taxa of which only one species, *Hydrobia ulvae*, is found at Hythe. Past studies show that mass deposition of sediment has deleterious effects on benthic fauna. The results suggest real variation, not just among species and ecological functional groups but also between the different sediment loads and types. The need to consider each case (and site) individually, and the difficulty of predicting the effects of recharge is clearly demonstrated.

The intra-specific variation obtained from these studies suggests the presence of 'tolerance thresholds'. For example, the Dungeness Crab (*Cancer magister*) appeared to survive burial in a 10 cm depth of sediment without apparent harm, but none of the experimental animals survived a 20 cm load (Chang & Levings, 1978). Nichols *et al.* (1978) found that most animals within a soft bottom community, including the

polychaete *Mediomastus ambiseta* and the bivalve *Nucula annulata* (details of other taxa were not given), could escape burial under 5 to 10 cm of sediment, but none attempted to crawl up though a burial layer 30 cm deep.

Maurer *et al.* (1981) undertook a series of experiments investigating the response of a range of benthic species to burial with simulated dredged material (the bivalves *Mercenaria mercenaria*, *Nucula proxima*, *Ilyanassa obsoleta*, the polychaetes *Scoloplos fragilis* and *Nereis succinea*, the mud crab *Neopanope sayi* and the amphipod *Parahaustorius longimerus*). They found that survival rates related to sediment depth, sediment type and temperature. Mortality generally increased with increased sediment depth, increased burial time, higher temperatures and non-native sediment (related to grain-size). Once again results were clearly species dependent. *Nereis* and *Mercenaria* were found to be the most resistant and *Parahaustorius* and *Scoloplos* the least resistant in silt-clay sediments, the order of resistance was reversed in 100% sand. The authors concluded that *Nereis* and *Mercenaria* were the least sensitive to the effects of oxygen depletion and increased ammonia concentration and *Parahaustorius* and *Scoloplos* the most sensitive.

A study that investigated the impact of sedimentation (resulting from the construction of a causeway) on *Corophium volutator*, *Macoma balthica* and *Mya arenaria* also concluded that effects were dependent on depth and rate of deposition and grain size (Turk & Risk, 1981). *Mya* was able to burrow upwards in accumulating coarse sand for which an LD50 was calculated at 24 cm, however, this decreased to 6 cm in fine sand and 3 cm in mud. *Macoma* was found to tolerate measured sediment rates of up to 10.2 cm per month without adverse effects, however, *Corophium* was found to be highly sensitive and just 2 cm of deposition per month had a negative effect on population density for this species. This was partially attributed to increased water content (associated with a shift in grain size) that was thought to inhibit the construction and maintenance of burrows, although it was concluded that this species was sensitive to sedimentation regardless of sediment type (Turk & Risk, 1981).

Experiments have also been carried out to investigate survival and vertical movement of the epibenthic gastropods; *Hydrobia ulvae* and *Littorina littorea* following burial under 5 cm of sediment (Chandrasekara & Frid, 1998). These species, both known to

have limited burrowing ability, were unable to regain the surface following burial in natural sediment (classified as high silt, low water content). It was found that *Hydrobia* regained the surface within a day in fluidised sediment (i.e. with an increased water content) due to the ability of this species to float. However, the snails were found to be inhibited by higher temperatures (20.3 $^{\circ}$C), which was attributed to oxygen stress Chandrasekara & Frid, 1998).

Further research is necessary to enable prediction of the impact of recharge activities on both benthic communities and their environment in terms of sediment dynamics and properties, particularly where the intention is to predict the effects of recharge for a particular site and a specific assemblage of fauna.

1.2.6 Sedimentological Properties

Bulk density and water content are used by oceanographers and biologists to describe sediments as they are of vital importance to organism and sediment behaviour. Wet bulk density (BD_w) reflects the relationship between the total water-saturated mass (M_t) and the volume of the water-saturated sample (V_t); $BD_w = M_t/V_t$ (after Flemming & Delafontaine, 2000). (Note, for comparison that dry bulk density reflects the relationship between the mass of the dry solids and the volume of the sample). Water content refers to the mass of the pore water in a sample, expressed as a percentage of the total water saturated sediment mass (Flemming & Delafontaine, 2000). Both bulk density and water content are therefore dependent on the ratio of sediment/particulate matter: interstitial water/solutes. In the past, bulk density has tended to be measured and utilised in the fields of sedimentology, engineering and sediment dynamics, whereas water content has been the parameter more commonly referred to by biologists. In an intertidal mudflat it is expected that, generally speaking, bulk density increases (and thus water content proportionally decreases) into the sediment as consolidation takes place (Amos, 1995). Bulk density is therefore a good measure of the consolidation of the sediment as consolidated sediments are more stable. An increase in density is often correlated with a decrease in erodibility (Amos *et al.*, 1998). It has been found that the top layer of unconsolidated sediment seasonally increases during the spring and summer (i.e. a decrease in bulk density/increase in water content) and decreases in winter where erosion tends to be the dominant process (Amos *et al.*, in press). This (seasonal) dominance of depositional processes gives rise

to an increased unconsolidated surface layer consisting of pseudofaeces and faecal pellets referred to as the 'fluff layer' (Amos, 1995). Dense populations of infauna which result in bioturbation may be expected to decrease the bulk density in the macrobiotic layer of the sediment (e.g. Willows *et al.*, 1998).

Significant changes in bulk density of the layer of sediments sampled (top 10 - 15 cm) may occur following recharge. Natural episodic events initiating erosion or deposition can also effect the bulk density, and this may be indirectly affected to some extent by fluctuations in biofilm development (Amos *et al.*, in press; Paterson, 1989). Major changes in bulk density (or water content) may also affect the burrowing behaviour of infauna (Nichols *et al.*, 1978; Maurer *et al.*, 1986). For example, Nichols *et al.* (1978) observed that a significant increase of bulk density appeared to inhibit burrowing activity in some species, believing the effect was due to compaction and increased cohesion of sediment. It is also possible that a significant decrease in sediment density may inhibit or effect burrowing in some species.

1.2.7 Environmental sediment conditions or physico-chemical parameters in Southampton Water

It is expected that temperature will vary on a seasonal basis as the area studied has a temperate maritime climate. Good (1996) recorded a range of fluctuating in-sediment temperature from 6 – 21 °C at Hythe between March and August. Temperature would also be expected to vary at the site on a diurnal basis and possibly more frequently due to tidal exposure. However, the infauna would not be exposed to such rapid temperature changes as occur on the surface of the sediments. Temperature is not predicted to be a major influencing factor in the case of recharge, as the material deposited would be taken from nearby and the difference in depth between the source and disposal site is not considered to be great enough to cause a gradient which would effect the fauna.

Salinity in Southampton Water ranges from 10 psu at the estuary head (Redbridge Viaduct) to 33.25 psu at the estuary mouth (Calshot Spit). However, salinity rapidly increases downstream, reaching 30 psu just 2 km from the estuary head (NERC, 1980). At Hythe, water salinity averages at around 30 psu, being marine rather than brackish or mesohaline (ABP, 2001). Salinity may be expected to increase during dry

weather due to reduced inputs of freshwater from the Rivers Test and Itchen, and decrease during periods of heavy rainfall. Fluctuations in salinity may also occur due to tides, however, the infauna will be protected to some extent from these fluctuations due to the 'buffering' effect of the sediments (McLusky, 1981). The salinity range found in Southampton Water may therefore be expected to support both true estuarine (euryhaline) and marine (stenohaline) species, typical of coastal infauna. It is expected that the distribution of sessile species, with limited osmoregulatory abilities (i.e. not euryhaline) within the estuary will be governed by the salinity range in the vicinity.

Cohesive sediments generally have a very thin oxygenated layer (several mm deep) close to the surface, due to the relatively high organic content and related high bacterial metabolism and the low diffusion rates of oxygen in water during emersion of the intertidal flat (Barnes & Hughes, 1988; Gray, 1981). This is true of the mudflat at Hythe where the sediment has a high clay/silt content and a relatively thin unconsolidated surface layer (Dransfeld, 1999; Day, 2000). Below this top layer the sediment rapidly becomes anoxic, except in localised areas due to the activities of tube builders and burrowers (Fenchel, 1996). On a temporal scale, fluctuations in oxygen concentrations may occur in relation to tidal coverage and temperature, due to the higher diffusion rate of oxygen in air than water and the higher saturation capacity of dissolved oxygen in water at low temperatures. Small-scale fluctuations may also occur which correspond to photosynthetic activity in the microphytobenthos (biofilms of microalgae such as diatoms on the sediment surface), algal blooms, and in relation to bacterial activity (e.g. seasonal succession due to fluxes in inputs of organic material from the saltmarsh) (Paterson, 1994, 1999b).

Significant changes in oxygen levels within the sediments are likely to occur in the event of recharge due to increased BOD (biological oxygen demand) and/or COD (chemical oxygen demand) in dredged material from increased levels of organic content and/or pollutants (Johnston, 1981; Rowlatt & Limpenny, 1987). Additionally, natural disturbances such as episodic sedimentation from storm discharge may cause a decrease in oxygen content of the biotic layer if a significant input of organic matter or smothering occurs (Diaz & Rosenberg, 1995).

The pH of cohesive sediment generally decreases with depth to the redox discontinuity layer (a zone of rapid transition in redox potential (Eh) between aerobic and anaerobic decomposition) along a gradient from oxygenated to reducing conditions. pH values typically quoted for littoral sediments range from approximately 7.5 at the surface to 6.5 at the redox discontinuity layer (Barnes & Hughes, 1988; Fenchel, 1996). Significant fluctuation in pH may occur following recharge subsequent to a decrease in oxygen concentration; for example, where die-off of organisms causes an increase in BOD and subsequent increase in CO_2 (Johnston, 1981). Significant pH changes may also occur in the event of recharge through chemical effects arising from sediment contamination (Maurer, *et al.*, 1985).

1.2.8 Previous studies on local area

Whilst there is extensive literature published on the Solent and Southampton Water generally, studies that focus on intertidal zone benthos are limited. Published material on the intertidal macrobenthic communities of Southampton Water is limited to a paper investigating the impacts of industrial pollution (including the oil refinery at Fawley) (Houston *et al.*,1983). Seven sites, including Hythe, were surveyed annually from 1975 to 1979 using grab samplers. The authors concluded that the fauna was of low diversity and species richness, and was highly dominated by a few species, particularly on the western side of the estuary. The cirratulid polychaete *Caulleriella caputesocis* was found to be the most abundant species, which dominated most of the sites sampled, except for where it occurred as co-dominant with another cirratulid *Tharyx marioni*. Other species commonly recorded were ragworm *(Nereis diversicolor)*, the catworm (*Nephtys hombergii*), a third cirratulid, *Cirriformia tentaculata*, the spionid polychaete, *Polydora*, the ampharetid *Melinna palmata*, *Capitella capitata*, oligochaetes, and the bivalves *Cerastoderma* spp. (cockles), *Mercenaria mercenaria*, and *Abra nitida*. Unpublished data on the macrofauna of the mudflat at Hythe (see chapter 2 for site description and location) were also obtained, from a baseline survey carried out by ABP Research (internal, unpublished data). These results, collected in 1998 indicated very similar assemblages of species and dominants to Houston *et al.*, (1983) indicating that the community at the site had changed little since the late 1970's. The results of an unpublished M.Sc. thesis investigating benthic community structure at Hythe (Good, 1996) were also in general agreement with the above observations.

Further publications on the intertidal fauna of Southampton Water are limited to two studies of the life history and population dynamics of polychaetes (George, 1964; Onyenekan, 1987), a study on bivalves (Barnes, 1973) and a report on the introduced species *Mercenaria (=Venus) mercenaria* (Ansell, 1963). Barnes (1973) concluded that, with respect to bivalves, the fauna of the estuary conformed to the widespread *'Macoma balthica* community' *op cit.*, common to sheltered, muddy shores (although *Macoma* itself was not present), but with the addition of a number of introduced species. He noted that only three species of bivalve, *Cerastoderma edule*, *Cerastoderma glaucum* and *Mercenaria*, were abundant members of the infauna. This is in agreement with Ansell (1963) who states that *Mercenaria* was common at several locations throughout the estuary including the intertidal flats at Hythe.

The ecology of 'widespread' species found at this site, such as the cockle (*Cerastoderma edule* and *Cerastoderma glaucum*) and the ragworm *Nereis diversicolor*, is relatively well known (e.g. Boyden, 1969; Flach, 1996; Smaal & Prins, 1992; Trevor, 1977). By contrast, the ecology of the species found to be numerically dominant at the site (the cirratulid polychaetes *Caulleriella caputesocis* and *Tharyx marioni*) has received little attention, and understanding of their effects on the sedimentary environment is based on a general knowledge of polychaete functional ecology (e.g. Fauchald, 1983; Giangrande, 1997; Gudmundsson, 1985; Woodin, 1974). It should be noted that any extrapolation or prediction of effects borrowed from studies carried out on other (similar) taxa should be undertaken with extreme caution, as the polychaetes are an extremely large and diverse order which display remarkable plasticity in life traits and functional ecology (Fauchald, 1983; Giangrande, 1997; Gudmundsson, 1985). Likewise, the effects of environmental change on these communities through human activities are undocumented and can only be predicted through extrapolation from other environmental impact studies carried out on parallel benthic assemblages in similar environments. For example, these species are expected to be relatively tolerant to environmental stress due to their occurrence and dominance in the intertidal zone of an industrialised, polluted estuary. This type of environment, which is known to be a demanding and stressful one for its inhabitants subjecting them to constantly fluctuating conditions (e.g., temperature, salinity, oxygen, turbidity etc.), is also believed to be subject to regular small scale physical disturbances under natural conditions (Brenchley, 1981; Hall, 1994; Hall *et al.*, 1992). However, the effects of

the large-scale episodic disturbance that recharge would infer on the community at Hythe could not be predicted with confidence.

Literature in the field of sediment dynamics at the site consists of a Ph.D. thesis on benthic diatoms (Dransfeld, 2000) and two unpublished M.Sc. theses. The first of these is a study carried out on the erodibility of the intertidal sediments of Southampton Water (Day, 2000). The project was carried out as a comparative study between three different sites within the estuary, including Hythe. The data were collected during the field programme for the current study (see chapter 2) and are also complementary to the laboratory work in this study relating to bed stability (see chapter 5). The second of these dissertations was about the design and construction of an instrument to monitor the impact of ship wake and pore pressure fluctuations on sediment resuspension (Broughton, 2000). The author concluded that a significant amount of re-suspended, fine material present in the water column over the mudflat is a direct consequence of passing ship traffic. This finding suggests that shipping activity influences the behaviour and transport of material following recharge. Furthermore, it has implications for environmental effects on both shallow water communities and local sediment transport were such activity to be significantly increased i.e. following port construction.

1.2.9 Erosion measurement instrumentation and technology

The complexity of factors controlling tidal flat stability necessitates the use of innovative technologies and methodologies (Amos, 1995) to investigate these environments and their processes. Much time and effort has been devoted to the design and development of innovative instrumentation and technology to collect accurate and precise data both *in situ* through field measurements and in the laboratory. The range of instruments available includes relatively large benthic annular flumes such as the Sea Carousel and Lab Carousel of Amos *et al.*, (1992) and smaller portable erosion devices such as the CSM (Paterson, 1989), EROMES (Schunemann & Kuhl, 1991), and the Instrument for Shear Strength *In situ* (ISIS) (Williamson & Ockenden, 1996). The field flumes and other field instruments are all deployed onto intertidal mudflats during exposure, except for Sea Carousel, which is deployed from a vessel during high water. The scale of data collected is dictated by the instrument used: the larger flumes provide a more integrated response, however, they are affected by the topography of

the sediment (ridges and gullies). The smaller instruments can provide more detailed information on smaller spatial scales relevant to individual animals, microalgae and sediment properties. For reviews and comparisons of these technologies the reader is referred to Black and Paterson (1997), Sutherland *et al.*, (1998) and Tolhurst *et al.*, (2000a,b).

1.2.10 CT scanning

Computer Tomography or CT scanning is a technique that has been used for some years in medicine, and more recently in geosciences (Duliu, 1999; Wellington & Vinegar, 1987). It is a radiological imaging technique that measures density and atomic composition of opaque objects (Wellington & Vinegar, 1987). The scanner and accompanying software produces high resolution images which can be used to investigate 3-D structure and morphology of a wide range of materials including human/animal tissues, rocks, tree trunks and sediments (Duliu, 1999; Ketcham & Carlson, 2001; Wellington & Vinegar, 1987). The digital images which are produced from the attenuation of X- or gamma-rays, are visual representations of values of Hounsfield Units (HU) which are directly proportional to the linear attenuation coefficient of the sample (Kenter, 1989). CT data are often expressed as CT numbers, which are normalised values derived from HU's (Orsi, 1994). These values reflect sedimentary characteristics such as density and atomic composition, allowing comparative analysis of sediment properties such as bulk density, water content and biological structures such as burrows and tubes (Wellington & Vinegar, 1987; de Montety *et al.*, 2000).

As a technique for non-destructive investigation of geological cores and material, CT has major advantages over traditional methods such as sectioning and X-ray techniques. CT provides accurate spatial information on how properties vary throughout heterogeneous material on a sub-millimetre spatial scale and is extremely sensitive to changes in properties such as density; attenuation differences as small as 0.1 % can be measured accurately (Wellington & Vinegar, 1987). The thickness of a section represented on a CT image can be reduced to around 0.5 mm (Duliu, 1999), and the horizontal pixel resolution 0.2 mm, so the resolution of details is much higher than traditional (X-ray) radiographs. The technique is totally non-destructive, cores can be scanned without prior removal from liners or even whilst frozen, leaving

samples intact for further analysis or curation (Wellington & Vinegar, 1987). The technique is also extremely rapid, with the latest generation of scanners producing images in seconds. This enables large numbers of images of serial sections of a sample to be produced in a matter of minutes (and without the time consuming processes of sample preparation or sectioning required for more traditional techniques of core analysis). As the data produced are in digital format they can easily be subjected to quantitative image analysis (Ketcham & Carlson, 2001).

The main disadvantage of the technique is the cost of purchasing and installing/maintaining the equipment, although this can be overcome if use of a existing scanner can be arranged. Otherwise, CT has few limitations one of which is the inability of the human eye to distinguish between shades of grey. Previously problematic effects known as beam hardening and absorption edges effects are now automatically corrected for in the current generation of scanners, and artifacts can be minimized through optimal useage of equipment for image production by experienced personnel.

CT has been utilized in the geological sciences for investigations into paleontology, fluid flow, soil science, sedimentology, petroleum science, reservoir lithology, meteorology, mineralology of coral structures and water circulation surrounding plant roots (Duliu, 1999; Ketcham *et al.*, 2001). CT scanner studies involving investigation of marine sediment cores include those published by Amos *et al.*, (1996), Boespflug *et al.*, (1995) Holler & Kogler (1990) and Orsi & Anderson (1999). The data obtained can be calibrated and converted into actual bulk density (in kgm^{-3}) (for full details of methodology see chapter 6) (Amos *et al.*, 1996; Orsi & Anderson, 1999). CT numbers have also been closely linked to parameters such as grain size and organic matter (Boespflug *et al.*, 1995). The results of these studies imply that CT scanning provides a rapid and accurate method for the collection of high resolution data of sediment properties usually only available through labour intensive sediment analysis techniques.

Recently, the potential of this method has been recognised as a possible technique for the analysis of biologically generated sediment structures by benthic organisms. The use of CT for the investigation of animal-sediment interactions in marine sediments is

still very much in its infancy. The extent of work published so far is one paper, which forms part of a conference proceedings (de Montety *et al.*, 2000). However, several further follow-up studies are currently underway as part of an environmental impact study following a large scale flood on the Saguenay Fjord, (Quebec), in which CT was used to identify and quantify different types and levels of bioturbation along a disturbance gradient. Another study (from the same event/site), which uses CT to analyse temporal changes of benthic community structure, is currently underway (Michaud *et al.*, *in prep*). These authors report successful use of CT as a new technique for analysing benthic biological processes and report the potential advantages of its use as a rapid and accurate technique for quantitative analysis of animal burrow structures and bioturbation.

1.3.1 Aims of research and scientific hypotheses

The overall aims and related hypotheses of this study were as follows:

1) To monitor and investigate the impact of recharge on a macrofaunal community on an intertidal mudflat, its associated sediment and environmental properties and dynamics. 'Recharge' refers to the controlled and intentional mass deposition of sediments (usually a by-product of dredging) in a defined area as a management strategy for the purpose of environmentally non-damaging disposal, and/or habitat restoration or creation. A biological community is defined as an area containing an assemblage of different species in different proportions (Begon *et al.*, 1996). For the purpose of this study the term macrofaunal community describes the assemblage of polychaete worms, bivalve and gastropod molluscs, amphipod crustaceans and oligochaetes retained on a 0.5 mm sieve which live within the sediment of the mudflat, also referred to as infauna. Sediment dynamics refers to the processes of erosion, transport and deposition of sediment over various time and space scales. The effects of recharge are investigated both in the field (chapter 2) and through complementary manipulative laboratory experiments (chapter 4) where mass depositional events are simulated under controlled conditions.

   It was hypothesised that, during recharge, benthic macrofauna in the area of deposition would be subjected to physical disturbance (through smothering)

and chemical effects such as oxygen stress which would have deleterious effects on a proportion of individuals. Deposition of a layer of sediment several centimetres thick during recharge would effectively bury the underlying infauna, it was therefore hypothesised that long-term survival depends on the ability of individuals to reposition themselves to reach the sediment surface and regain access to oxygen and food. It is suggested that, as the migratory ability of fauna is related to individual morphology and life habits, the effects of recharge would be species dependent and that highly motile, active burrowers would be less impacted than more sedentary species. It was also hypothesised that survival would be related to burial depth, and that increased burial depth would increase deleterious effects on fauna. Two possible outcomes for the impacts of recharge on the community were suggested; that effects would be temporary and the community would return to its prior state during the days, weeks or months following recharge through recolonisation, or that impacts would persist, resulting in a modified community structure.

2) To investigate interactions between the benthic macrofauna and their immediate, surrounding sedimentary environment. Particular attention was given to the stability of the sediment – water column interface and the layer of sediments inhabited by the macrofauna. Potential relationships between macrofauna and bed stability, were investigated through laboratory experiments which provide quantitative measurements of erosion thresholds and rates. Knowledge of the relationships between the infauna and bed stability/erodibility at Hythe would provide a basis for predicting the effects of future recharge activities at the site. Stabilising or destabilising effects attributable to the benthos may be expected to influence the consolidation and therefore the potential erosion of the deposited material following recharge.

It was hypothesised that the infauna and their activities may bring about modification of the erosive properties of the bed, either causing net stabilisation (biostabilisation) or net destabilisation (through bioturbation). It was suggested that this effect may be density dependent (related to the number of individuals present per surface area unit of sediment).

3) To apply Computer Tomography (CT), a radiological scanning technique, to studies of sediment structure and properties such as wet bulk density, biological structures (animal burrows) and bioturbation within the sediment. Cores taken from Hythe both prior to and following the recharge were analysed using CT, to identify and map finescale changes in bulk density downwards through the sediment column, and provide information on the effects of recharge.

It was thought that the modification of sediment structure and its properties by infauna could be visually resolved and identified using CT, including individual burrows and the localised effects of these structures on sediment density. It was hypothesised that the activities of the macrofauna would result in a distinct bioturbated layer detectable by characteristic density, which, as a measure of consolidation relates to bed stability (and the experiments carried out in chapter 5). It was suggested that the effects of recharge on sediment structure, bulk density, faunal distribution and burrowing activities could also be identified using CT. It was hypothesised that recharge would result in the deposition of a distinct sediment layer characterised by relatively low density which would increase in bulk density with time following consolidation. The process and rate of consolidation may be modified through bioturbation by surviving or recolonising fauna. It was suggested that any significant changes in faunal distribution or community structure from recharge would be confirmed and their effects on sediment structure analysed using CT.

1.3.2 Scientific objectives

The overall aims of the research (above) are broken into the specific objectives listed below. They are divided into *in situ* fieldwork that incorporates the collection of background data and monitoring the effects of the recharge, and laboratory studies which include the controlled experiments on smothering and erosion experiments, and the CT scanner study.

The specific objectives of the field study were:

- To obtain data on macrofauna community structure at the site through a field sampling programme, including the collection of *in situ* physico-chemical data on

environmental characteristics such as temperature, salinity, oxygen content and pH of interstitial water.

- To obtain quantitative data and monitor changes in bed elevation at selected sampling stations at the field site.

- To collect quantitative data on the effects of recharge on the macrofaunal community and associated sediments at the site.

The specific objectives of the laboratory studies were:

- To design and build a self-contained microcosm facility, incorporating tidal simulation, for laboratory investigations.

- To use the above system to run manipulative, controlled and replicated laboratory experiments on animal-sediment interactions (outlined below) that could not be undertaken at the field site where access is severely limited.

- To examine macrofauna response to controlled smothering within the microcosm facility installed at ABP. For this purpose experiments were designed and run to investigate the effects of episodic deposition (smothering) on individual species, and to investigate the effects of smothering on natural mixed species assemblages.

- To examine the effect of the macrofauna community at Hythe on bed stability, and investigate interactions between sediment erodibility and macrofauna community structure. Experiments were carried out on sediment cores in order to investigate the effects of macrofauna density on erosion threshold and erosion rates.

- To compare and evaluate the CSM (Cohesive Strength Meter, Tolhurst *et al.*, 1999) and EROMES (an instrument for investigating the erosive properties of sediments, Quaresma *et al.*, 2002; Schunemann & Kuhl, 1996) as a means for measuring erosion threshold and erosion rates in the laboratory on 'natural' biological sediment.

- To investigate sediment structure using a CT scanner (Wellington & Vinegar, 1987), to map sediment bulk density (as a measure of sediment consolidation) on a fine scale (mm) using the data and images obtained from the scanner.

- To identify modification of sediment structure and properties (bulk density) by macrofauna (i.e. burrow structure and bioturbation) through the analysis of CT images.

- To compare cores sampled pre and post-recharge and identify any effects (of recharge) present using the CT images.

2 The results of a trial recharge dredge disposal experiment, Hythe, Hampshire

2.1.1 General introduction

This chapter describes the field investigation carried out as part of a trial recharge experiment carried out by ABP on the foreshore at Hythe, Southampton Water, England on 11th and 12th October, 2000.

The fieldwork programme was designed to collect background data on the site of the experiment, four months prior to recharge, and to monitor the site following disposal. This field site, chosen for sampling throughout this study, was determined by the intended location of recharge. Site location, methods, sampling technique and instrumentation, details of the experiment, and the results obtained from the field data are presented below. The findings of this research are discussed both in the specific context of the recharge experiment and in a more general ecological context.

Full details of additional results and data obtained from the experiment by ABP are given in their report (ABP, 2001).

The scientific objectives of this study were as follows:

- To collect baseline data and background information on macrofauna abundance and community structure, environmental parameters and bed elevation changes at the site in the months preceding recharge.

- To investigate the effects of the recharge on the benthic macrofauna at the site through a field-monitoring programme devised for this purpose.

- To identify and monitor environmental effects of the recharge at the site, which may affect the benthic community. For this purpose, *in situ* measurements of dissolved oxygen concentration, salinity, temperature and pH of the interstitial waters of the sediment were made.

- To monitor the effects of recharge on bed elevation at the site and identify sediment deposition (accretion) or erosion prior to and following the recharge experiment.

2.1.2 Site description - the field site on the intertidal mudflat at Hythe

The intertidal mudflat at Hythe (Figure 2.1) is situated between the military base to the north, and Fawley to the south (OS coordinates: SU435072: 443500,107200). It is bordered to the west by salt marsh, intersected by several creeks. The whole site covers approximately 275 ha of which about 200 ha is intertidal mudflats and the other 75 ha is saltmarsh. At the interface between the saltmarsh and the mudflat there is a vertical cliff approximately 1.5 m high. The intertidal flat extends seawards to the east for approximately 140 m (Figure 2.1). The tidal regime of Southampton Water is characterised by a double high water, lasting around 3 hours which is preceded by a young flood stand of up to 2 hours on spring tides and followed by relatively high ebb velocities due to tidal asymmetry. Sediment deposition over the tidal flats is facilitated by the young flood stand and relatively long periods of slack water during the double high waters (Price & Townsend, 2000). However, in common with much of the Southampton Water, the Hythe shoreline is undergoing net erosion overall, with a recorded salt marsh recession of approximately 2 m yr^{-1} (V. Quaresma - personal communication, 2003). Sea level rise of 4-5 mm yr^{-1} has been recorded in the estuary during the last century (compared to 1.2 mm yr^{-1} previously and globally), which has been ascribed to regional changes in the tidal regime (Cundy & Croudace, 1996). The sediment at Hythe is highly cohesive with a large proportion of clay, and is extremely soft in places. Relict clays have recently been exposed at the top of the flat as the saltmarsh is eroded, whereas the middle and lower flats are covered by modern deposits with a soft unconsolidated top layer. Seasonal growths of diatom films and red macroalgae (attached to debris) have been periodically observed at the site (predominately on the upper flats), and the surface is at times scattered with shell debris, primarily *Cerastoderma* (personal observation).

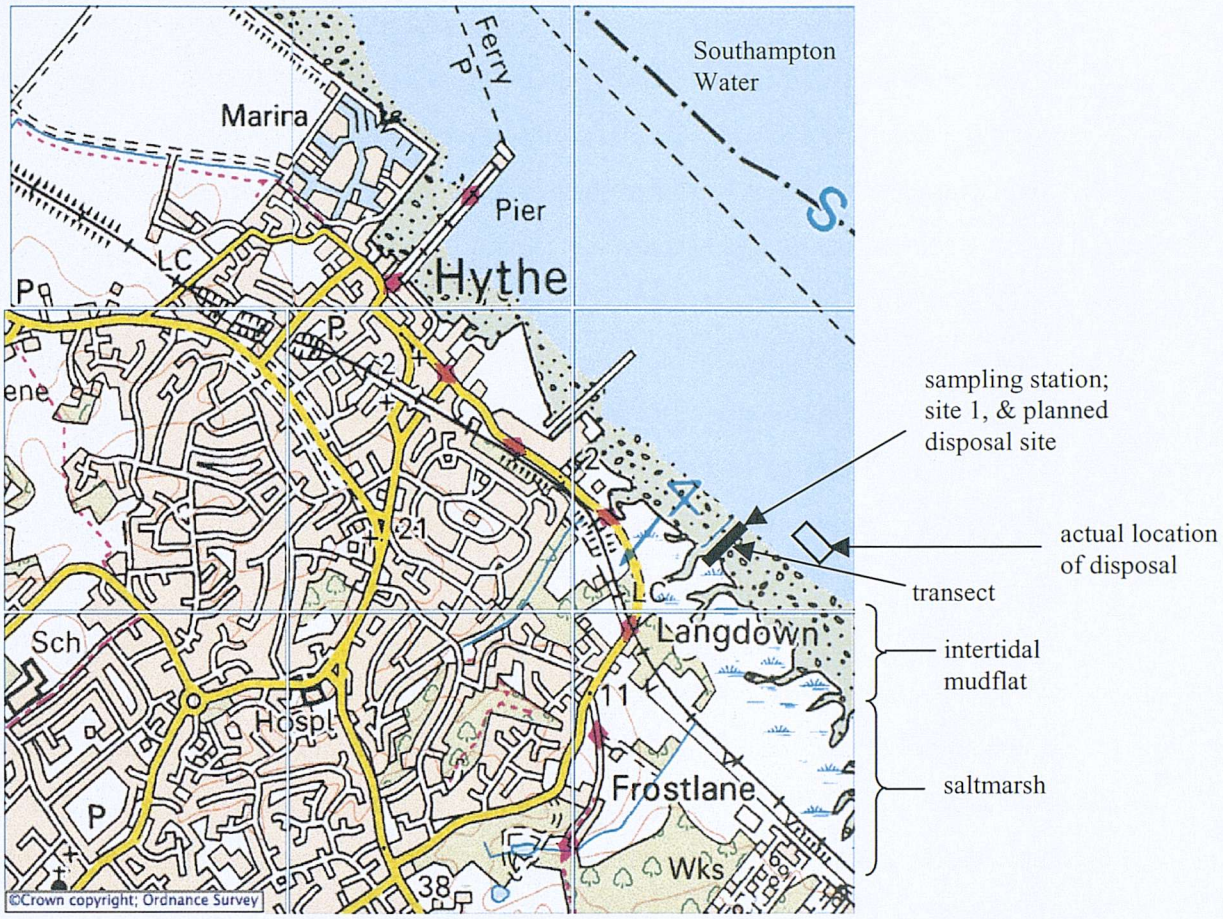


Figure 2.1 Map of Hythe town and waterfront showing locations of sampling station (site 1), transect (see profile, Figure 2.2) and intertidal mudflat. Scale: 1 square to 1 km, Landranger 196, 2002.

2.1.3 Recharge experiment at Hythe

On 11th and 12th October, 2000, 1050 cubic metres of dredged material taken from the mouth of the River Itchen (the largest river to drain into Southampton Water) was discharged onto the foreshore at Hythe. Due to restrictions imposed by water levels and the type of vessels used - split bottom barges for disposal (and a Bluefin suction hopper for dredging), the sediment was placed in five separate loads of between 100 m^3 and 256 m^3. The discharges took place at 09:42 and 10:42 on October 11th, and at 11:23, 12:30 and 13:35 on October 12th and coincided with the high water stand stage of the tide. (ABP, 2001). The material was discharged in liquid slurry form, and had a measured bulk density of 1260 kgm^{-3} (ABP, 2001).

The original site located at the northern end of the intertidal flats at Hythe between neap and spring MLW, was selected using sediment transport pathway predictions from the MIKE21 PARTICLE predictive mathematical sediment transport model by

ABP (ABP, 2001). This site was chosen in order to allow the deposited material to disperse over the intertidal flats, but to avoid deposition on the upper saltmarsh, a nature reserve managed by the Hampshire Wildlife Trust. However, the recharge location was slightly modified due to difficulties with access by the dredge vessel. In practice the material was disposed lower down the shore and some distance downstream; around MLWS, 443988-444082 Easting; 107028-107051 Northing. Originally planned to occur during the neap cycle of the tides (in an attempt to minimise transport of the material away from the site), disposal was actually carried out three days after the neap on the first tide that the vessel could obtain access due to restrictive shallow water levels. The discharges were carried out during the high water stand of the tide (ABP, 2001).

2.1.4 Experimental sampling station location (and set up)

The sampling station, site 1, was located at post 4 on the shore profile (Figure 2.2). The site was situated at Mean Low Water Neap Tide Level (MLWNTL) according to published charts of the estuary. However, repeated trips to the site revealed that it was much closer to the Mean Low Water Spring Tide Level (MLWSTL) and only exposed when the predicted height of low water was less than 0.8 m (spring tides reach a maximum of 4.0 m, relative to chart datum, in Southampton Water). Site 2 was located at post 3 on the shore profile (Figure 2.2), 40 m shorewards of site 1.

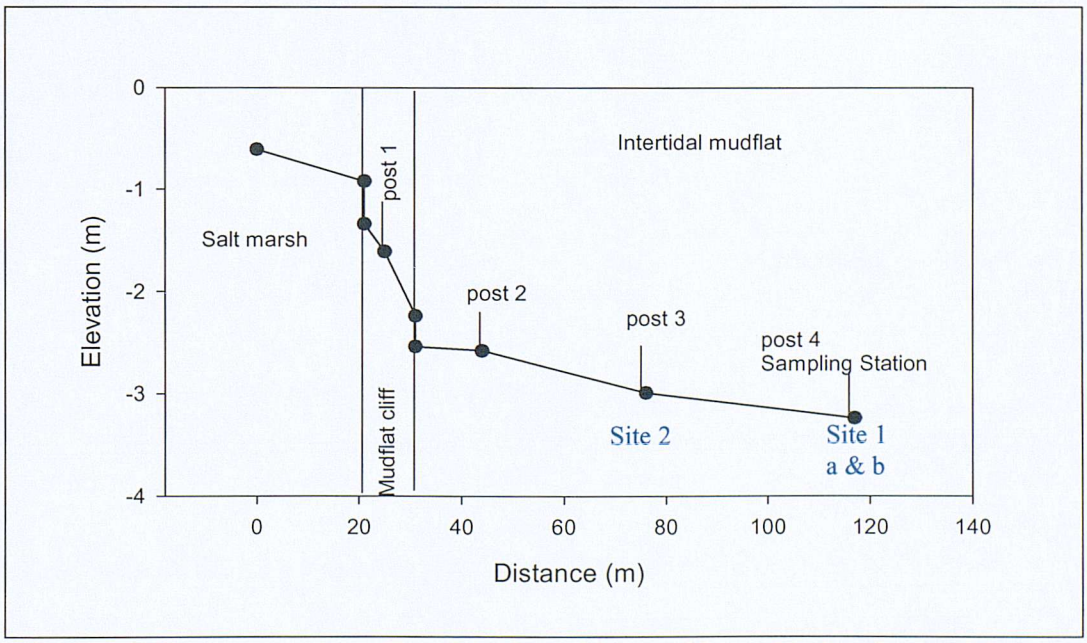


Figure 2.2 Shore normal profile of the intertidal mudflat at Hythe obtained from survey data. Sampling station (site 1) is shown. Elevation is relative to chart datum.

2.2 Methods and Instrumentation

2.2.1 Bed elevation measurements

Accretion-erosion bars were set up on the tidal flat at MLWSTL and mid-shore (Figure 2.2), to measure changes in shore elevation before and after tidal flat recharge. At each station, pairs of 2 m steel poles, 1 m apart, were driven 1.5 m into the underlying clay (to avoid loss or sinking of the poles), arranged in a shore parallel direction (after O'Brien, 1998). A second pair of poles was set up at site 1. The poles were arranged so that they formed a quadrat, spaced 1.0 m apart.

Bed elevation measurements were taken at nine equally spaced intervals between each pair of poles using a steel bar designed to fit over the top of the poles. Calipers were used to record the vertical distance between the top of the bar and the sediment surface at each point, giving a vertical resolution of ±0.1 mm. This method has been found to be inexpensive, reliable and accurate (O'Brien, 1998), the main disadvantage being the limited amount of data collected as measurements are time consuming.

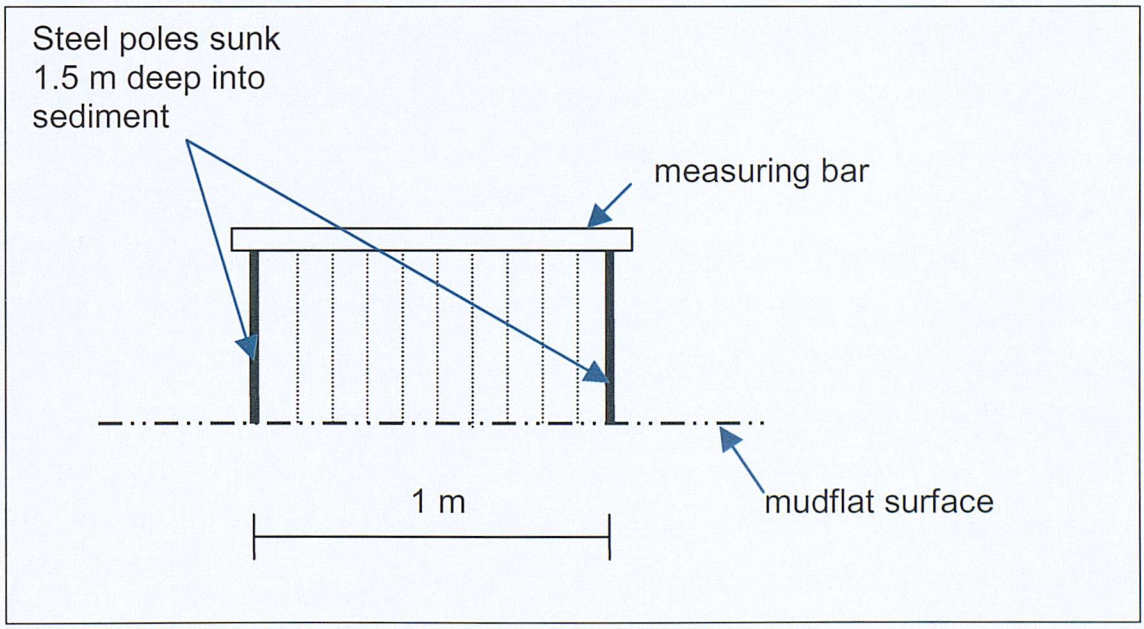


Figure 2.3 Schematic diagram showing a pair of bed elevation measurement poles. The 9 dotted lines represent the axes of measurement between the bar and the sediment surface

Bed level height recordings were taken at monthly intervals from April 2000 to October 2000, and in January 2001 and means were calculated for each date and site. Further data collection was prevented by a lack of access to the site, through a lack of

tidal retreat on several occasions, and due to restrictions to access imposed by Hampshire County Council following the national outbreak of Foot and Mouth Disease.

2.2.2 Physicochemical data collection

In situ measurements of sediment oxygen, pH, temperature, salinity were measured using calibrated Jenway® probes at site 1. Replicate readings of each parameter were taken at the sediment surface, and then at 1 cm intervals into the sediment down to a depth of 10 cm. Dissolved oxygen was measured using a Jenway® Model 9150 waterproof dissolved oxygen meter giving a resolution of 0.1 % or 0.01 mg/l, and an accuracy of +/-2 % within 10 °C of calibration temperature. Saturated calibration was undertaken in the laboratory at 15 °C using a 2 % sodium sulphite solution. pH data were collected using a Jenway® Model 3150 waterproof pH meter with a resolution of 0.01 pH units, and an accuracy of +/- 0.02 pH units. Calibration was performed in the laboratory prior to use using buffer solutions at a pH of 4, 7 and 10.05, supplied with the instrument. Temperature and salinity data were recorded with a Jenway® Model 4200 waterproof conductivity meter supplied with a modified probe designed for use in sediments. The instrument has a resolution of 0.1 °C, accuracy of +/-0.5 °C for temperature, and a resolution of 0.1 psu and is accurate to within +/-1 psu for salinity. The instrument was calibrated using 0.746 g of research grade potassium chloride dissolved in 1 litre of deionised water. These probes were chosen as they were suitable for field use and allowed relatively rapid *in situ* measurement of properties sensitive to disturbance. Resolution and accuracy on a microscale would be improved by the use of microprobes (e.g. for measuring DO), however, as these are more time consuming to use they would not have been suitable due to the very short periods of tidal exposure at the site.

2.2.3 Macrofauna sampling

Five replicate push cores, 75 mm diameter, 150 mm depth, were taken monthly at station 1, from June to October 2000, and on several additional dates immediately following recharge. These were sieved in 50 mm vertical sections, with a 0.5 mm sieve. The sieve contents were then stained with Rose Bengal and fixed in 5 % formalin. All animals retained on the sieve were identified to the highest taxon possible using a binocular microscope and counted. The sampling date, and number of

specimens found of each species were recorded for each core. This sampling was terminated prematurely, at the end of October 2000, due to the lack of deposition of recharge material at the site and prohibition of access due to the Foot and Mouth crisis.

Cores of this size (75 mm diameter, 150 mm depth) were chosen to maximise representativeness of samples and labour efficiency, being large enough to sample the largest infauna present, in this case mature *Cerastoderma* bivalves. The number of replicates was selected to ensure the best possible statistical validity bearing in mind the limitations imposed by sampling and analysis procedures. No macrofauna occurred at depths greater than 100 mm, and very few animals were present below 50 mm. The use of a 0.5 mm mesh sieve allowed for the retention of the smaller species of macrofauna (commonly reported in estuarine muds subject to industrial contamination) which, may have been lost if a 1 mm sieve was used. The chosen methods for processing, staining and fixing of the samples allowed the animals to be collected sufficiently intact to be successfully identified under a dissection microscope. This is particularly important with polychaetes, many species of which are extremely fragile and often lose appendages important for identification. Samples were retained to confirm the identification at species level using a high powered microscope and dissection.

2.2.4 Sediment core collection

Replicate cores were collected at the same site for analysis of sedimentary properties (e.g. bulk density). These cores were flash frozen in the field using liquid nitrogen to preserve sediment fabric and prevent the formation of ice crystals and disturbance during transportation (after Amos *et al.*, 1996). They were then transferred to a freezer and stored at -20 ^{o}C until later analysis using the CT Scanner at GeoResources, INRS, University of Quebec (Chapter 6).

2.2.5 Data analysis and statistical methodology

Multivariate Techniques

The software package PRIMER v5 (Plymouth Routines in Multivariate Ecological Research was used for multivariate community analyses to provide cluster analysis in the form of dendrograms and MDS (Multi-dimensional scaling):

Hierarchical Clustering (CLUSTER)

Based on a computed triangular similarity matrix, square root-transformed similarity coefficients (after Bray & Curtis, 1957) were calculated between every pair of samples. The coefficient is an algebraic expression of how close the abundance levels are for each species, averaged over all species; 100% = total similarity, 0% = total dissimilarity (Clarke & Warwick, 1994). The communities are then represented visually by a dendrogram, which links the samples in hierarchical groups on the basis of the level of similarity between each cluster (Figure 2.16).

Multi-dimensional scaling (MDS)

A similarity matrix is computated as for CLUSTER. The output is then in the format of a 'map', where the rank order of the distances between the samples agree with the rank order of the matching (dis)similarities from the matrix (Clarke & Warwick, 1994). The output is usually a two-dimensional ordination (Figure 2.17).

2.3 Results of Field Investigations

2.3.1 Bed elevation results – accretion/erosion

Graphs showing changes in bed elevation at sites 1 and 2, Hythe are presented in Figures 2.4 to 2.6. Measurements were taken on 5th June (day 1), 5th July (day 31), 4th August (day 61), 29th September (day 117), and the 12th (day 131), 13th (day 132), 16th (day 135) and 18th October 2000 (day 137). Data points represent the mean change recorded between consecutive dates relative to recorded surface level (zero) on 5th June.

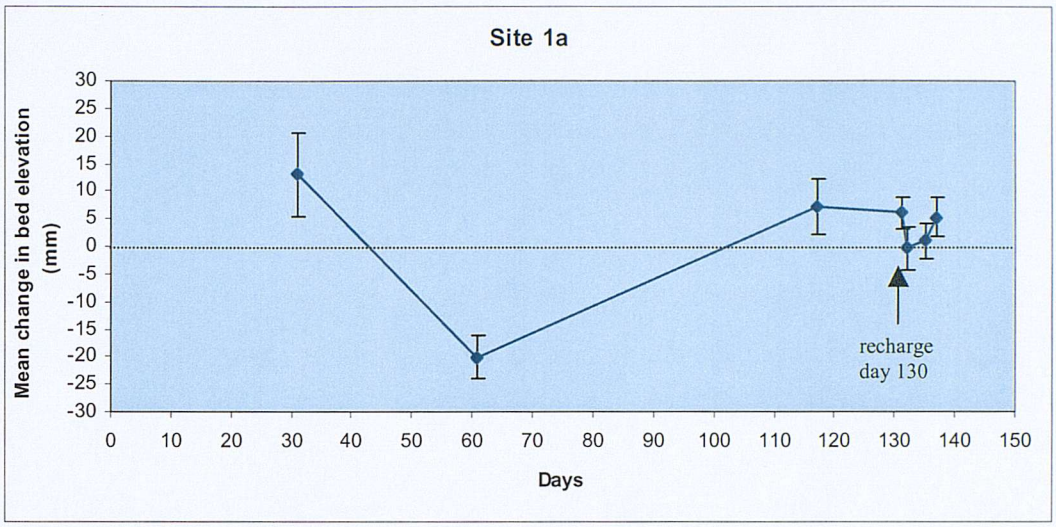


Figure 2.4 Bed elevation data from Hythe: Changes in mean bed elevation at site 1a between 5th June and 18th October, 2000. Error bars represent σ_{n-1}.

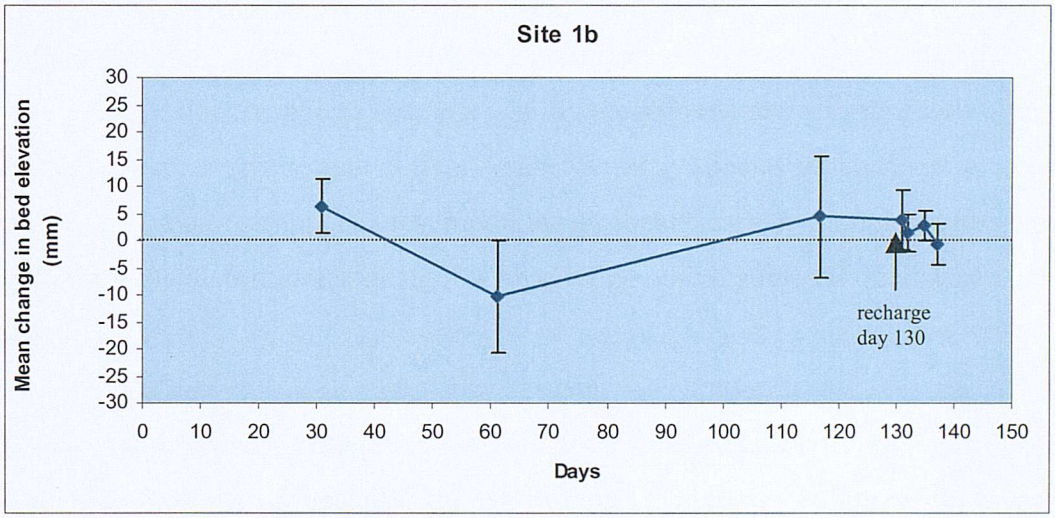


Figure 2.5 Changes in mean bed elevation at site 1b between 5th June and 18th October, 2000. Error bars represent σ_{n-1}.

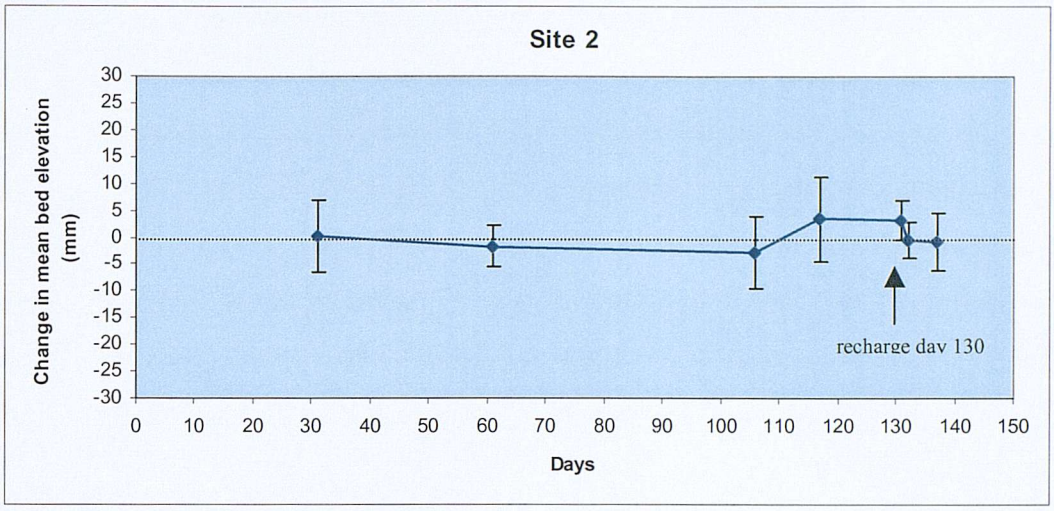


Figure 2.6 Changes in mean bed elevation at site 2 between 5th June and 18th October 2000. Error bars represent σ_{n-1}.

The observations show that the surface of the mudflat is dynamic, with elevations varying by several cm at each measuring position between sampling dates. These changes are the result of interactions between current and wave action, run-off and drainage from the salt marsh and biological activity. The changes in level recorded at the sites appear to be random with no clear unidirectional trend. Slight accretion is indicated at site 1a that corresponds to visual observations at the site, but little significant change is suggested overall. The data suggest that if the 'typical' seasonal trend of deposition in the summer and erosion during the winter documented for other intertidal estuarine mudflats occurs at Hythe, the erosional stage of the cycle had not yet begun despite heavy rainfall during the autumn.

No deposition is suggested on the days immediately following recharge (day 130, 11th October 2000). Slight erosion is indicated during this period, although this does not appear to be significant and is probably seasonal rather than as a direct result of recharge.

2.3.2 Sediment profiles obtained from *in situ* measurements of temperature, salinity, oxygen and pH

Figure 2.7 shows mean sediment temperature profiles for site 1 between June and October 2000. As expected, the profiles show that the sediment temperature was at its highest in July and August with values around 18 – 19 °C. The insulatory effect of the sediment is clearly indicated as the highest temperature recorded was just short of 19 °C, and the lowest temperature recorded (October 16) was above 12 °C. The profiles indicate that there is very little variation in temperature with depth (<1 °C).

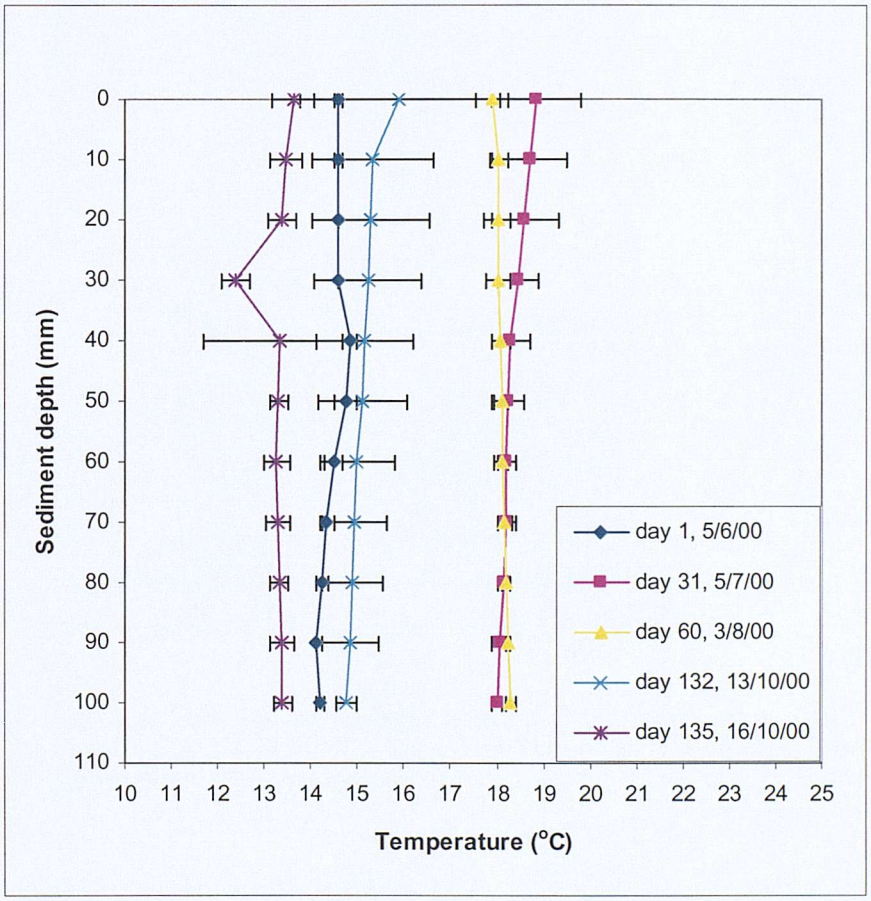


Figure 2.7 Sediment profiles showing *in situ* mean recorded measurements of temperature at Hythe in 2000. Error bars represent σ_{n-1} (where n = 3).

The salinity profiles (Figure 2.8) show greater variability in the upper 5 cm of the sediment. The values for sediment salinity recorded *in situ* at site 1 ranged from 6 to 26 psu. The vast majority of the measurements fell between 10 and 20 psu indicating the sediment salinity in this part of the estuary is lower than that of the water column (≈30 psu) and can be classified as mesohaline (McLusky, 1981).

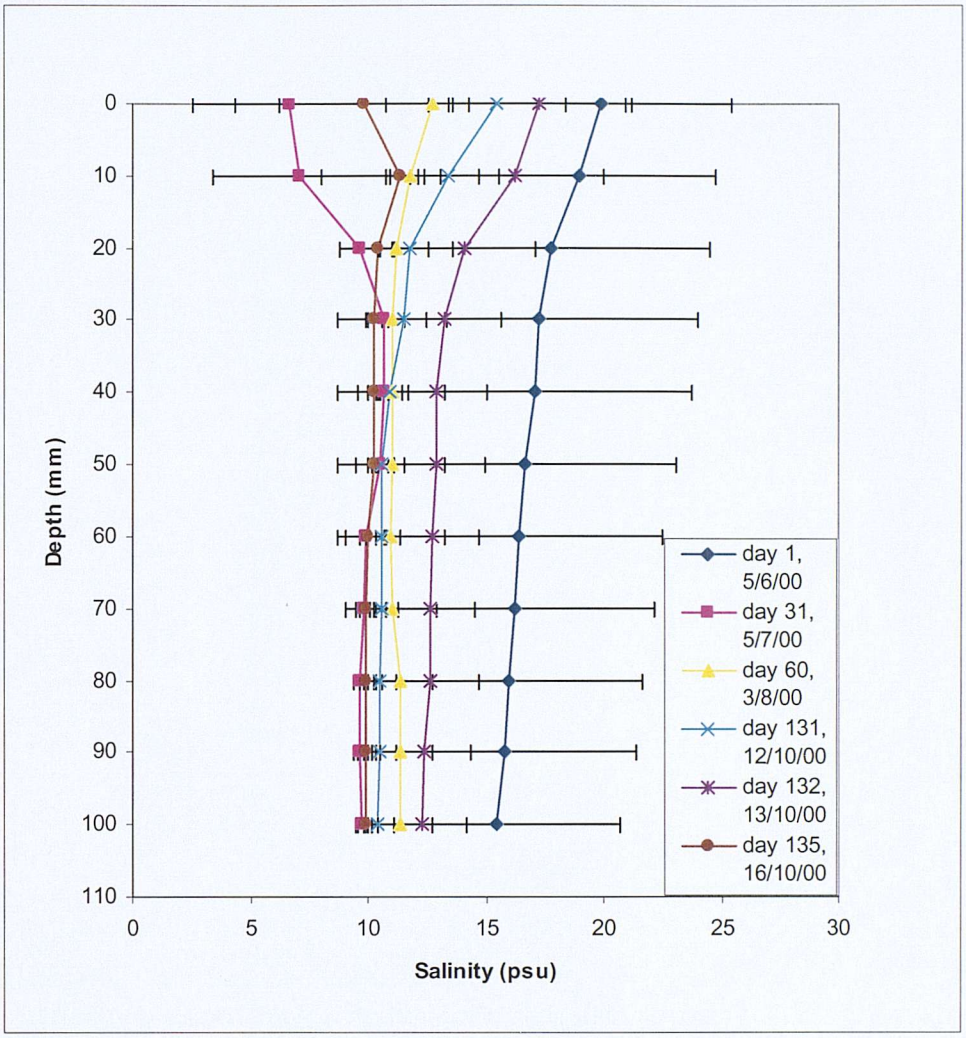


Figure 2.8 Sediment profiles showing *in situ* mean recorded measurements of salinity at Hythe in 2000. Error bars represent σ_{n-1} (where n = 3).

Profiles of mean dissolved oxygen values are given in Figure 2.9. The data imply that the oxygenated zone of the sediment extends to 3-5 cm. The values in the oxic layer of the sediment are variable (1-10 $mgl^{-}1$), but are within the range quoted for estuarine mudflats (e.g. Barnes & Hughes, 1988). The data suggests a decrease in oxygen concentration in the upper sediments during the autumn, which probably reflects a seasonally increased influx of organic matter from the saltmarsh.

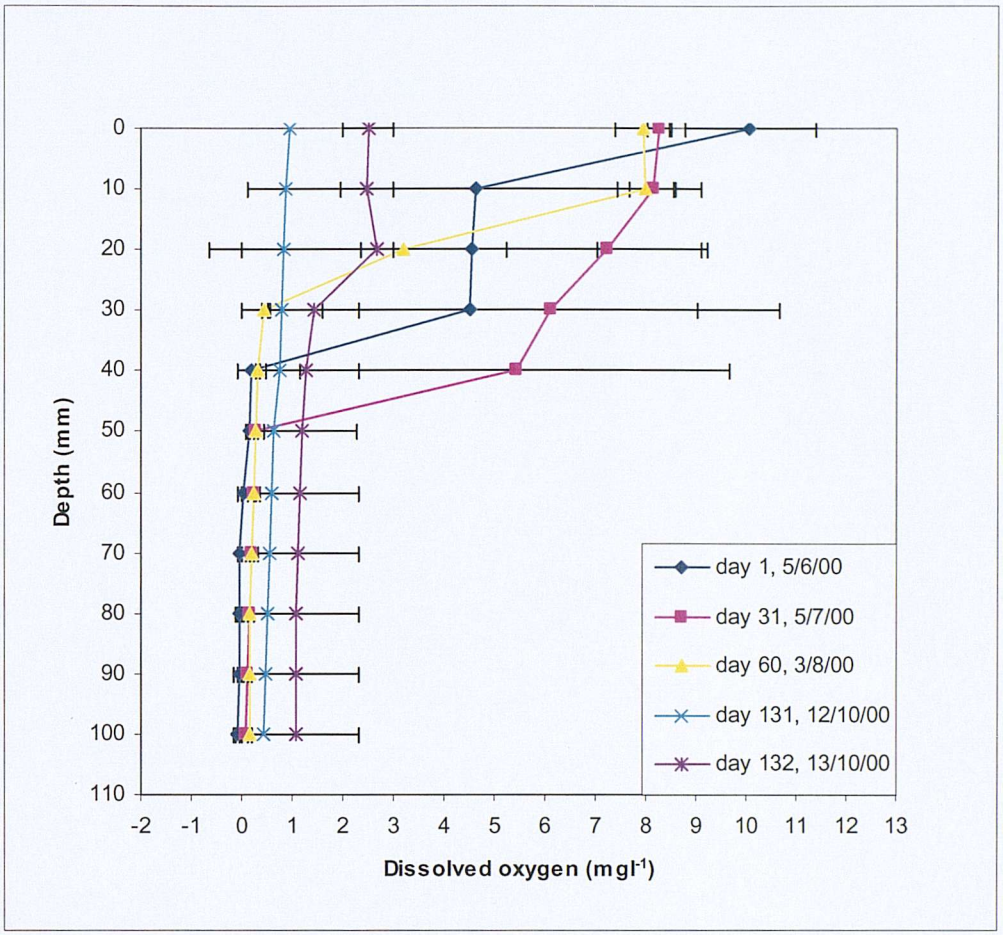


Figure 2.9 Sediment profiles showing *in situ* mean recorded measurements of dissolved oxygen at Hythe in 2000. Error bars represent σ_{n-1} (where n = 3).

Profiles for the same period were also plotted from the averaged pH readings (Figure 2.10). pH was generally within the range of 6 - 8, showing a gradual overall decrease with depth as reducing conditions occur, as would be expected for marine sediments (Barnes & Hughes, 1988). The data suggest that the greatest variability is present in the upper 20 mm of the bed.

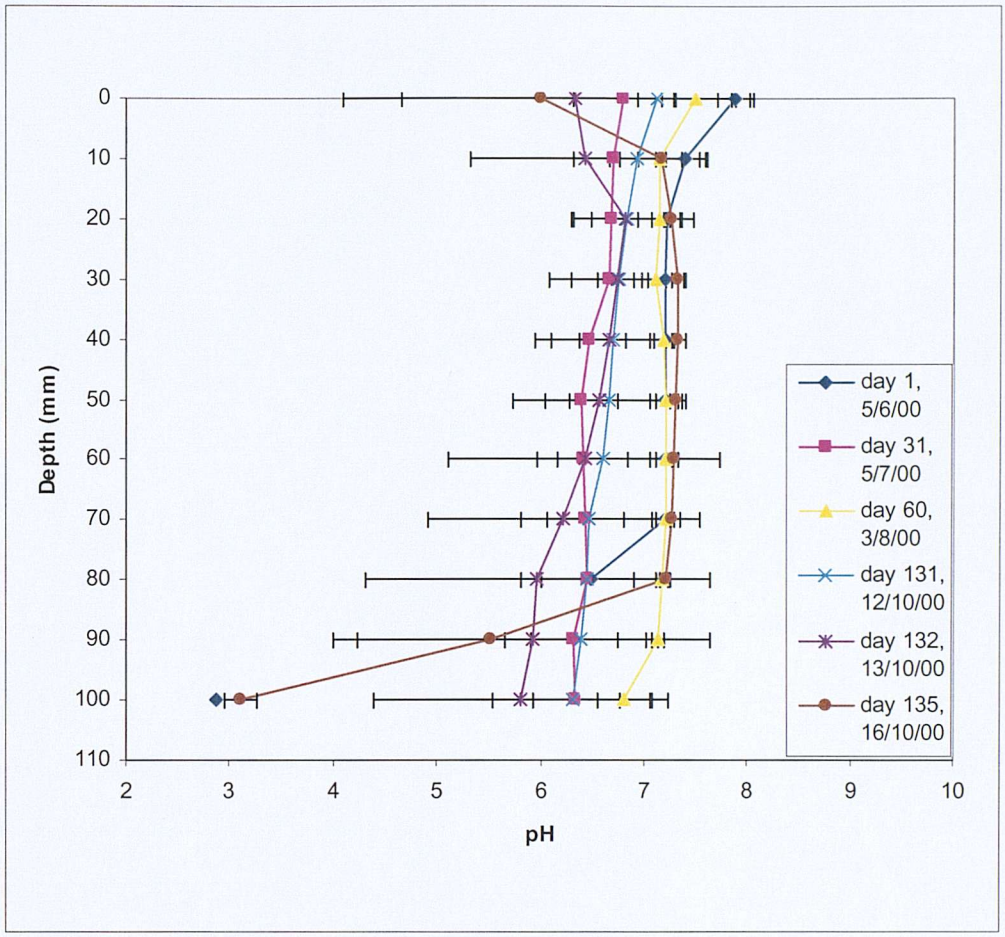


Figure 2.10 Sediment profiles showing *in situ* mean recorded measurements of pH at Hythe in 2000. Error bars represent σ_{n-1} (where n = 3).

2.3.3 Macrofauna abundance and community structure

Macrofauna samples were collected from Hythe site 1 on the following dates; 5th June (day 1), 5th July (day 31), 3rd August (day 60), 18th September (day 106), 3rd (day 122), 11th (day 130), 12th (day 131), 13th (day 132), 16th (day 135) and 18th (day 137) October (Figure 2.11).

The data collected from the Hythe Samples during this period show that the macrofauna at Hythe consists of cirratulid, nereid, nephtyd, spionid, ampharetid and capitellid polychaetes, tubificid oligochaetes, the gastropod *Hydrobia*, bivalves and amphipod crustaceans.

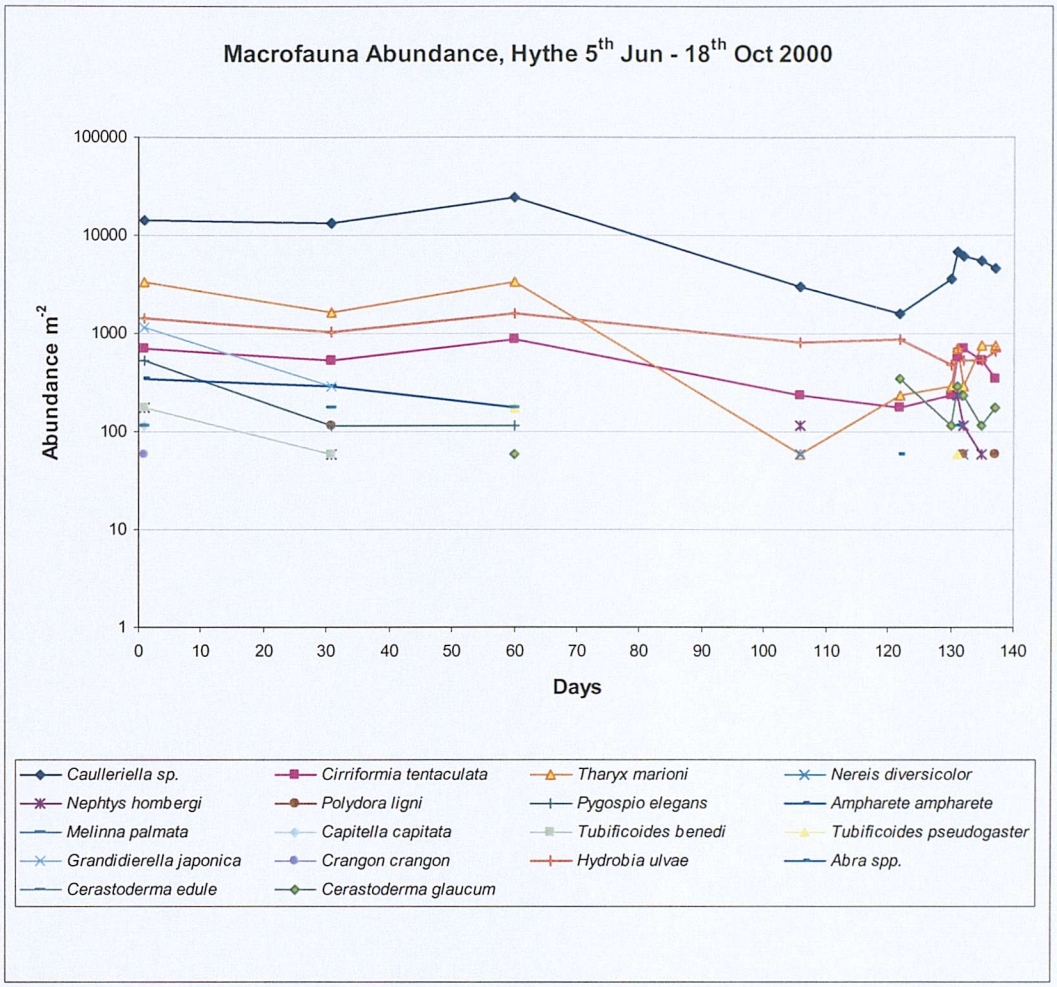


Figure 2.11 Plot showing mean abundance (mean density – numbers m^{-2}), and community structure of macrofauna (retained on a 0.5 mm sieve) from Hythe, sampled between June and October 2000.

Numerically (biomass was not calculated), the macrofauna was dominated by small cirratulid polychaetes (approximately 2 - 5 mm in length) of the genus *Caulleriella*. Averaged data expressed as abundance per m^2 (Figure 2.11) show that, whilst other species (including larger animals) are present in densities of between 10 – 3,000 m^{-2}, this genus occurs in densities of 1,000 – 24,000 m^{-2}. A sub-sample of individuals identified (Worsfold, 1996) using high-powered microscopy suggested that they were of the species *Caulleriella caputesocis* (Saint-Joseph, 1894). The population dynamics and secondary production of this species in Southampton Water is given in Oyenekan (1987). It is worth noting, however, that the taxonomy of this genus is currently considered to require revision (Howson & Picton, 1997) and there is considerable taxonomic dispute with regard to individual species. A photograph of a cirratulid polychaete is shown in Plate 2.1 below for illustrative purposes (a picture of

Caulleriella was not available). Note the characteristic morphology including the profusion of tentacles and the pointed prostomium at the anterior end and much-reduced parapodia.

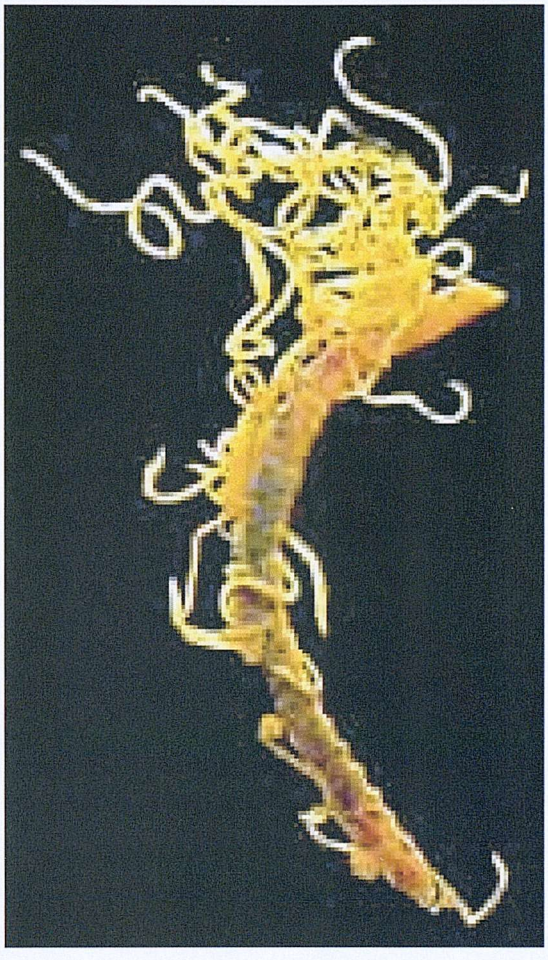


Plate 2.1 Photograph (Harris, 1999) of cirratulid polychaete (specimen identified to family level only) for illustration purposes. Individuals of the genus *Caulleriella* found at Hythe were observed to be 2 - 5 mm in length.

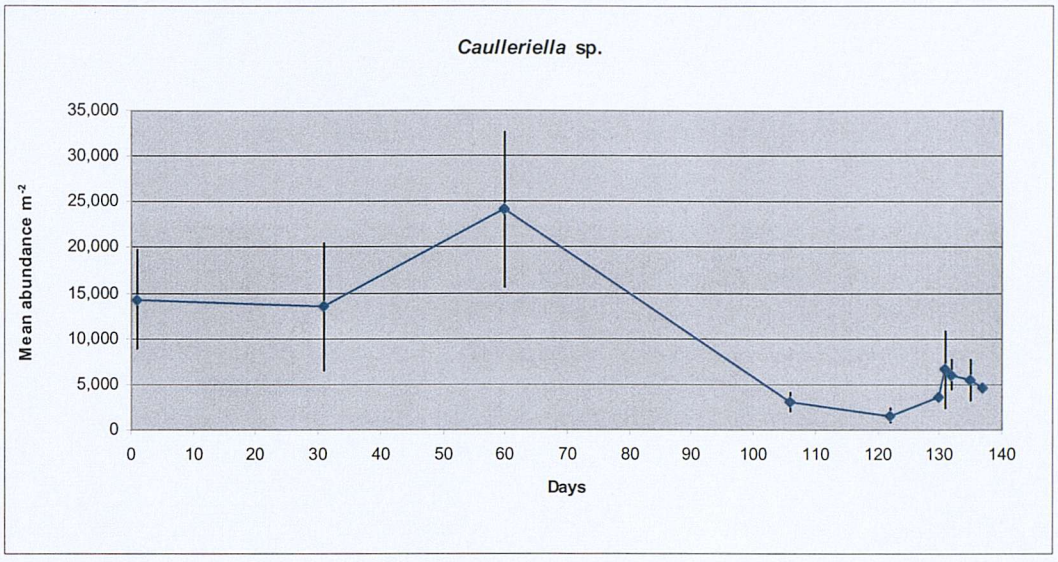


Figure 2.12 Mean abundance of the cirratulid polychaete *Caulleriella* at Hythe, June – October, 2000 (error bars represent σ_{n-1}, n = 5).

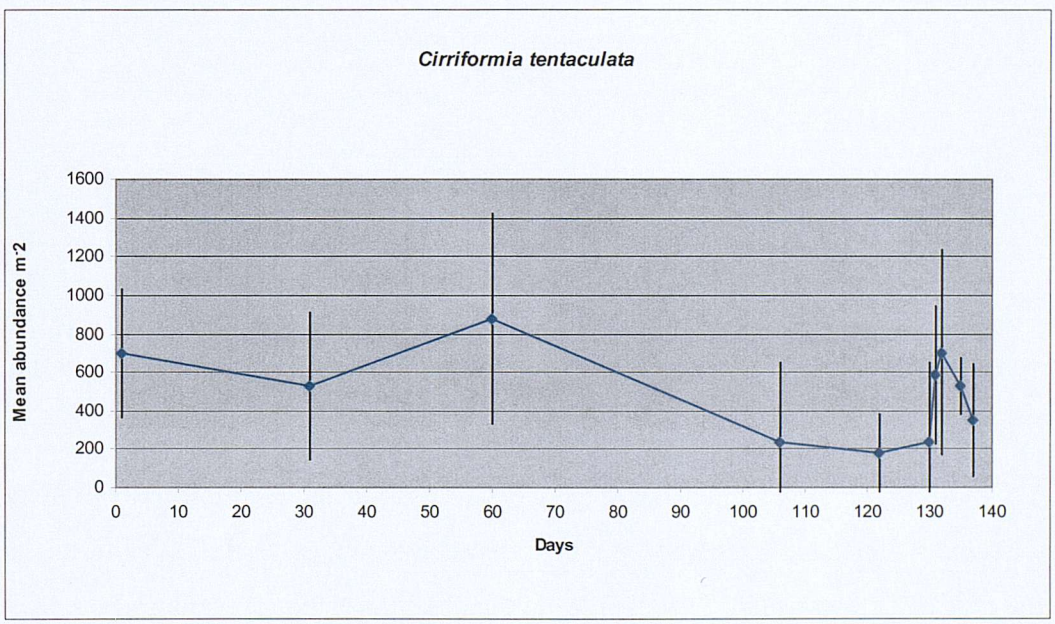


Figure 2.13 Mean abundance of the cirratulid polychaete *Cirriformia tentaculata* at Hythe, June – October, 2000 (error bars represent σ_{n-1}, n = 5).

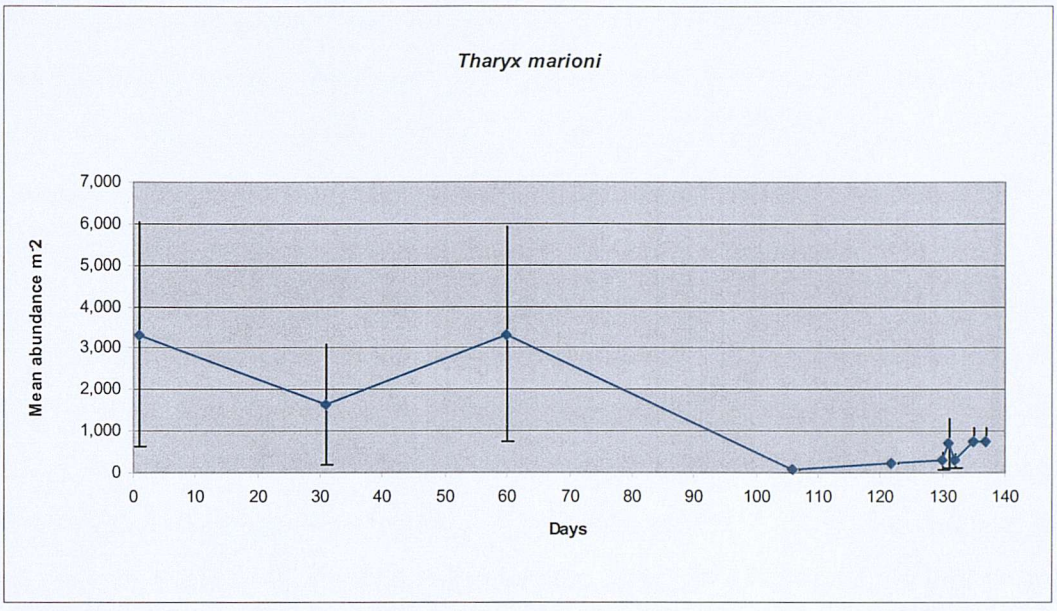


Figure 2.14 Mean abundance of the cirratulid polychaete *Tharyx marioni* at Hythe, June – October, 2000 (error bars represent σ_{n-1}, n = 5).

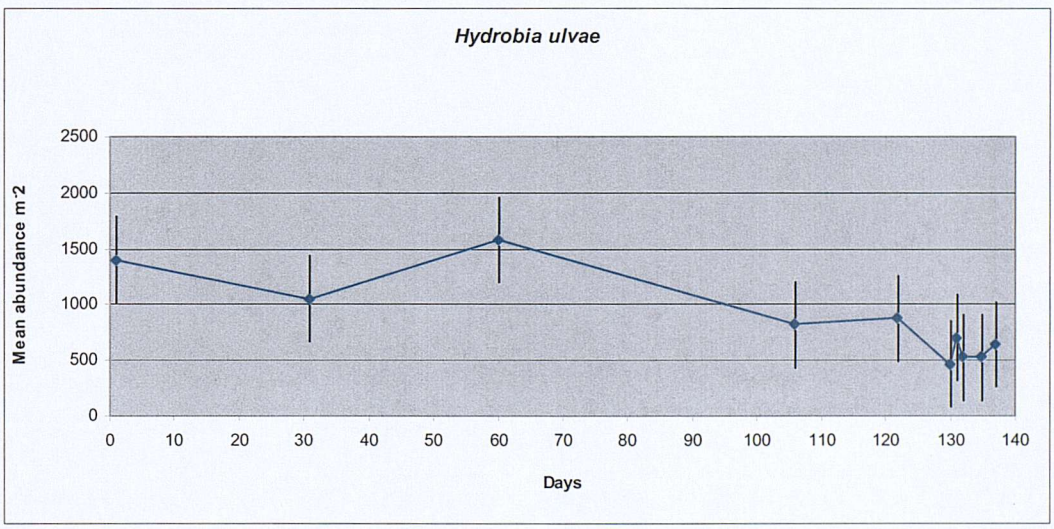


Figure 2.15 Mean abundance of the gastropod *Hydrobia ulvae* at Hythe, June – October, 2000 (error bars represent σ_{n-1}, n = 5).

Other abundant species include *Cirriformia tentaculata* (Montagu), *Tharyx marioni* (Saint-Joseph) and *Hydrobia ulvae* (Pennant) (Barnes, 1994; Worsfold, 1996). The results show peaks of abundance occurring for most species in the July – August period (days 31 and 60), with decreased thereafter (Figures 2.13, 2.14 and 2.15). Several of the less abundant taxa were absent in the autumn samples. The exception to this was the lagoon cockle, *Cerastoderma glaucum* (Poiret), which was more common in samples from 23rd September (day 111) onwards. This may be due to 'patchiness'

as these are the largest animals found at the site and when present in samples, occurred individually or as 2 or 3 per core. The apparent increase in abundance of *Cirriformia* in October (day 130 onwards, Figure 2.13) may be the result of increased sampling frequency during this period combined with the patchy distribution of these relatively larger species (see Discussion).

The diversity of the macrofaunal community was low, and was dominated by *Caulleriella.* It appears that the differences observed in sample composition are due to spatial heterogeneity or 'patchiness', an ecological phenomenon well documented in soft sediment communities (Thrush, 1991). The data also indicate that, although a seasonal decrease in abundance was observed, no corresponding decrease in diversity occurred. No clear impact of recharge on the macrofauna or its community structure was detected.

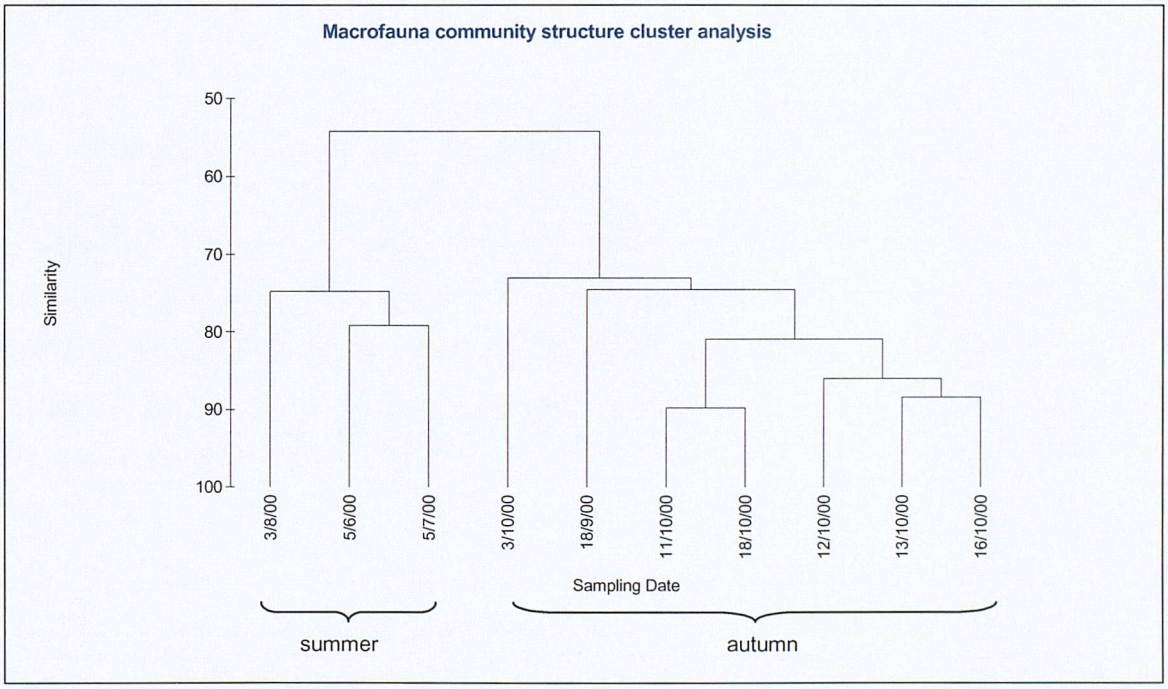


Figure 2.16 Dendrogram of the macrofauna community at Hythe, June – October, 2000.

A dendrogram of averaged monthly data for the Hythe macrofauna based on output from Primer V.5 (Figure 2.16) shows the similarity/dissimilarity between the data from each month's sampling. Samples collected in the summer months were distinct from those collected during the autumn, which suggests a clear seasonal pattern in

macrofauna community structure and abundance. There is, however, no clear indication of effects from recharge.

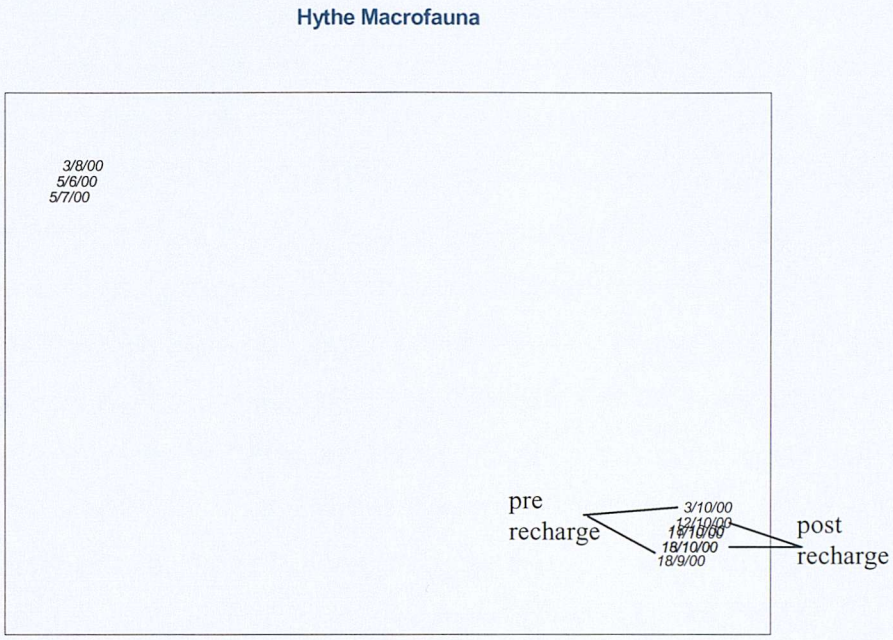


Figure 2.17 MDS plot of the macrofauna community at Hythe, June – October, 2000. Produced from mean abundance data using PRIMER v.5.

A Multi-Dimensional Scaling (MDS) cluster analysis plot, plotted from calculated similarity indices, confirms that there was significantly less difference between samples collected during the same season. The division of the data into two components that appears to reflect seasonal correlation (summer and autumn samples) is clear in the MDS plot.

Neither of these plots shows a significant signal from the recharge as would be expected if the infauna at the site were significantly affected. The more detailed appearance of the right hand side of the dendrogram depicting samples collected in autumn (Figure 2.16) is partially due to a greater number of samples were taken during the period immediately following recharge. The similarities/dissimilarities in community structure in the samples show little clear distinction between pre- and post recharge dates and are likely to represent the patchiness inherent in soft sediment communities and a seasonal trend towards reduced biomass in the winter.

2.4 Discussion

2.4.1 Trial recharge experiment

The lack of impact upon the macrofauna community at site 1 suggests that that the recharge experiment was unsuccessful. This is supported by the other data collected from the site. If recharge were successful, a significant layer of sediment in the order of 10's of centimetres thick would have been deposited at the site that would have been easily observed and created a clear signal in the bed elevation measurements. The bed elevation data clearly indicate that there was no significant deposition following recharge, implying that the majority of the material was not deposited upon this area of the tidal flats. Changes in sediment pH and dissolved oxygen concentration resulting from anthropogenic sediment deposition may also be expected following recharge. However, these data showed no trend of significant change following the experiment. The lack of deposition at the site was further confirmed by tracer data collected by ABP (2001), and the general conclusion reached was that due to the change in location of the dumping site, the prevailing hydrodynamics and state of the tide, the majority of the sediment was transported away from the mudflat at Hythe and carried towards the estuary mouth (ABP, 2001).

The experience gained from this project highlights the difficulties inherent in implementing such an experiment, and the impact that later modification of details such as deposition times and location, can make. It underlines the necessity of proper planning and prior knowledge of local hydrodynamic conditions and sediment transport to maximise chances of success. It also emphasises the sensitivity of such operations and the need to rigorously stick to schedule and operate within narrow limits when working in such dynamic environments if such trials are to prove useful in the future.

The lack of deposition from the trial recharge meant that the field studies undertaken were not able to form a basis for investigating the effects recharge would have on the macrofauna community and its environment. Due to the difficulties in obtaining access to and working at the site it was deemed impractical to implement a field experiment to simulate recharge or to carry out manipulative experiments at the site. Therefore, laboratory studies were planned and carried out (see chapter 4) and the field

data served the purpose of background information characterising the site and describing its community.

2.4.2 General observations at field site

Visual observations made at Hythe during visits also provided relevant information: It was noted that the surface of the mudflat was a complex system of ridges and troughs (shore normal) on the scale of several centimetres and that the 'trough' areas, estimated as covering approximately 50 % of the site, did not drain and were still covered with water during exposure. Checks were made following 'random' sampling to ensure that samples incorporated roughly equal parts of these surface features to prevent bias from inherent differences. The effects of this heterogeneity on the distribution of macrofauna were not examined. However, as the flats are a highly dynamic environment, surface topography is expected to be variable on a tidal timescale. As these features do not form a permanent or long-term structure at any particular point, and would be expected to shift within a timescale of hours, they would not be expected to cause differences in community structure or differential colonisation between the two types of structures. It was also noted that whilst the entire mudflat could be described as extremely cohesive and composed of fine, soft sediment with a high proportion of clays and silt, there was noticeable cross-shore variation with the middle flats being the 'softest' and the upper flats relatively the firmest. Changes were also observed throughout the field programme and during later sampling trips. Overall, during the period of June to the end of October 2000 the sediment at site 1 appeared to become increasingly soft. This change corresponds to the overall trend of deposition recorded by the bed elevation data and may reflect a general trend for deposition to occur during summer months (coupled with erosion during the winter) recorded on other mudflats (e.g. Amos *et al.*, 1995). In contrast however, over a two year period from spring 2000 – 2002, the upper flats became firmer and showed signs of erosion, and the lower flats also seemed to be slightly firmer.

Repeated visits to the site also led to the observation that the seasonal blooms of diatoms seen on the upper flats, did not appear lower down the shore at site 1 (or at site 2 - mid-shore site). This is most likely a reflection of the lack of exposure at this site as during submersion the water column would be too turbid to allow enough light to penetrate to the sediment surface for photosynthesis. On two occasions in spring small

pieces of red macroalgae were found on the sediment surface, but a closer look showed that they were attached to small stones and shell debris which indicated that they had been washed up from another site. On most occasions the mudflat surface was littered with shell debris (mostly from the bivalve *Cerastoderma*, with some broken *Mytilus* and littorinid shells occurring intermittently), which appears to be transported across the mudflats and deposited on the saltmarsh in large quantities in the form of mobile banks known as cheniers (Augustinus, 1989).

It was discovered that the positions of the mean low water marks for both spring and neap tides, as determined by charts of Southampton Water were incorrect. It was found that both these positions appeared to have migrated inshore (it was not within the scope of this program to precisely quantify this due to the variable nature of the tides and effects of local weather conditions). This suggests that the tidal flats at Hythe, and possibly the entire estuary, have undergone significant morphological changes since the charts were drawn up (source data for the charts was found to be collected between 1967 and 1973 for the Hythe area (Admiralty Chart No.1905, (1994)). In terms of this study, whilst the sampling site (which was selected to correspond to the predicted location of recharge) was thought to coincide with MLWNTL, in practice it was found to be significantly lower down the shore. Repeated field observations showed that the site was only exposed during spring tides, when predicted tidal heights were less than 0.8 m, and was only exposed for more than a couple of minutes during tides of less than 0.5 m. Furthermore, exposure of the site was greatly affected by local weather conditions and during several predicted spring tides of less than 0.6 m during low pressure or if the prevailing wind direction was onshore, the site was not exposed at all and remained submerged. It is therefore suggested that site 1, in practice, approximates the mean low water spring mark.

These features of the site proved to be a real hindrance to the field programme, as during the winter months, nearly all predicted low water times occurred during darkness -either in the early hours of the morning, or the early evening. Furthermore, on several occasions, even on the lowest predicted spring tides of the month, the site was not exposed due to the weather therefore preventing the collection of samples and data. This situation also meant that the periods of exposure at the site were much shorter than envisaged giving a very limited sampling window, and therefore that

samples and recordings had to be taken extremely rapidly and efficiently before the tide returned. This led to fewer recordings being made than desired on some occasions.

Access by boat was also extremely limited due to conditions at the site and the only means of sampling from the seaward side was to collect grab samples at high water. This type of sampling was unsuitable due to the disturbance/destruction it causes, and the need for precise quantitative samples (access from the shore was prohibited during the spring of 2001 due to Foot and Mouth disease).

2.4.3 Implications of field results

The in-sediment temperature ranged from 12 to 19 °C between the months of June and October. With respect to the infauna, the sediment acts as protection from the steeper aerial temperature gradients experienced above the surface of the mudflat where rapid fluctuations can occur due to exposure and inundation of the tides and solar radiation. This is especially important during the summer months when animals already subjected to a drop in oxygen concentration/hypoxic conditions due to increased temperatures could become fatally stressed if the available oxygen became further depleted.

The greatest variability in salinity within the sediment was found in the top 50 mm. This was due to freshwater runoff and subsurface drainage from the saltmarsh as well as fluctuations in the overlying water column due to tidal mixing, and variation in river discharge. It is possible that the values recorded during the summer and autumn were somewhat lower than average (cf. salinity of water column at Hythe of approximately 30 psu) due to sustained periods of heavy rainfall. Recorded *in situ* sediment salinities varied between 6 and 30 psu. Salinity levels this low, and subject to this amount of fluctuation would restrict colonisation to euryhaline organisms and may be expected to exclude stenohaline, marine organisms. This helps to explain the relatively low diversity of the macrofaunal community, which is characteristic of estuaries with a mesohaline or brackish water column.

The dissolved oxygen data collected *in situ* within the sediment at Hythe (site 1) indicated that the oxygenated zone extended to a depth of between 30 – 40 mm. This appears to correspond to the depth inhabited by the (macro) infauna at the site, which

construct tubes and burrows to approximately this depth (~95 % of macrofauna processed from this site were found in the top 50 mm of the sediment). This level of oxygen penetration is relatively deep for cohesive sediments that are commonly anoxic within a few millimetres of the surface (Diaz & Rosenberg, 1995). Oxygen concentrations within the top 20 mm of the sediments remained within the typical values quoted as 'normoxia' for shallow marine waters and estuaries at 2 - 10 mgl^{-1} (Diaz & Rosenberg, 1995). Below 20 mm oxygen concentrations fell into the range defined as hypoxia (0 - 2 mgl^{-1}) (Diaz & Rosenberg, 1995). This extension of oxidised sediments was probably due to the activities of the infauna as the irrigation of burrow stuctures can distribute oxygen deeper into the sediment and inhibit reducing conditions around the burrows (Fenchel, 1996). It may also reflect some error induced by the probe, which may introduce oxygen as it is pushed downwards through the sediment (as previously discussed). The upper 30 - 50 mm layer of sediment also corresponds to increased variability in salinity, which also reflects a more open structure and suggests that animal burrows and reworking by the infauna may play an important role in determining the physico-chemical properties in this layer of the sediments.

Although there was considerable scatter, the data suggests that the greatest variability in pH occurred in the top 20 mm of the sediment before decreasing with depth as would be expected due to the reducing conditions present below the surface sediments which are caused by bacterial activity (e.g. Fenchel, 1996). This also corresponds to the trends seen in the dissolved oxygen content above. The variation in pH recorded on 5th June and 13th October may have been due to chemical contaminants from local industry (e.g. Monsanto Chemicals at Hythe or the nearby Fawley oil refinery). Overall these results indicate that macrofauna inhabiting the top few centimetres of the sediment are generally unlikely to be lethally stressed by pH changes. However, if rapid fluctuations such as those mentioned above were common, this would in all likelihood form a selection pressure for pollution-resilient taxa able to survive the effects of chemical contaminants, and would be a cause for low biological diversity in the intertidal fauna of Southampton Water.

2.4.4 Discussion of macrofauna data

Recharge experiment aside, the biological field data collected at Hythe provides valuable information on the structure and short-term dynamics of the infaunal macrofauna community of the lower intertidal zone. Eighteen taxa were recorded from the site between spring and autumn 2000. Of these only four were common in abundance in all, or nearly all of the samples; these were the cirratulid worms *Caulleriella*, *Tharyx marioni* and *Cirriformia tentaculata*, and the gastropod *Hydrobia ulvae*. Cockles (*Cerastoderma edule* (Linné) and *C. glaucum*), the ragworm *Nereis diversicolor* (O.F. Müller) and the catworm *Nephtys hombergi* (Savigny) were present in many of the samples in low numbers. Polychaetes of the family Spionidae; *Polydora ligni* (Bake & Maciolek) and *Pygospio elegans* (Claparède), and Ampharetidae; *Ampharete ampharete* (Grube) and *Melinna palmata* (Grube) were found infrequently along with several individuals of the juvenile bivalve genus *Abra*, oligochaete worms of two *Tubificoides* species, *Capitella capitata* (Fabricius) and an introduced amphipod recently recorded in Southampton Water; *Grandidierella japonica* (Stephensen).

The results of this study indicate the Hythe mudflats are characterised by low biological diversity. This matches the conclusions of much earlier surveys of the site (Barnes, 1973; Houston *et al.*, 1983) suggesting that little has changed in the overall community structure since then. Barnes (1973) suggested that these assemblages correspond to a modified *Macoma balthica* community (after Thorson, 1957), with several introduced species, including *Mercenaria mercenaria* and *Crepidula fornicata*, superimposed. These species were not recorded from the area sampled during this study, which probably reflects the wider range of areas surveyed by Barnes.

The results from this study indicate a stressed fauna, commonly found in the dynamic and fluctuating environment of the intertidal zone in estuaries. Additionally some of the taxa found (e.g. *Polydora*, *Nereis*) have been described as pollution indicator organisms of organic enrichment in the past (e.g. Pearson & Rosenberg, 1978). This is not surprising perhaps when considering nearby industry and Southampton Waters' history (see Chapter 1), although interestingly, Oneyekan (1987) lists the most dominant species, *Caulleriera*, as being intolerant of organic pollution. Houston *et al.* (1983) reported that a gradient of sediment contaminants decreasing with distance away from the Fawley oil refinery is reflected in the fauna. The fauna at Hythe was

less depauperate compared to the shore further south towards the refinery, which was described as 'almost a biological desert', although the authors were unable to make conclusions about the water quality of the site except that it was oxygen saturated (Houston *et al.*, 1983). It is beyond the scope of this study to examine pollutants as a cause of impoverished biodiversity, however, given the long history of industrial effluent discharges together with elevated hydrocarbon concentrations (100 - 500 ppm; Phillips, 1980) measured in the sediments nearby it seems probable that the fauna are affected. In general it seems most likely that the relatively low biodiversity and species richness of the macrofauna at Hythe result from the stresses imposed from the combination of naturally dynamic, fluctuating environmental parameters and anthropogenic pollutants.

In agreement with previous studies (George, 1964; Onyenekan 1987); the most abundant taxa at the site were the cirratulid polychaetes; *Caulleriella*, *Tharyx*, and *Cirriformia*. These are believed to be fairly sedentary burrowing or tubiculous species, which deposit feed by extending a long pair of feeding palps and mass of tentacles from their burrows (George, 1964; Onyenekan 1987). Relatively little is known about these genera, which have attracted sparse attention. In general, animals of the family Cirratulidae are extremely fragile and difficult to sample intact because they tend to disintegrate when removed from the sediment, as well as being difficult to identify at the higher taxonomic levels. Probably as a reflection of this, publications have not progressed beyond taxonomy and an incomplete understanding of their lifecycle/distribution (George, 1964; Onyenekan 1987).

The only published account of the ecology of *Caulleriella* was also carried out on Southampton Water populations, but was restricted to the sublittoral zone. It suggested that the species' distribution was related to silt content (showing positive correlation with a silt content > 60%) and was depressed where sediment copper concentration exceeded 50 ppm. It was found to be monotelic (mass mortality prevelant following spawning) with low level breeding occurring year round in animals over 1 year of age, and a peak breeding season from May to September (Oyenekan, 1987). This supports the conclusion from the current data that the significantly higher numbers of this species recorded during the summer months were a seasonal trend and corresponded to recruitment.

A study into the life cycle and reproductive ecology of *Cirriformia tentaculata* was carried out on the opposite shore of Southampton Water in the early 1960's (George, 1964). This appears to be the only published paper of its kind on this genus, a fact which the author notes as being surprising considering the widespread distribution reported for this species in the British Isles. George described *Cirriformia* as being one of the commonest sedentary polychaetes on British shores. It was reported from the widest range of intertidal habitats; including mud, sand/gravel and mud mixtures, sand, gravel and in rock pools and crevices (probably in localised muddy accumulations) on rocky shores. A deposit feeder like the other cirratulids mentioned above, *Cirriformia* feeds 'indiscriminately on the detritus in the surface layers of the mud' (George, 1964). As with *Caulleriella*, the breeding season in Southampton Water was determined as May – September, also when the greatest population densities were recorded (George, 1964). This trend was reflected in the samples collected from Hythe during this study, where a decrease in density was recorded from summer to autumn. The slightly higher densities found in the latter part of October 2000 may reflect this.

Of the other species found in the lower intertidal sediments at Hythe, *Nereis* (ragworms) and *Nephtys* (catworms) are both mobile polychaetes believed to utilize a number of different feeding strategies depending on resources available, mostly deposit feeding or filtration feeding and less commonly predation (Fauchald, 1983). *Nephtys* has been found to preferentially feed on meiofauna where sufficiently abundant (Fauchald, 1983). The spionids (Polychaeta) *Pygospio* and *Polydora* are both tubiculous deposit feeders that are commonly described as displaying opportunistic or r-selected life traits (Grassle & Grassle, 1974; Gudmundsson, 1985; Zajac & Whitlach, 1991). The tube-building, sedentary, ampharetid polychaetes, *Ampharete* and *Melinna* are also sedentary tube-dwellers that feed on the sediment surface by the extension of palps.

Two species of *Cerastoderma; Cerastoderma edule*, the common cockle and *Cerastoderma glaucum,* the estuarine or lagoon cockle, appear to co-exist at this shore level at Hythe. This is in agreement with the surveys carried out by Barnes (1973), which concluded that both of these species co-occurred. Barnes also reported that, despite being the dominant bivalve in Southampton Water, the cockles were sparsely distributed at Hythe, in the order of 10 - 40 m^{-2}. This compares well to the results from

this study where the distribution of this species was found to be extremely patchy, ranging from zero to several hundred per metre square, which averages out to a comparable figure. It is interesting therefore that, thirty years ago Barnes suggested that one of the two species may be expected to die out; as in his opinion the Southampton Water sites were less than optimal habitats for *Cerastoderma*, possibly because of the nature of the substratum. During this study it was observed that virtually all specimens were large, adult individuals with no sign of recent year classes or juveniles. This was further supported during measurements for laboratory experiments on cockles (see chapter 4), where it was found that all live animals were several years old. This raises the following questions: Is this merely a coincidence, a corresponding stage of a long-term cycle of the Hythe populations that is not understood, or are the populations in the final stages of decline following some unknown environmental changes or forcing factor within the estuary? Further studies would be needed throughout the estuary to test these hypotheses. Unlike the other taxa described above, *Cerastoderma* is a filter feeder – a strategy typically associated with coarser sediments and lower turbidity environments. It is also a genus found to prefer coarser sediment (Flach, 1995), so it seems likely that the individuals at Hythe are living at the limits of survival in such soft mud.

It was noted during laboratory processing that the core samples containing cockles contained few numbers of the other species, including the dominant cirratulids. Flach (1995) reported that the presence of cockles was found to significantly reduce densities of juveniles of many other species including juvenile *Cerastoderma edule, Pygospio elegans, Nephtys hombergi and Tharyx marioni.* The reason suggested for this inhibition was the crawling and 'shaking' movements undertaken by *Cerastoderma edule* that significantly disturbed the surrounding sediment. The lack of smaller species coexisting with *Cerastoderma* in the cores may therefore be more than just an effect of sample size. Flach suggested that a two year old cockle, 3 cm long disturbs around 20 cm^2 of sediment to a depth of 3 cm per week, and that at a density of 500 m^{-2}, 29 % of the sediment is disturbed each week.

The macrofauna at the site was consistently dominated by surface detritus feeders of the genera of *Caulleriella*, *Tharyx*, and *Cirriformia.* This is also in agreement with the community data collected in the 1970's (Houston *et al.*, 1983), and unpublished survey

data collected by ABP. Although relatively few studies are available for comparison, results indicate that the community has remained relatively stable over the last three decades with little indication of change. The term 'climax community' in the sense of ecological successional theory cannot be applied here as, the end community may be considered as a mosaic of several highly tolerant taxa inhabiting a dynamic fluctuating environment subject to constant disturbance and pollution. It is well accepted that shallow water and intertidal soft sediment assemblages are characterized by 'patchiness' (Hall *et al.*, 1992) due to their being regularly subjected to disturbance on varied spatial and temporal scales from predation (shorebirds, large epifauna and fish), the actions of waves, currents and storms, and by human activities such as fishing, bait digging and dredging activities. These disturbances may effectively defaunate a localised area of substrate giving a mosaic effect with patches of sediment (on scales of centimetres to hundreds of square metres, depending on the nature of the disturbance) then becoming available for re-colonisation (Hall *et al.*, 1992; Hall, 1994). This results in a mosaic of patches generated at different stages in a successional sequence (in the broadest sense of the term) and a patchy or over-aggregated distribution of species. With reference to this study, its seems probable that much of the variation between individual samples can be attributed to this condition, as the results of the statistical tests imply that there were no significant differences in community structure throughout the sampling period. Some seasonal difference was indicated in the data that appeared to reflect the main breeding season (May to September) previously reported for the cirratulids at this site (George, 1964; Onyenekan, 1987). A corresponding significant shift in dominance, community structure or diversity was not observed.

2.5 Summary and Conclusions

A field programme was undertaken to monitor the effects of a trial recharge experiment on the macrofauna and associated sediments of a lower shore site on an intertidal mudflat at Hythe, Hampshire. Biological sampling, bed elevation measurements, and *in situ* physico-measurements were carried out, and sediment cores were collected for later analysis with a CT scanner (chapter 6) in the months prior to and following recharge. The main conclusions obtained from the results of the study are summarised below:

The field site studied did not appear impacted by the trial recharge experiment because the material was transported away from the site and not deposited upon the tidal flats. The field monitoring was terminated the month after the recharge due to the lack of deposition and subsequent prohibition of shore access due to Foot and Mouth disease.

The macrofauna community of the lower intertidal zone on the mudflat at Hythe is an impoverished community typical of physically and anthropogenically stressed environments characterized by low diversity and species richness. It corresponds with results obtained in previous studies (e.g. Barnes, 1973; Houston *et al.*, 1983) suggesting that in the long-term it is of a stable nature.

The community appeared to be consistently dominated by deposit feeding, tubiculous, cirratulid polychaetes, although the distribution was inherently patchy.

The mudflat surface was found to be dynamic although no clear trends were identified by the bed elevation data. In sediment measurements of temperature, salinity, pH and dissolved oxygen concentration were found to be within ranges typically quoted for this type of environment (seasonal temperature changes were recorded as would be expected).

3 The design and construction of an experimental microcosm system for evaluating the effects of smothering on benthic macrofauna

3.1 Introduction

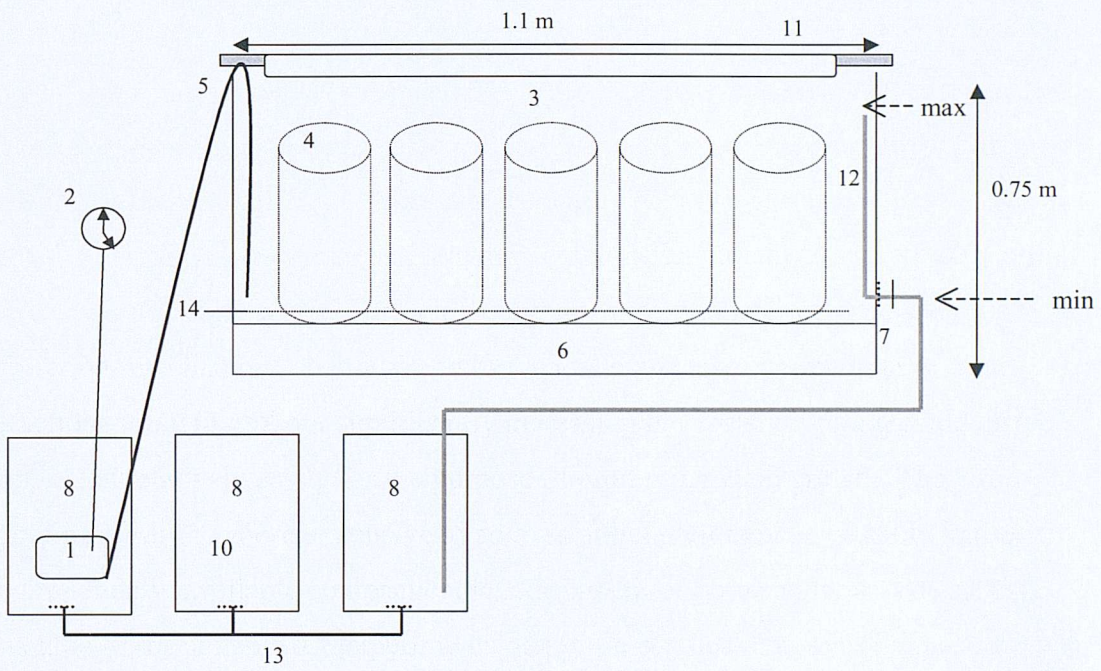
A relatively large capacity microcosm system (≈ 1000 litres) was developed to conduct controlled laboratory smothering experiments that aim to replicate the effects of physical burial upon infauna (as may occur in the event of recharge). The design and building of the microcosm was undertaken in the physical modelling laboratory at ABPMER (Associated British Ports Marine Environmental Research). This chapter outlines the design and specifications of the experimental system, and the procedures undertaken in building, setting up and testing it. Maintenance of water quality, which was assessed through the measurement of parameters such as dissolved oxygen, salinity, temperature and pH, and the stability of the system are also addressed. Details of experimental results are given in Chapter 4.

The experimental system needed to incorporate holding tanks in which sets of replicate sediment cores (containing macrofauna) could be placed, and periodically inundated to represent tidal differences. As there was no access to a continuous seawater supply in the laboratory, a closed, re-circulatory system was required that could be filled at the beginning of the experiments. The holding tanks needed to be deep enough to accommodate deposition of up to 0.5 m of sedimentary material on top of the initial substrate and fauna. The tanks also needed to be large enough to hold several replicate cores for each level of the variable (e.g. smothering depth). Regular water circulation and oxygenation was also deemed essential to prevent deterioration of water quality deleterious to fauna (it was also necessary to install an integral but discrete water reservoir system, in which the water could be fed and temporarily stored during the simulated 'tidal exposure' periods before being returned to the tanks). Major constraints to the project and the system specifications are described below.

3.2 Experimental System Design and Construction

The entire microcosm system had to be constructed within a tight budget, using a design based on readily available and inexpensive materials and components.

The finished system consisted of four large, 250 l, glass, holding aquaria in which replicate cores containing sediments and organisms could be placed (Figure 3.1, Plates 3.1 and 3.2). Each tank was connected to a programmable Powerhead® pump and a series of drainage reservoirs, giving continuous recirculatory flow through the tanks and reservoirs when the pumps were in operation. Additionally, the pumps are programmable allowing controlled inundation and exposure of the sediment within the system. Control of the pumps enabled regular inundation and exposure of the cores.



KEY

1 Powerhead pump
2 24 hour timer
3 Glass aquarium (1.1 m x 0.3 m x 0.75 m)
4 Cylindrical microcosm chambers (0.65 m x 0.2 m Ø)
5 Water inlet (plastic tubing (25 mm Ø)
6 Sand bed
7 Outlet pipe to reservoir (40 mm Ø) with filter and tap
8 Reservoirs (plastic) ($1m^3$ combined capacity)
9 Inlet to pump (25 mm Ø)
10 Mesh filters
11 Fluorescent growth light connected to timer
12 Standpipe/overflow pipe
13 Hosepipe connecting water reservoirs
14 Perforated airline (connected to air pump)

Figure 3.1 Schematic diagram of experimental system setup. For illustration purposes only one of four tanks is shown. Minimum and maximum water levels are shown by the broken arrows on the right.

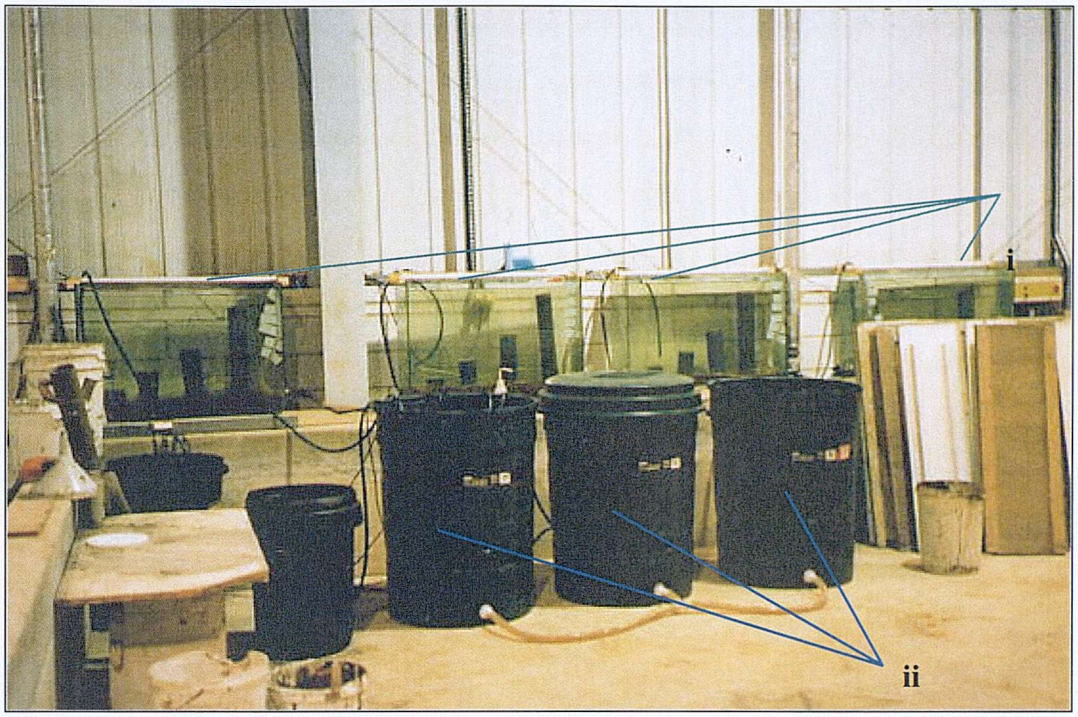


Plate 3.1 Photograph of experimental system showing the four glass tanks (i) and three water reservoirs in basin (ii)

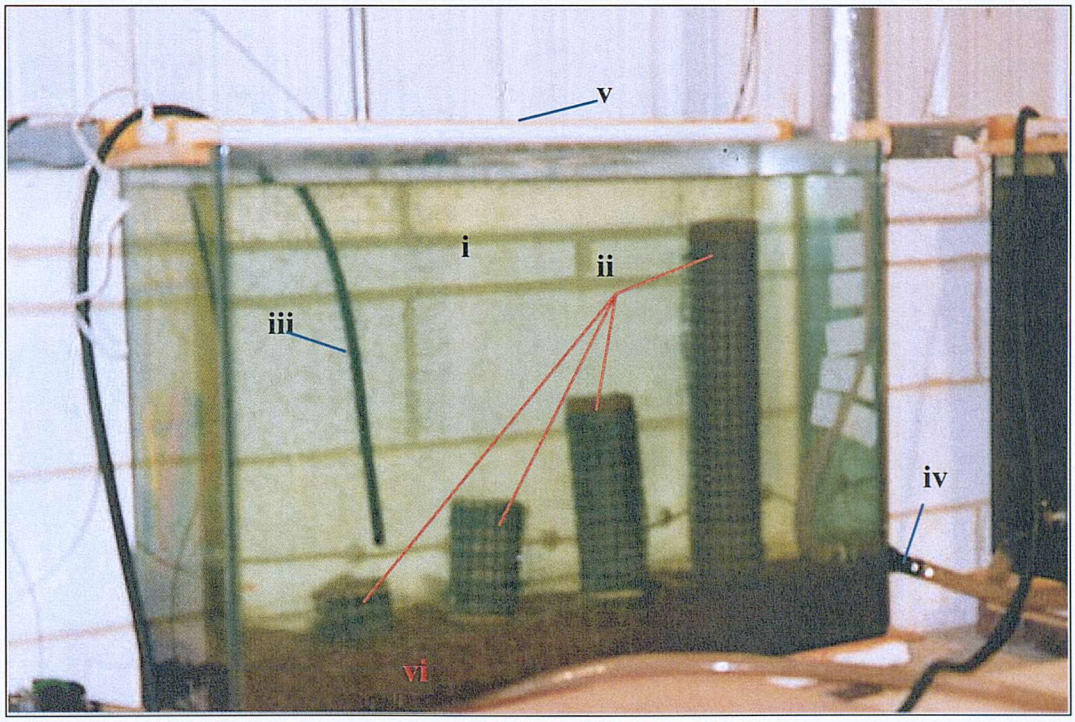


Plate 3.2 Close-up photograph of experimental system showing one of the glass tanks (i) and four experimental cores (ii). The inlet hosepipe (iii), outlet (iv), light tube (v) and coarse sand base (vi) at the bottom of the tank can also be seen.

The glass tanks were 1.1 m long, by 0.3 m wide and 0.75 m deep, thus providing a capacity of around 250 litres (allowing for displacement) and a bed surface area of

around 0.36 m^2. They were constructed to order from 10 mm thick glass to withstand the pressure from the weight of the water (approximately 250 kg) without the need for additional fitted bracing which would obstruct access. A 25 mm circular hole was cut in the end of each tank, approximately 150 mm from the bottom to allow the drainage system to be fitted (described below). During installation the tanks were placed upon brick feet and plywood and were cushioned by thick polystyrene to prevent strain from an uneven floor surface.

Each glass tank was connected to three 350 litre plastic water butts, or reservoirs, with 40 mm diameter clear plastic hosing. These were positioned approximately 1 metre below the bottom of the tank in an existing basin to allow drainage by gravity. The drainage holes on the tanks were fitted with 25 mm diameter skin fittings (supplied for use in the yacht fitting industry) connected to a reservoir. Each hose was fitted with an isolation tap to allow independent control over drainage and improve access for maintenance, and a filter constructed from a section of 200 µm nylon plankton mesh to prevent the escape of sediments en masse. An overflow standpipe was also fitted to each outlet to increase control over drainage and as a further precaution against flooding when the pumps were in operation. Care was taken to ensure that each piece of hosing was of identical length and gradient. The water reservoirs were connected in series using hose pipe connectors and further 40 mm hosing. Drainage from the tanks to the reservoirs was gravity driven.

Reservoir number one contained the four submersible powerhead pumps used to supply the holding tanks. Four Hozelock 1400 Cascade pond pumps were used, each with a maximum flow rate of 1400 lh^{-1} or 1050 lh^{-1} at 1 m head. The total head on the system was 1.7 m (the pumps used were designed to operate with a maximum head of 2.4 m). The outlet of each of these pumps was connected to one of the glass tanks. Once again, care was taken to ensure that each length of hosing and its gradient was identical to minimize variation in filling times and flow through the system. The pumps were fitted with a digital programmable timer to allow controlled filling and flow into the tanks at set periods defined by the user.

Lighting was provided in the form of four Interpet 'Daylight plus' 38 w fluorescent aquarium tubes (one over each tank), giving a spectrum approximating natural daylight. This lighting was provided to prevent modification of behaviour by fauna due to unnaturally low light conditions and inhibit dieback of photosynthetic microorganisms naturally present in the surface of sediment. The lights were fitted above each of the tanks onto custom built wooden frames and connected to fluorescent starter kits. The light units were also connected to programmable timers to allow manipulation of the photo period as required.

The laboratory in which the tanks were sited was unheated. Windows were limited to several small panes for light under the roof several metres above the tanks ensuring that direct solar radiation would not be a problem. Ambient room temperature followed outside air temperatures (personal observation).

Oxygenation of the water column was achieved in part by the tanks being left open at the top to allow gaseous exchange at the water surface. Turbulent mixing created by the pumps in the reservoir and tanks, and continual circulation of the water through the system also increased oxygen exchange. Oxygen was also supplied to each glass tank by an electrical air pump and lengths of submerged airline. Within each tank the end of the airline was connected to air tubing which was perforated at cm intervals along its length. This was fitted along the entire length of the tank at the back, creating an 'air curtain', and thus further increasing turbulence and surface area for oxygen diffusion.

For safety purposes the laboratory was fitted with a 120V mains supply. This necessitated all electrical components (which ran on 240V) being connected to a transformer, safely sited above ground several metres away from the tanks. All visible cables were firmly secured to a ledge behind the tanks or taped to the floor and housed under a wooden platform. Lengths of hosing were also treated in this way. To minimize flood damage the system was installed in a laboratory designed for the purpose of large-scale physical coastal models and, so was already fitted with extensive floor level drainage.

3.3 Filling and Testing of the System

Once the tanks and all connections were found to be fully watertight, the system was drained for the setting up and testing of flow rates and drainage rates. A 'bed' of coarse sand (≈ 10 cm deep) was installed in the tanks for the microcosms/cores to rest on. Coarse sand was chosen as a suitable substrate as it would not cause major resuspension which would be expected to result in significant increases in turbidity and/or clogging of the hoses.

Initially, only one pump was connected to a manifold to fill all four tanks. However, it was soon discovered during testing that significant variation between flow rates could not be controlled within this type of set up. It was thus decided to fit pumps to each tank individually. The tanks were filled and timed and the pump outlets readjusted, so that each tank had approximately the same fill rate, and so flow rates into the tanks were as equal as possible. Flow rates were measured and the pump outlets set to approximate typical current speeds found at the sampling site of up to 0.2 m s^{-1} (personal observation).

The drainage flows were tested and timed by manipulation of the outflow taps and pipes. Some random variation in drainage rates appeared to be insurmountable, but this was minimised by adjusting the diameter of the outflows so that the tanks drained within several minutes of each other to ensure that experimental cores within the system were subjected to the same periods of exposure/submersion. The system was set up so that it could be filled over 30 minutes and drained over about 1.5 hours as required. The timer on the pumps was programmed to fill the tanks at the start of the experiment (zero hours), commence drainage at 10 hours and refill at 12 hours. This provided a approximated tidal inundation of 10 hours, followed by 30 minutes to an hour of exposure twice a day. This was similar to the conditions of inundation found on the lower intertidal zone of the sampling site at Hythe (due to the double high waters occurring in Southampton Water). The additional oxygen supply and lights were then fitted and tested, and the timer on the lights programmed to approximate the natural photo period.

Custom-built core liners were designed and constructed for use in the tanks during the proposed smothering experiments. These needed to be rigid enough to stand unsupported on the sand bed at the base of each tank. 100 mm was chosen as a suitable diameter for the cores. It was estimated that this would provide adequate surface area for several adult bivalves (e.g. *Cerastoderma edule*) to bury themselves, large enough to avoid edge effects such as core compaction, yet manageable in terms of the amount of sediment required for burial, and processing (retrieval of specimens). Lengthwise, the core liners needed to have the capacity to accommodate an additional depth of up to 0.5 m of cohesive sediment added to the existing sediment core during the smothering phase of the experiment. This meant that the longest cores needed to be a minimum length (depth) of 600 mm to allow for this. The idea of using pre-formed standard acrylic core liner used for field sampling was discarded as it was felt that the lack of permeability of the cores may result in localised oxygen depletion and/or local accumulation of toxic metabolites due to inhibited circulation and flushing within the cores. Instead the cores were constructed from plastic garden mesh (25 mm diameter mesh size), lined with plankton net (200 μm mesh size). This meant that the resulting cores were self-supporting within the system, yet water and dissolved oxygen could slowly permeate the sediment by diffusion and flushing, whilst keeping the sediment within the cores and preventing significant amounts of mud from escaping or eroding. Horizontal oxygenation of the sediment through the core liner was restricted to several millimetres and did not affect faunal behaviour. The tops of the cores were capped with plankton mesh and the bottoms lined with plastic sheeting to prevent the fauna escaping. This design also allowed the core liners to be cut to length as required for the experiments depending upon the depth of smothering sediment to be added

3.4 Results and Discussion of System

Environmental conditions within the system were frequently monitored both during preliminary tests and during experimental runs (for graphical plots and full data sets obtained see Chapter 4). During the experiments, water and sediment oxygen content, temperature, salinity and pH were monitored using calibrated probes (instruments as described in Chapter 2). The results for water quality recorded during smothering experiment 2 are given in Table 3.1.

	Day 1	Day 5	Day 7
	mean salinity (psu)		
Tank 1	33.2	28.3	28.7
2	33.3	28.2	28.6
3	33.2	28.1	28.5
4	33.1	28.3	28.5
	mean temperature (°C)		
Tank 1	13.3	11.2	10.5
2	12.9	11.2	10.6
3	13.1	11.2	10.6
4	13.2	11.2	10.6
	mean pH		
Tank 1	7.2	7.9	8.0
2	7.3	7.3	7.9
3	7.1	6.7	7.4
4	7.6	7.5	6.8
	mean dissolved oxygen concentration (mgl^{-1})		
Tank 1	9.5	9.9	10.2
2	9.8	10.3	10.6
3	10.0	10.4	10.4
4	9.8	10.1	10.1

Table 3.1 Mean salinity, temperature, dissolved oxygen content and pH of water column from smothering experiment 2.

3.4.1 Oxygen

Oxygenation is a major factor for consideration in any system designed to contain living aquatic organisms. Brackish and marine organisms are sensitive to fluctuations in oxygen concentration and even those infaunal species relatively tolerant to hypoxic conditions will suffer if the water column is not sufficiently oxygenated (Gray, 1981; Diaz & Rosenberg, 1995). Therefore oxygen concentration was carefully monitored both when testing the system, prior to adding biota, and during experimental runs. The method used in this system of supplementing oxygen supply using an air curtain directly installed into the holding tanks was found to be efficient at maintaining water oxygen concentration at or close to saturation at all times (Table 3.1). Detecting depleted oxygen concentration in the fine sediment of the cores held within the system was more difficult as, except for localised small scale areas such as oxygenated worm burrows, cohesive sediment is often naturally anoxic within a few millimeters of the surface (Diaz & Rosenberg, 1995). As microprobes for measuring oxygen at a fine scale were not available, measurements were taken with a standard calibrated probe, manufactured to withstand use within sediment (see Chapter 2 for details of equipment). Readings which ranged from 0 to 2.4 mgl^{-1} (see Table 4.7, Chapter 4) were found to be comparable to those achieved in the field with the same instrument (see Figure 2.7, Chapter 2). It was therefore concluded that, taking into account the

values of oxygen in the surrounding water body and the relatively low mortality rates in control animals (Table 3.2 below), the organisms held within the system were not unduly stressed with regard to oxygen.

Species	% Mortality in control animals	Smothering experiment no.
Nereis diversicolor	0	3
Cerastoderma spp.	3.7	4
Tapes philippinarum	12.5	2
Cerastoderma spp.	19	1

Table 3.2 Mortality rates (expressed as percentages) recorded for control animals from smothering experiments 1 – 4.

3.4.2 Salinity

Water within the system was made up to the desired salinity of 30 psu, using freshwater to which was added Instant Ocean® salt mix. This gave the advantage that the large amounts of water needed to fill the system (approximately 1000 litres) could be added directly from the mains using a hose pipe. Using a commercially manufactured salt mix designed for long-term use in marine aquaria also avoided many of the impurities and pollutants found in natural water bodies (e.g. Armannsson *et al.*, 1985; H.R. Wallingford, 1997; Hiscock, 1998), and ensured that a balanced suite of minerals essential to organisms was present. Frequent measurements taken in the system indicated that salinity remained stable throughout the experiments (e.g. Table 3.1). The cause for the apparent decrease in salinity recorded during experiment 2 was unknown and may be due to instrument malfunction (see Chapter 2 for discussion). This did not reoccur during the other experiments. Any significant increase in salinity due to evaporation was compensated for by the addition of freshwater, as necessary.

3.4.3 Temperature

During all experiments the laboratory central heating system was switched off. Frequent temperature monitoring showed that temperatures remained close to external ambient conditions and the system was not subject to rapid fluctuations (e.g. Table 3.1). As all experiments carried out within this system were run during winter months (November – March), it is unlikely that temperature *per se* would have posed a problem to the species involved as they were not exposed to significantly elevated temperatures.

3.4.4 pH

Regular checks showed that the pH of the water within the system was stable and remained around neutral (7). pH within the sediment of experimental control cores was slightly lower (as is characteristic of fine sediment conducive to reducing conditions) but well within the ranges previously recorded from the field site where samples originated (see Chapter 2, results). It is unlikely that control organisms were subjected to deleterious or toxic pH levels, a conclusion that is supported by high survival rates recorded for control fauna (Table 3.2).

3.4.5 General water quality and filtration

The nylon mesh filters installed over the outlet pipes from the tanks provided physical filtration, and care was taken to ensure that these were easily accessible for cleaning. Biological (bacterial) or chemical filtration was not deemed necessary as experiments were designed to run for no more than two weeks each, and the accumulation of waste products would not be expected to reach toxic levels within this time (the water was changed for each experiment). It was believed that, due to the constant water circulation, the size and capacity, constant aeration, and natural nitrogen cycle capacity of the system, metabolites such as ammonia, and nitrite would not accumulate to levels that would impose stress upon experimental fauna within the timescale of each experiment. This was subsequently supported by experimental survival rates.

3.4.6 Cores

The design used for the cores holding the organisms and sediments allowed cores to be made up to any required length and diameter. They could also be dismantled with ease at the end of the experiment and cleaned for re-use. Low levels of mortality recorded in control animals (Table 3.2) confirmed that the suitability of the design and construction.

3.5 Conclusions and Future Recommendations

Measurements collected during the experiments and pilot tests showed that values of temperature, oxygen, salinity and pH remained within ranges in which the species used in the experiments would not be unduly stressed (see Table 3.1, also Tables 4.1- 4.7, Chapter 4). These parameters were shown to remain stable and were not subject to

rapid fluctuations. Other water quality parameters such as ammonia and nitrate levels were not tested during this study, although this would be a simple procedure to implement in the future, as manufactured kits are readily available commercially. However, the low levels of mortality recorded in control animals (Table 3.2) within the experimental timescale indicated that overall, conditions within the system were good. It seems likely that the few control organisms that did not survive succumbed to stress occurring from sampling and transportation from the field rather than laboratory conditions *per se*, and the set-up was concluded to be successful and appropriate for the type of experiments it was designed for.

The capacity and dimensions of this system mean that it has great potential use for laboratory study and experimentation. The current set-up was found to be sufficient for the short-term holding of intertidal, estuarine benthic organisms. It would be relatively simple to upgrade the system to cater for less tolerant stenohaline marine species and pelagic species, or to hold organisms for a longer period by adding power filtration (mechanical, chemical and biological) and a protein skimmer or foam fractionation unit without significantly increasing the costs. Lighting could also be upgraded relatively easily for the purpose of promoting algal growth. The issue of cooling would be more expensive to address although it may be possible to use commercially available compact coolers designed for home aquaria. The system would also be suitable for experiments of a more physical nature as long as relevant flow conditions could be generated where required (e.g. by the employment of internal pumps).

4 Vertical migration and mortality in macrofauna as a response to smothering – a series of controlled laboratory experiments

4.1 Introduction

The aim of these experiments was to identify the ability of different species and assemblages of macrofauna to survive a single episodic event of smothering (i.e. burial with sediment), as would occur during successful recharge. Effects on macrofauna from smothering were analysed in terms of observed mortality and vertical migration. Each species or assemblage was subjected to several levels of burial up to fifty centimetres and estimated 'tolerance thresholds' were calculated. Environmental parameters were monitored in the sediment and the water column.

These experiments were designed solely to investigate the effects of physical burial, and were therefore carried out with sediment of a similar grain size, bulk density (water content) and chemical composition to that of the animals' natural habitat.

4.2 Methods

4.2.1 Single species smothering experiments

Fauna selected for the single species experiments were *Nereis diversicolor*, *Cerastoderma* species (*Cerastoderma edule* and *Cerastoderma glaucum*) and the manila clam, *Tapes philippinarum*. Individuals of these species are relatively large and can be readily separated (live) from the sediment. Specimens were added to the experiments at known densities and removed at the end of each experiment, enabling accurate calculation of survival and migration.

4.2.1.1 Experiment 1

Sixteen cores were filled to a depth of 5 cm with fresh sediment collected from site 1, at Hythe (location as for Chapter 2), which had been sieved through a 500 μm mesh to remove the macrofauna. Three adult cockles (*Cerastoderma* spp.) were carefully placed on top of each core, the tops of which were then closed to prevent any animals from escaping. The cores were labelled, and four were placed into each of the four holding tanks of the experimental system. The pumps were switched on and the tanks were filled. Measurements of dissolved oxygen content, temperature and salinity of the water were recorded, and the cores were then left for 48 hours to allow the animals to acclimatize and take up normal positions (i.e. buried) within the sediment.

A large volume (approximately 150 kg) of sediment was collected at Hythe (close to site 1) by grab sampler at high water for use as smothering material. The sediment was sieved in the laboratory using a 500 μm sieve to remove macrofauna, placed in sealed freezer bags and frozen to kill juveniles, then stored at -20 °C until required. It was then thawed and mixed with seawater to a bulk density of approximately 1200 kgm^{-3} (to account for water lost due to drainage and drying during freezing, and to ensure that the sediment was uniform) as required for use in the experiments.

One core in each tank was 'smothered' to a depth of 10, 25 and 50 cm (four of each in total). The remaining four cores (one in each tank) were left unsmothered to act as controls. Dissolved oxygen, salinity, temperature and pH measurements were taken

from the water and the sediment (at a depth of 1 cm), using calibrated probes (discussed in Chapter 2), and repeated after 2, 4, and 6 days.

The cores were removed from the tanks after six days and the liners carefully undone. Each core was then vertically sectioned, taking care to ensure that as little disturbance as possible was caused to the sediment, and that no cockles were moved or dislodged. The position of each cockle within the core was measured and noted. The animals were then carefully removed one by one, gently rinsed in seawater and examined to determine if they were alive or dead (the latter were easily distinguished by gaping valves partially filled with sediment and a strong odour of decomposition). The length of each cockle, measured with callipers was recorded along with general observations on the appearance of the sediment.

4.2.1.2 Experiment 2

The above experiment was repeated using slightly smaller manila clams (*Tapes philippinarum*) collected from Round Island, Poole Harbour* but with the following modification; four animals were added to each core. Environmental parameters were monitored as before. After one week the cores were removed, processed and analysed as described above.

*NB The sediment for the experiment with *Tapes* was collected from Poole Harbour as this species is not found at Hythe. This sediment was also used for cores containing *Cerastoderma* spp. from this site to ensure that results were not confounded through the introduction of non-native sediment.

4.2.1.3 Experiments 3 & 4

Further experiments were carried out using sixty-four large ragworms (4 per core), *Nereis diversicolor,* collected by a local bait digger for experiment 3, and one hundred and ten *Cerastoderma* of varied sizes (6-7 cockles per core), collected from Poole Harbour for experiment 4. In the latter experiment, the cockles were subjected to smothering with just 5, 7.5 and 10 cm (cf. 10, 25 and 50 cm in previous experiments).

4.2.2 Experiment 5; mixed species smothering experiment

For the mixed species assemblages, cores were collected from the mudflat at Hythe and transferred to the experimental system with minimal disturbance. This was necessary as it proved to be impossible to separate the much smaller individuals of the genera *Caulleriella* and *Tharyx* live from the sediment (prior experimentation with an elutriation column constructed for this purpose proved unsuccessful, and chemical techniques were unsuitable due to the need for live and unharmed specimens). The larger cirratulid, *Cirriformia,* could not be separated from the sediment whole, as the appendages of these worms were extremely fragile.

7.5 cm diameter core liner was used for this experiment, cut to a length of 20 cm. Forty-eight cores were collected, to a depth of 10 cm, from site 1 at low water (as described in chapter 2). These were transported back to the laboratory immediately, capped with muslin and 12 cores were placed in each of the four tanks. As before, water measurements were taken of dissolved oxygen, temperature and salinity, before being left to acclimatise.

'Smothering' was performed on 36 of the cores after 2 days by adding 2, 5 and 10 cm of slurried sediment, as described above, to a third each of the cores, equally distributed between the four holding tanks. The other cores (3 in each tank) were maintained in the tanks without further treatment (to act as controls).

Water parameters were measured (as above) every two days. Sediment measurements of these parameters were not taken from each core individually, as due to the smaller size of the core, this may have caused significant changes to conditions within the experiment. However, sediment measurements were recorded within the slurried material just prior to smothering.

It was the intention during this experiment to remove 12 cores, including 3 controls, at 48 hour intervals following deposition. Unfortunately, due to a weekend power cut which stopped water aeration and flow for up to 48 hours, the experiment had to be abandoned before the second batch of cores could be removed for analysis.

The twelve recovered cores were analysed as follows: Each core was sectioned vertically into 2 cm sections (110 sections in all). Each slice was individually placed in a labelled jar and covered with 10 % formalin containing rose bengal stain. The jars were shaken to disperse the formaldehyde evenly throughout the sediment to ensure that all macrofauna present were preserved. Each section was then sieved (500 μm), and the contents of the sieve carefully backwashed into another sample jar; 4 % formalin was added to the jar before being sealed and labelled.

The fauna in each sample was then identified and counted under a dissection microscope. The vertical position of each individual was recorded by section, and the vertical migration distance, relative to the original sediment versus the smothering material, was calculated (see section 4.2.3.1 below). It was not possible to record mortality or survival directly for this experiment as this necessitates the extraction of cirratulids live from the sediment. Therefore, as information about the rate of mortality could not be obtained directly, the absence of vertical migration (in response to smothering) was used as an indirect measure of mortality.

4.2.3 Data processing and statistical analyses

4.2.3.1 Survival versus mortality

For experiments 1 - 4 (single species experiments), mortality was directly recorded and was plotted as percentage of total individuals found at each depth. The data were subjected to Probit analysis, a statistical test designed for the analysis of quantal–response assay data, using Minitab v13.30. Commonly used for dose-response experiments in medical studies, Probit analysis also has a wide range of environmental applications (Sokal & Rohlf, 1995; Wardlaw, 2000). In this case, the depth of smothering was treated as the 'doseage' and the response (i.e the lethal effects of smothering) was calculated from mortality. During analysis, the data were first transformed by Probit transformation from a cumulative response (sigmoid) curve to a linear response probit 'curve' (with a log normal distribution). Linear regression analysis was carried out, and a probability distribution plot drawn up from the obtained 'probit' values: A probit of 3 corresponds to 2.5 % of the population (to the left hand side of the normal distribution curve), a probit of 5 corresponds to approximately 50 % of the population, and a probit of 7 corresponds to approximately 97.5 % of the

population (Wardlaw, 2000)). The resulting graph was then interpolated to give estimates of parameters such as LD50 (= lethal dose 50; the amount of 'treatment' required to kill 50% of the population).

For experiment 5 (mixed species) potential survival was estimated from the distribution data as there was no means of directly assessing survival or mortality in this experiment for reasons discussed earlier. For this purpose it was assumed that an individual recovered less than 6 cm for the surface at the end of the experiment could potentially survive. Consequently, individuals remaining buried under more than 6 cm of sediment were presumed dead due to a potential lack of access to food and long term effects of oxygen depletion. Based on these assumptions, cumulative curves were plotted for estimated mortality.

4.2.3.2 Vertical migration as a response to 'smothering'

Vertical migration was calculated for each species using the recorded vertical positions of individuals within the sediment at the end of the experiments. For all experiments, the given depth notes the position of the uppermost edge of the shell (bivalves) or prostomium (polychaetes). An individual whose final position (dead or alive) was below the interface between the original core surface and the zone of deposition from smothering was taken as zero. These positions were then plotted as a percentage of the total number of individuals on histograms (Figures 4.8 - 4.22 & 4.27 - 4.29).

Note that positive values for vertical migration represent the measured amount of burrowing upwards through the zone of deposited or 'smothered' material. Hypothetically, if an animal placed on the surface of the original sediment at the start of the experiment buried itself 4 cm deep during the period of acclimatisation, and was then recovered 1 cm below the start of the deposited zone, it was classed as zero migration. In fact, if this were the case, the animal would have had to complete 3 cm of upwards migration to reach its final position. As, however, there was no way of recording the depth of individual animals immediately prior to smothering, only migration through the deposited material was assigned a positive value and counted for the purpose of quantitative analysis.

The migration data from all five experiments was tested for normality (Anderson-Darling, Kolmogorov-Smirnov and Ryan-Joiner tests), for homogeneity of variance (Bartlett's and Levene's tests), and for interaction (a statistical term tested to identify lack of independence or crossover between samples) prior to running Analyses of Variance in Minitab (Moore & Cobby, 1998; Wardlaw, 2000). The migration data from experiments 1 - 4 were subjected to two-way factorial design, general linear model ANOVA between smothering depth and tanks, on square root transformed data, as significant interaction between samples was detected (this type of ANOVA was selected as it is resistant to departure from normality, heterogeneity of variance and interaction). Data from experiment 5 for individual species were analysed using the Kruskal Wallis test, a non-parametric alternative to one-way ANOVA, as significant departure from normality and heterogeneity of variance were detected. As no significant tank effects were detected, and species was found to be largely statistically insignificant as a source of significant variance for experiment 5, migration values were then calculated as percentages, and the data pooled and analysed by ANOVA* to increase the power of the analysis over the non-parametric tests.

(* factorial design, two-way general linear model between species and smothering depth, subjected to normality, homogeneity of variance and interaction tests as before)

4.2.3.3 Environmental data

The data collected on environmental parameters during these experiments (water and sediment dissolved oxygen, pH, temperature and salinity) were tabulated (Tables 4.1-4.8), and subjected to statistical analysis as described below:

The water parameter data from Experiments 1 - 5 were tested for normality and homogeneity of variance as before. They were subsequently tested with the Friedman's test, a non-parametric equivalent of two-way ANOVA (as the data displayed significant departure from a normal distribution), to identify significant differences between the four tanks and in water quality over the duration of the experiments. The sediment data were tested for normality and homogeneity of variance before being analysed using Kruskal Wallis tests for variation between experimental runs, and between the base (core) sediment and the sediment used for smothering.

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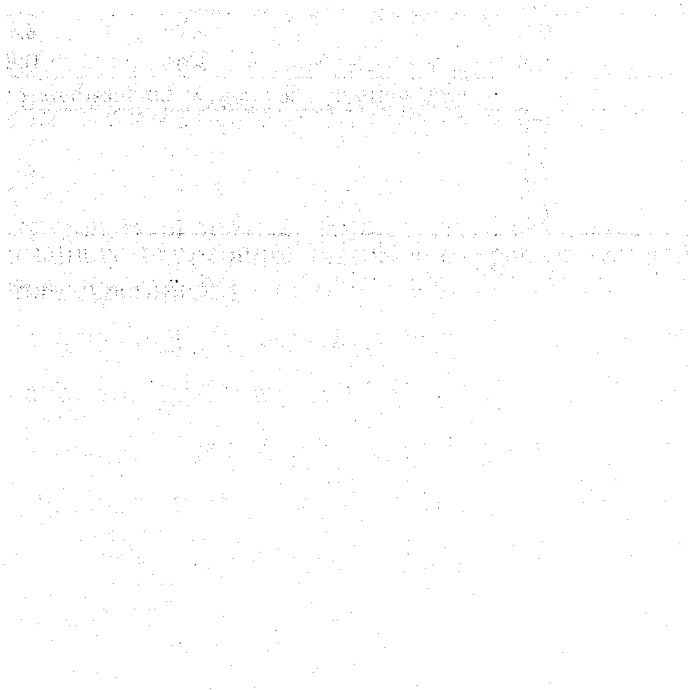


4.3 Results

4.3.1 Environmental conditions within experimental system – water quality data

The water quality data for experiments 1-5 are given in Tables 4.1 to 4.5. Each value is the mean of 3 readings. Blanks in the tables indicate missing values due to instrument malfunction.

The results indicate that in terms of salinity, temperature, pH and dissolved oxygen content, the system was relatively stable with only slight fluctuation of these parameters occurring between the tanks or during the experiments.

Tank	Day 1	Day 3	Day 5	Day 7
		mean salinity (psu)		
1	31.4	31.8	-	32.3
2	31.6	31.7	-	32.3
3	31.6	31.6	-	32.2
4	31.7	31.8	-	31.5
		mean temperature (°C)		
1	13.9	15.3	-	13.5
2	13.9	15.3	-	13.6
3	14.0	15.6	-	13.6
4	14.0	15.7	-	13.8
		mean pH		
1	7.9	7.7	-	7.6
2	7.8	7.8	-	7.7
3	7.8	7.9	-	8.0
4	8.0	8.1	-	8.2
		mean dissolved oxygen concentration (mgl^{-1})		
1	-	-	-	9.7
2	-	-	-	9.6
3	-	-	-	9.5
4	-	-	-	9.2

Table 4.1 Mean salinity, temperature, dissolved oxygen content and pH of overlying water column from experiment 1

Tank	Day 1	Day 3	Day 5	Day 7
		mean salinity (psu)		
1	33.2	-	28.3	28.7
2	33.3	-	28.2	28.6
3	33.2	-	28.1	28.5
4	33.1	-	28.3	28.5
		mean temperature (°C)		
1	13.3	-	11.2	10.5
2	12.9	-	11.2	10.6
3	13.1	-	11.2	10.6
4	13.2	-	11.2	10.6
		mean pH		
1	7.2	-	7.9	8.0
2	7.3	-	7.3	7.9
3	7.1	-	6.7	7.4
4	7.6	-	7.5	6.8
		mean dissolved oxygen concentration (mgl^{-1})		
1	9.5	-	9.9	10.2
2	9.8	-	10.3	10.6
3	10.0	-	10.4	10.4
4	9.8	-	10.1	10.1

Table 4.2 Mean salinity, temperature, dissolved oxygen content and pH of overlying water column from experiment 2

Tank	Day 1	Day 3	Day 5	Day 7
		mean salinity (psu)		
1	29.0	28.9	-	29.0
2	27.3	28.9	-	29.1
3	26.3	28.9	-	29.0
4	28.6	28.8	-	29.0
		mean temperature (°C)		
1	13.9	13.9	-	14.2
2	13.9	13.9	-	14.0
3	13.9	13.9	-	14.1
4	14.0	13.9	-	14.1
		mean pH		
1	7.3	6.5	-	7.0
2	7.2	6.5	-	7.0
3	7.3	6.6	-	6.9
4	7.1	6.5	-	7.0
		mean dissolved oxygen concentration (mgl^{-1})		
1	9.5	8.8	-	9.0
2	9.4	8.7	-	9.2
3	9.9	8.8	-	9.1
4	9.5	8.9	-	9.2

Table 4.3 Mean salinity, temperature, dissolved oxygen content and pH of overlying water column from experiment 3

Tank	Day 1	Day 3	Day 5	Day 7
mean salinity (psu)				
1	31.6	31.4	31.5	31.8
2	30.7	31.5	31.6	31.8
3	30.7	31.7	31.5	32.0
4	30.9	31.6	31.7	31.9
mean temperature (°C)				
1	13.4	13.4	12.8	12.6
2	13.7	13.3	12.8	12.6
3	13.6	13.5	12.8	12.6
4	13.6	13.7	12.9	12.6
mean pH				
1	7.0	7.4	7.3	7.1
2	7.1	7.4	7.4	7.0
3	6.9	7.5	7.4	7.1
4	7.0	7.4	7.5	7.0
mean dissolved oxygen concentration (mgl^{-1})				
1	9.0	9.1	9.9	9.7
2	9.4	9.1	10.0	9.8
3	9.2	10.1	9.9	9.9
4	9.8	8.9	9.9	9.8

Table 4.4 Mean salinity, temperature, dissolved oxygen content and pH of overlying water column from experiment 4

Tank	Day 1	Day 4	Day 6
mean salinity (psu)			
1	30.1	30.2	31.6
2	30.2	30.4	31.8
3	30.2	30.4	31.5
4	30.0	30.3	31.3
mean temperature (°C)			
1	12.8	12.9	12.3
2	12.7	13.0	12.1
3	12.8	13.0	12.2
4	12.8	13.1	12.2
mean pH			
1	7.5	7.6	7.3
2	7.6	7.7	7.1
3	7.4	7.5	7.4
4	7.6	7.7	7.0
mean dissolved oxygen concentration (mgl^{-1})			
1	9.8	9.9	9.8
2	9.6	9.6	9.2
3	9.9	9.5	8.9
4	9.8	9.8	9.0

Table 4.5 Mean salinity, temperature, dissolved oxygen content and pH of overlying water column from experiment 5

The results of the Friedman's tests (Table 4.6), indicate that the data do not display statistically significant variation (the P-values are all >0.05, indicating that variation can be attributed to random sampling differences). In other words, the statistical

analysis indicates that the specimens were not subjected to significant tank effects (arising from differences between tanks) during these experiments.

Water parameter	Experiment	Grand median	*DF*	*s*	*P*
Salinity (psu)	1	31.725	3	1.22	0.748
	2	28.575	3	3.78	0.286
	3	28.863	3	2.45	0.484
	4	31.546	3	2.54	0.468
	5	30.313	3	6.21	0.102
Temperature (°C)	1	13.975	3	7.96	0.047
	2	11.200	3	0.75	0.861
	3	13.913	3	3.00	0.392
	4	13.143	3	7.11	0.068
	5	12.775	3	3.37	0.337
pH	1	7.875	3	6.31	0.097
	2	7.375	3	3.80	0.284
	3	7.013	3	1.29	0.733
	4	7.000	3	0.69	0.875
	5	7.525	3	1.07	0.784
Dissolved oxygen content (mgl^{-1})	1	28.575	3	3.8	0.286
	2	10.175	3	6.2	0.081
	3	9.100	3	3.00	0.392
	4	10.000	3	5.70	0.127

Table 4.6 Results of Friedman's test for water parameter data from experiments 1-5. *DF* = degrees of freedom, *s* = an approximation of chi square distribution determined by the statistic, *P* = level of significance.

4.3.2 Sediment data

The ranges of values obtained from the sediment readings during the experiments are presented in Table 4.7. The base sediment values are for the core sediment into which fauna were seeded or collected (measurements taken at the beginning and end of the experiments). Smothering sediment values were taken from the deposited material (prior to, post smothering and at the end of experiments).

	Day	Salinity (psu)	Temperature (°C)	pH	DO (mgl^{-1})
Experiment 1	Base sediment	5.9-14.4	14.2-15.6	6.7-7.4	-
	Smothering sediment	7.2-8.0	13.1-14.5	7.0-7.3	-
Experiment 2	Base sediment	8.2-8.6	8.4-8.5	6.81	2.4
	Smothering sediment	10.3-15.6	10.8-11.0	2.9-5.2	1.4-4.5
Experiment 3	Base sediment	8.4-11.2	13.7-14.6	7.7-7.9	0-0.1
	Smothering sediment	2.3-10.2	10.1-14.6	5.8-6.4	0.1-0.9
Experiment 4	Base sediment	8.7-9.6	12.6-13.2	6.6-6.8	0.6-0.8
	Smothering sediment	9.1-9.4	12.6-12.8	6.7-6.9	0.4-0.7
Experiment 5	Base sediment	7.2-8.9	10.6-11.0	6.3-7.0	0.1-0.6
	Smothering sediment	7.2-8.4	12.1-12.9	5.9-7.3	0.1-0.4

Table 4.7 Ranges of values recorded for sediment parameters (base and deposited sediment) during experiments 1-5.

Table 4.8 gives the results of the statistical analyses on the sediment data. Kruskal Wallis tests were performed on the data to test for variance between experiments and between base sediment and smothering material in terms of salinity, temperature, pH and dissolved oxygen content. No significant variation was detected in sediment salinity either between the experiments ($P = 0.094$) or the two types of sediment ($P = 0.254$). The test results indicate that sediment temperature did vary significantly between experimental runs ($P = 0.000$), and slightly within the sediment used ($P = 0.011$), indicating that the smothering material was at a lower temperature than the cores in the experimental system. Significant variance in sediment pH was also present, both between experimental runs and sediment types ($P = 0.000$). Variability in dissolved oxygen content was shown to be significant between the experiments ($P = 0.000$), but not between sediment types ($P = 0.2970$).

Sediment parameter	Source	Median	*z*	*DF*	*H*	*P*
Salinity (psu)	Experiment					
	1	9.100	0.66	4	7.94	0.094
	2	9.450	1.14			
	3	8.900	-0.62			
	4	9.165	0.84			
	5	7.300	-2.53			
	Sediment type					
	base	9.000	1.14	1	1.30	0.254
	smothering	8.980	-1.14			
Temperature (°C)	Experiment					
	1	14.750	5.26	4	35.7	0.000
	2	9.650	-3.54			
	3	14.600	-0.11			
	4	12.800	-1.56			
	5	11.550	-2.32			
	Sediment type					
	base	14.60	2.53	1	6.42	0.011
	smothering	12.80	-2.53			
pH	Experiment					
	1	7.135	4.37	4	20.50	0.000
	2	6.015	-2.27			
	3	6.240	-0.92			
	4	6.725	-1.86			
	5	6.600	-0.86			
	Sediment type					
	base	7.020	3.94	1	15.51	0.000
	smothering	6.680	-3.94			
Dissolved oxygen content (mgl^{-1})	Experiment					
	1	1.975	2.76	4	24.60	0.000
	2	1.905	3.23			
	3	0.150	-2.21			
	4	0.645	-0.39			
	5	0.240	-2.49			
	Sediment type					
	base	0.585	-1.10	1	1.22	0.270
	smothering	0.760	1.10			

Table 4.8 Summary of results from Kruskal Wallis tests on sediment data (experiments 1-5). z = standard deviations from the mean, DF = degrees of freedom, H = an approximation of chi square distribution (with k-1 degrees of freedom where k is the number of samples), P = level of significance.

4.3.3 Single species experiments: Experiment 1, *Cerastoderma* spp. (Hythe)

Figure 4.1 indicates that, during the experiment, all control animals had buried themselves to approximately the same depth in the sediment; in this case about 3 cm. Out of the 16 individuals used for the controls, 13 survived (giving a mortality rate of 19 %).

Following smothering with 10 cm of sediment, the depth distribution of the cockles was significantly different from control animals, ranging between 7 and 13 cm.

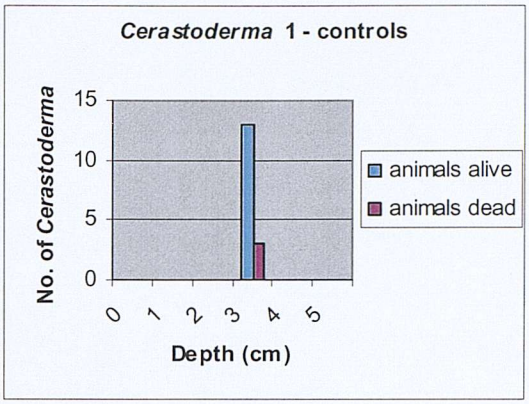


Figure 4.1 Distribution and survival of control animals (cockles) at the end of experiment 1.

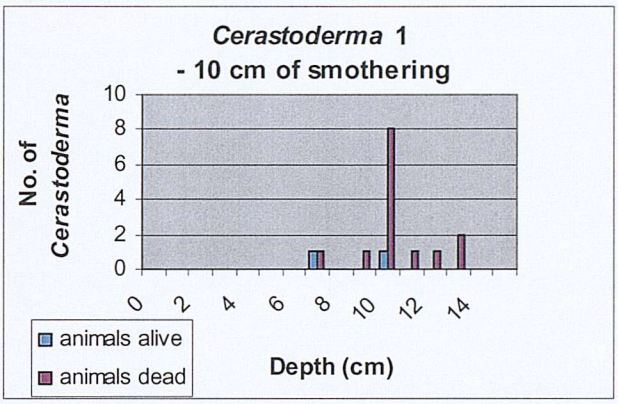


Figure 4.2 Position and condition of cockles at the end of experiment 1, following smothering with 10 cm of sediment.

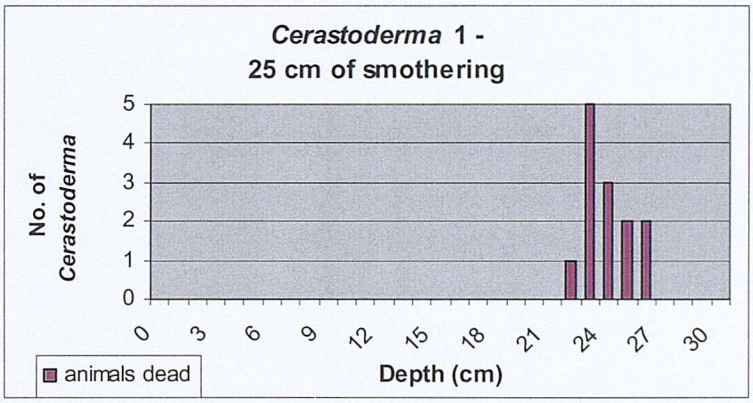


Figure 4.3 Distribution and state of cockles following 25 cm of deposition.

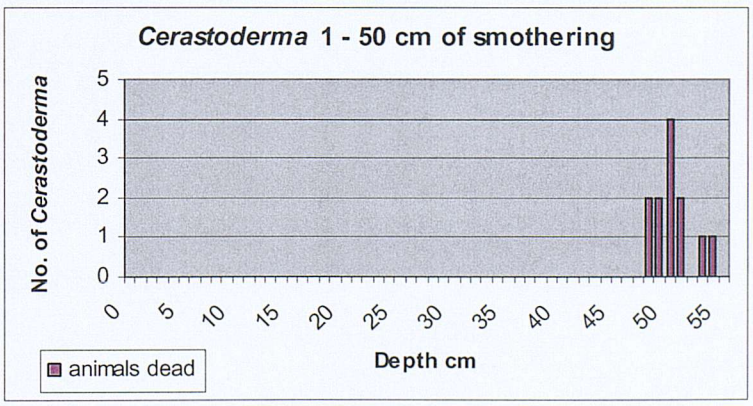


Figure 4.4 Distribution and state of cockles following 50 cm of smothering.

Figure 4.2 shows that 4 of the 16 individuals were recovered from where they had buried themselves into the orginal sediment, 10 cockles remained at the interface between the original material and the deposited layer, and just 2 animals had burrowed upwards into the new sediment. Compared to the controls, mortality was greatly increased following smothering, just 2 out of 16 cockles (13 %) were recovered alive.

Following 25 cm of deposition, mortality reached 100 % before the end of the experiment. All 16 cockles were removed dead from the bottom 7 cm of the cores with little indication of vertical migration into the deposited layer towards the surface (Figure 4.3).

100 % mortality was recorded for the cockles subjected to 50 cm of smothering. All the animals were retrieved from the lower 7 cm of the cores (Figure 4.4), indicating little ability to burrow upward when covered with this depth of sediment.

4.3.4 Experiment 2: *Tapes philippinarum* (Poole Harbour)

Figure 4.5 shows the vertical distribution of manila clams used as controls for smothering experiment 2. A relatively high survival rate was achieved with 14 out of 16 clams recovered alive at the end of the experiment. Figure 4.5 shows that individual clams had buried themselves to different depths, and were distributed throughout the sediment.

Figure 4.6 shows the distribution of clams smothered with 10 cm of sediment. The mortality rate was low, with 14 clams recovered alive (as in the controls). However, little evidence of vertical migration was seen, with the highest animals being found 7-8 cm from the sediment surface.

All clams were recovered alive following deposition of 25 cm. However, little evidence of upwards migration was shown with all clams being recovered from depths of between 22 and 26 cm (Figure 4.7).

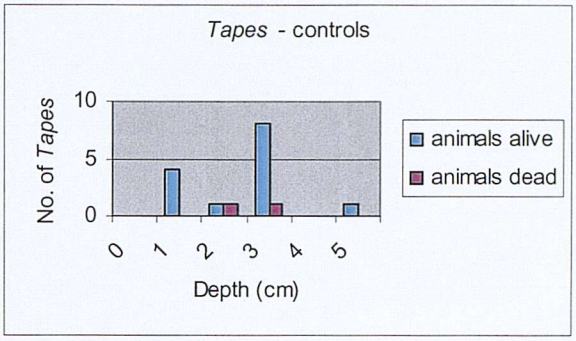


Figure 4.5 Distribution and state of clams used as controls for experiment 2.

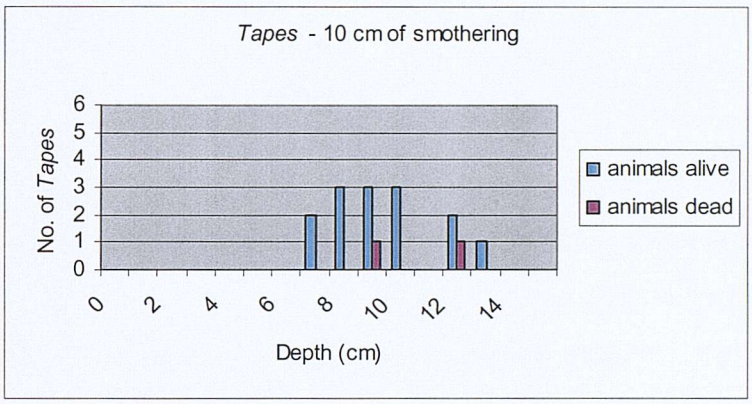


Figure 4.6 Position and state of clams following 10 cm of smothering.

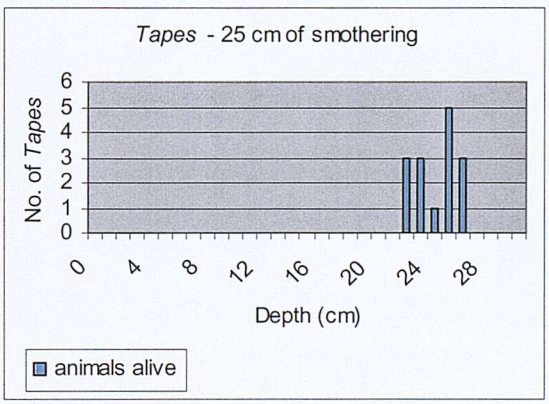


Figure 4.7 Position and state of clams following 25 cm of smothering.

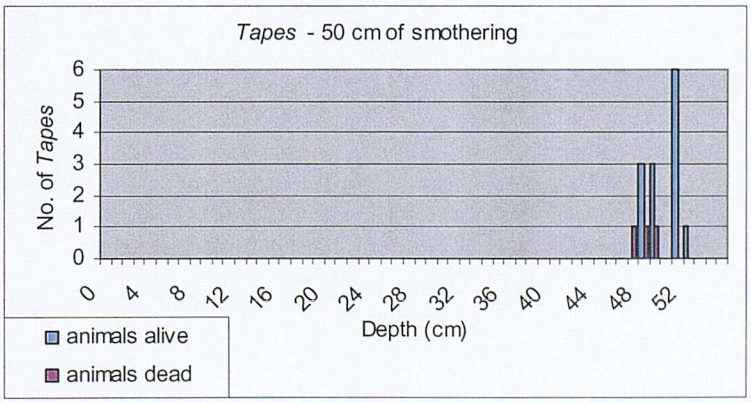


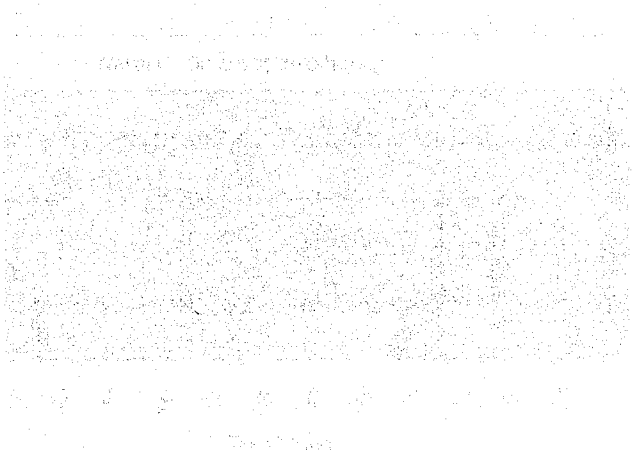
Figure 4.8 Position and state of clams following 50 cm of smothering.

Following 50 cm of deposition, no migration into the deposited layer was indicated (Figure 4.8), and all individuals were recovered from the bottom 6 cm of the cores. A slight increase in mortality (3 out of 16 clams) was also observed compared to controls and other treated cores.

4.3.5 Experiment 3: *Nereis diversicolor*

Figure 4.9 shows that, at the end of experiment 3 all 16 worms used were recovered live from the bottom of the cores.

Figures 4.10 – 4.12 give the results of the smothered cores from experiment 3. *Nereis diversicolor* was extremely resilient to both the treatment and laboratory conditions with all worms recovered live at the end of the experimental period. With respect to vertical migration following smothering, individuals of this highly motile species appeared to be randomly distributed throughout the cores; from the surface of the deposited layer, to the base of the original sediment without deleterious effects, even when smothered with half a metre of sediment.



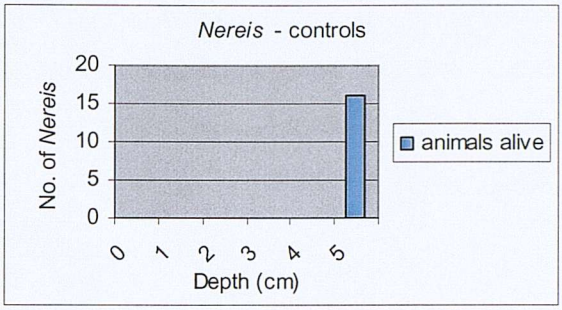


Figure 4.9 Distribution and state of control animals from experiment 3.

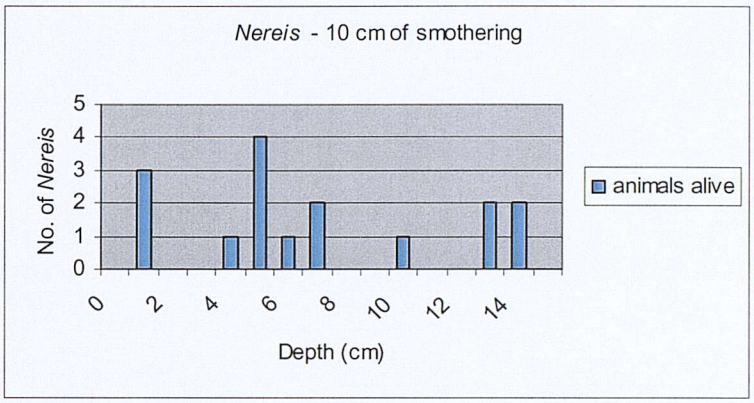


Figure 4.10 Position and state of worms following 10 cm of smothering.

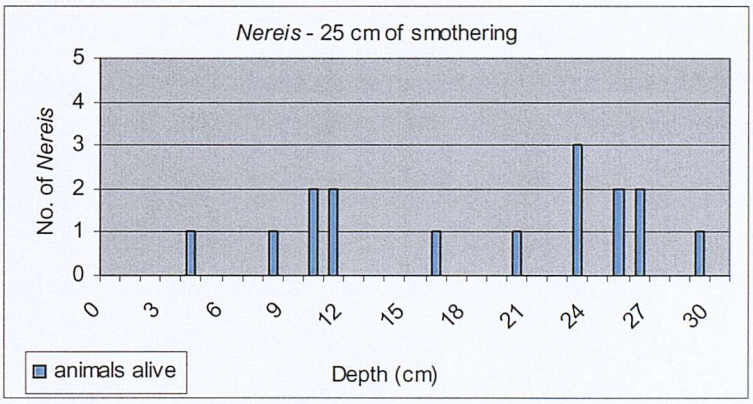


Figure 4.11 Position and state of worms following 25 cm of smothering.

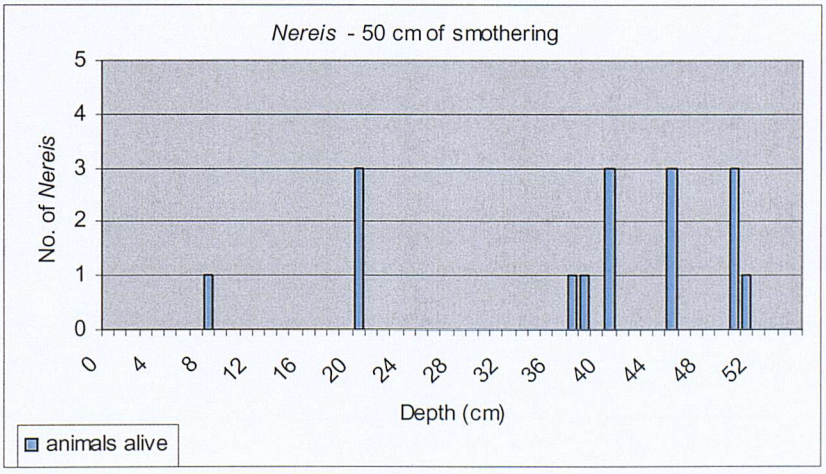


Figure 4.12 Position and state of worms following 50 cm of smothering.

4.3.6 Experiment 4: *Cerastoderma* spp. (Poole Harbour)

Figure 4.13 shows the distribution of individuals used as controls from experiment 4. The cockles were found buried just below the sediment surface at the end of the experiment. Recorded mortality was low, with 26 out of the 27 cockles recovered alive.

Increased mortality (relative to controls) was observed following 5 cm of deposition. At this level of treatment, more than half the cockles had migrated into the smothered material (Figure 4.14). All the animals were recovered from the top 6 cm of the core, with none found in the bottom 4 cm.

Figure 4.15 indicates that after 7.5 cm of deposition mortality increased to 100 %. Some indication of migration was seen with the majority of animals being found in the middle of the cores, but all were dead by the end of the experiment.

All *Cerastoderma* were recovered dead from the cores when smothered with 10 cm of material. All individuals were recorded at depths ranging from 6 to 13 cm except for one, which had migrated to 3 cm below the surface (Figure 4.16).

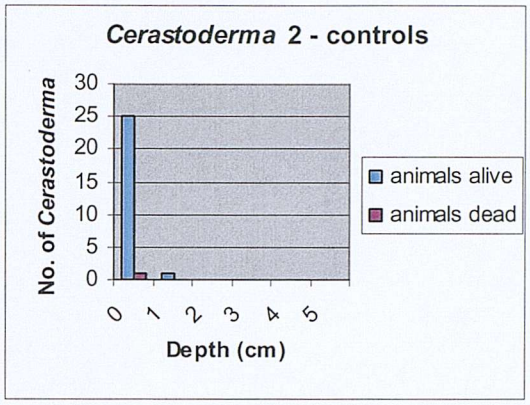


Figure 4.13 Distribution and state of control animals (*Cerastoderma* spp. from Poole Harbour) from experiment 4.

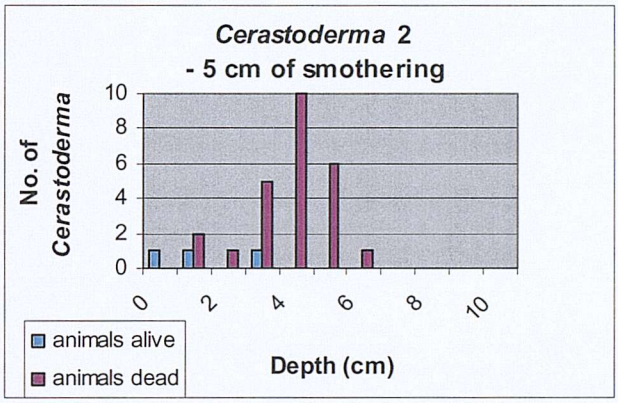


Figure 4.14 Position and state of cockles following 5 cm of smothering.

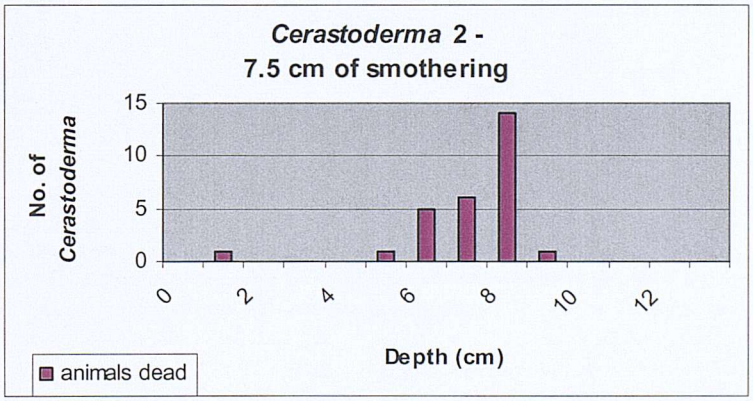


Figure 4.15 Position and state of cockles following 7.5 cm of smothering.

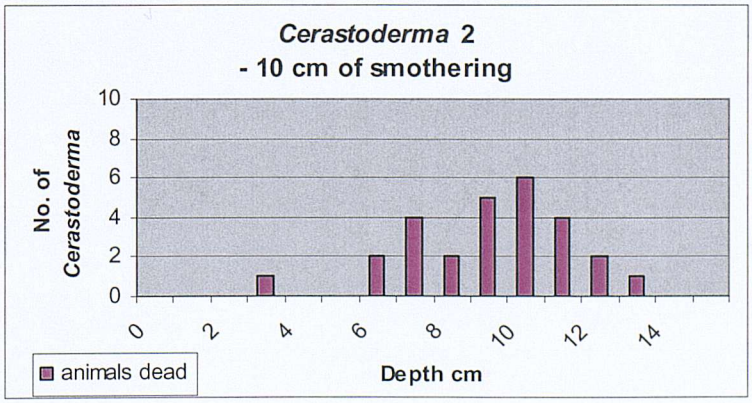


Figure 4.16 Position and state of cockles following 10 cm of smothering.

4.3.7 Mortality (experiments 1 - 4)

Figure 4.17 shows the mortality data plotted as a percentage for each experiment, against the depth of deposition (smothering). Mortality occurred as a result of smothering in two out of the three genera investigated. The dissimilarity between the shape of the curves for each species suggests that mortality, as a result of smothering fauna with simulated dredge spoil was species dependent.

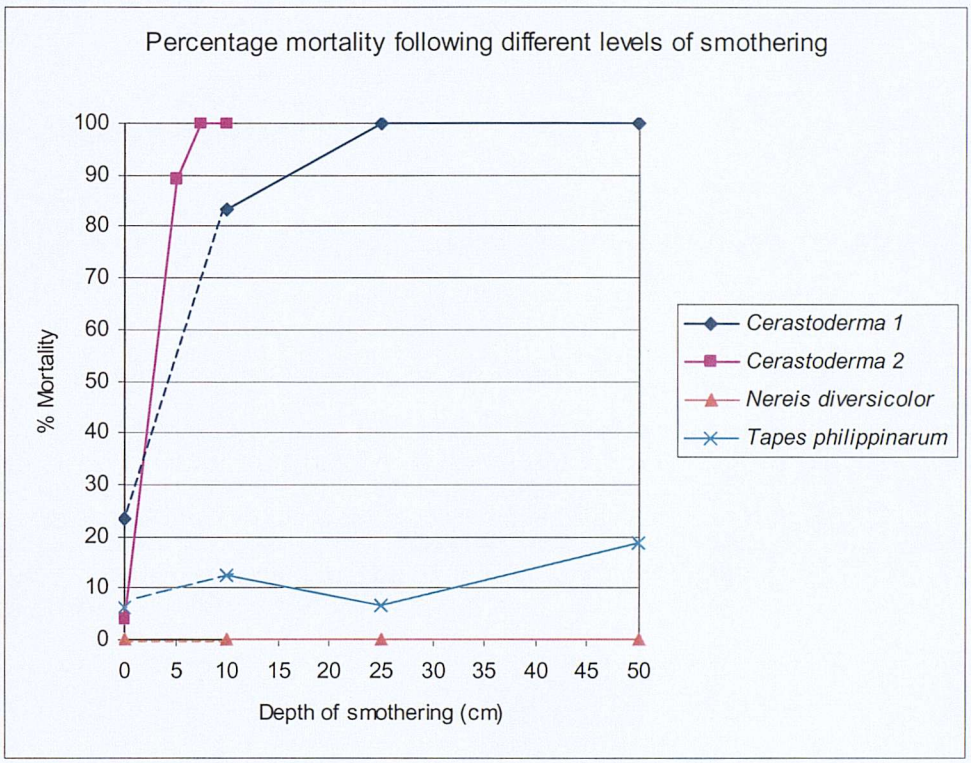


Figure 4.17 Mortality of macrofauna following smothering (as a function of depth). Dotted lines represent projected curves where no data was available (for species not tested at a smothering depth of 5 cm).

The results from these experiments imply that *Nereis* was highly tolerant of smothering (100 % survival), whereas cockles, *Cerastoderma*, appeared to be much more sensitive to smothering with high levels of mortality occurring under just 5 cm of sediment. The manila clam, *Tapes*, seemed more tolerant of the conditions imposed by smothering, although experimental animals were still found to have higher mortalities (8 – 20 %) compared to control animals (3 %), $P < 0.05$.

4.3.8 Experimental controls (experiments 1 - 4)

Figures 4.1, 4.5, 4.9 and 4.13 show the condition and depth of the control animals at the end of each experiment. The relatively higher survival of control individuals compared to treated animals indicates that increased mortality in the latter was an effect of the experimental treatment; i.e. smothering. It was observed that all animals, including the controls, had buried themselves in the core sediment prior to deposition.

	% Mortality	
Fauna	**Controls**	**Experiment**
Cerastoderma exp1	24	95
Cerastoderma exp 4	4	96
Tapes philippinarum	6	13
Nereis diversicolor	0	0

Table 4.9 Percentage mortalities recorded from controls versus smothered cores from experiments 1 – 4.

Table 4.9 summarises the variance in mortality between control and experimental fauna. Specimens of *Cerastoderma* used as controls experienced 24 % and 4 % mortality compared to 95 % and 96 % in experimental animals. For *Tapes* 6 % mortality was experienced in controls compared to13 % in experimental animals. No mortality was recorded for *Nereis*.

Species (Experiment)	**Estimated Probit response parameter**	**Log_{10}+1 depth**	**Depth (cm)**	**Standard error**
Cerastoderma spp. (1)	LD_{10}	0.74	0.55	± 0.12
	LD_{50}	1.39	2.46	± 0.11
	LD_{90}	2.04	11.01	± 0.12
Tapes philippinarum	LD_{10}	1.89	7.68	± 0.19
(2)	LD_{50}	5.94	*(86816.06 ‡)*	± 5.29
	LD_{90}	9.99	*(981747943.00 ‡)*	± 341.04
Nereis diversicolor	LD_{10}	-	-	-
(3) *	LD_{50}	-	-	-
	LD_{90}	-	-	-
Cerastoderma spp. (4)	LD_{10}	1.12	1.31	± 0.11
	LD_{50}	1.40	2.49	± 0.11
	LD_{90}	1.67	4.73	± 0.11

Table 4.10 Summary of Probit analysis results for mortality data. Confidence limits are 95 % ($P < 0.05$). * zero mortality was recorded for this species. ‡ Estimated statistical values returned by the Probit test for *Tapes* (in italics) were clearly outside possible depth ranges, which reflects the low mortality rates recorded for this species.

Table 4.10 gives the values for parameters estimated by Probit analysis. LD50 represents the depth at which an estimated 50 % of the population would die, LD10

represents the estimated depth for lethal effects on 10 % of the population and LD90 is the estimated depth at which 90 % of the population would succumb. The statistics confirm that differences in response between species are highly significant. The results also confirm that *Cerastoderma* was significantly less tolerant to smothering than *Tapes* during these experiments, (with an estimated LD50 of around 2.5 cm compared to several metres). Similar values were obtained for the cockles in both experiments 1 and 4. Values could not be estimated for *Nereis* as no mortality was recorded, the zero response for this species indicates high tolerance to smothering.

4.3.9 Vertical migration (experiments 1 - 4)

Figure 4.18 shows the measured vertical migration of *Cerastoderma* following smothering with 5 cm of sediment. The positive values given represent migration in an upwards direction through the deposited material in centimetres, measured from the interface between the depositional and original sediment. Measurements were taken from the upper surface of the bivalve. Migration plots include data for all individuals from that experiment (including those that were recovered dead at the end of the experiment).

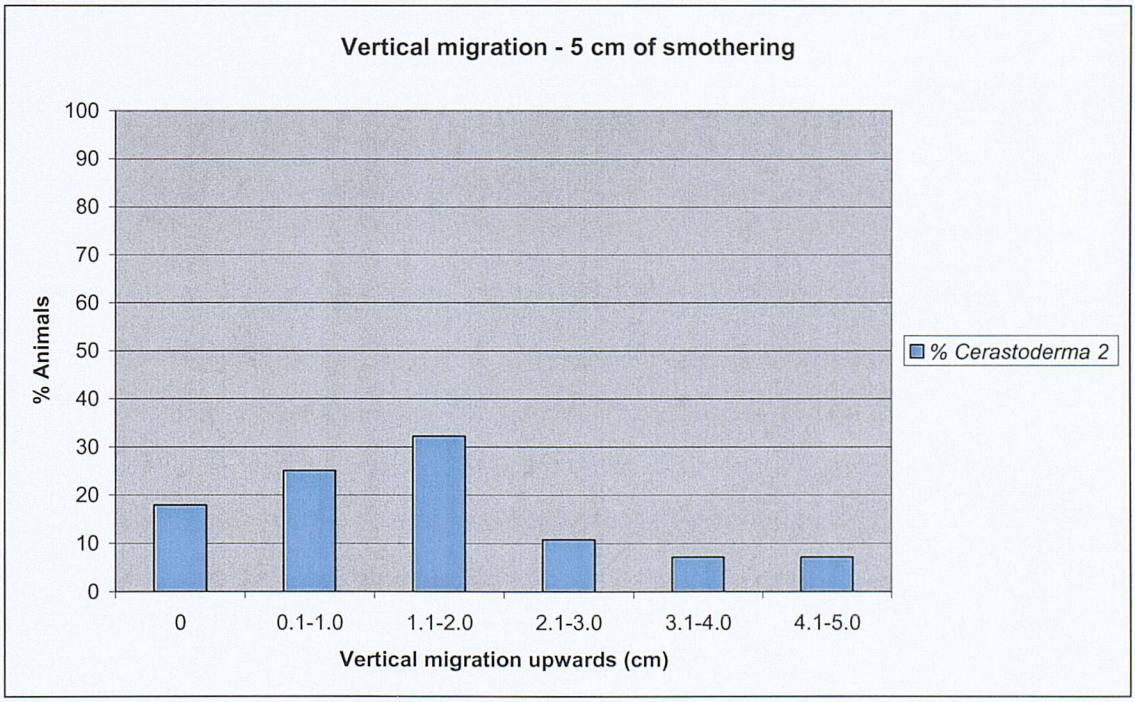


Figure 4.18 Histogram showing vertical migration in *Cerastoderma* (experiment 4; *Cerastoderma* 2) after being smothered with 5 cm of sediment.

The plot shows that over 80 % of cockles underwent some migration. Approximately one third of the animals (32 %) migrated up to 2 cm into the deposited material, 25 % migrated up to 1 cm, 11 % migrated between 2 - 3 cm, 7 % 3 - 4 cm, and 7 % migrated the maximum 5 cm (= to the surface of the deposited sediment). Around 18 % of animals did not appear to have moved into the deposited layer at all and were recovered from the underlying sediment at the end of the experiment.

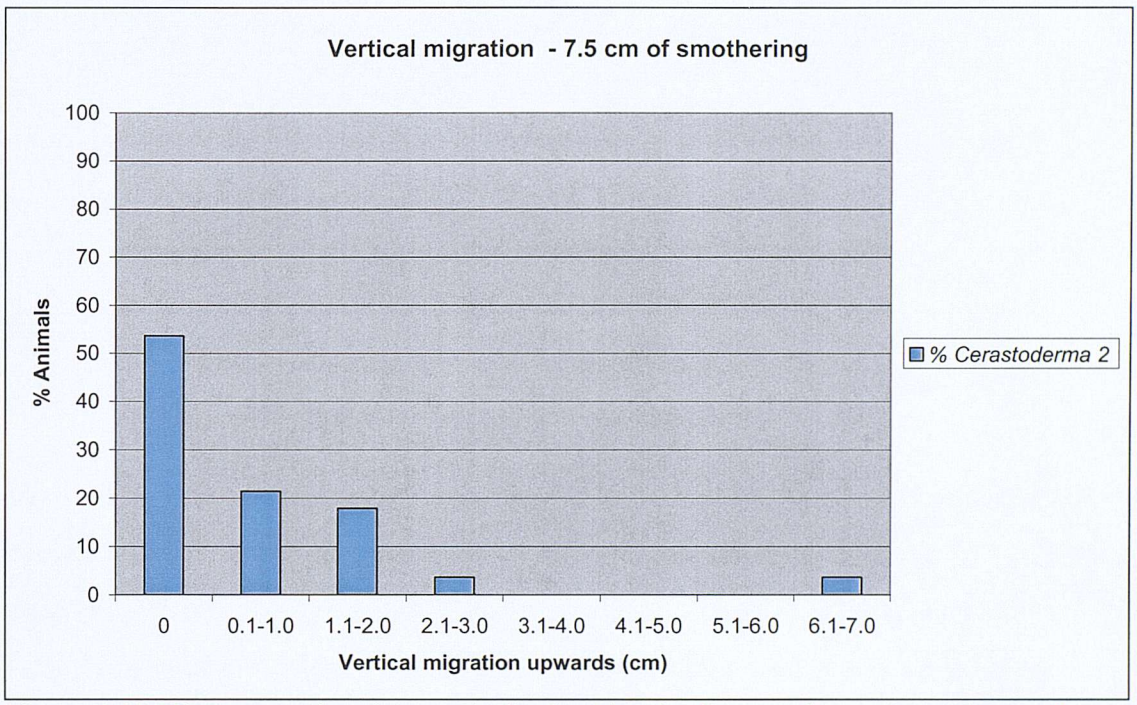


Figure 4.19 Histogram showing vertical migration in *Cerastoderma* (experiment 4; *Cerastoderma* 2) after being smothered with 7.5 cm of sediment.

Figure 4.19 shows the response of *Cerastoderma* to being smothered with 7.5 cm of sediment. At this level of deposition, only 45 % of cockles migrated into the deposited material by the end of the experiment. Of these, only 40 % appeared to move up into this layer by a maximum 2 cm (inferring that they were still buried by more than 5 cm of sediment). Measured migration of cockles for this level of sediment load was significantly less than for 5 cm deposition, and reflects the higher mortality rate (100 %) observed during this experiment.

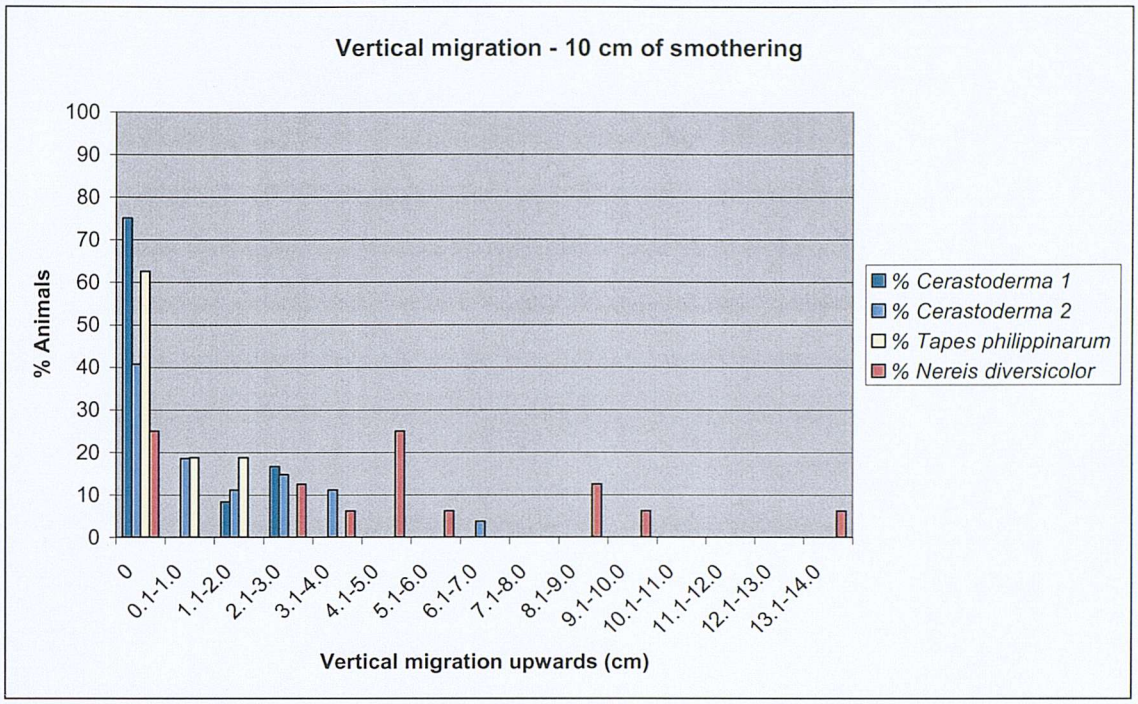


Figure 4.20 Histogram showing vertical migration in macrofauna after being smothered with 10 cm of sediment (experiment 1; *Cerastoderma* 1, experiment 2; *Tapes*, experiment 3; *Nereis*, experiment 4; *Cerastoderma* 2).

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Figure 4.20 shows the response of the fauna from experiments 1 - 4 for a depositional depth of 10 cm. *Cerastoderma*, shows little migratory capacity when buried under this depth of material; 75 % of cockles from experiment 1 were still found to be in the underlying sediment when recovered. Some migration was recorded for 60 % of cockles in experiment 4, however, a further 30 % of individuals were shown to have moved upwards less than 3 cm and therefore remained more than 7 cm below the sediment surface. The limited ability to reach the upper levels of the sediment was also reflected in an increased rate of mortality (Figure 4.17). The second bivalve used in these experiments, *Tapes*, gave similar results. When buried under 10 cm of additional sediment nearly two thirds of individuals did not reach the deposited sediment. No clams were recovered at a depth of less than 8 cm, indicating that the maximum migration into the smothered material recorded was just 2 cm. In contrast, the results obtained from *Nereis* were quite different from those of the bivalves; this highly motile species appeared to be randomly distributed throughout the entire sediment column.

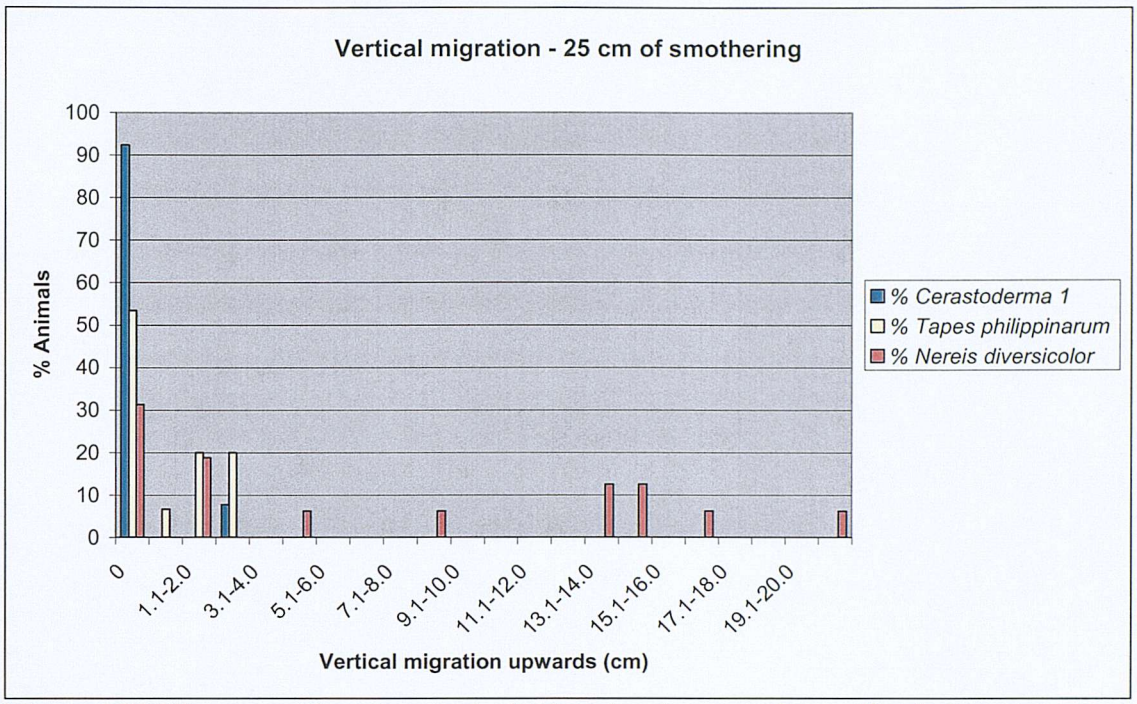


Figure 4.21 Histogram showing vertical migration in macrofauna after being smothered with 25 cm of sediment (experiment 1; *Cerastoderma* 1, experiment 2; *Tapes*, experiment 3; *Nereis*).

Under 25 cm of deposition (Figure 4.21), over 90 % of *Cerastoderma* were recovered from the base sediment with little movement towards the surface. Almost half (around 45 %) of the clams had attempted migration, although the maximum distance moved into the smothering material was just 3 cm, indicating that this genus did not possess the motility to burrow upwards such a distance. As before the ragworms (*Nereis*) appeared to be scattered throughout the cores at varying distances from the surface.

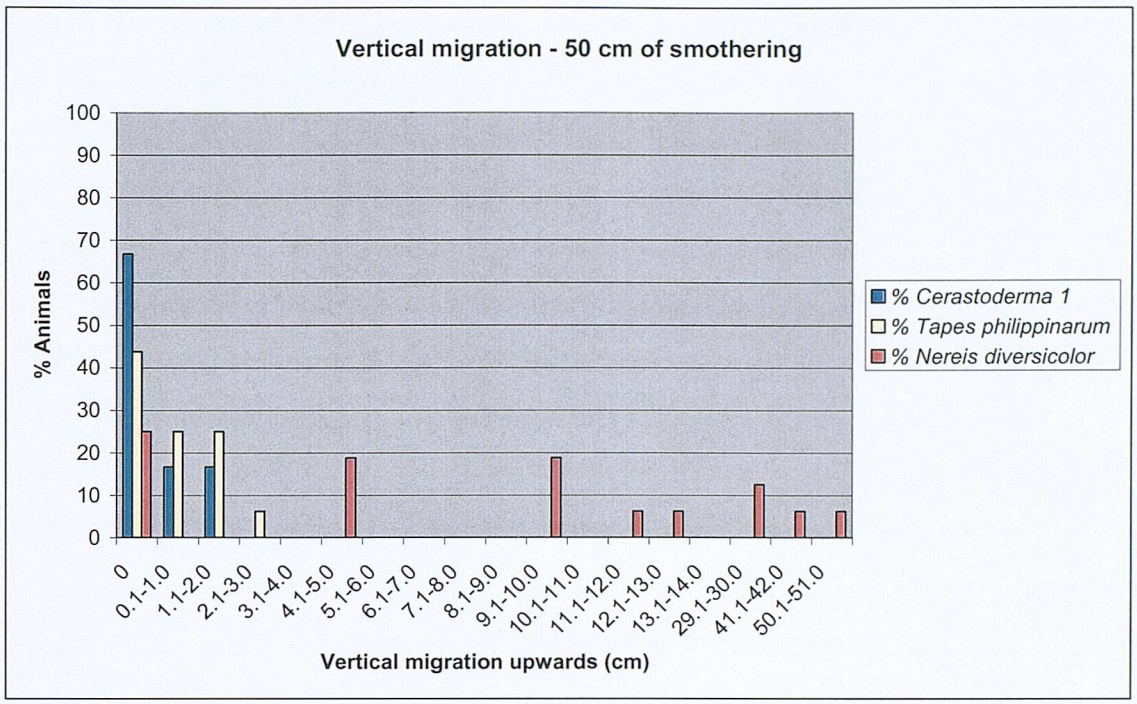


Figure 4.22 Histogram showing vertical migration in macrofauna after being smothered with 50 cm of sediment (experiment 1; *Cerastoderma* 1, experiment 2; *Tapes*, experiment 3; *Nereis*).

The results in Figure 4.22 show that, under a deposited layer of half a metre, *Nereis* was the only species able to reach to the new sediment surface. As before, the vertical burrowing ability of the bivalves appeared to be insufficient or inhibited at this depth and was limited to a few centimetres at maximum. *Tapes* appears to display slightly better burrowing ability under these conditions than *Cerastoderma*, with just over half the clams reaching the deposited sediment, compared to just one third of cockles.

Experiment	Species	Source	*DF*	*ss*	*ms*	*F*	*P*
1	*Cerastoderma* spp.	Smothering depth (cm)	2	0.48	0.24	1.21	0.241
		Tank *	3	1.30	0.43	2.71	0.068
		Interaction	6	6.20	1.03	6.42	0.000
2	*Tapes philippinarum*	Smothering depth (cm)	2	0.61	0.30	0.64	0.54
		Tank	3	1.99	0.66	1.38	0.264
		Interaction	6	1.92	0.32	0.67	0.675
3	*Nereis diversicolor*	Smothering depth (cm)	2	13.56	6.78	2.22	0.124
		Tank	3	6.06	2.02	0.66	0.581
		Interaction	6	35.02	5.84	1.91	0.106
4	*Cerastoderma* spp.	Smothering depth (cm)	2	6.09	3.05	6.30	0.003
		Tank	3	0.67	0.22	0.46	0.711
		Interaction	6	15.55	2.59	5.36	0.00

Table 4.11 Summary of the factorial design ANOVA (Two-Way, General Linear Model) on the migration data from Experiments 1-4 (*DF* = degrees of freedom, *ss* = sum of squares, *ms* = mean square, *F* = *F*-ratio statistic between treatment variance/within treatment variance or treatment *ms*/error *ms*, *P* = level of significance). * tests for significant variance resulting from individual tanks as a factor.

The results from the ANOVAs carried out on the migration data from experiments 1 to 4 are given above in Table 4.11. The statistical analyses were carried out to establish whether variation between the different depths of smothering tested was significant, and whether statistically significant experimental effects existed between the tanks (i.e. a 'tank factor'). The results of the ANOVA indicate that variation resulting from smothering depth as a source was not statistically significant as a factor ($P > 0.05$): In other words; the different levels of smothering depth applied during these experiments did not significantly affect migration distance *per se*. Interaction in both smothering depth and tanks was found to be significant for experiments 1 and 4. The test results also indicate that no significant variation was present between the tanks.

4.3.10.1 Experiment 5: Smothering of mixed species assemblages

Figure 4.23 shows the position of macrofauna, given as a percentage for each species, identified from untreated cores (controls) at the end of experiment 5. It shows that, in agreement with previous observations (Chapter 2), individuals of the genera *Caulleriella*, and *Tharyx* were recovered from the top few centimetres of the sediment. The majority of individuals of the larger cirratulid, *Cirriformia*, were found to be slightly deeper. The oligochaete, *Tubificoides pseudogaster*, was also recovered from the top 4 cm of the sediment ($n = 6$), and the spionid, *Pygospio*, was found throughout the cores ($n = 9$). No individuals were found at a depth greater than 6 cm of sediment, supporting earlier observations that the maximum depth of colonization for macrofauna at Hythe is around 5 cm.

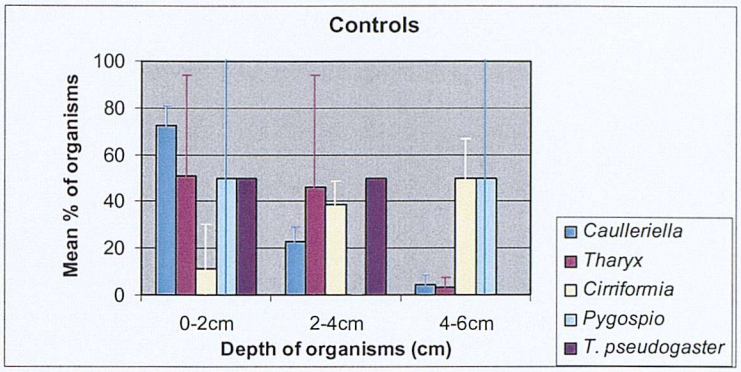


Figure 4.23 Histogram showing averaged depth distribution of fauna in untreated cores (controls) at the end of the mixed species assemblages experiment (experiment 5). Error bars represent standard deviation from the mean.

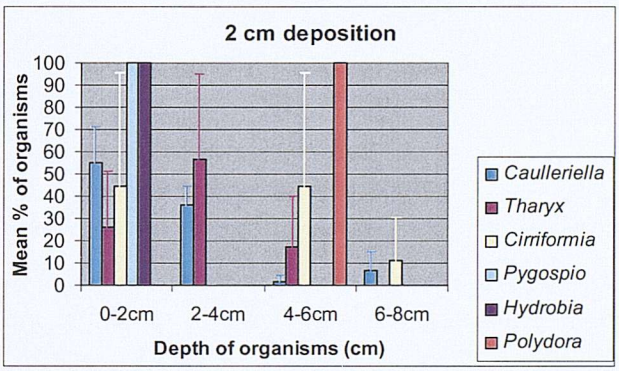


Figure 4.24 Mean depth distribution of fauna from experiment 5, retrieved from cores subjected to 2 cm of smothering. Error bars represent σ_{n-1}.

Figure 4.24, gives the results of the four cores subjected to 2 cm of deposition. It indicates that the majority of individuals responded to the treatment by migrating upwards into the deposited layer (*Hydrobia*, 100 %; *Pygospio*, 100 %; *Caulleriella*, 55 %; *Cirriformia*, 44 %; *Tharyx*, 27 %). Only 11 % of *Cirriformia*, and 7 % of *Caulleriella* were found more than 6 cm below the new surface.

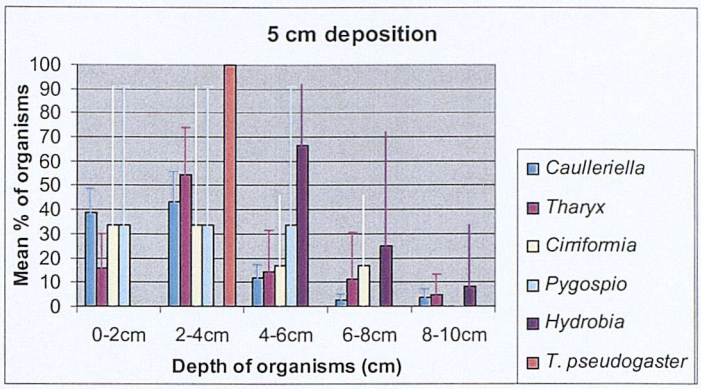


Figure 4.25 Mean depth distribution of fauna from experiment 5, retrieved from cores subjected to 5 cm of smothering (error bars represent σ_{n-1}).

Figure 4.25 gives the vertical position of animals within the sediment following 5 cm of smothering. Again, the majority of individuals appear to have responded to the treatment by migrating into the deposited material (*Tubificoides*, 100 %; *Caulleriella*, >85 %; *Tharyx*, 70 %; *Cirriformia*, 65 %; *Pygospio*, 65 %). In contrast, the gastropod, *Hydrobia ulvae*, did not migrate up into the deposited layer but remained at a depth of 4 - 6 cm, suggesting tolerance limits to burial were exceeded at this smothering depth.

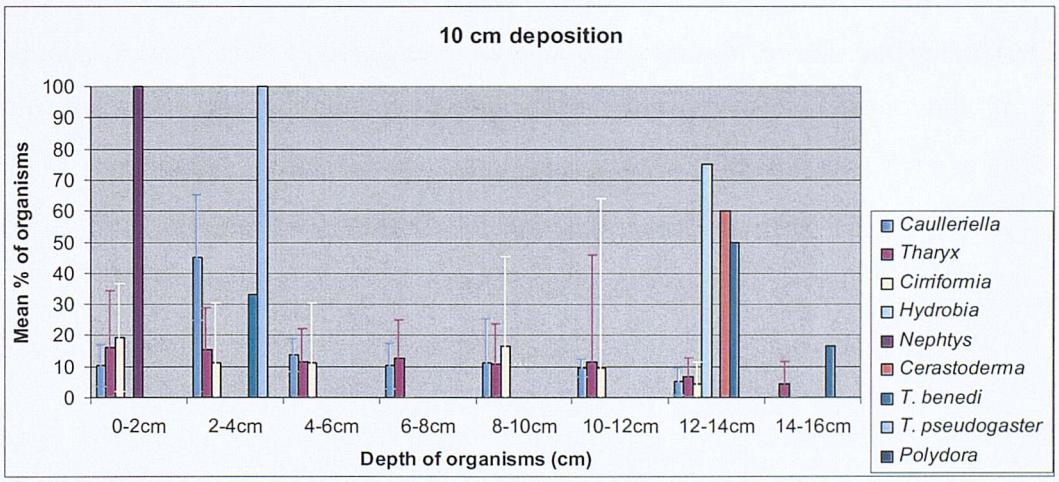


Figure 4.26 Mean depth distribution of fauna from experiment 5, retrieved from cores subjected to 10 cm of smothering (error bars represent σ_{n-1}).

Figure 4.26 shows that, under a smothered depth of 10 cm, migration towards the surface was limited to some extent. At this depth of deposition around 65 % of *Caulleriella* were recovered from the top 6 cm of sediment. However, only 43 % of *Tharyx* and 42 % *Cirriformia* reached this level and no *Hydrobia* were found above a depth of 12 cm. The oligochaete *Tubificoides pseudogaster* was found between 2 - 4

cm deep, although the majority of individuals of a second species of this genus; *Tubificoides benedi*, were found at a depth of 12 - 16 cm.

4.3.10.2 Vertical migration (experiment 5)

Figures 4.27 to 4.29 give the results from experiment 5 plotted as (frequency distribution) histograms of upwards vertical migration following smothering of the dominant three species of Cirratulids from Hythe (*Caulleriella caputesocis*, *Tharyx marioni* and *Cirriformia tentaculata*) with 2, 5 and 10 cm of sediment. Figure 4.27 shows that after just 2 cm of deposition, approximately 40 % of the polychaetes had migrated into the deposited layer. Following 5 cm of smothering, the majority of the smaller genera *Caulleriella* and *Tharyx* (around 80%) had migrated upwards into the deposited sediment, compared to 50 % of the larger *Cirriformia* (Fig. 4.28). Following the addition of 10 cm of deposited material to the cores (Figure 4.29), 10 % of *Caulleriella*, 25 % of *Tharyx* and 30 % of *Cirriformia* remained in the original sediment, with most of the worms retrieved from the deposited layer. 73 % of the *Caulleriella*, 50 % of *Tharyx* and 41 % of *Cirriformia* were found to be at a depth of less than 6 cm from the sediment surface at the end of the experiment. These results indicate that significant migration was recorded by these taxa when smotherered by up to 10 cm of sediment.

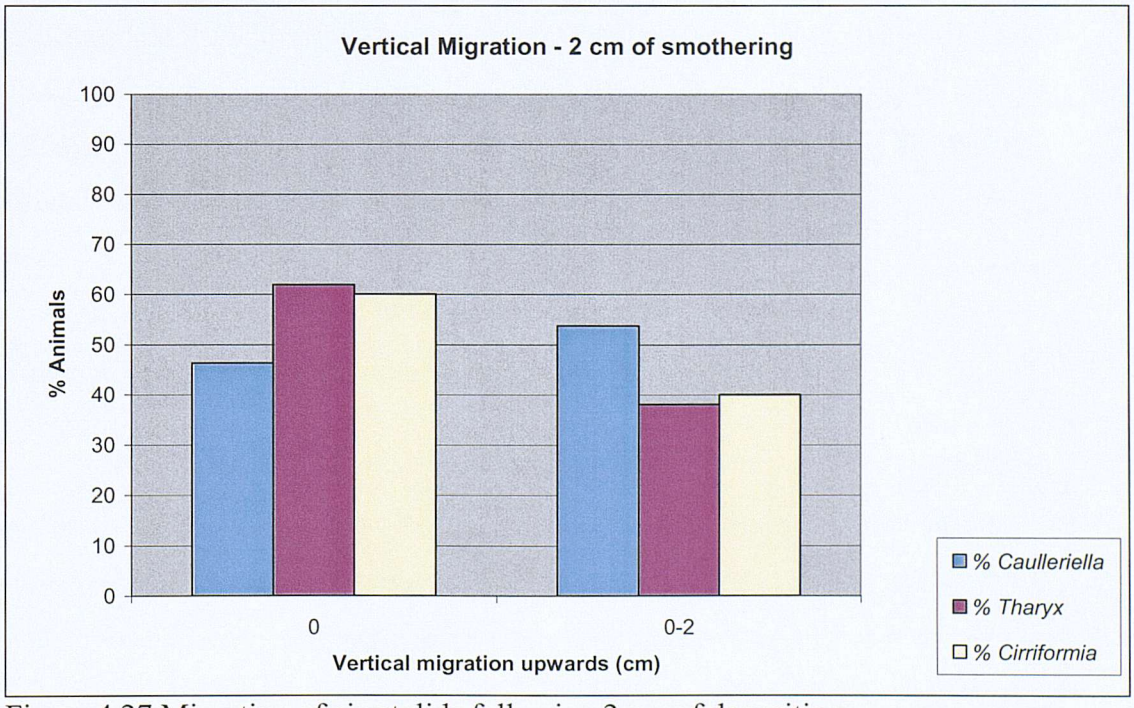


Figure 4.27 Migration of cirratulids following 2 cm of deposition

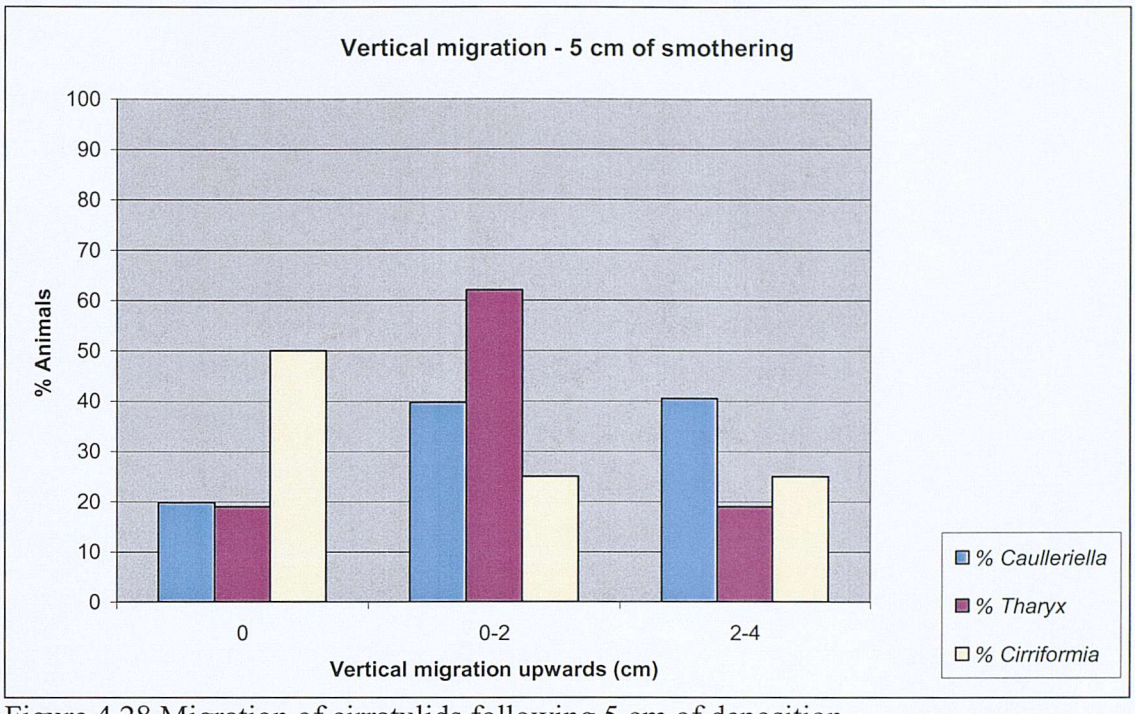


Figure 4.28 Migration of cirratulids following 5 cm of deposition

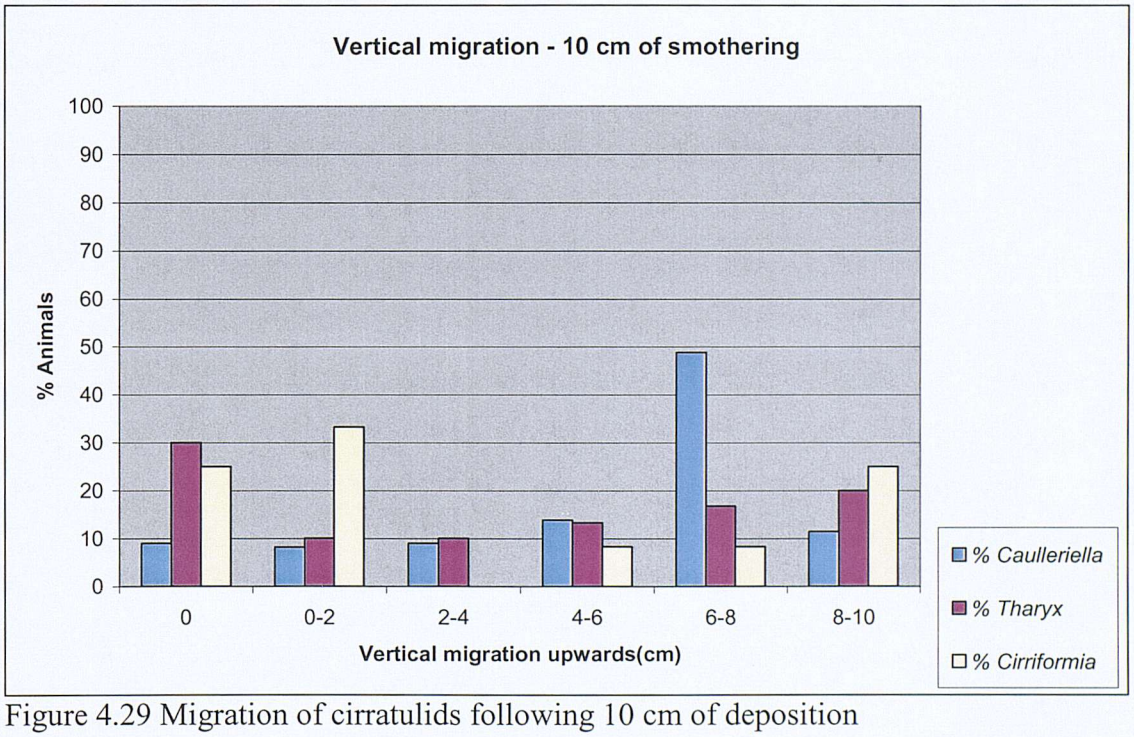


Figure 4.29 Migration of cirratulids following 10 cm of deposition

The results of the ANOVA carried out on the pooled migration data for the three species of cirratulids are given in Table 4.12. The analysis was carried out on percentage data in order to allow comparison of data where *n* varied. The results indicate that, for this experiment, there was no significant difference in migration as a response to smothering between the three cirratulid species (P >0.991), although

smothering depth was a highly significant factor affecting migration (P = 0.000). The interaction factor is a statistical term, which is tested to identify lack of independence (crossover) between the samples. No significant interaction was implied by the data.

Experiment	Source	*DF*	*ss*	*ms*	*F*	*P*
5	Species	2	0.04	0.02	0.01	0.991
	Smothering depth (cm)	2	54.16	27.08	1.78	0.000
	Interaction	4	0.38	0.10	0.04	0.997

Table 4.12 Summary of the factorial design ANOVA (Two-Way, General Linear Model) on the migration data from *Caulleriella*, *Tharyx* and *Cirriformia,* experiment 5.

Table 4.13 gives the results of the Kruskal Wallis tests on the migration data from experiment 5 for individual species. The test statistic and P-values obtained (P = 0.00) imply that vertical migration in *Caulleriella* and *Tharyx,* as a response to smothering, is very significantly affected by the depth of smothering. The test results imply that migration distance in *Cirriformia*, was not affected by the depths of deposition (2, 5 and 10 cm) applied during this experiment.

Species	Smothering depth (cm)	Median	*z*	*DF*	*H*	*P*
Caulleriella	2	2.00	-7.52	2	135.62	0.000
	5	2.00	-4.96			
	10	8.00	10.82			
Tharyx	2	0.00	-3.51	2	16.01	0.000
	5	2.00	0.05			
	10	5.00	3.19			
Cirriformia	2	0.00	-1.45	2	3.44	0.179
	5	0.71	-0.58			
	10	1.41	1.71			

Table 4.13 Summary of results from Kruskal Wallis tests on smothering depth from migration data of individual species from experiment 5

4.3.10.3 Estimated survival; experiment 5

Estimated potential survival was calculated, for individuals reaching the top 6 cm of the sediment. The results of calculations for the polychaete *Caulleriella* are given in Figure 4.30. The graph indicates that estimated survival was high for lower levels of smothering (95 % at 5 cm), and decreased to around 70 % when deposition was increased to 10 cm.

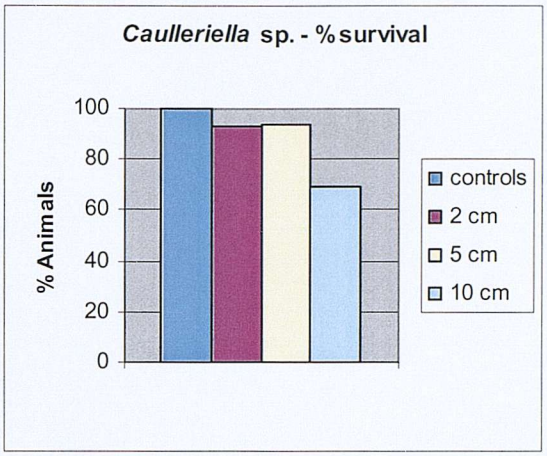


Figure 4.30 Calculated potential survival of *Caulleriella* estimated following smothering (n = 378).

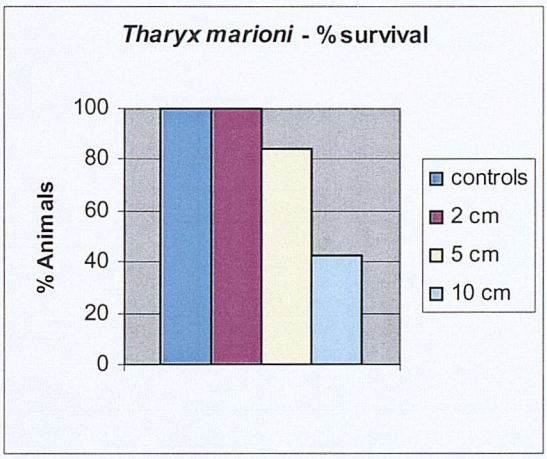


Figure 4.31 Calculated potential survival of *Tharyx* estimated following smothering (n = 93).

Estimated or calculated potential survival for *Tharyx* is shown in Figure 4.31. The data implied that this species is more sensitive than *Caulleriella* to smothering, with estimated mortality increasing to 42 % at 10 cm deposition.

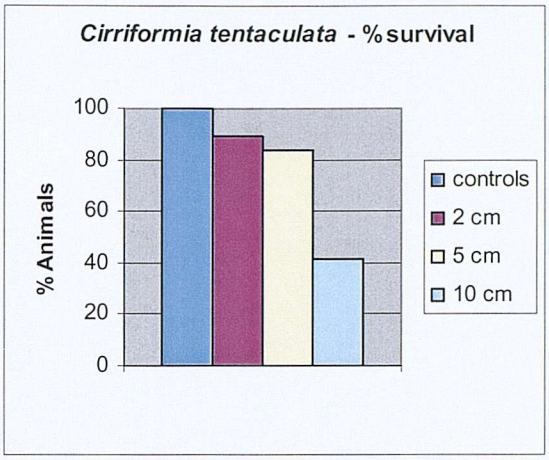


Figure 4.32 Calculated potential survival of *Cirriformia* estimated following smothering ($n = 29$).

The above graph indicates that mortality in *Cirriformia* may have steadily increased in response to smothering, increasing to 41 % following 10 cm of deposition.

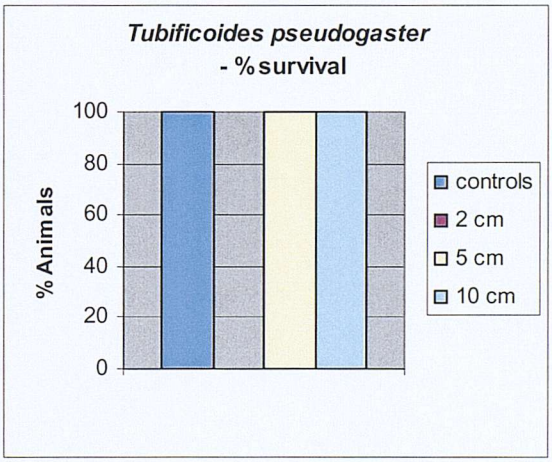


Figure 4.33 Calculated potential survival of *Tubificoides* estimated following smothering ($n = 6$).

The data implied that *Tubificoides pseudogaster* was more tolerant to smothering than the cirratulids (Figure 4.32). Note that the lack of bar for 2 cm of smothering is due to this species being absent from these cores (which were randomly sampled – this species has been infrequently found in samples from Hythe), and does not imply mortality.

These results should be interpreted with caution as, they rely on the assumption that individuals reaching a depth of <6 cm from the surface would survive, and those lower down would perish. In addition *n* is significantly smaller for *Tubificoides* which is

present in lower densities leading to a greater expected error. This is also the case for *Hydrobia* and *Pygospio* (Figures 4.34 & 4.35), which were also present in low densities.

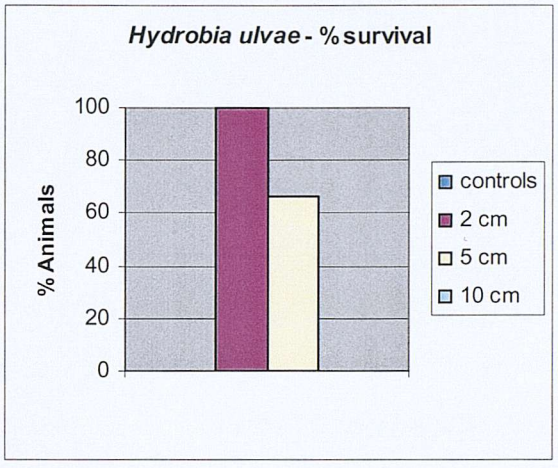
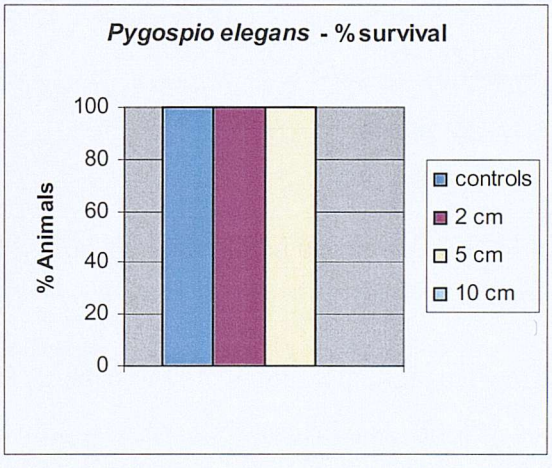


Figure 4.34 Calculated potential survival of *Hydrobia* estimated following smothering ($n = 12$).



4.35 Calculated potential survival of *Pygospio* estimated following smothering ($n = 9$).

Figure 4.36 shows estimated survival plotted against depth of deposition. It indicates that, in agreement with the single species experiments, response to smothering appears to be species specific, and is also sensitive to the depth of the deposited material.

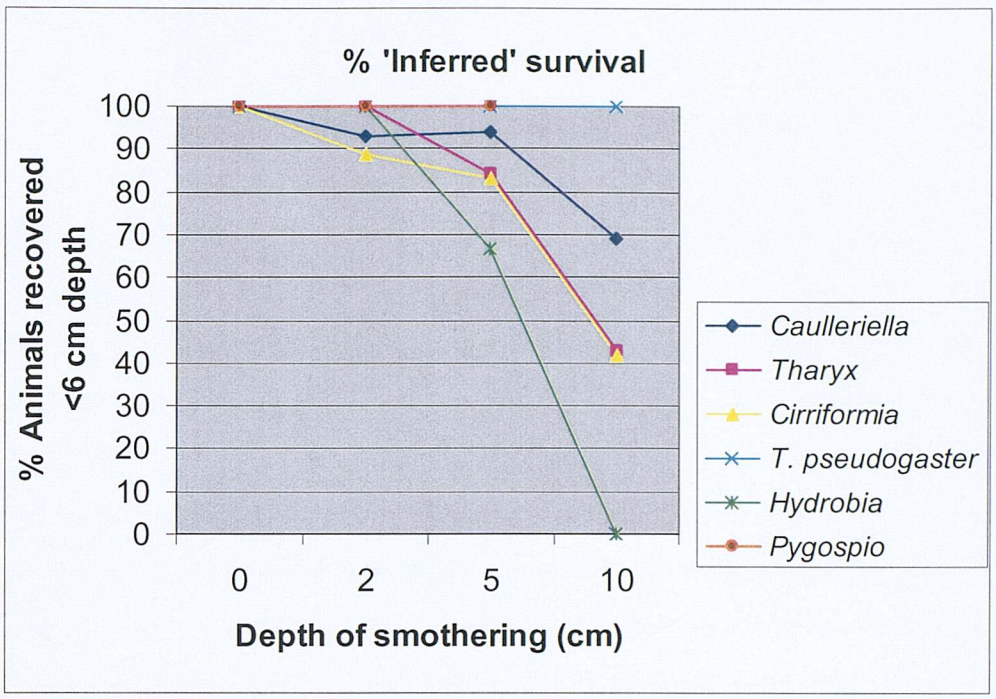


Figure 4.36 Estimated mortality calculated as a percentage from experiment 5, plotted against depth of smothering (survival is inferred here by the organism being recovered from a depth of <6 cm at the end of the experiment)

The gastropod, *Hydrobia ulvae*, appears to be the most sensitive to smothering amongst the taxa included, and the spionid polychaete, *Pygospio elegans*, and oligochaete, *Tubificoides pseudogaster*, the most tolerant. The dominant taxa at the site, the cirratulids of the genera *Caulleriella*, *Tharyx* and *Cirriformia*, appear to be relatively tolerant of lower levels of deposition, but indicated increasing sensitivity to increasing sediment load.

4.4 Discussion

The effects of burial under natural sediment on macrofauna have been investigated in the laboratory (e.g. Brenchley, 1981; Maurer *et al.*, 1986). These results suggest that tolerance to smothering varies widely between individual species. However, these studies focus on just a few benthic species and do not include results for the genera found at Hythe and selected for this study. It was suggested by Brenchley (1982), that larger, more mobile animals are more able to tolerate burial by burrowing upwards through sediment before conditions (depletion of oxygen etc.) cause mortality, i.e. that their motility enables them to avoid deleterious effects by escaping. Consequently it was suggested that smaller, more sedentary fauna, of restricted motility and confined to the top layer of sediments, may be less able to survive burial due to a lesser ability to burrow through deposited material before suffocation occurs (Brenchley, 1981). Thus it is hypothesised that the larger, deeper, active burrowers and more motile species will be able to survive burial at greater depths than smaller, more sedentary subsurface tube builders.

Whilst experiments on individual species will give valuable information and may allow estimations of tolerance and survival to smothering from activities such as recharge, interspecific biological interactions (i.e. inhibition or facilitation, see below) may modify what actually occurs in the field. Many communities possess attributes that are greater than the sum of individual species' abilities due to mutualistic or commensalistic interspecific or biotic interactions of facilitation (Begon *et al.*, 1996). Facilitation is a common phenomenon in community dynamics, where one species alters conditions or the availability of a resource so that another species is able to colonize and survive, which would be unable to compete or survive without the modifications made by the first species (Whitlatch & Zajac, 1985). In the event of recharge, rapid burrowing as a means of escape by larger, more motile organisms may, to some extent, facilitate the survival of smaller weaker burrowers by providing enough oxygen through bioturbation to facilitate survival until the upper sediment layers can be reached. This also, however, depends upon the ability of the less motile animals to migrate vertically upwards, as if this were not sufficient, they would be unable to survive in the long term due to a lack of food resources.

4.4.1 Vertical migration of infauna as a response to smothering

The responses of several species of infauna to smothering or burial were investigated during these experiments by quantitative measurement of vertical migration: Migration of infauna has been recorded in previous studies as a response to oxygen stress and /or increased concentrations of reduced compounds (Diaz & Rosenberg, 1995; Maurer *et al.*, 1980, 1981, 1982).

Vertical migration upwards may be seen as an escape mechanism or survival strategy to avoid unfavourable conditions imposed by mass deposition and the resultant smothering such as oxygen stress. A layer of deposition would be expected to block burrow entrances, inhibiting irrigation and causing rapid depletion of oxygen. The development of anaerobic conditions would also trigger the production of hydrogen sulphide and other reduced compounds, which are products of anaerobic bacterial metabolism and are highly toxic to the infauna (Diaz & Rosenberg, 1995). In cases of recharge, where a time-lapse of hours to days may be present between dredging and deposition, anaerobic bacterial metabolites may be present in the dredged material in elevated concentrations following the decomposition of fauna (although it seems probable that these would largely be dispersed during recharge). It has been suggested that it may be the presence of chemicals, such as hydrogen sulphide, which trigger migration following burial under a layer of sediment. However, Maurer *et al.* (1985) found that migration was generally undertaken within a day of burial and completed by the eighth day, whereas hydrogen sulphide concentrations did not accumulate so rapidly (and often not for fourteen days following burial). The author therefore concluded that it was unlikely that increased concentrations of hydrogen sulphide was the chemical factor stimulating the onset of migration, but hypothesised that it could be a combination of low dissolved oxygen concentration and high ammonia concentration which occurred rapidly (Maurer *et al.*, 1985).

The ability of infauna to burrow and the rates at which burrowing occurs is directly related to animal morphology. Bivalves with a large cylindrical foot are well equipped to burrow whereas those with a reduced foot or one modified for byssal attachment are less adapted to prolonged or extended movement through the sediment. Gastropod molluscs are generally not as well equipped for burrowing as bivalves (Chandrasekara & Frid, 1998; Stanley, 1970), whereas polychaetes with a well-developed proboscis

and parapodia are good burrowers (Trevor, 1977). Brenchley (1981) found that the species most sensitive to sedimentation during laboratory experiments were dense, shallow-dwelling, tubiculous deposit feeders. Numbers of the tubiculous amphipod *Corophium volutator* were found to be considerably decreased by as little as two centimetres of deposition per month in a field study investigating the impacts of sedimentation (Turk & Risk, 1981).

The experimental results obtained from the current study indicate differences between three species which appear to relate to their functional morphology and motility: *Nereis*, which is known to be a 'free-living' and highly motile species appears to be unrestricted by the sediment and was found to migrate/burrow freely during this study without any apparent adverse effects. Maurer *et al.*, (1986) suggested that *Nereis succinea* would be able to burrow upwards through up to 0.9 metres of fine sediment following burial, and Saila and Pratt (1972 in Nichols *et al.*, 1978) determined that *Nephtys incisa* (another highly active polychaete with well-developed proboscis and parapodia) was capable of moving up through 21 centimetres of mud in less than 24 hours.

In contrast, the bivalves *Cerastoderma* and *Tapes* appear to have been unable to migrate more than a few centimetres through the imposed sediment load. These are both shallow or near-surface dwelling species. They possess relatively short siphons and, as protection from predation they posses thick, heavy shells, which may also confer limited burrowing ability (Boyden, 1969; Chang & Levings, 1978; Lee, 1999; Maurer *et al.*, 1986). Bivalves burrow by forming a penetration anchor with the foot when the shell is pressed against the substratum and the abductor muscle is relaxed. The foot then extends into the sediment and, as the siphons close and the abductor muscle contracts, water is forced out around the foot, which then presses against the sediment forming a terminal anchor. The anterior and posterior retractor muscles then contract which pull the shell into the sediment aided by a rocking action (Raffaelli & Hawkins, 1996). For the bivalve to burrow upwards in this way, it must first turn itself upside down. Detailed descriptions of *upwards* burrowing in bivalves and the mechanisms used did not appear to be present in the literature.

It is important to note that, although many of the clams were recovered alive at the end of the experiment (4 days post smothering), animals found at a depth greater than approximately 10 cm would be extremely unlikely to survive indefinitely as they would be prevented from feeding or aerobic respiration due to the depth of burial. The experiments were terminated at this point to avoid complications that may arise due to animals being kept in the laboratory for more than a week without a replenishable food supply or the accumulation of toxic waste products, as these factors would both be expected to elevate mortality and interfere with results.

Total or partial closing of the valves and reduced metabolic rate has been reported to be a common response in bivalves to hypoxic conditions (Sobral & Widdows, 1997). However, in a situation where the animal is buried under a considerable depth of sediment (such as would be expected to occur following successful recharge), it is believed that long term survival would rely upon the ability to 'escape' to surficial oxygenated sediments, and construct an oxygenated burrow within the deposited layer.

The results of the ANOVA tests carried out on the migration data indicate that variance in vertical migration distance as a function of smothering depth was only statistically significant in experiment 4 (*Cerastoderma*). This experiment was the second experiment carried out using cockles. It was conducted with decreased levels of smothering (5, 7.5 and 10 cm), because of the high sensitivity to deposition indicated in experiment 1, in which nearly all the cockles died. Sensitivity to varied depths could only be evaluated from the results of experiment 4, as the depths of burial used in experiments 1 and 2 (10, 25 and 50 cm) were beyond the bivalves' tolerance, and therefore inhibited migration equally. This result is not unexpected as both *Cerastoderma* and *Tapes* are shallow burrowers that typically live near the surface. In addition, a previous study that investigated the tolerance of *Cardidae* bivalves to burial or deposition found that high mortalities occurred when the sediment load approached 10 cm (Chang & Levings, 1978). By contrast, *Nereis* is apparently unaffected by any of the depths of smothering tested (experiment 3), and it appears that this actively burrowing polychaete is highly tolerant to burial.

It appears conclusive therefore that the results of these experiments provide further evidence that migration ability following smothering is directly related to functional

morphology and motility, with *Nereis* being by far the most adept at escaping the effects of burial. It appears that both *Cerastoderma* and *Tapes* lack the ability to escape more than 10 cm of burial by rapid burrowing. However, *Tapes* displayed a greater tolerance to conditions in the sediment caused by smothering, and was able to remain alive for several days after, whereas *Cerastoderma* was not. This indicates that the clams are more tolerant than cockles to short-term burial, but that both species are sensitive to long-term burial. This implies periodical smothering by shallow depths of sediment from recharge may have less of an impact on these bivalves if given long enough time intervals between disposal events to recover. However, for this strategy to be successful the depth of sediment deposited would need to be closely controlled and maintained well below 10 cm, preferably less than 5 cm. In contrast, a one-off disposal of dredge spoil, resulting in a deposited layer thicker than 10 cm would be likely to have a severely deleterious effect on the bivalves, and could result in wiping out all individuals in the area of deposition.

The analysis of variance of the migration data from experiment 5 indicated that there was no significant variation between the migratory response of the three dominant cirratulids, *Caulleriella*, *Tharyx* and *Cirriformia*. This result also supports the above observations that migration, as a response to burial, is related to motility and functional morphology: As these three genera are closely related, it is perhaps not surprising that their response in terms of migration was similar. The subsequent one-way Kruskal Wallis test did however indicate slight variation in depth distribution between *Cirriformia* and the other two species. This variation may be explained by the fact that *Cirriformia* is much larger than *Caulleriella* or *Tharyx* (a total body length of <10 cm compared to a length of <1 cm), and would therefore be able to construct and live in deeper burrows.

The results of the ANOVA and Kruskal Wallis tests indicated that smothering depth was the most significant factor as a cause of variance in the migration data from experiment 5. The proportion of organisms undergoing migration, as well as the amount of migration measured, varied with smothering depth. Following 2 cm of deposition, many individuals (around 40 – 50 %) had migrated into the deposited layer (a vertical distance of up to 2 cm). Whereas following 5 cm of deposition, around 80 % of the two smaller cirratulid species and 50 % of the larger *Cirriformia* had migrated

into the new layer. In contrast, after 10 cm of smothering the polychaetes appeared to be more randomly scattered throughout the sediment with 60 % of *Caulleriella*, nearly 40 % of *Tharyx* and approximately 35 % of *Cirriformia* remaining in the original sediment, and more than 6 cm from the new surface. Given that both in previous field samples and the controls, the maximum depth of fauna for this site was found to occur at around 5 cm, individuals remaining below this depth may be prevented from feeding at the surface and from respiration. The implications of these results are that these cirratulids may be tolerant to up to 5 cm of smothering, but that they would be seriously impacted at levels above 10 cm.

Some results were also obtained from experiment 5 for the following species; *Hydrobia ulvae, Pygospio elegans, Tubificoides pseudogaster & Tubificoides benedi, Nephtys hombergi* and *Polydora ligni*. These species were all previously recorded from the field site at Hythe either periodically or sporadically at low densities (see Chapter 2 for field data). It was not possible to carry out individual statistical analysis of the results for these species from this experiment, as they were present in the cores in low numbers. However, the depth distribution data indicate that *Pygospio* and *Hydrobia* had migrated into the deposited layer following smothering with 2 cm. Following deposition of 5 cm, *Pygospio* and *Tubificoides pseudogaster* had moved into the deposited sediment, although all individuals *of Hydrobia* remained buried in the base sediment. After burial under 10 cm of sediment, *Nephtys, Tubificoides benedi* and *Tubificoides pseudogaster* were found to have migrated a depth of 6 - 10 cm into the deposited layer, but all the *Hydrobia*, and some *Tubificoides benedi* remained buried at a depth greater than 10 cm. The numbers of individuals of these taxa were too small for these results to be statistically viable or conclusive. However, they do indicate that *Nephtys*, and the oligochaetes (*Tubificoides* spp.) displayed the greatest response of migration, and that *Hydrobia* and the spionids were able to migrate following the lower levels of deposition, but were inhibited by a depth of 10 cm.

These results appear to agree with the results from the single species experiments as *Nephtys* is a highly motile species, most closely resembling *Nereis* in function, whereas the spionids are smaller and more sedentary. *Hydrobia* is a gastropod that has adapted to living in muddy sediments but it is usually restricted to the surface. Other epifaunal species have been found to be highly sensitive to smothering in previous

studies (Stanley, 1970). It was not possible to identify any 'community effects' either facilitative (positive) or inhibitory (negative) on migration, as there were not enough *Cerastoderma* and no *Nereis* present in the cores from experiment 5. The only examples of *Cerastoderma* in experiment 5 were retrieved from the underlying base sediment in the cores smothered with 10 cm of material, providing further evidence that this species is intolerant to being buried under this depth of sediment and does not possess the ability to migrate to the surface under such conditions.

4.4.2 Survival and mortality following smothering of infauna

Mortality would be expected to occur as a result of burial if conditions in the immediate vicinity of an organism became unsuitable for maintaining respiration for a prolonged period, and if avoidance through migration or the construction of a new burrow was not possible. The most likely environmental impact of smothering is oxygen depletion as discussed previously.

Infauna which inhabit fine sediments in estuaries are well adapted to life in environments which are typically low in oxygen (hypoxic) (Diaz & Rosenberg, 1995; Sobral & Widdows, 1997). The many species of polychaete and oligochaete worms, bivalves and crustaceans which make up the infauna, have evolved strategies such as burrow irrigation (where oxygenated water is drawn through the burrow by the animal), to extend the oxygenated zone several centimetres into the sediment and therefore sustain aerobic respiration in sediments which would otherwise be anoxic (Aller,1982). In an environment that is highly dynamic and regularly exposed to disturbance by predators and from the actions of waves and currents (Brenchley, 1981; Hall *et al.*, 1994; Maurer *et al.*, 1981, 1986; Probert, 1984) individuals must regularly repair, re-open, and reconstruct their burrows to maintain supplies of food and oxygen. Previous studies have shown that some species have evolved an alternative mechanism to tolerate toxic conditions. The closing of the valves and reducing metabolic rates is commonly found in bivalves, which is less costly in terms of energy than escape by extensive burrowing and migration (Sobral & Widdows, 1997). However, where conditions prohibitive to survival are sustained, or where escape by migration is not possible, due to the extent of burial and limited motility, mortality will occur. Mass mortalities of infauna following exposure to hypoxia and anoxia have been well-documented worldwide, and no macrofauna occur in persistent anoxic environments

(Diaz & Rosenberg, 1995). Large-scale burial, such as could potentially occur during dredge spoil disposal, may therefore cause mass mortalities, with large areas of sediment being defaunated.

4.4.2.1 Control treatments

The distribution of the fauna in the control cores from experiments 1 to 4 show that the animals, which were placed on the sediment surface at the setting up stage, had all burrowed into the sediment and resumed typical buried positions in burrows. This observation suggests that the individuals had resumed something resembling 'normal' behaviour, and was supported by visual observation of the bivalves with their siphons extended into the overlying water column in feeding position. The significantly higher levels of live fauna retrieved from control cores compared to those from smothered cores supports this conclusion.

Some mortality is inevitable when using live marine or estuarine organisms in laboratory studies due to stress imposed on them during sampling and transport, particularly when individuals are extracted from the sediment. In the case of the cockles (*Cerastoderma*) from experiments 1 and 4, mortalities of 95 % and 96 % in treated animals compared to 24 % and 4 % in control animals clearly indicate that mortality was induced by the experimental treatment or smothering. The trend was less distinct for the clams (*Tapes*), although it is still evident, as more than twice as many smothered clams were recovered dead than control animals (13 % versus 6 %). No mortality was recorded for the ragworms (*Nereis*), which proved to be highly tolerant of both laboratory conditions and experimental treatment.

4.4.2.2 Mortality in cockles following smothering

Cockles (*Cerastoderma*) were found to be the least tolerant species to burial. The migration results suggest that this intolerance is due to a lack of ability to migrate rapidly upwards through thick layers of sediment, and therefore be unable to escape the unfavourable conditions imposed during burial. A proportion of the cockles displayed some ability to reposition themselves if the deposited layer was less than 10 cm, and preferably less than 5 cm thick, but were severely inhibited if the sediment load was increased beyond this limit.

4.4.2.3 Mortality in manila clams (*Tapes philippinarum*) following smothering

The results suggest that the clams were more tolerant of smothering than the cockles. However, the fact that a greater than 100 % increase in mortality occurred in smothered animals compared to controls, indicates that the clams' tolerance to smothering was exceeded. In terms of burial depth, the highest mortality rates were recorded from the cores smothered with the maximum burial depth tested (50 cm), indicating that impacts on the clams were more severe (indicated by increased mortality rates) as sediment load increased. However, analysis of the data on migration from this experiment suggests that the ability of *Tapes* to escape the effects of smothering was similar to that of *Cerastoderma*. Also, taking into account that the clams were only maintained in the experimental system for a few days following smothering (for the reasons discussed earlier), it seems likely that they survived by employing the mechanism of partial metabolic shutdown. A closely related species of clam, *Ruditapes decussatus*, was found to reduce its respiration rate by up to 65 % as a response to severe hypoxia (Sobral & Widdows, 1997).

It is not possible to determine whether these individuals would have remained alive in the long term. However, as their recorded/observed position in the sediment was well beyond reach of oxygen and food supplies, it is extremely unlikely that they would have survived indefinitely following burial. As these animals appeared unable to reach the surface following smothering, if it had been feasible to maintain the experiment for a longer period (several weeks to months), the mortality recorded for this species would have been greatly increased. Further studies in microcosms better equipped for keeping fauna long-term would be needed to prove this hypothesis with certainty, however the results obtained from this study strongly suggest that *Tapes* would not survive long-term burial of more than ten centimetres and possibly as little as five centimetres.

In terms of recharge, it is thought that this species may be tolerant of several successive shallow burial events as long as adequate time was allowed between each for repositioning and reconditioning. However, it is predicted that a population would suffer significant loss of individuals if a single, permanent deep layer were deposited.

4.4.2.4 Mortality in *Nereis diversicolor* as a result of smothering

The ragworm, *Nereis* proved to be highly tolerant to smothering with 100 % survival recorded for this species. As discussed previously, it is thought that this tolerance is due to the high motility of this species, as worms showed no inhibitory effects in coping with half a metre of sediment deposition. Similar results were obtained in a previous study for *Nereis succinea* (Maurer *et al.*, 1986). It was found that *Nereis succinea* was the most tolerant of the species tested to burial with fine sediment, and was suggested by the authors that this species may be able to survive burial with up to 90 cm in its native (fine) sediment.

4.4.3 Probit analysis

Probit analysis was applied to the data from experiments 1 - 4. This is a quantal response assay, commonly used to investigate a wide range of environmental as well as medical effects (Sokal & Rohlf, 1995; Wardlaw, 2000). For these experiments it was used to identify statistically significant variation in response between the different species included. It also enabled predictions of smothering depth as an estimated LD50 for each species within 95 % confidence limits (see Table 4.10). The LD50 predictions, in agreement with the rest of the results and analysis, imply that *Cerastoderma* is less tolerant to smothering than *Tapes*. LD50s of 2.46 cm (experiment 1) and 2.49 cm (experiment 2) were estimated for the cockles, compared to a value of 868 m estimated by the statistic for the clams! These results clearly highlight the increased sensitivity of *Cerastoderma* to smothering. The extremely large depth predicted for *Tapes* is a result of the short term low mortality recorded for this species following smothering, and is interpreted as indicating greater tolerance to smothering, rather than as a definitive burial depth (it can be seen from Table 4.10 that the standard error associated with this prediction is considerably higher). However, it is interesting to note how close the two values produced by the statistical analysis for *Cerastoderma* are (2.46 and 2.49 cm) despite being obtained from different experimental runs and different populations. LD10 and LD90 values are also estimated by the Probit analysis. However, it can be seen from the table that the error limits for these estimates are much greater than for the LD50 and must therefore be interpreted with caution.

4.4.4 Estimated survival (discussion of results from experiment 5)

Estimated survival of an individual from experiment 5 (the mixed species experiment) was calculated from the faunal depth distribution (taken from field samples collected from Hythe between June 2000 and May 2002), and the animal's recorded vertical position in the sediment at the end of the experiment. For example, if a worm was found in the top 2 cm of the smothering material in a core on which was deposited 5 cm of material, it had vertically migrated at least 3 cm. As it had reached the top 2 cm of the core this was deemed to be an indication of potential survival, as at this depth access to food resources and oxygen would become available. The weakness of this reasoning is that it assumes that all macrofauna would have access to the sediment surface (and therefore could theoretically carry out deposit feeding, respiration and burrow irrigation uninhibited) if they were within 2 cm of it. This may not always be the case for smaller individuals (e.g. *Caulleriella* and *Tharyx*), and so may lead to an overestimation of 'success' for these species. However, this may be balanced to some extent by larger individuals such as *Cirriformia*, which would be expected to construct slightly deeper burrows. As these polychaetes could not be extracted from their burrows alive, no direct means of assessing mortality was possible and so estimating survival from the distribution data was the only option.

The results suggest that the cirratulid polychaetes *Caulleriella*, *Tharyx*, and *Cirriformia* were sensitive to the level of smothering, as estimated mortality was increased with increasing depth of deposition. The data indicate that these species were relatively tolerant of up to 5 cm of sediment, but would be seriously impacted by 10 cm (estimated mortality 35 – 60 %). These estimates should be interpreted with care as mortality was not directly measured and for this reason no statistical analysis is presented. However, they are still thought to be useful in predicting the effects of mass deposition on the populations. It is expected that accuracy is greatest for *Caulleriella* which has the highest number of observations of individuals ($n = 378$). Reasonably large sample sizes were also obtained for *Tharyx* ($n = 93$) and *Cirriformia* ($n = 27$). Observations on the other species present in the cores in experiment 5 could be made on just a few individuals. The data obtained suggests that *Hydrobia* ($n = 12$) was inhibited by just 2 cm of deposition, and that the spionid *Pygospio* ($n = 9$) and oligochaete *Tubificoides pseudogaster* ($n = 6$) were more tolerant. These observations on *Hydrobia* agree with the results of a study carried out by Chandrasekara and Frid

(1998) who found that the snails were unable to regain the surface or survive after burial under 5 cm of sediment with a high silt-low water content, probably due to oxygen stress. Stanley (1970), suggested from his investigations into relationships between morphology and burrowing ability/life habits that epifauna are generally far more susceptible to negative effects of burial than either shallow-dwelling infauna or deeper burrowers.

Future investigation using a direct measurement of mortality is required to test these predictions. It is possible that, in the future, a technique enabling individual organisms (and their response to smothering) to be filmed *in situ* could be employed for this purpose. One such possibility is the real-time video technique of Sediment Profile Imagery (Keegan *et al.*, 2001) if technological advances meant that the necessary spatial resolution could be obtained. In terms of the implications for future recharge operations, it seems possible that the tolerance of these species to smothering may be similar to that of the bivalves discussed above. The results obtained from these experiments suggest that smothering with a depth of more than five centimetres would have severe impacts on the macrofauna community and result in mass mortalities

4.4.5 Environmental data

Environmental data were collected during these experiments: a) To ensure that environmental conditions (particularly water quality) within the experimental system were maintained at a level which did not pose a risk to experimental organisms (i.e. to significantly stress or enhance mortality); b) to identify any experimental or tank effects, which may have confounded results; and c) to identify any distinct trends in these parameters which may have occurred as a response to the experimental treatment (i.e. were independent of controls) and potentially relate to the biological results obtained. It was beyond the scope of this study to examine and monitor sediment geochemistry in full detail and provide data on parameters such as porewater ammonia or sulfide content etc.

The results of the water and sediment data and the corresponding statistical analyses are discussed below in relation to conditions within the system for sustaining the macrofaunal species studied during these experiments, the biological results of the experiments, and implications for future experiments and dredge disposal activities.

4.4.5.1 Salinity

The use of artificial seawater resulted in relatively good control of water salinity (which was adjusted to 30 psu prior to starting the experiments). Salinity values remained within ±4 of 30 psu throughout the experiments, which is well within the range tolerated by the estuarine species being held in the tanks. Slight increases in salinity during experimental runs may be attributed to evaporation. Decreases may have been caused by the addition of the sediments to the system (which were of lower salinity than the overlying water column despite the smothering sediment being mixed with seawater once thawed to replace water lost due to freezing).

The results of the Friedman's tests on the water salinity data indicate that there was no significant difference in salinity between the four tanks - i.e. that there were no 'tank effects' due to salinity variations which may have affected the results of the experiments (the system was recirculatory). The statistics also confirm that there were no significant trends or gradients in salinity during the experiments, suggesting that the fauna did not experience significant stress or enhanced mortality as a result of fluctuations in water salinity.

The results of the Kruskal Wallis tests indicted that there was no significant variation in sediment salinity either between the base sediment and the smothering material, or between the five experiments. This implies that fluctuations in salinity of the sediments within the experimental system can be attributed to small scale local variations within the sediment and sampling and/or instrumentation induced error. The statistically insignificant fluctuation observed in sediment salinity within individual experiments may be explained by the periodic drainage and inundation of the sediments within the system.

It is generally believed that infauna (in estuaries) experience less salinity variation than epifauna because of the buffering effects conferred by the interstitial water which is less variable than the overlying water mass (Boaden & Seed, 1985). In laboratory experiments it is generally easier to maintain relatively constant salinity in the water. The greater variation in salinity recorded in the sediment during these experiments may be due to instrumentation error (i.e. inaccuracy of the probe used) or to sampling

error as implied by the statistics. The sediment salinity data recorded *in situ* in the field (chapter 2) displayed a similar range of values. Another possibility is that the processing of the sediments (sieving, mixing, freezing and thawing) reduced the natural buffering capacity of the sediments. Either way, as the species used in this study were sampled from an estuarine environment and are known to be tolerant of fluctuations in salinity (e.g. *Cerastoderma spp.),* or euryhaline (*Nereis diversicolor, Hydrobia ulvae)* (Boaden & Seed, 1985, McLusky, 1981), it is concluded that observed salinity fluctuation would not have affected the mortality rates.

4.4.5.2 Temperature

These experiments were conducted in an unheated laboratory during the winter. Monitoring indicated that fluctuations in water temperature were restricted to a couple of degrees over a week, and followed ambient temperatures as expected. The statistical analysis of the temperature data (Friedman's tests) confirms that any differences in temperature between the tanks and during the experimental runs were not significant i.e. were slight enough to be attributed to sampling variation.

Sediment temperature was more variable than water temperature despite measures taken to equilibrate the temperature of both the base and the smothering material prior to use. Despite transferring the smothering material to the laboratory several hours prior to use following thawing, it was slightly colder (by 1 – 2 ^{0}C) than the cores containing the animals for experiments 1, 3 and 4. For experiments 2 and 5, the reverse was true and the sediment used for smothering was 2 - 3 ^{o}C warmer than the underlying core material. This is likely to reflect the fact that the temperature of the core sediment was taken at the beginning of the experiment when the core had been constructed using sediment recently taken from the field, which would have warmed up slightly during the acclimatization period in the laboratory. It is unlikely that this variation was present throughout the duration of the experiment (cf. water temperature data). The statistical tests also reflect this variation between the base and smothering sediment, but do not take the timescale into account. Statistically significant variation in temperature between the five experiments was also identified, reflecting seasonal (i.e. natural) changes in ambient temperature rather than short-term (laboratory-induced variation).

Therefore, no adverse effects of temperature on mortality data would be expected. It would not be recommended to undertake experiments such as these in the summer as opposed to winter if the temperature could not be controlled, as it is likely that higher temperatures would cause significant effects on experimental results. In a previous study investing sedimentation on infauna with temperature controlled experiments, evidence was found to suggest that mortality following burial was significantly increased in the summer compared to winter (Chandrasekara and Frid, 1998). There is also some evidence that migration in some species is affected by temperature (Maurer *et al.*, 1986). However, significant effects on biological results by temperature fluctuation as a laboratory effect *per se* are considered unlikely here. Estuarine fauna from temperate regions are frequently subjected to a wide range of temperatures and are well adapted to accommodate this (Boaden & Seed, 1985).

4.4.5.3 pH

The pH of the water in the system remained around neutral to slightly alkaline as would be expected for natural seawater. No extreme fluctuations were observed during the experiments. Friedman's tests showed no significant variation in water pH during the experiments, either between tanks or during individual experimental runs.

The pH of the sediment used in the experiments was found to be slightly lower than that of the overlying water. However, values did not show significant departure from those obtained in the field (see Chapter 2) and expected for cohesive sediments that are characterised by reducing conditions arising from anaerobic bacterial activity (e.g. Barnes & Hughes, 1988). The lowest values recorded were from the smothering sediment, perhaps due to increased bacterial activity following thawing. However, they were within the range recorded in the field (with the same instrument). It is likely that some sampling error was introduced by the electronic probe (details given in Chapter 2). It is also likely that localised fluctuations of pH occurred during these experiments due to the die-off of organisms (see mortality data). This is indicated by the statistics which imply that the greatest variation occurs between experiments 1 and 2 which also corresponds to the greatest variation in mortality. Given the nature of these experiments, this is probably an effect of the experimental treatment rather than being causal, as pH was found to be comparably variable during the fieldwork (see results section of chapter 2).

4.4.5.4 Dissolved oxygen content

Monitoring indicated that the dissolved oxygen concentration of the water within the tanks remained relatively constant and close to saturation throughout the experiments. Variability in the data was within the limits of sampling accuracy. Therefore the mortality of experimental organisms cannot be attributed to anoxic or hypoxic conditions arising from inadequate oxygenation of the water in the system.
Levels of dissolved oxygen in the sediment were generally within ranges observed in the field, and were of an order of magnitude lower than values for the overlying water. This is to be expected for cohesive sediments in which permeability of oxygen is low and the demand for oxygen by bacterial metabolism outstrips supply (Barnes & Hughes, 1988; Fenchel, 1996). The smothering sediment appeared to have a slightly lower dissolved oxygen content than the base sediment. Perhaps due to a lack of burrow irrigation, and potential increased bacterial metabolism following the die-off of micro and meiofauna (too small to be removed by sieving) killed during freezing. A similar increase in BOD may also be expected to occur in dredge spoil due to die-off of indigenous fauna. Statistical analysis of the data (Kruskal Wallis tests), however, indicated that the difference in oxygen content between smothering material and base sediment was not significant.

Variation in sediment oxygen content between experiments, however, was found to be statistically significant, and this is likely to reflect differences in the biological activity and thus BOD of the sediment. The smothered fauna was probably subjected to significant decreases in oxygen concentration due to experimental treatment (i.e. smothering), but not due to independent laboratory incubation effects. Oxygen concentration of the water in the holding tanks and control cores were maintained at a level adequate to sustain benthic infauna of this type. Previous studies on environments where the overlying water body was subjected to hypoxia (<2 ml l^{-1} dissolved oxygen) showed that *Nereis diversicolor* was relatively tolerant of hypoxia and that *Cerastoderma* spp. displayed similar resistance to low oxygen concentrations (Diaz & Rosenberg, 1995). It was not possible to gain a time series of oxygen data from within the sediments of the smothered cores over the duration of the experiments as this would have caused significant disturbance. However, in future experiments, time and resources permitting, this could be achieved through the use of microprobes and would be valuable in gaining more detailed information in monitoring the effects

of smothering upon localised sediment properties affecting the fauna. Continuous readings by reliable oxygen electrodes would also be preferable to measurements taken at intervals, as periodic and aperiodic oscillations in oxygen concentrations have been found to be the norm in many habitats (Diaz & Rosenberg, 1995). Spatial patchiness in oxygen concentration is also prevalent in shallow-water sediments, where the upper 2 - 10 cm of sediment has been found to exist as a mosaic of oxic and anoxic microhabitats on the scale of millimetres due to the presence of small (0.5 - 2 mm diameter) worm burrows (Fenchel, 1996).

4.4.6 Implications of experimental results for intertidal recharge

The results obtained from these experiments imply that cockle populations would be seriously impacted by recharge events, and are likely to be wiped out should the layer of sediment deposited exceed 5 - 10 cm. The high sensitivity of cockles to smothering matches the results obtained by Chang & Levings, (1978) where another Cardidae bivalve (the heart cockle, *Clinocardium nuttallii*) was found to be seriously impacted by burial under 10 cm of sediment. In the event of defaunation through recharge, subsequent recovery of the species in the area would therefore rely upon colonisation by juveniles following reproduction by animals in surrounding areas. It is unlikely that adult migration from adjoining areas would be significant, as adults are relatively sedentary. In the case of the Hythe population of *Cerastoderma*, it is likely that were a large area (100's of square metres) of the mudflat to be subjected to deposition of more than a few centimetres through recharge, recovery would either be extremely slow or absent. This is hypothesised as, several years of sampling on the lower mudflat at Hythe has indicated little evidence of reproduction in the cockle population (personal observation).

On the basis of this study, it is unlikely that recharge would cause long-term impacts on populations of *Nereis*, unless the sediment dumped was significantly coarser than the indigenous sediment or contained toxins. In the case of small patches or weakened individuals being wiped out by deposition, recovery is likely to be more rapid than for the bivalves due to the ability of surrounding adults to migrate (horizontally) into the area vacated

It appears that, as with the bivalves investigated here, recharge could only be carried out without severe deleterious effects on the cirratulids *Caulleriella*, *Tharyx* and *Cirriformia*, if the layer of deposition was 5 cm or less. Again it is important to note that additional adverse environmental factors (e.g. pollutants, changes in bulk density) would be expected to decrease faunal tolerance to smothering (Maurer *et al.*, 1985; Turk & Risk, 1981).

Observations on *Hydrobia* and the other species included in experiment 5 were too few to be conclusive. However, together with the results of other studies, it appears that *Hydrobia* is very sensitive to deposition.

Previous studies have indicated that smothering with non-native sediment of distinctly different types also directly affects burrowing ability and mortality of infauna following burial. In particular it was found that the burial of organisms from muddy environments under significantly coarser material (i.e. sand) or vice versa inhibited migration ability (Maurer *et al.*, 1985; Nichols *et al.*, 1978; Turk & Risk, 1981). For example, in a study of the effects of tidal barriers, Turk and Risk (1981) found that the LD50 for the bivalve *Mya arenaria* (which inhabits sandy sediments) was 24 cm for coarse sand, 6 cm for fine sand and 3 cm for mud. During these experiments the effects of different sediment types on migration ability were not investigated, and the density of the smothering material was deliberately controlled in order that it did not significantly depart from that of the base sediment (approximately 1200 kgm^{-3}). This means that bulk density was unlikely to have affected the migration ability of individuals.

Long-term effects and recovery rates of the infaunal community following recharge cannot be predicted with confidence for this site (Hythe) without further investigation (see also Chapter 1 for review, and Chapter 7 for further discussion). The results of previous studies (Arntz & Rumohr, 1982; Drucker, 1995; Essink & Beukema, 1986; Frid, 1989; Gunther, 1992; Hall, 1994; Johnston, 1981; Levin, 1984; Levings *et al.*, 1985; Newell *et al.*, 1998; Pearson & Rosenberg, 1978; SOAEFD, 1996; Valdes-Cogliano *et al.*, 1988; Vale *et al.*, 1989; Van Dolah *et al.*, 1979; Warwick & Clarke, 1993; Wildish & Thomas, 1985; Zajac & Whitlatch, 1982 a&b) showed that recovery depends upon several factors: The ability of existing fauna to survive burial and

recolonise through vertical migration, the availability of migratory adults in adjoining areas (for motile species), and the availability of juveniles through recruitment in adjacent areas. Thus seasonality (i.e. the time of recharge) and the size of the area affected will also play a role in determining the rate of recovery.

It is believed that the results obtained from these experiments allow for some prediction of effects on the species tested in the event of intertidal recharge at Hythe (Southampton Water). Accurate prediction would rely upon the following assumptions: i) The sediment deposited is of a similar type (grain size) to that at the site of disposal; ii) disposal sediment is not chemically contaminated; and iii) temperature ranges are similar to those recorded during these experiments. However, if in the event of future recharge, there were major differences in sediment type between the disposal material and the indigenous sediment at the disposal site, this may be expected to significantly affect the response of the infauna.

4.5 Summary and Conclusions

A series of controlled laboratory experiments designed to investigate the effects of smothering with native sediments were carried out in the experimental system described in Chapter 3. Four experiments were carried out on individual species (*Cerastoderma edule*, *Cerastoderma glaucum*, *Nereis diversicolor*, and *Tapes philippinarum*) and a fifth experiment was undertaken on cores containing natural, mixed species assemblages from Hythe. Animals in experimental cores were buried under 2 - 50 cm of sediment, whilst those in control cores were left unsmothered. Response to smothering was analysed in terms of survival versus mortality, and observed vertical migration. The effects of burial depth were analysed, and tolerance thresholds were calculated for each species. Major conclusions from these experiments are listed below:

Cockles (*Cerastoderma* spp.) appear to be highly sensitive to smothering, and mass mortality would be expected if this species were buried under more than 2 cm of sediment.

The manila clam (*Tapes philippinarum*) is reasonably tolerant to short-term i.e. temporary burial, but long-term survival of these animals would be unlikely if burial were extended for prolonged periods (i.e. if deposited material persisted in the long-term as would occur during successful recharge).

Nereis diversicolor appears to be highly tolerant of burial and no negative effects were observed when individuals of this species were buried under half a metre of mud.

The cirratulid polychaetes *Caulleriella caputesocis*, *Tharyx marioni* and *Cirriformia tentaculata* displayed some tolerance to shallow burial. However, results suggest that populations may incur severe losses at a burial depth of just 5 to 10 cm.

The mud snail, *Hydrobia ulvae*, appears to be extremely sensitive to deposition and significant deleterious effects on populations may occur with as little as 2 cm of sedimentation.

Long-term survival of all species following burial depends on this ability to reach the upper layer through vertical migration. Migratory ability is species dependent, and is closely related to individual morphology and life habits.

Highly motile, active burrowers such as nereid and nephtyd polychaetes are far more likely to survive burial than are relatively less mobile, weaker burrowers such as near-surface living suspension feeding bivalves or small tubiculous species. Epifauna are likely to be the most sensitive to burial with severe consequences occurring with just one or two cm of deposition.

Intertidal recharge at Hythe is likely to deplete significantly the macrofauna community. A resulting layer more than ten cm thick may be expected to essentially defaunate (with the exception of *Nereis*) the area affected.

The impacts on the macrofauna could be reduced if recharge was limited to no more than 2 cm in thickness at one time, and intermittent periods for recovery and repositioning of fauna were allowed. The species tested during these experiments, except cockles, would be expected to survive burial under a layer 5 cm thick (although numbers would be seriously depleted).

5 The influence of Hythe macrofauna on sediment erodibility; laboratory experiments using EROMES and the CSM

5.1 Introduction

The primary aim of these experiments was to investigate and identify potential relationships between macrofauna density and bed stability at site 1, Hythe. Investigations into the stability of cohesive sediments of intertidal mudflats have been the subject of much attention due to the economic and environmental impact of sediment transport and erosion in ports, estuaries and coastal areas. As a result of this interest, an array of instrumentation is available for investigations of the erosion properties of intertidal sediments.

The primary factors controlling the stability of abiotic cohesive sediments are water content and bulk density (Amos *et al.*, 1998). However, many authors have reported that the presence of biota has a significant influence on stabilisation and destabilisation processes (e.g. Grant & Daborn, 1994; Meadows *et al.*, 1989; Paterson, 1989; Widdows *et al.*, 1998). Biostabilisation of the bed by microphytobenthos such as diatom biofilms has been found to play a major role seasonally (Paterson, 1989; Paterson, 1994), whereas macrofauna mediated processes such as bioturbation and faecal pellet production (biodeposition) can have a destabilising effect (Austen *et al.*, 1999). Dense populations of tube building fauna have also been reported to stabilise the sediment through the cementation of tube and burrow structures (Meadows & Tait, 1989; Meadows *et al.*, 1990). In addition, physical factors such as grain size, water content, bulk density, organic matter content, aerial exposure time, elevation, temperature and salinity are all believed to contribute to the erosive properties of intertidal cohesive sediments (Amos *et al.*, 1988; Black, 1989; Paterson *et al.*, 1990). Many previous studies have focused on investigating the influence of one or more of these factors along a shore-normal and/or shore-parallel direction. The focus is on a localised region of the mudflat, around low water, with attention to small scale variability in factors such as elevation, aerial exposure, grain size, biological community structure, water content and bulk density. The aim of these experiments therefore was to identify the effects of individuals or patches of individuals of the infauna on the stability of the surrounding sediment. At Hythe the infauna community

is dominated by numerous small cirratulid polychaetes (see chapter 2) about which virtually nothing is known. A continuation of this study is included in the next chapter, where a CT scanner was used to examine burrow structures and fine-scale bulk density patterns. More generally, the results of these experiments have wider implications for recharge at this site, and for tidal flat stability.

Two different instruments were used to collect erosion data; the EROMES system (Schunemann & Kuhl, 1991) and the Cohesive Strength Meter (CSM) (Paterson *et al.*, 1989). The EROMES used in this study is a new portable version, recently built by Sediment Dynamics Research (SDR, University of Southampton), based on the original design produced by Schunemann and Kuhl (1991) (see Plate 5.1). Modifications were carried out to facilitate *in situ* investigation into the erosion of cohesive sediment in the field. The instrument consists of an erosion chamber containing a motor driven propeller for producing controllable horizontal bed shear stress (eroding pressure), and an Optical Backscatter Sensor (OBS) for measuring water turbidity (arising from the eroded sediment). The instrument is connected to a portable lap-top computer which houses the software used to control the motor and define appropriate programming for erosion experiments. Full descriptions of the SDR version of EROMES and software, and the original version are given in Quaresma *et al.*, (2002) and Schunemann and Kuhl (1991) respectively.

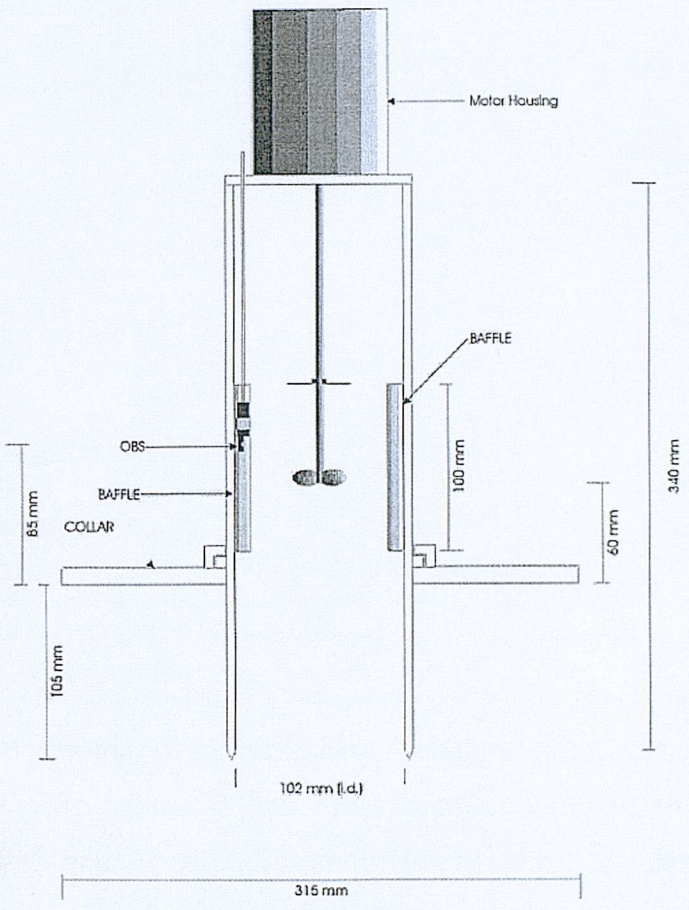


Plate 5.1 Diagram of EROMES (Quaresma *et al.*, 2003)

The CSM is a portable instrument designed for rapid measurements of sediment erodibility in the field (plate 5.2). It consists of a small (6.6 cm^2) cylindrical erosion chamber containing a vertical jet which fires water to erode the sediment, together with an infra red transmitter and receiver diodes to measure water turbidity (as a percentage of optimal light transmission). The chamber is connected to a waterproof box containing a computer and battery, a liquid crystal display (LCD), pressurised air canister and pressure gauge, and a refillable water bottle. During an erosion experiment, jet pressure is increased incrementally (depending on the program selected), whilst light transmission is logged by the computer. An empirical calibration based on the equations of Bagnold (1966), modified by McCave (1971), is used to transform eroding pressure (i.e. the pressure of the water jet) into an equivalent horizontal bed shear stress (Paterson, 1989; Tolhurst, 1999).

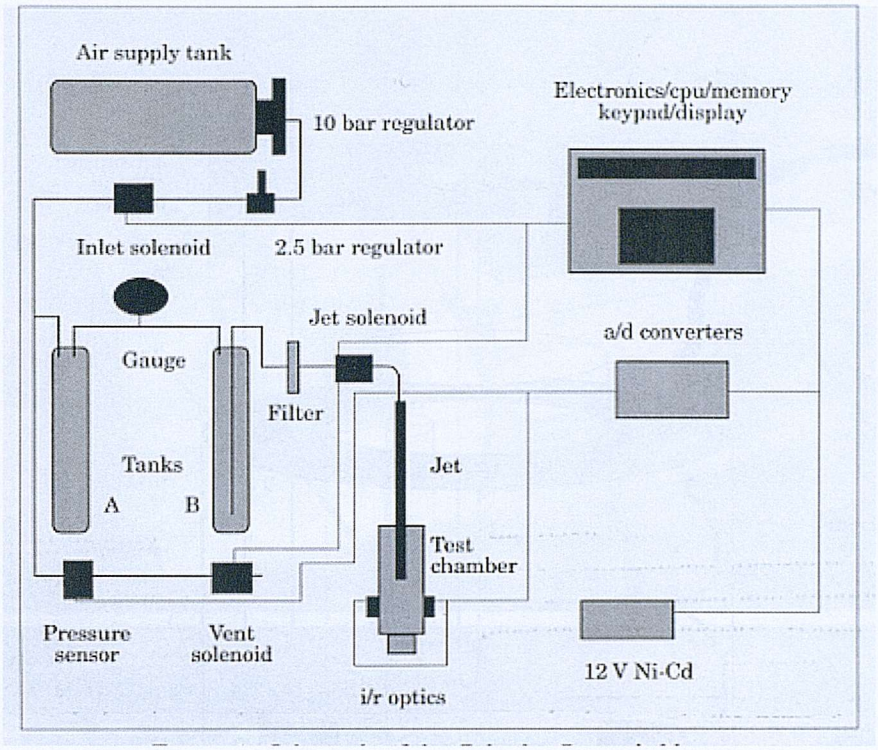


Plate 5.2 Schematic diagram of the CSM after Tolhurst *et al.*, (1999)

It was not possible to collect EROMES data in the field due to the extremely restricted access to the field site, and the length of time required to run individual erosion experiments using EROMES (approximately 1 hour each). Additionally, previous attempts to use the CSM at the site had proved unsuccessful due to the water-logged and extremely soft nature of the sediment surface during the short exposure periods of the tidal cycle (the instrument can only be used if the sediment surface is exposed). Therefore it was decided that the only way of carrying out the experiments was on relatively undisturbed cores transported back to the laboratory.

5.2 Methods

5.2.1 Core collection

At spring tide low water, thirty-six 6.6 cm diameter push cores and twelve 10 cm diameter push cores were collected from site 1, on the lower intertidal zone on the mudflat at Hythe. The bases of the cores were capped before being transported to the laboratory, taking care to minimise disturbance during sampling and transport.

5.2.2 Experimental set up

In the laboratory the cores were labelled with a reference number before being set up into four large holding tanks. Half of the smaller cores were placed into the first tank for use with the CSM, half of the larger cores were placed into the second tank for use with EROMES, the remaining cores were then divided between tanks 3 and 4 in the same way.

The tops of the cores were first covered with bubble wrap to prevent disturbance of the sediment surfaces, and seawater (28 psu) was added to the first two tanks at a slow trickle. Once the tanks had filled above the level of the core surface the bubble wrap was floated off without coming into contact with the sediment surface. The final water level was approximately 10 cm above the sediment surface.

Seawater containing 4 % formaldehyde (final concentration) was added to tanks 3 and 4 to kill the macrofauna in half the cores, using the same method (the formaldehyde solution (formalin) was used as a biocide in order to compare treated and untreated cores to identify the impacts of the fauna on sediment stability and erodibility). These two tanks were then covered and sealed with heavy-duty polythene to prevent the escape of toxic fumes from the formalin.

An air supply was provided, via perforated airlines fixed around the perimeter of the tanks, to ensure that the water did not become oxygen-depleted, and the tanks were insulated against temperature changes with thick layers of fibreglass. The laboratory was well ventilated and the windows blacked out to prevent solar heating. The water temperature in the tanks was recorded periodically.

5.2.3 Erosion experiments

Erosion experiments were carried out on days 1, 2 and 5. Each of the 6.6 cm diameter cores were subjected to erosion tests using the CSM (Cohesive Strength Meter), and the larger 10 cm cores were subjected to erosion experiments using EROMES.

5.2.3.1 EROMES experiments

The water temperature and salinity in the tanks was recorded. The water covering the surface of the cores was clear with no visible signs of turbidity, so it was decided that siphoning out all the water to the sediment surface and refilling once the instrument was in place would carry an increased risk of disturbance. The water level was dropped to the top of the core liners, and enough water to account for displacement by the instrument was carefully removed from above the sediment surface of the first core (E1, tank 2) using a pipette.

EROMES was carefully positioned over the first core (following pre-measurement and adjustment of the propeller shaft) ensuring that the bottoms of the propeller blades were set at a distance of exactly 4 cm above the sediment surface. The required programme (see below) was then selected from the controlling software, and the erosion experiment was initiated, noting the start-time.

The program used to run the erosion experiments with EROMES was written to incorporate three phases: I) an initial 3 minute period of still water to ensure equilibrated conditions in the chamber; II) an erosion phase in which flow was increased (by increasing the propeller rotation speed) in a series of 8 steps, each of 5 minutes duration; and III) a still water settling period lasting 15 minutes.

The above procedure was repeated with the second core (E2), and then cores E3 and E4 from tank 3. Care was taken to ensure that the tank was fully covered during the latter experiments to prevent escape of fumes from the formalin. At the end of each experimental run the instrument was carefully removed ensuring that no suspended sediment was allowed to contaminate the other cores in the tank, which was immediately resealed. The parts of the instrument which had come in contact with the water, were then carefully rinsed to remove any traces of formalin or sediment.

Erosion experiments as described above were repeated on cores E5, E6 (untreated) and E7 and E8 on day 2, and on the remaining cores (E9 – E12) on day 5. It was determined that none of the surfaces of the cores were disturbed prior to data collection, and visual observations on surface appearance were noted for each core.

30 ml water samples were collected during the first two erosion experiments for calibration of the OBS output data. Samples were taken via the sampling port on the side of the instrument using cut-off syringe samplers in which were placed pre-weighed glass fibre filters. Samples were taken 1 min 30 sec after the start of the program, then at 5 minute intervals throughout the duration of the experiment. The filters were then dried and re-weighed and the suspended sediment concentration (SSC in mgl^{-1}) was calculated using the formula:

$SSC = (M_2 - M_1)/V$, where M_1 = filter mass, M_2 = mass of filter and sample, V = volume of water sample.

5.2.3.2 CSM erosion experiments

On each of the three days, the CSM was used to erode the surface sediment of the smaller cores. The chamber was carefully positioned on the sediment surface, ensuring that no sediment was suspended, before being filled with water and the appropriate program run. Starting times and initial jet pressure and transmission values were recorded on paper along with the test reference to aid in identification and processing of the data. The tests were terminated manually when the light transmission value remained consistently below 50 % following firing of the jet.

Several of the cores had to be discarded as the surface sediment was disturbed during 'deployment' of the instrument (as is commonly experienced with this instrument when working with very soft sediment). Over the three days, data sets were collected from 26 cores in total; 13 from untreated cores, and 13 from formalin treated cores. Once again, great care was taken to ensure that contamination of samples and transfer of formalin solution was avoided.

5.2.4 Core processing

At the end of each experiment the cores were carefully removed from the system for enumeration of fauna. Untreated cores were suspended in and covered with 10 % formalin to prevent deterioration in the interim. Each of the 38 cores was sieved in seawater through a 500 μm mesh. The sieve contents were backwashed into labelled jars and covered with 4 % formalin for storage until the macrofauna were identified and counted using a dissection microscope.

5.2.5 Data processing

The data sets were downloaded from the lap-top running the EROMES software, and from the CSM data storage facility for processing. Details of methods and calibrations used are given below:

5.2.5.1 CSM data

Eroding pressure (i.e. jet pressure) was converted to equivalent horizontal shear stress using the following equation (after Tolhurst *et al.*, 1999):

$\tau_o = 67((1-\exp(-x/310))-195(1-\exp(-x/1623))$ Pa, where x is the eroding pressure and τ_o is equivalent bed shear stress (Pa). The constants are empirically derived values obtained from laboratory experiments.

Critical erosion thresholds (τ_c) were identified for each data set being defined as the first pressure step at which light transmission decreased below 90 % (after Tolhurst, 1999).

Erosion profiles were plotted for each data set. Linear regression was carried out of % light transmission on bed shear stress (Pa). τ_c was located on each plot, and the regions of erosion and no erosion were identified and noted. The gradient is related to erosion rate (a steeper gradient denotes a higher erosion rate as indicated by a greater increase in suspended sediment concentration). The goodness of fit of the regression is given by the R^2 value (correlation coefficient).

The obtained critical erosion threshold values (τ_c) for each core (C1 – C26) were summarized in Table 5.1.

5.2.5.2 EROMES data

OBS output data (V) were calibrated to suspended sediment concentration, SSC (mgl^{-1}) by linear regression, using the (dry weight) sediment mass data obtained from the filter samples from the first two cores (Figure 5.27).

EROMES propeller speed (R) was calibrated to horizontal bed shear stress (Pa) using the following equation obtained during laboratory calibration (after Quaresma *et al.*, 2002):

$\tau = 3.3*10^{-3}R - 0.105$ Pa, for a propeller height of 4 cm

Calculated SSC values (mgl^{-1}) and bed shear stress values, τ (Pa), were plotted against time (seconds) to give erosion time series plots for each core (Figures 5.28-39). Periods of no erosion, erosion and still water settling were identified, and the region of erosion was classified as Type I erosion (benign or self-limiting erosion) and Type II (chronic erosion), *sensu* Amos *et al.*, (1992) (more detailed explanations of these terms are given in section 5.4.2.).

Erosion profiles were plotted (with data from the erosion phase of each experiment) as $\log_{10}$ transformed SSC on untransformed or $\log_{10}$ transformed shear stress (as appropriate to approximate a linear relationship: Figures 5.40-51) and linear regression was carried out for each core. Critical erosion threshold values (τ_c) were calculated for each core by extrapolating backwards and solving the equation (obtained through regression) using averaged, subcritical SSC values (taken from the pre-erosion phase of the data set from the experiment).

Calculated τ_c values were tabulated (Table 5.2). The obtained values for the gradient or slope of the regression equations were also included as they represent a measure of the internal friction coefficient (ϕ), a parameter used to relate to the consolidation process

within the bed, increasing ϕ indicates increasing consolidation, and therefore potential increases in bulk density and shear strength (Amos *et al.*, in press).

Erosion rates (E_m) ($mgm^{-2}s^{-1}$) were calculated from the SSC data for each 20 second period using the following equation:

$$E_m = (\Delta SSC*V) / (\Delta T*\alpha)$$

where SSC is suspended sediment concentration (mgl^{-1}), V is the volume of the erosion chamber (0.00098 m^3), T is time (in sec) and α is the sediment surface area (0.0082 m^2). Mean erosion rates (and corresponding standard deviations) were then calculated with the data from the erosion phase of each experiment.

5.2.5.3 Biological data and statistical analysis

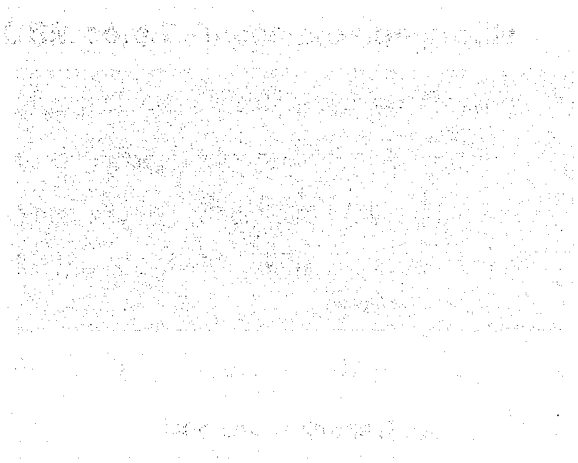
Macrofauna abundances per m^2 were calculated for total individuals, total polychaetes, bivalves, cirratulids, spionids, combined *Nereis* and *Nephtys* spp. (i.e. large, highly mobile polychaetes) and *Caulleriella* (the dominant species). Numbers for other taxonomic groups were considered too small for statistical analysis. Scatterplots, equations and R^2 values for linear regressions between macrofauna abundance and τ_c are given in Figures 5.51 – 5.53.

Critical erosion threshold values obtained from EROMES were also regressed on τ_c values obtained from CSM data in order to compare the data from the two different instruments, and identify significant variation present arising from this source (see Figure 5.54).

Statistical tests for normality showed that the data departed significantly from normal distributions, even following transformation, therefore analysis of variance was carried out by using the non-parametric Kruskal Wallis test. Tests were performed on the data using Minitab 13.30 to investigate effects of the formalin treatment, sampling date (time effects), instrument type and biological densities on erosion threshold. Tests were also run to identify significant effects of the formalin treatment and time series on biological density. Results are presented in Table 5.4.

Principal Component Analysis (PCA), a multivariate statistical analysis commonly used to identify and rank environmental factors affecting biological populations or communities was carried out on the biological (faunal densities), sediment (τ_c) and physical data (day of experiment, instrument type, formalin treatment) from the erosion experiments using PRIMER 5.0. PCA was used to identify any experimental/laboratory effects on the cores and underlying factors (results and bubble plots are presented in Tables 5.5 – 5.6 and Figures 5.56 – 5.60).

Further regression analyses were then carried out on mean erosion rates and faunal density data (mean erosion rates on total faunal density, total polychaete density, total bivalve density, cirratulid density, spionid density, motile polychaete density and *Caulleriella* density), to identify any correlation or laboratory effects as before. Kruskal Wallis tests were run on the erosion rate data and faunal data as described above. The regression analyses were carried out firstly on the combined data from all twelve cores, and then additionally on the formalin treated and untreated cores separately following the results of the statistical tests.



5.3 Results

5.3.1 CSM erosion data

Erosion profiles plotted from the CSM data for cores C1 to C26 are given above in Figures 5.1 to 5.26. The profiles show % light transmission (% LT) on the y-axis, plotted against calculated equivalent horizontal bed shear stress on the x-axis. The critical threshold for erosion (τ_c), defined as the first pressure increment after light transmission (tx) dropped below 90 % (after Tolhurst *et al.*, 1999), is shown by the arrow on each graph. The bed shear stresses to the left of this point were non-eroding, and those to the right were characterised by erosion. The line of best fit and equation determined by linear regression are also marked on each plot. The 'goodness' of fit is noted by the R^2 value, and the gradient (given in the equation) is related to erosion rate. A steep slope (gradient) indicates a relatively higher erosion rate.

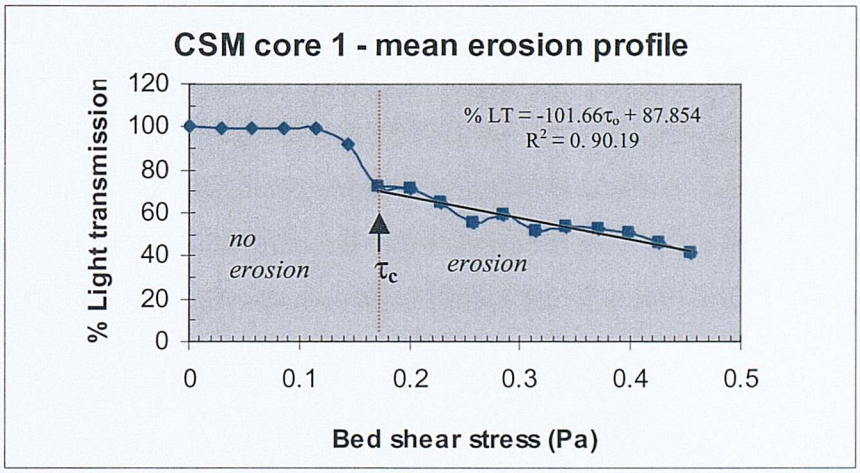


Figure 5.1 Core C1, untreated, day 1

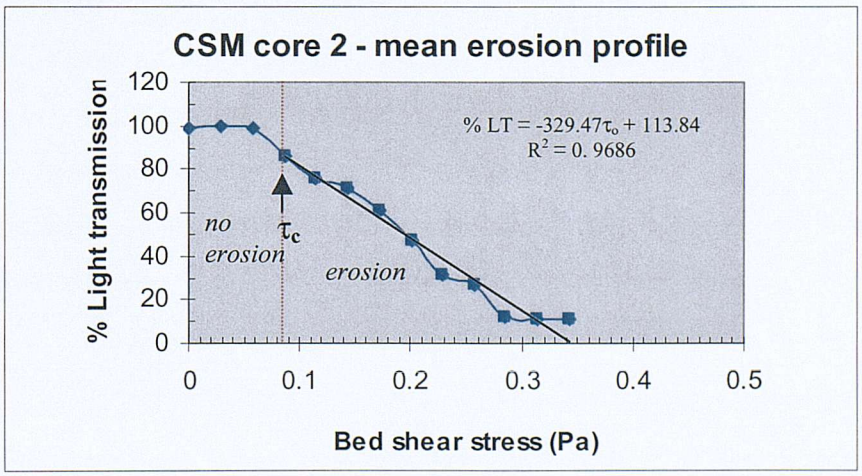


Figure 5.2 Core C2, untreated, day 1

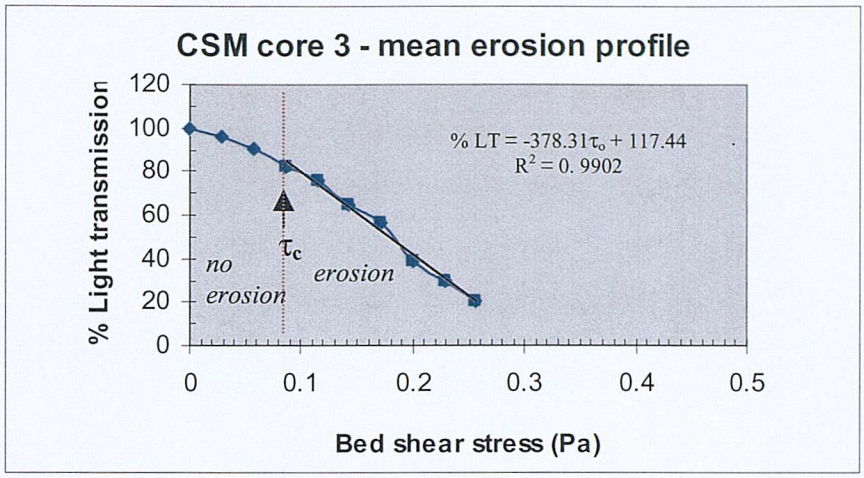


Figure 5.3 Core C3, untreated, day 1

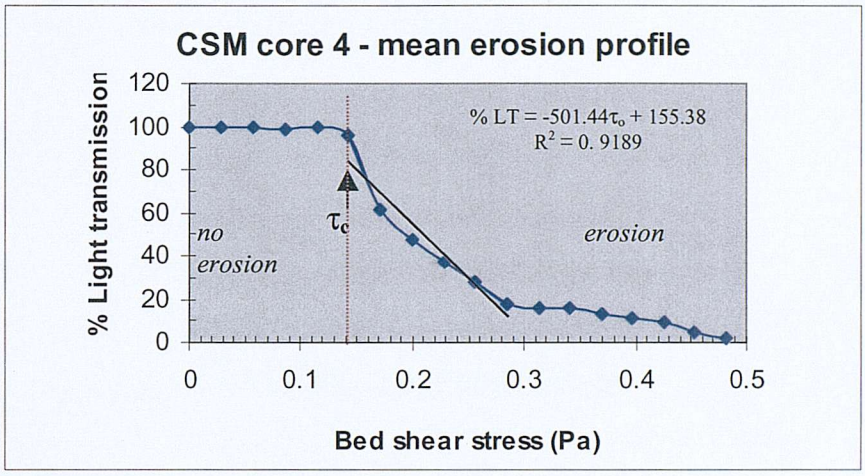


Figure 5.4 Core C4, untreated, day 1

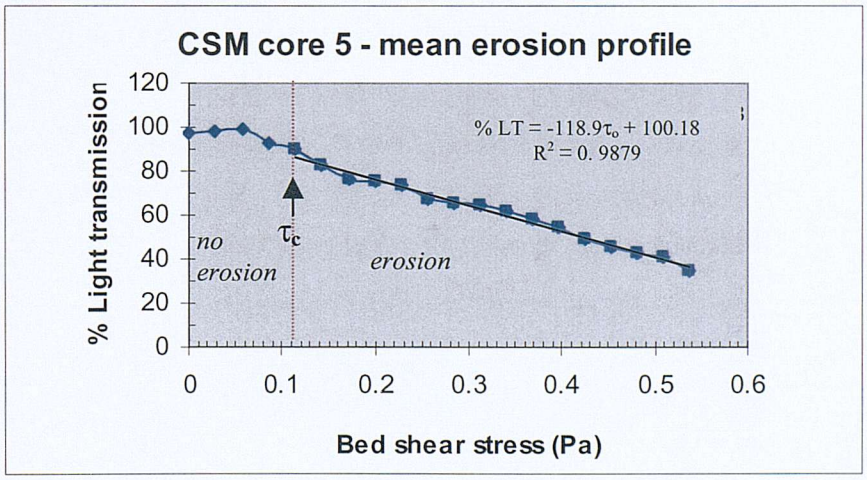


Figure 5.5 Core C5, formalin treated, day 1

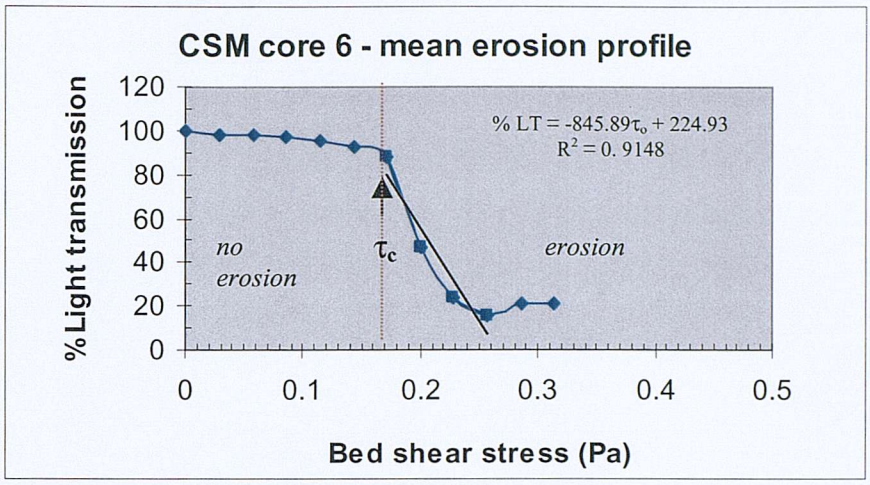


Figure 5.6 Core C6, formalin treated, day 1

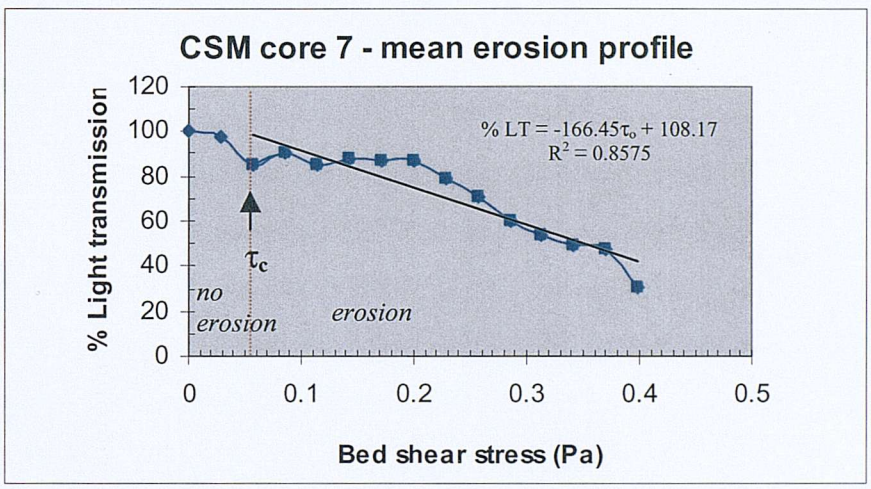


Figure 5.7 Core C7, formalin treated, day 1

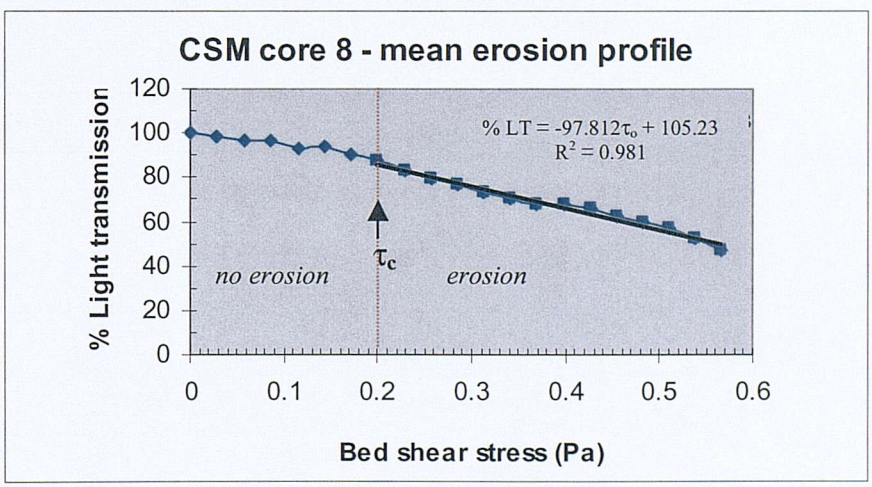


Figure 5.8 Core C8, formalin treated, day 1

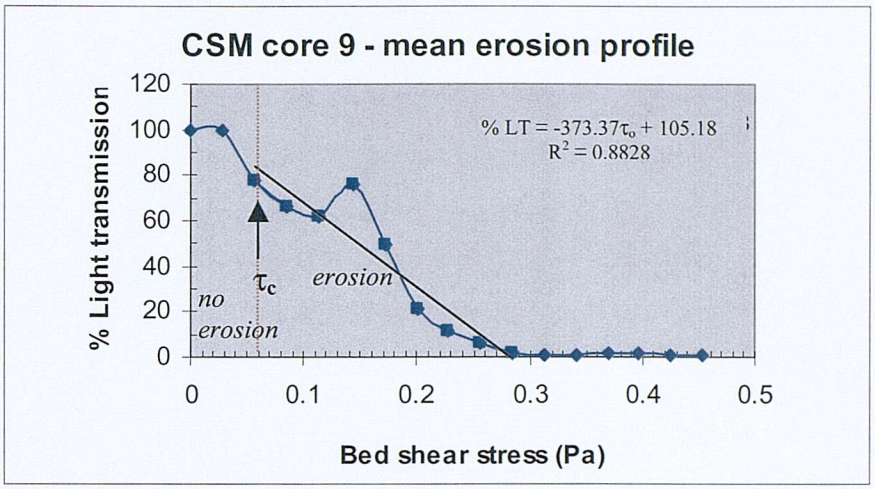


Figure 5.9 Core C9, untreated, day 2

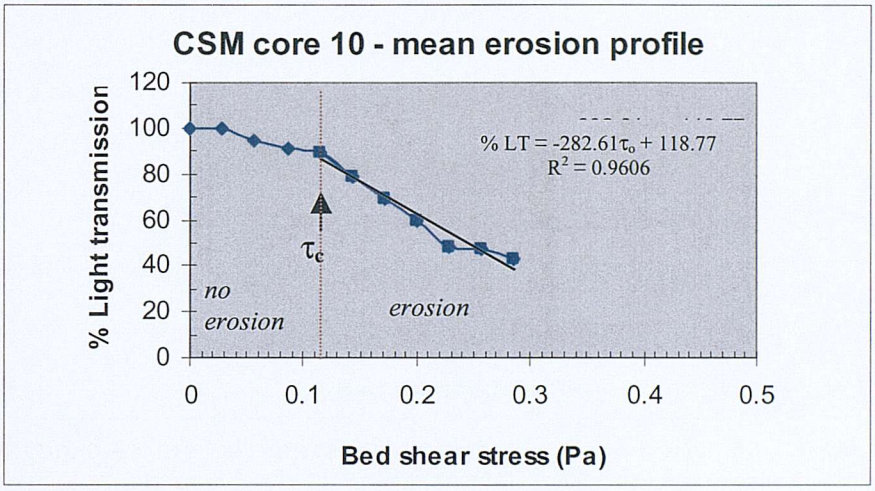


Figure 5.10 Core C10, untreated, day 2

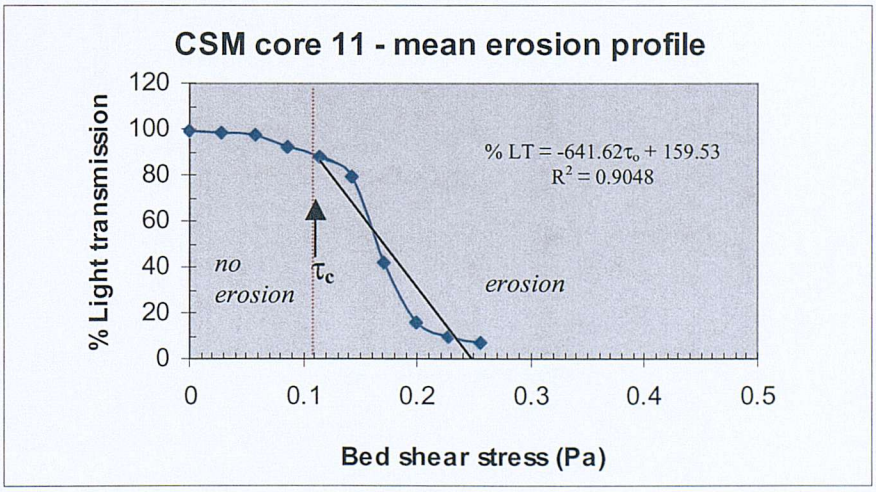


Figure 5.11 Core C11, untreated, day 2

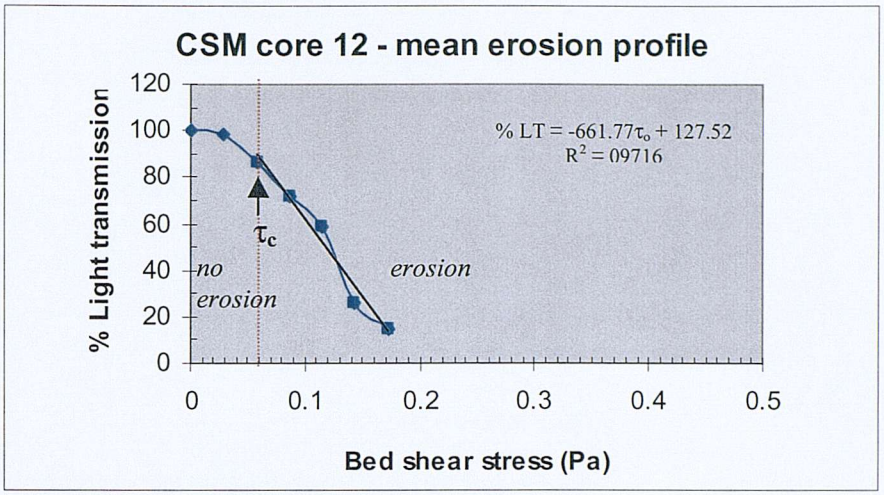


Figure 5.12 Core C12, untreated, day 2

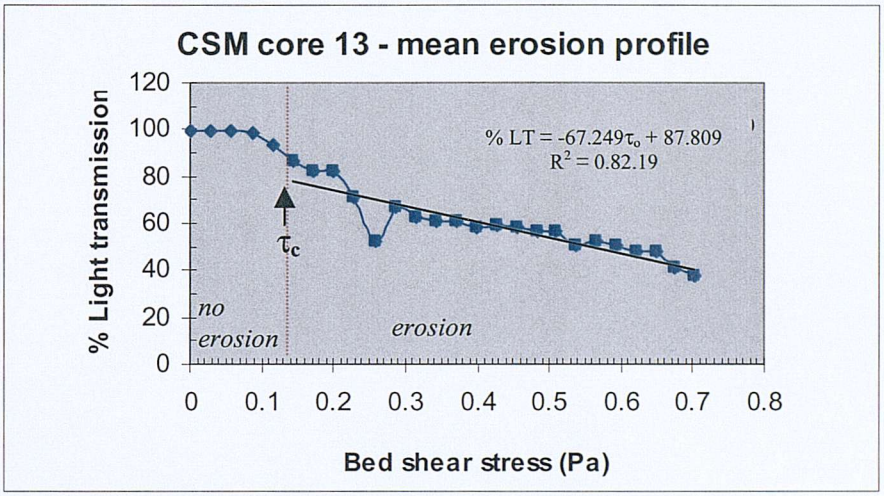


Figure 5.13 Core C13, formalin treated, day 2

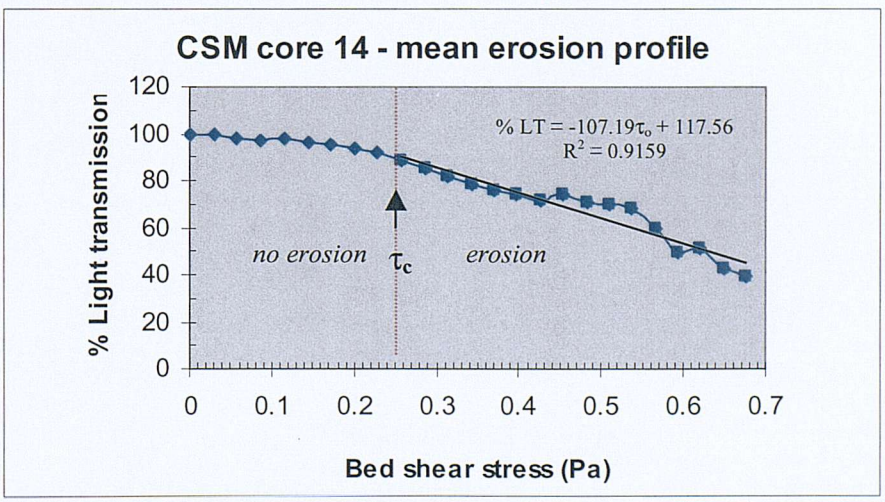


Figure 5.14 Core C14, formalin treated, day 2

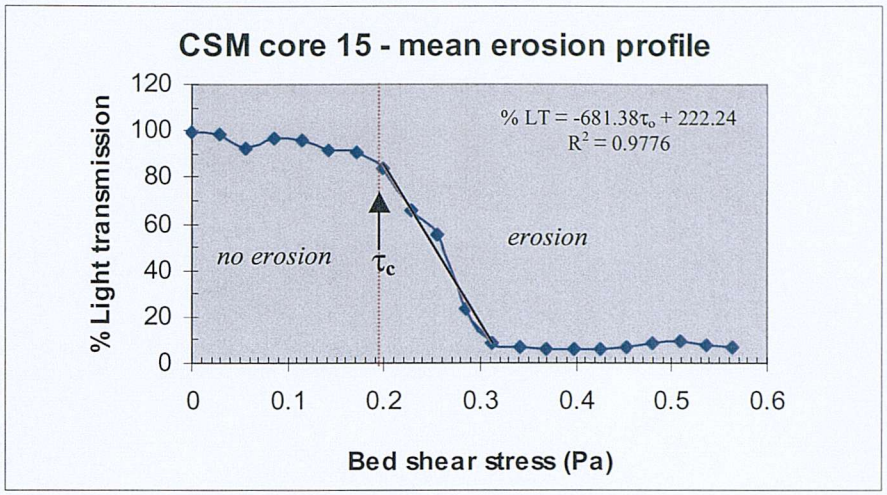


Figure 5.15 Core C15, formalin treated, day 2

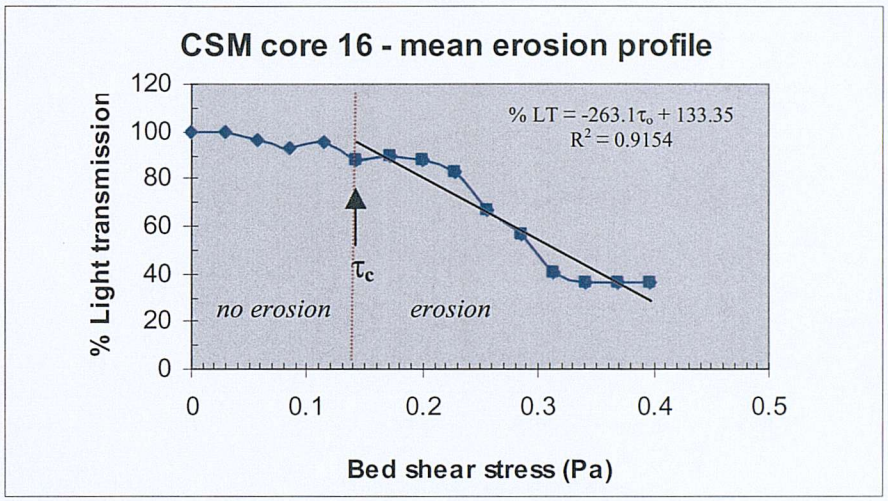


Figure 5.16 Core C16, formalin treated, day 2

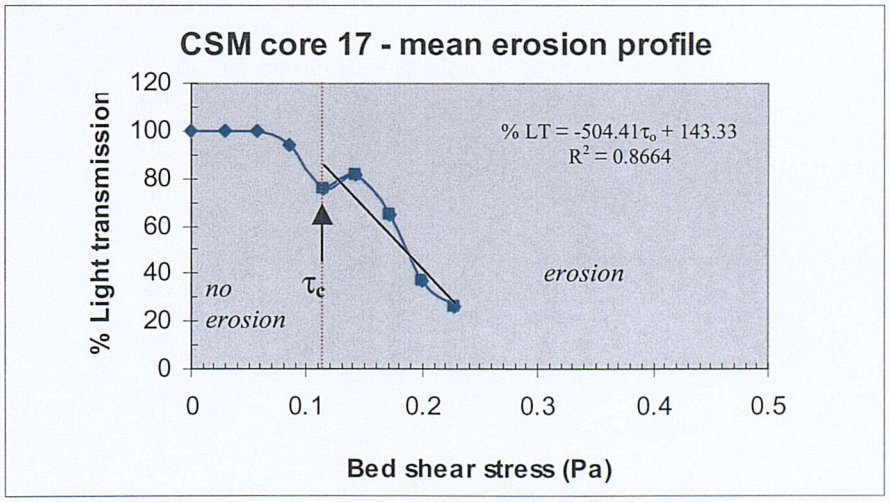


Figure 5.17 Core C17, untreated, day 5

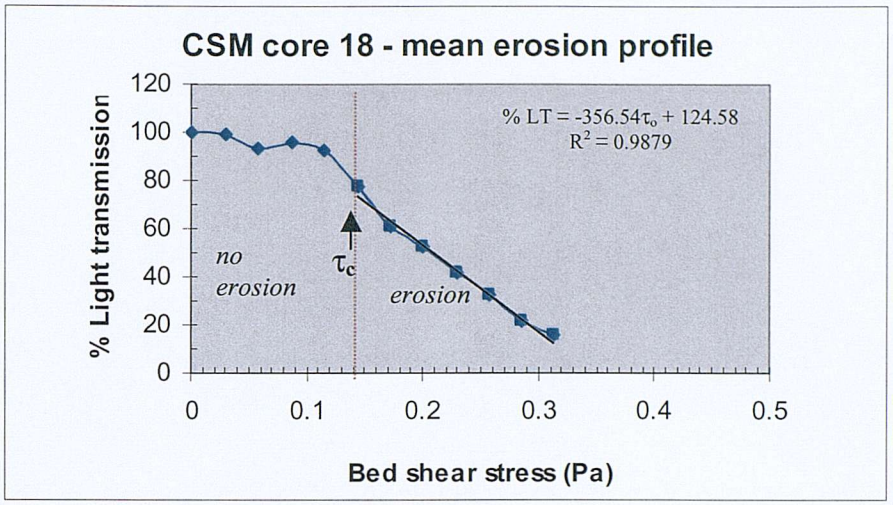


Figure 5.18 Core C18, untreated, day 5

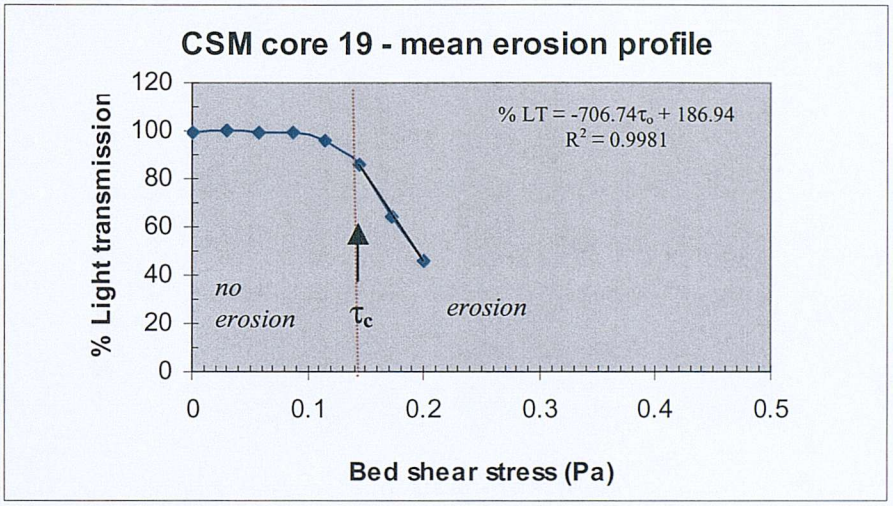


Figure 5.19 Core C19, untreated, day 5

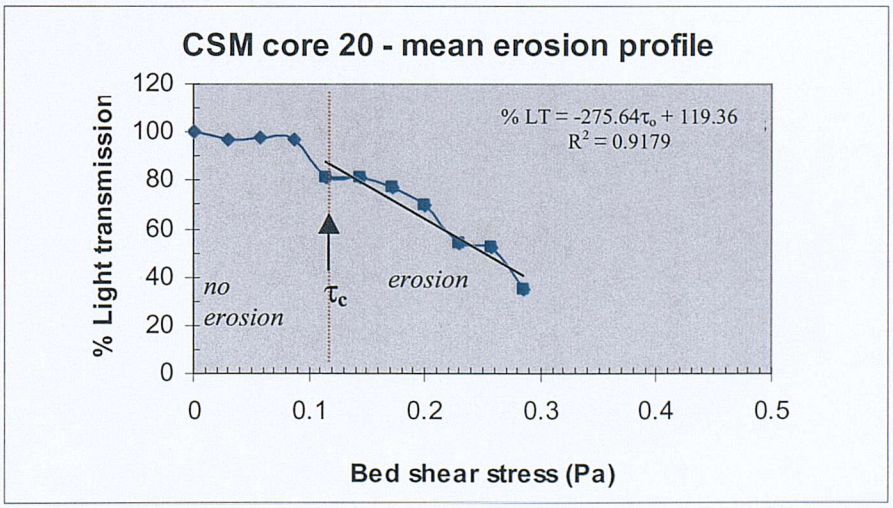


Figure 5.20 Core C20, untreated, day 5

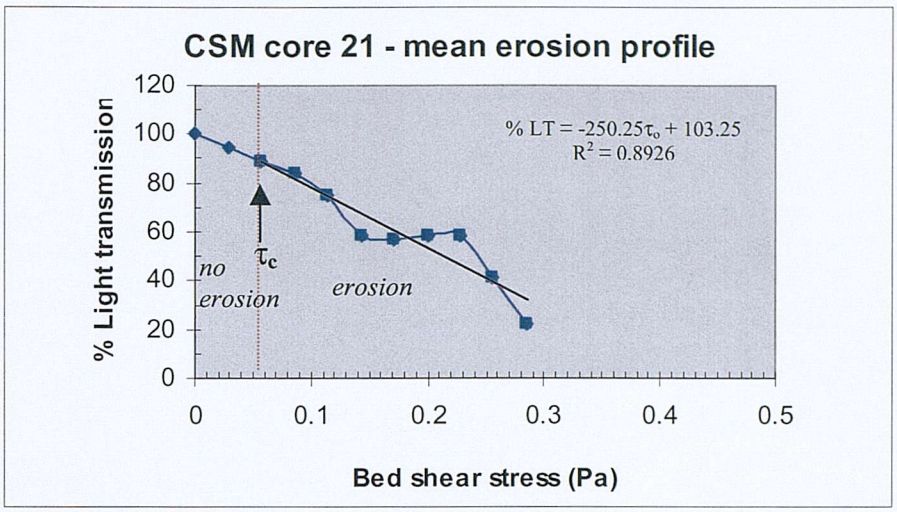


Figure 5.21 Core C21, untreated, day 5

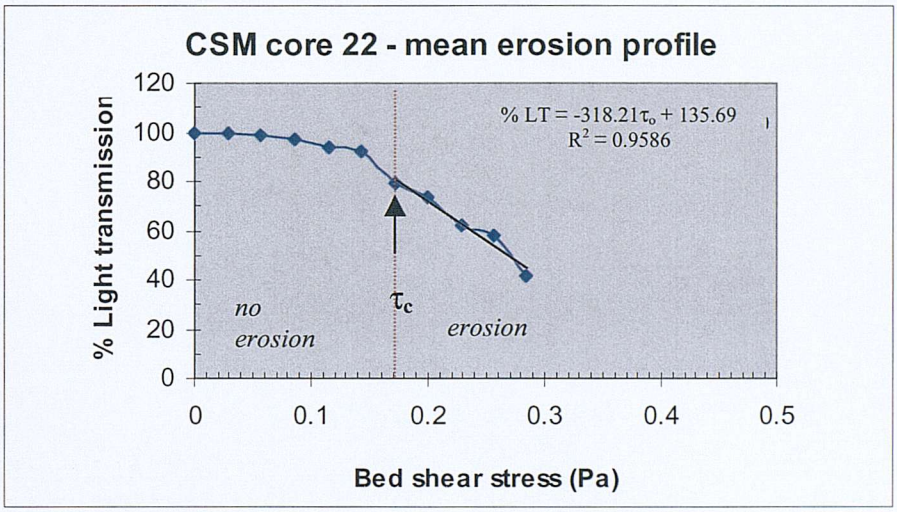


Figure 5.22 Core C22, formalin treated, day 5

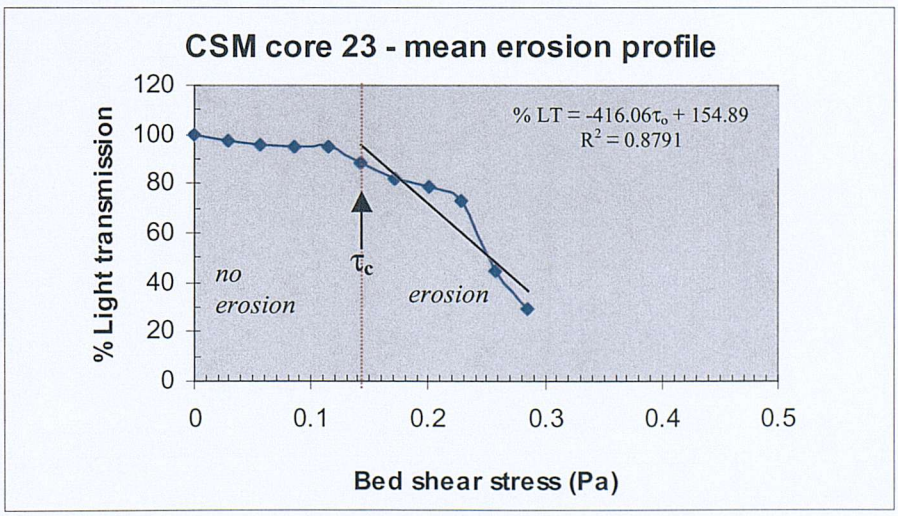


Figure 5.23 Core C23, formalin treated, day 5

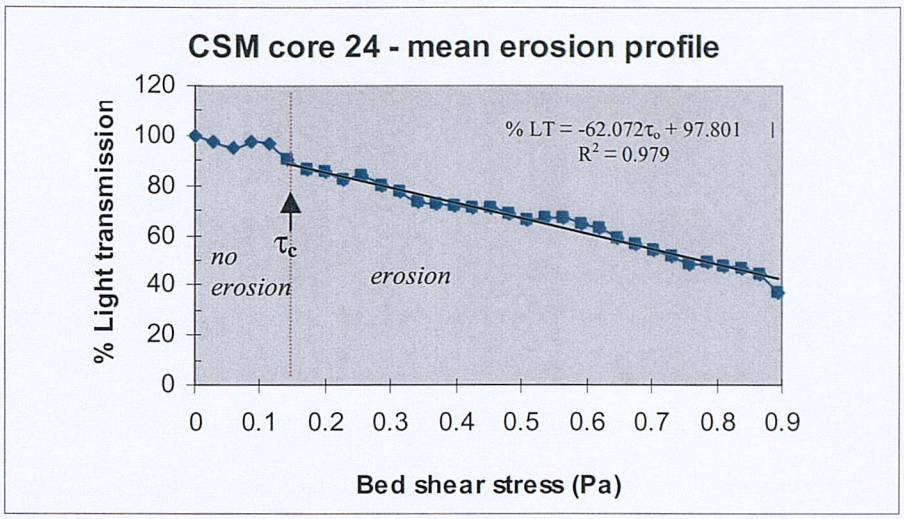


Figure 5.24 Core C24, formalin treated, day 5

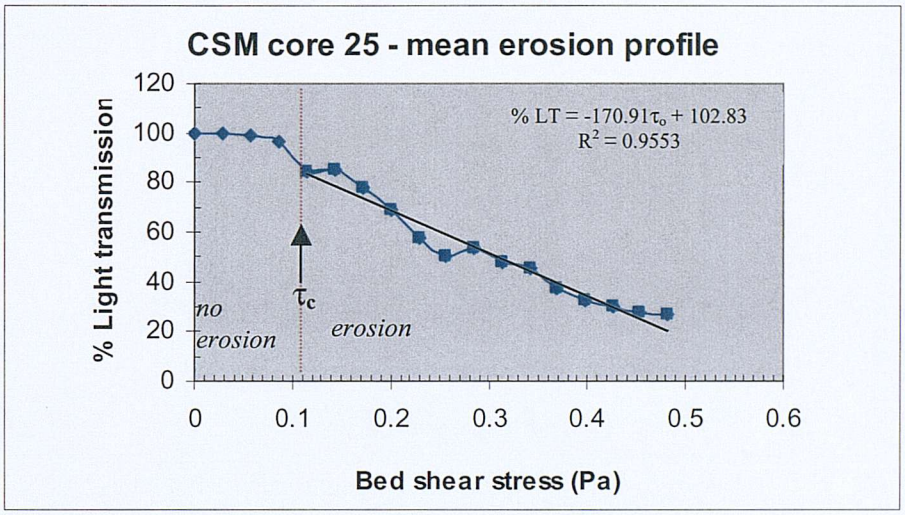


Figure 5.25 Core C25, formalin treated, day 5

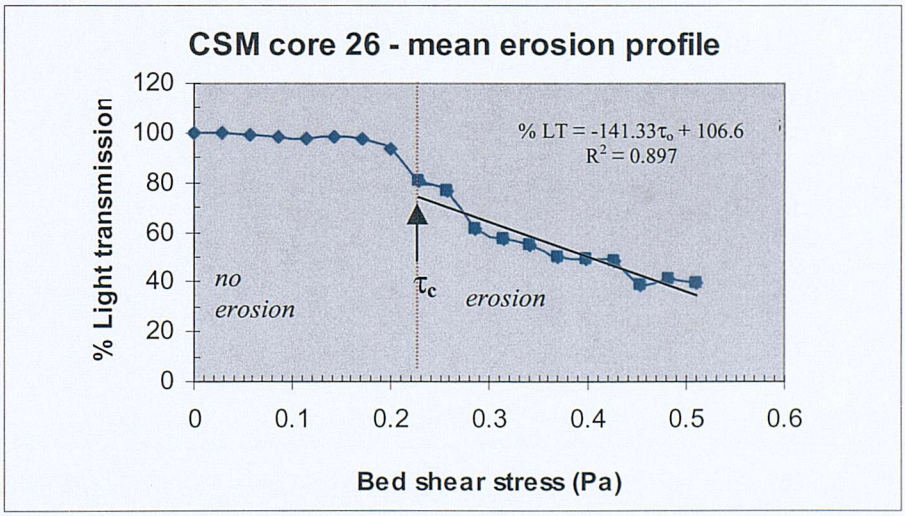


Figure 5.26 Core C26, formalin treated, day 6

Variations in the shape of the profiles and values for τ_c reflect differences in the sediment properties between the cores. High R^2 values indicate a consistently linear relationship between bed shear stress and erosion rate.

5.3.2.1 EROMES OBS calibration

The calibration plot from the OBS output data is shown above. The scatter is increased at higher suspended sediment loads, suggesting that the accuracy of the calibration is reduced at suspended sediment concentrations over around 400 mgl^{-1}. The upper limit of the OBS was found to occur at a corresponding SSC of around 1600 mgl^{-1}. The data are indicative of the natural variation found in fine or cohesive sediment (e.g. floc size) even where sediment samples are taken from the same localized site and area.

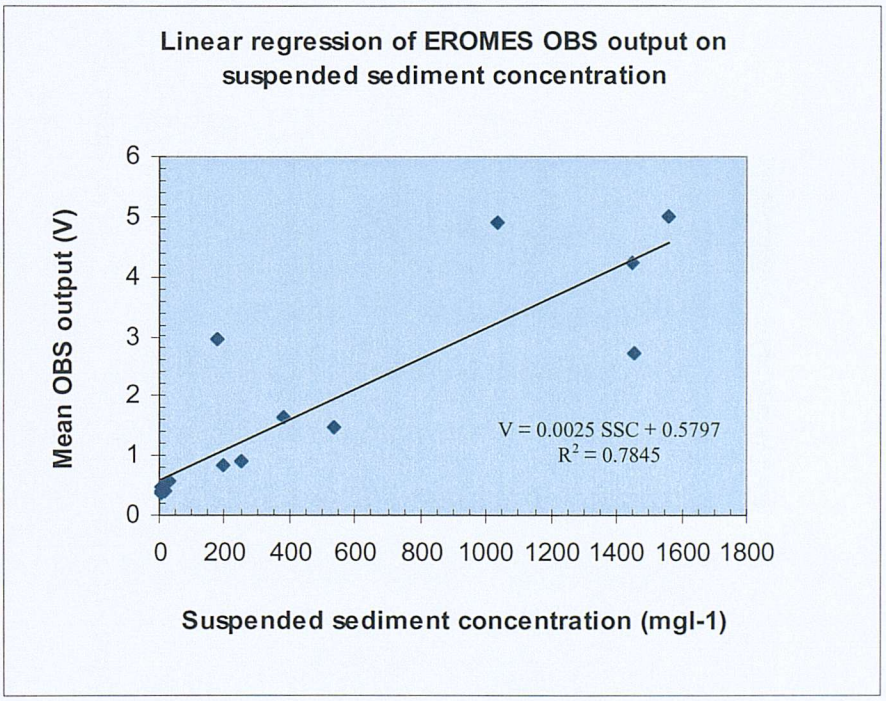


Figure 5.27 Linear regression used to calibrate the OBS for the erosion experiments with EROMES

5.3.2.2 EROMES erosion data

Figures 5.28 – 5.39 show the time series plots obtained from the erosion experiments with EROMES (cores E1 to E12). Calculated SSC (in gl^{-1}, dark blue line) and calculated horizontal bed shear stress (Pa, pink line) were plotted against time. The graphs show the stepwise increases in shear stress produced by the increased propeller rotation speeds during the experiment, and the still water equilibration and settling

periods at the beginning and end of each time series. Erosion is indicated by peaks in SSC following an increase in shear stress. Maximum OBS output is shown by a plateau following the erosion peaks (i.e. cores 2, 3,5,6,9 and 10). No erosion was indicated for core E8 (Figure 5.35), as confirmed by visual observation; the sediment surface appeared undisturbed and the water clear at the end of the experiment. The OBS appeared to be fully functioning upon inspection at this time, so the reason for this lack of erosion is unclear. The plots have been annotated to show regions of erosion and no erosion. The type of erosion predicted is also marked where possible, indicated by an asymptotic curve (Type I) or an unrestricted increase in SSC (Type II) *sensu* Amos *et al.* (1992). Significant variation in erosion, erosion type and erosion rate was indicated between the 12 cores. Variation in τ_c was also indicated between the cores: Increased τ_c is suggested by erosion peaks occurring later in the time series and corresponding to relatively higher bed shear stresses. Peaks in SSC occurring near the beginning of the time series suggest a lower erosion threshold (e.g. Figures 5.29 & 5.33).

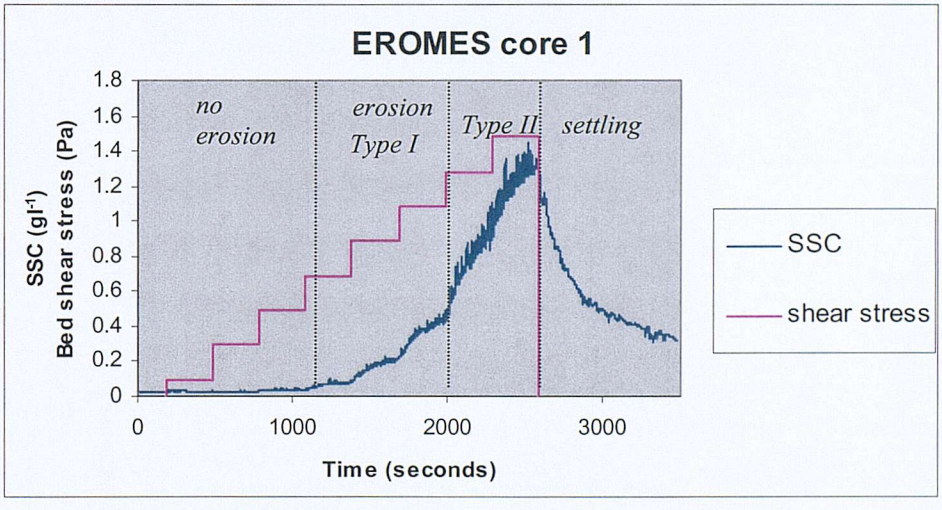


Figure 5.28 Core E1, untreated, day 1

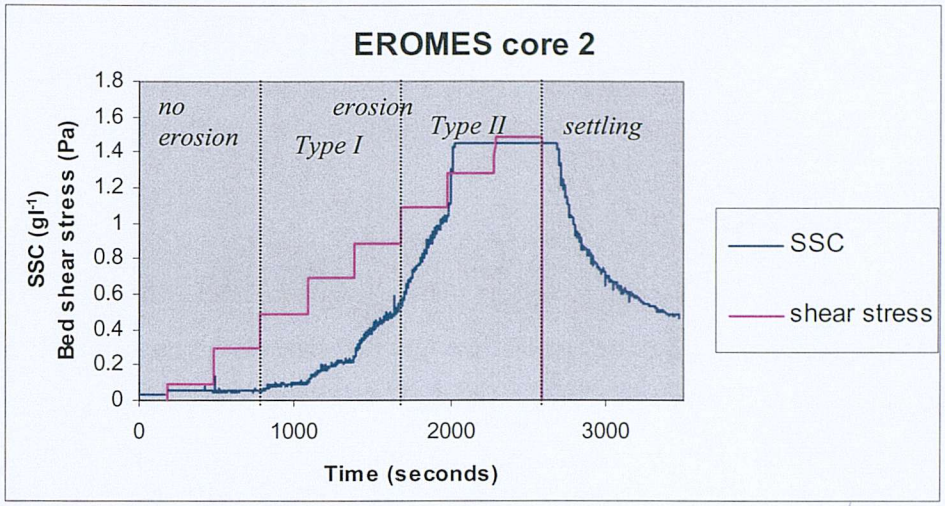


Figure 5.29 Core E2, untreated, day 1

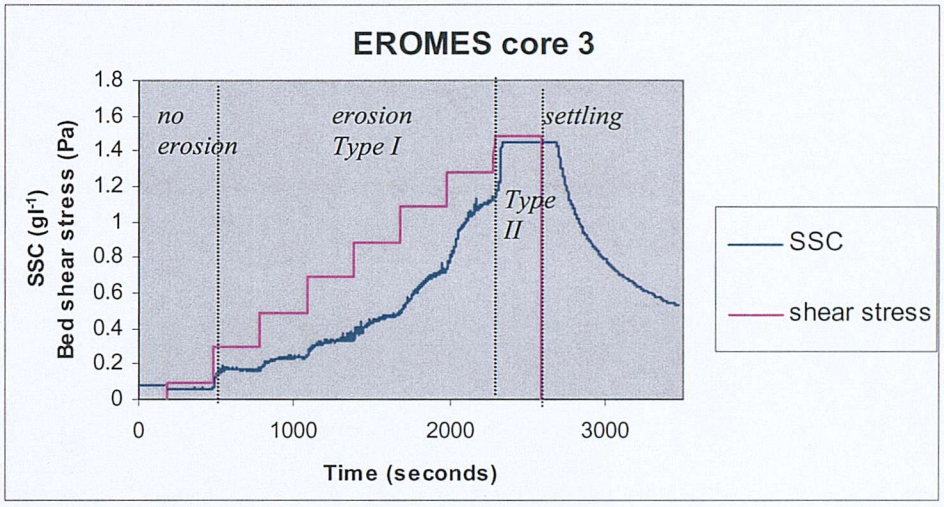


Figure 5.30 Core E3, formalin treated, day 1

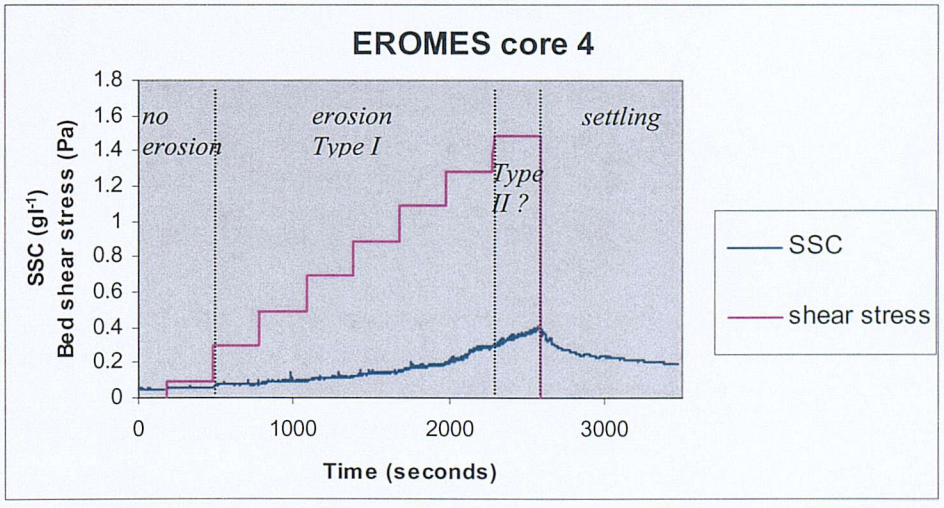


Figure 5.31 Core E4, formalin treated, day 1

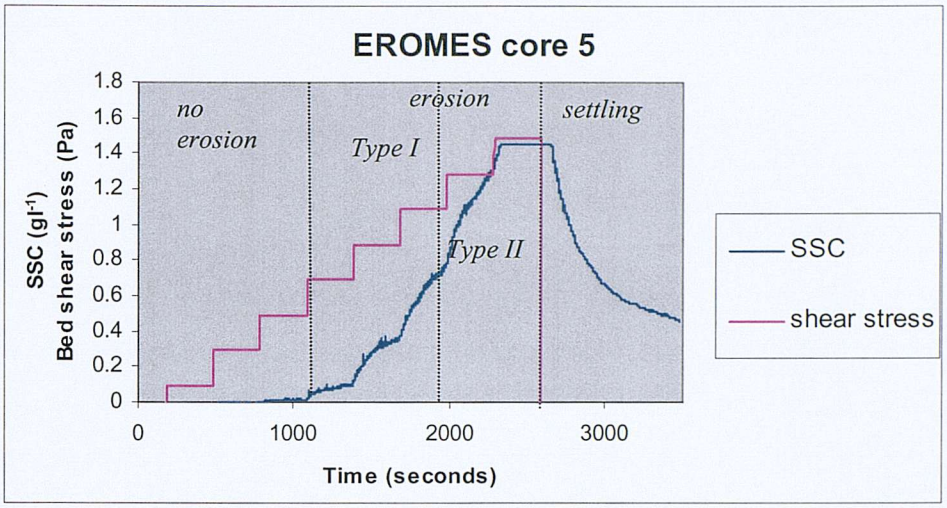


Figure 5.32 Core E5, untreated, day 2

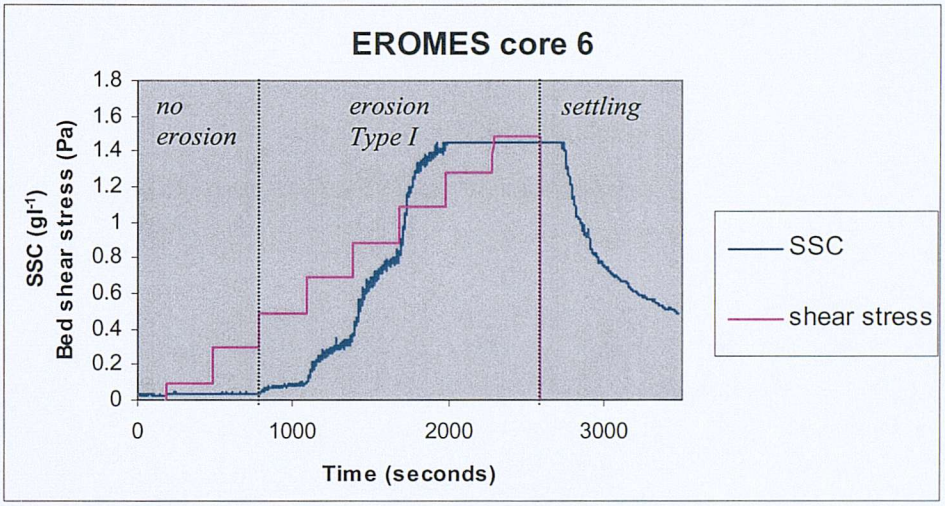


Figure 5.33 Core E6, untreated, day 2

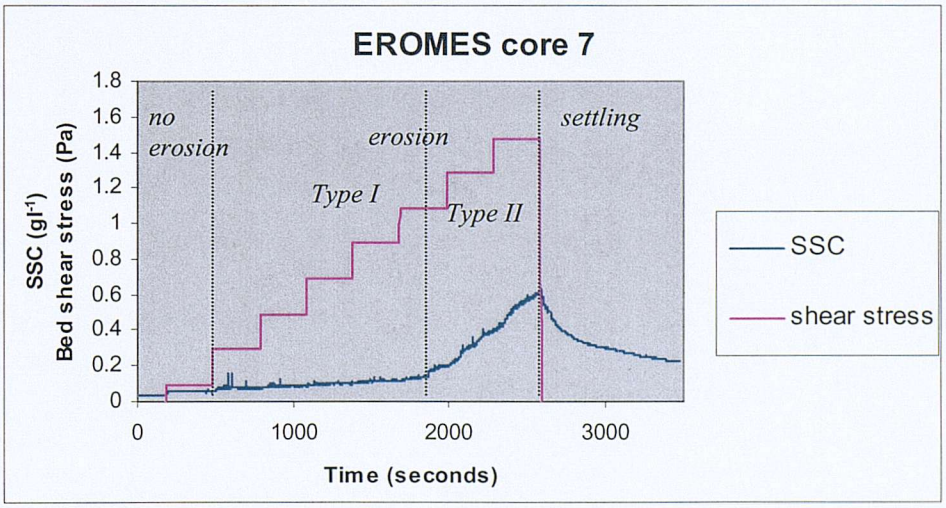


Figure 5.34 Core E7, formalin treated, day 2

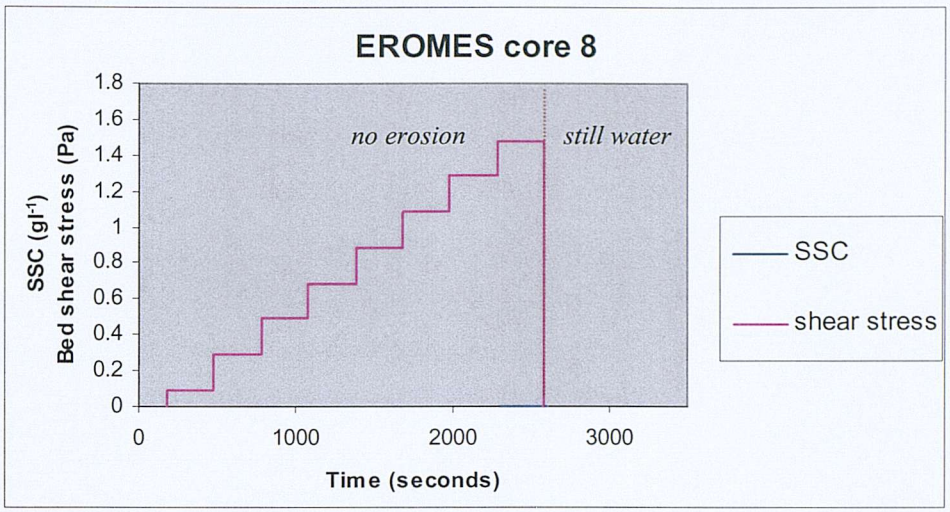


Figure 5.35 Core E8, formalin treated, day 2

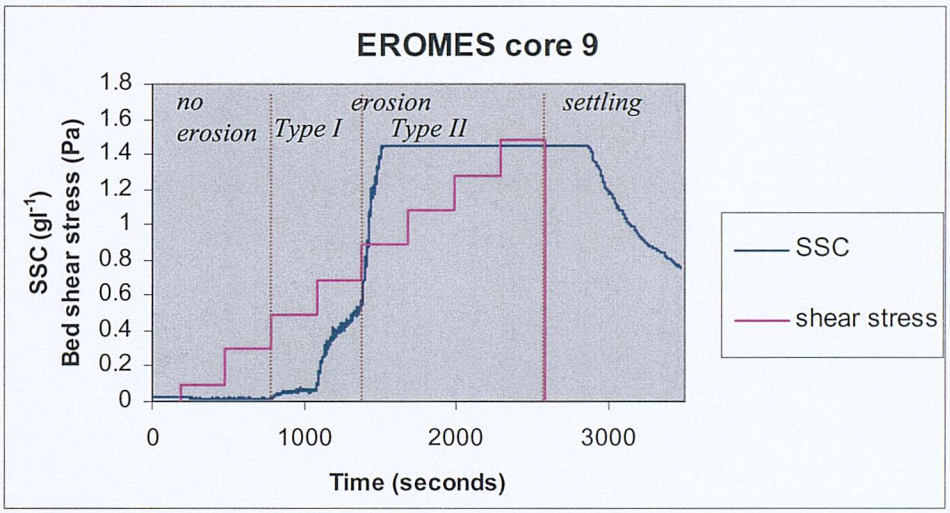


Figure 5.36 Core E9, untreated, day 5

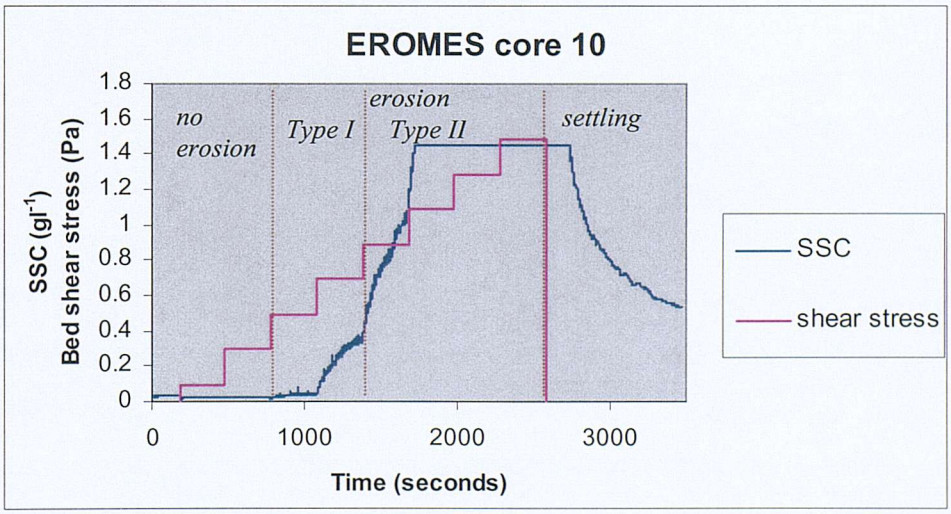


Figure 5.37 Core E10, untreated, day 5

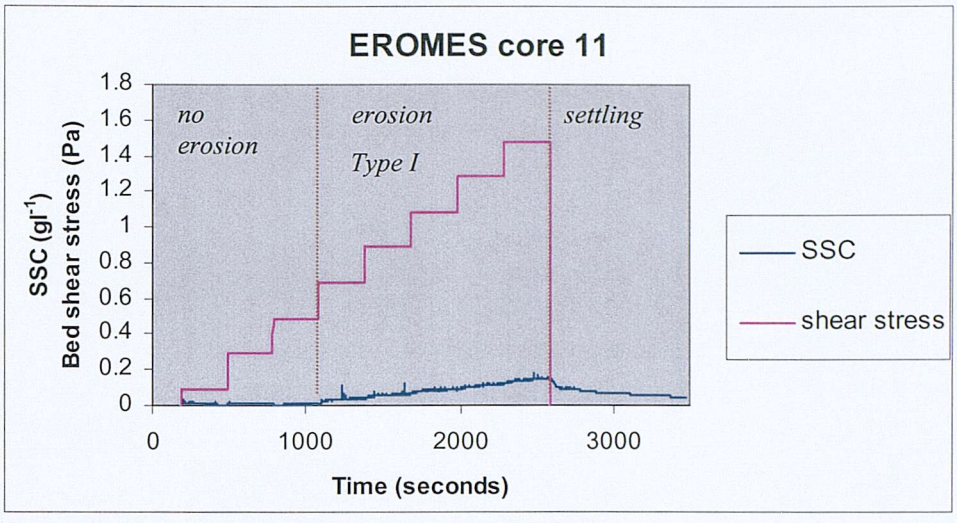


Figure 5.38 Core E11, formalin treated, day 5

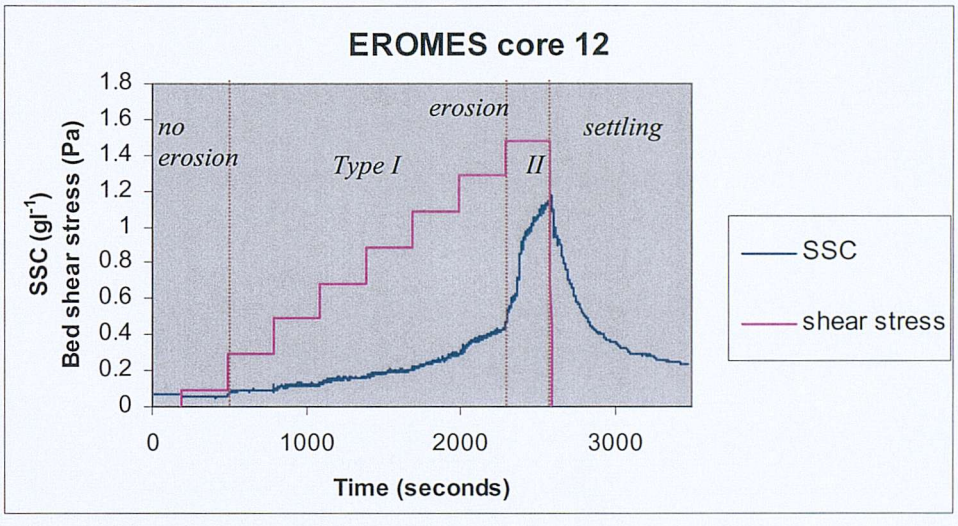


Figure 5.39 Core E12, formalin treated, day 5

Erosion profiles are presented in Figures 5.40 to 5.50. Log_{10} transformed SSC is plotted against untransformed or log_{10} transformed bed shear stress as appropriate for linear regression (as determined by the highest R^2 value). The values for the slopes (gradients) given by the equations are interpreted as a relative measure of friction coefficient (ϕ) (see discussion below). As before, the variation present in the curves represents variation present in the erosion characteristics of the sediment between the cores, a high R^2 value indicates a linear relationship. The relationships obtained from the regressions were extrapolated backwards to the sub-critical ranges of bed shear stress (i.e. the range immediately prior to erosion which is indicated by a sharp increase in SSC), as described in section 5.2.5.2, to estimate single values for τ_c which are listed in Table 5.2.

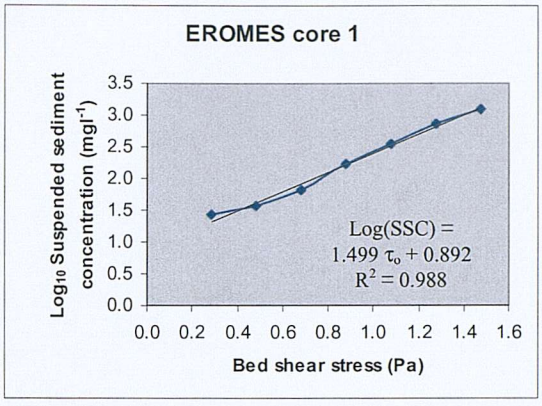


Figure 5.40

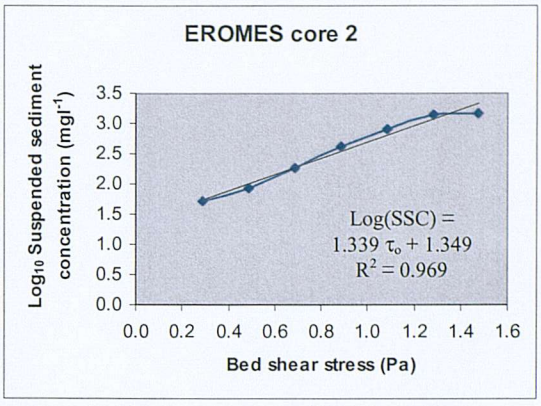


Figure 5.41

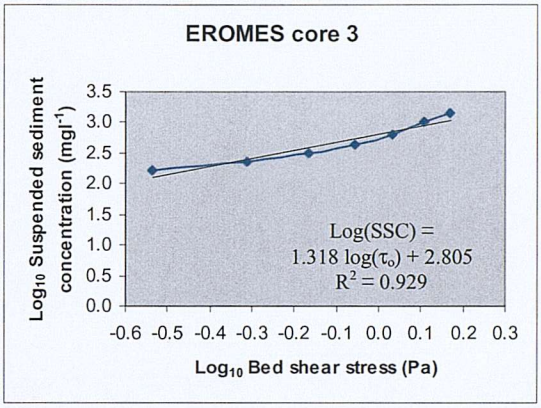


Figure 5.42

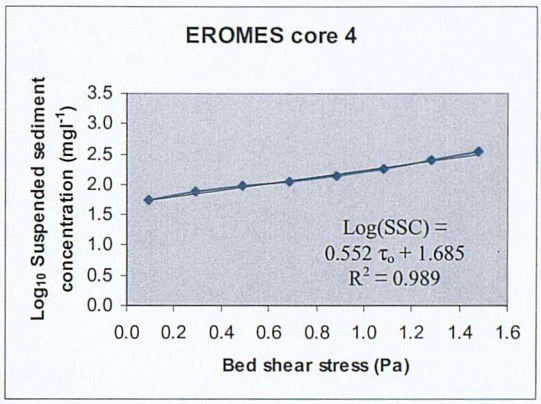


Figure 5.43

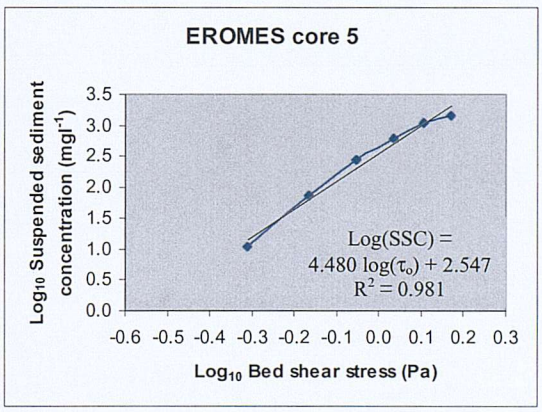


Figure 5.44

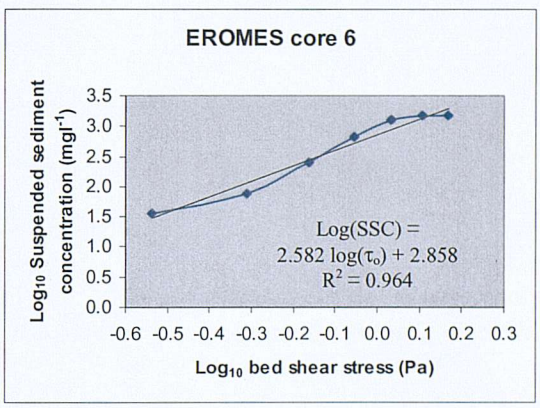


Figure 5.45

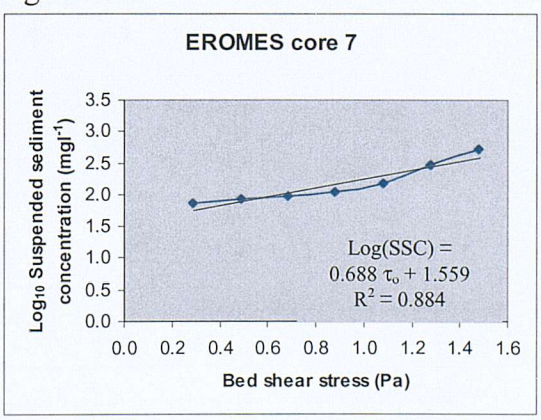


Figure 5.46

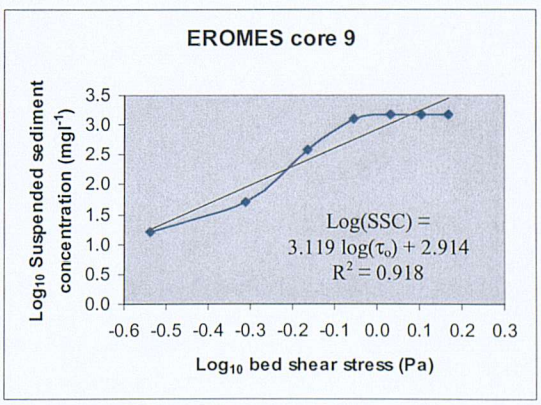


Figure 5.47

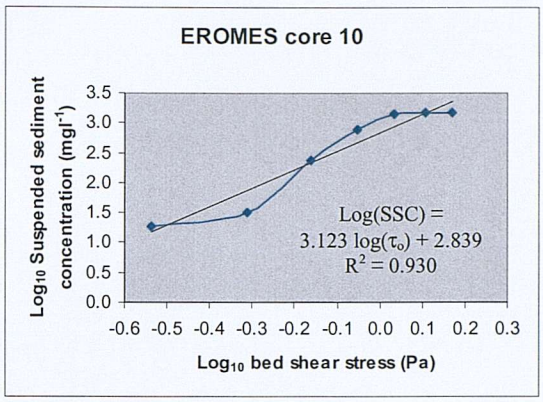


Figure 5.48

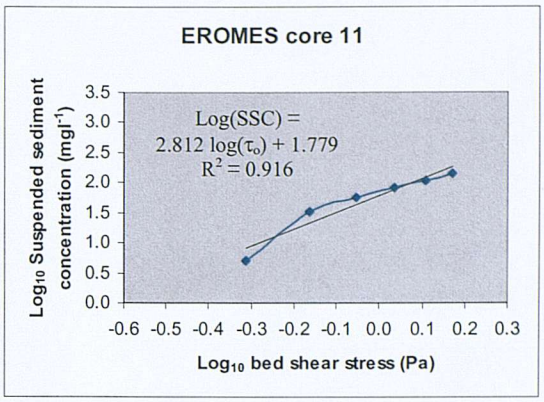


Figure 5.49

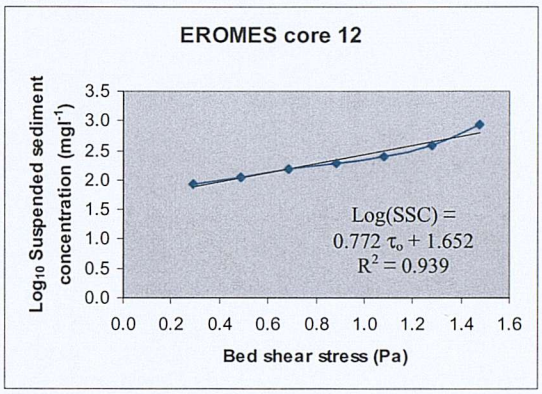


Figure 5.50

5.3.3 Critical erosion threshold results

Tables 5.1 and 5.2 give the range of values calculated for τ_c from these experiments. The data show that generally higher values were estimated for the cores eroded by EROMES than the CSM. Possible reasons for this variation are addressed later. Results from cores investigated with the CSM suggest that the critical erosion threshold of the sediment ranges up to 0.21 Pa (Table 5.1) and values obtained from EROMES fall between 0.03 and 0.55 Pa.

CSM Core	*Day of experiment*	*Formalin Treatment*	**Erosion threshold τ_c (Pa)**	*Gradient**
1	1	-	**0.11**	-149.750
2	1	-	**0.07**	-310.540
3	1	-	**0.06**	-322.850
4	1	-	**0.08**	-250.070
5	1	+	**0.10**	-121.120
6	1	+	**0.09**	-324.350
7	1	+	**0.10**	-154.870
8	1	+	**0.16**	-93.374
9	2	-	**0.00**	-245.280
10	2	-	**0.08**	-229.110
11	2	-	**0.07**	-427.880
12	2	-	**-0.15**	-533.190
13	2	+	**0.09**	-85.453
14	2	+	**0.21**	-86.450
15	2	+	**0.09**	-217.350
16	2	+	**0.12**	-188.060
17	5	-	**0.07**	-330.290
18	5	-	**0.08**	-302.000
19	5	-	**0.09**	-246.560
20	5	-	**0.09**	-220.630
21	5	-	**0.05**	-242.910
22	5	+	**0.10**	-199.480
23	5	+	**0.10**	-216.280
24	5	+	**0.16**	-64.476
25	5	+	**0.09**	-176.340
26	5	+	**0.15**	-147.280

Table 5.1 Critical thresholds for erosion (τ_c) and gradients* obtained from CSM data by linear regression. * as a relative measure of erosion rate.

EROMES core	*Day of experiment*	*Formalin Treatment*	**Erosion threshold τ_c (Pa)**	*Gradient**
1	1	-	**0.37**	1.499
2	1	-	**0.29**	1.339
3	1	+	**0.16**	1.319
4	1	+	**0.03**	0.552
5	2	-	**0.27**	4.480
6	2	-	**0.32**	2.582
7	2	+	**0.24**	0.688
8	2	+	-	-
9	5	-	**0.29**	3.120
10	5	-	**0.32**	3.123
11	5	+	**0.55**	2.812
12	5	+	**0.10**	0.772

Table 5.2 Critical thresholds for erosion (τ_c) and gradients* obtained from EROMES data by linear regression. * as a proxy of ϕ (friction coefficient).

5.3.4 Results of linear regressions of critical erosion thresholds on faunal abundance, and instrument comparison

Figures 5.51 – 5.53 show the results of the linear regressions carried out between τ_c and faunal density. No significant relationships are indicated by these plots which show considerable scatter and R^2 values close to zero. Possible causes for these results are discussed in section 5.4.3 below.

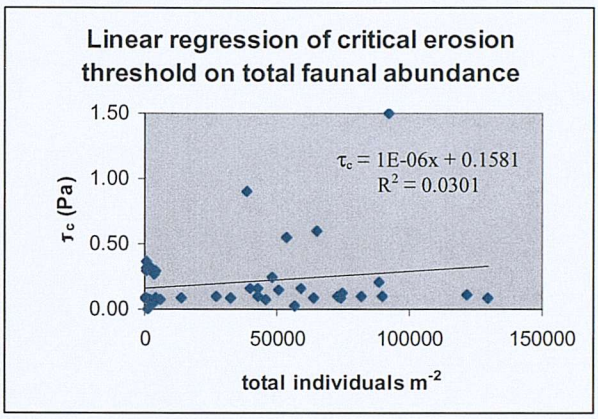


Figure 5.51

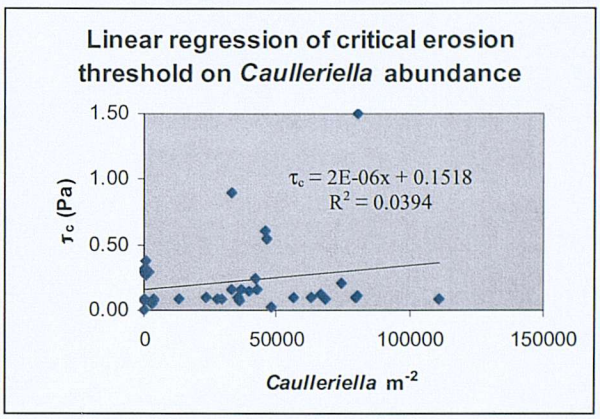


Figure 5.52

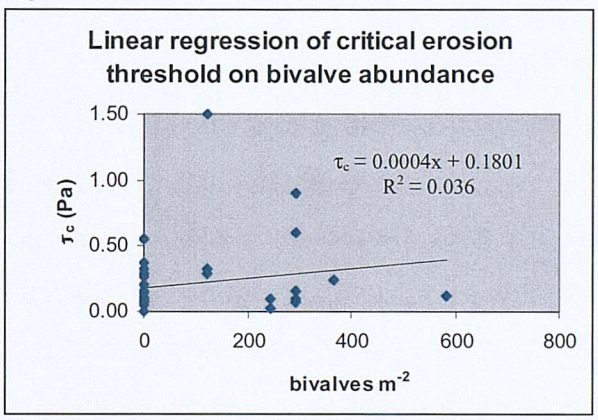


Figure 5.53

Figure 5.54 shows the result of a linear regression carried out to compare the critical erosion threshold data from the two instruments, EROMES and the CSM (after Tolhurst *et al.*, 2000). No significant relationship is indicated with much scatter present (the computed equation and R^2 value are shown for illustration). Possible reasons for this are discussed later in the chapter (section 5.4.7).

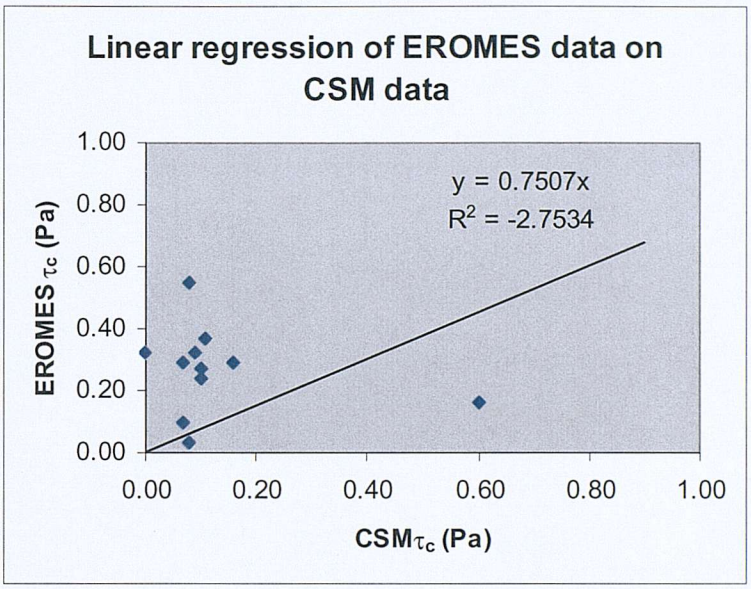


Figure 5.54

5.3.5 Results of faunal analysis

Core ref.	Day/ Formalin		Total Macrofauna individuals	Total fauna per m^2	Cirratulids per m^2	Spionids per m^2	Polychaetes per m^2	Bivalves per m^2	*Nereis* & *Nephtys* per m^2	Caulleriella per m^2
E1	1	-	7	854	854	0	854	0	0	610
E2	1	-	6	732	366	0	366	0	0	122
E3	1	+	324	39512	36829	976	39024	0	1098	33293
E4	1	+	464	56585	52683	1341	56098	244	1585	47927
E5	2	-	30	3659	1463	244	3415	0	1707	854
E6	2	-	5	610	366	122	610	0	122	0
E7	2	+	395	48171	44024	1951	47561	366	1220	41707
E8	2	+	755	92073	85610	3293	91341	122	976	80366
E9	5	-	37	4512	2683	122	4268	122	1341	1585
E10	5	-	20	2439	1220	610	2317	122	0	854
E11	5	+	440	53659	50000	2439	53293	0	366	46098
E12	5	+	349	42561	40244	1220	42195	244	488	35244
C1	1	-	417	121894	117802	976	120433	0	0	80386
C2	1	-	156	45601	42385	1462	45016	0	1169	36247
C3	1	-	223	65186	63139	877	64601	292	585	45893
C4	1	-	254	74247	73370	292	74247	0	585	68693
C5	1	+	308	90032	85063	2338	87986	0	292	79801
C6	1	+	218	63724	62262	585	63724	0	877	29231
C7	1	+	280	81847	78047	1462	81263	292	585	56416
C8	1	+	202	59047	53493	2923	58170	292	1754	42678
C9	2	-	5	1462	1169	0	1169	0	0	292
C10	2	-	0	0	0	0	0	0	0	0
C11	2	-	7	2046	877	292	1169	292	0	0
C12	2	-	0	0	0	0	0	0	0	0
C13	2	+	443	129494	122479	4092	127740	0	877	111079
C14	2	+	303	88571	83894	877	87986	0	2631	74540
C15	2	+	111	32447	32154	0	32154	0	0	27770
C16	2	+	255	74540	70155	1462	73078	585	585	66355
C17	5	-	20	5846	4677	292	5846	0	585	3800
C18	5	-	15	4385	4092	0	4385	0	292	3800
C19	5	-	3	877	877	0	877	0	0	0
C20	5	-	47	13739	13446	0	13739	0	292	13154
C21	5	-	13	3800	3508	0	3508	0	0	3215
C22	5	+	249	72786	71032	1462	72786	0	292	62847
C23	5	+	93	27185	26600	585	27185	0	0	23385
C24	5	+	146	42678	39462	2923	42678	0	0	36539
C25	5	+	131	38293	36247	0	38001	292	1169	33031
C26	5	+	173	50570	44431	4385	50278	0	1169	39462

Table 5.3 Actual and calculated faunal densities from counts for individual cores

5.3.6 Results of Kruskal Wallis tests (non-parametric analysis of variance)

Response variable	Source	Median	*z*	*DF*	*H*	*P*
Total macrofauna density (individuals m^{-2})	Day of experiment					
	1	61386	1.95	2	3.83	0.147
	2	18053	-0.72			
	5	20462	-1.18			
Erosion threshold (τ_c, Pa)	Day of experiment					
	1	0.1050	0.14	2	0.18	0.912
	2	0.1050	-0.42			
	5	0.1000	0.27			
Total macrofauna density (individuals m^{-2})	Treatment					
	untreated	3659	-3.84	1	14.74	0.000
	formalin treated	56585	3.84			
Erosion threshold (τ_c, Pa)	Treatment					
	untreated	0.09000	-1.36	1	1.85	0.173
	formalin treated	0.12000	1.36			
Erosion threshold (τ_c, Pa)	Instrument					
	CSM	0.09000	-3.20	1	10.32	0.001
	EROMES	0.29000	3.20			
Erosion threshold (τ_c, Pa)	Total macrofauna density (individuals m^{-2})					
	0 - 129494	-0.035 - 900	-	36	36.74	0.434
Erosion threshold (τ_c, Pa)	Total polychaete density (individuals m^{-2})					
	0 - 127740	-0.035 – 0.09	-	34	36.34	0.360
Erosion threshold (τ_c, Pa)	Cirratulid density (individuals m^{-2})					
	0 - 122479	-0.035 – 0.09	-	34	36.52	0.352
Erosion threshold (τ_c, Pa)	*Caulleriella* density (individuals m^{-2})					
	0 - 111079	0.08 – 0.09	-	31	32.13	0.410
Erosion threshold (τ_c, Pa)	Spionid density (individuals m^{-2})					
	0 - 4385	0.09 – 0.15	-	17	21.78	0.193
Erosion threshold (τ_c, Pa)	Mobile polychaete density (*Nereis* & *Nephtys*, individuals m^{-2})					
	0 - 2631	0.09 – 0.21	-	15	14.58	0.482
Erosion threshold (τ_c, Pa)	Total bivalve density (individuals m^{-2})					
	0 - 585	0.095 – 0.12	-	5	8.97	0.110

Table 5.4 Summary of results from Kruskal Wallis tests on erosion experiment data. *z* = standard deviations from the mean, *DF* = degrees of freedom, *H* = an approximation of chi square distribution (with k-1 degrees of freedom, where k is the number of samples), *P* = level of significance.

Table 5.4 summarizes the results obtained from the Kruskal Wallis tests. The statistical analyses confirm that there was no significant correlative relationship between macrofauna densities and critical erosion threshold in the data obtained from these experiments in relation either to total faunal density or to individual taxonomic groupings. The variance between total macrofauna density and the time (date) of the erosion experiments was also shown to be statistically insignificant at a 95 % confidence level (P = 0.147).

The Kruskal Wallis test results confirm the presence of highly significant variance in the erosion threshold values between the two instruments used (P = 0.001), which is too great to be attributed to sampling error alone. Highly significant variance in faunal density was also found between the untreated and treated cores, which can be attributed to the formalin treatment (P = 0.00). One possible reason for this is core deterioration, the implications of which are discussed later (section 5.4.8). Treatment of the cores with formalin was not found to be a significant factor for the variance in erosion thresholds (P = 0.173). No significant relationship was found between erosion threshold and the time (day) of the erosion experiments (P = 0.912).

5.3.7 Results of PCA analysis

The results of the Principal Components Analysis (PCA) carried out on the biological, sediment and physical data (see Table 5.6) obtained from the erosion experiments are shown in Tables 5.5 - 5.6 and Figures 5.55 – 5.59.

Table 5.5 shows the Eigenvalues, variation and cumulative variation (expressed as percentages) for each of the principal components identified by the test. Principal components are linear compounds of correlated variables identified by the statistical analysis. Eigenvalues are ratios of total variability in the data and are variances of the uncorrelated data produced by the analysis. The results indicate that principal component 1 accounts for 55.4% of the observed variation in the data, and principal components 2, 3, 4 and 5 account for 12.3%, 10.7%, 8.5% and 5.6% respectively.

PC	Eigenvalues	% Variation	Cumulative % variation
1	5.54	55.4	55.4
2	1.23	12.3	67.7
3	1.07	10.7	78.4
4	0.85	8.5	86.9
5	0.56	5.6	92.5

Table 5.5 Eigenvalues obtained from Principal Components Analysis. PC = principal component, Eigenvalues = a measure of variance produced by PCA, % Variation = percentage of total variation in data calculated by PCA, Cumulative % variation = calculated cumulative percentage of total variation.

Eigenvectors obtained from the PCA are shown in Table 5.6. Eigenvectors are correlation coefficients calculated during PCA in the form of weightings (or loadings) that are applied to the variables (the variables used in the analysis are given in the first column of Table 5.6).

Variable	PC1	PC2	PC3	PC4	PC5
Day of experiment	0.107	-0.534	-0.638	0.320	-0.126
Treatment untreated /formalin	-0.307	-0.003	-0.182	0.496	-0.129
Erosion threshold (τ_c, Pa)	-0.207	-0.402	-0.187	-0.729	0.137
Total macrofauna density (individuals m^{-2})	-0.343	0.418	-0.099	0.084	0.030
Cirratulid density (individuals m^{-2})	-0.416	0.108	-0.049	-0.039	0.028
Spionid density (individuals m^{-2})	-0.368	-0.147	-0.193	-0.041	-0.035
Total polychaete density (individuals m^{-2})	-0.418	0.085	-0.043	-0.042	0.012
Total bivalve density (individuals m^{-2})	-0.192	-0.447	0.441	0.324	0.665
Caulleriella density (individuals m^{-2})	-0.415	0.047	-0.064	-0.042	0.014
Mobile polychaete density (*Nereis* & *Nephtys*, individuals m^{-2})	-0.210	-0.370	0.525	0.019	-0.709

Table 5.6 Eigenvectors (correlation coefficients or weightings) from PCA. PC1 – PC5 = principal components.

The data suggests that the variability ascribed to PC1 is due to intercorrelations resulting from differences in faunal density, in particular cirratulid density, and experimental treatment (formalin) (see also Figure 5.57). PC2 is described by loadings that are similar for several variables including day, τ_c, total faunal density and total bivalves, and the data suggests that PC2 reflects instrument differences between EROMES and the CSM (Figure 5.55). For PC3 the eigenvectors are loaded towards day of the experiment, mobile polychaete density and bivalve abundance (i.e. large

individuals) appear to be the factors influencing variablility. This suggests that PC3 can be interpreted as intercorrelation influenced by incubation time. PC4 and PC5 only account for 8.5 and 5.6% of total variability and are therefore not considered significant.

Figures 5.55 – 5.59 show the sample plot and bubble plots obtained from the PCA (PRIMER 5.0). Each of the scatter plots are composed of the same data (as shown in Figure 5.55), but show a different variable as an overlay with the values as labels to illustrate correlation (or lack of) between the variables and PC1 and PC2 (which form the axes of the graphs). The variation present in the data from the two different instruments and which corresponds to PC2 is clearly shown in Figure 5.55. The variation in faunal density, which corresponds to PC1, is illustrated in Figure 5.57. Figure 5.59 illustrates the variation in the data sets that can be ascribed to the treatment of half of the cores with formalin. The correlation with PC1 can be clearly seen with very little overlap present between untreated and treated samples. No clear relationship is suggested between the two principal components (PC1 and PC2) and critical erosion threshold data (Figure 5.56) where the distribution of values and bubble size is apparently random with much scatter.

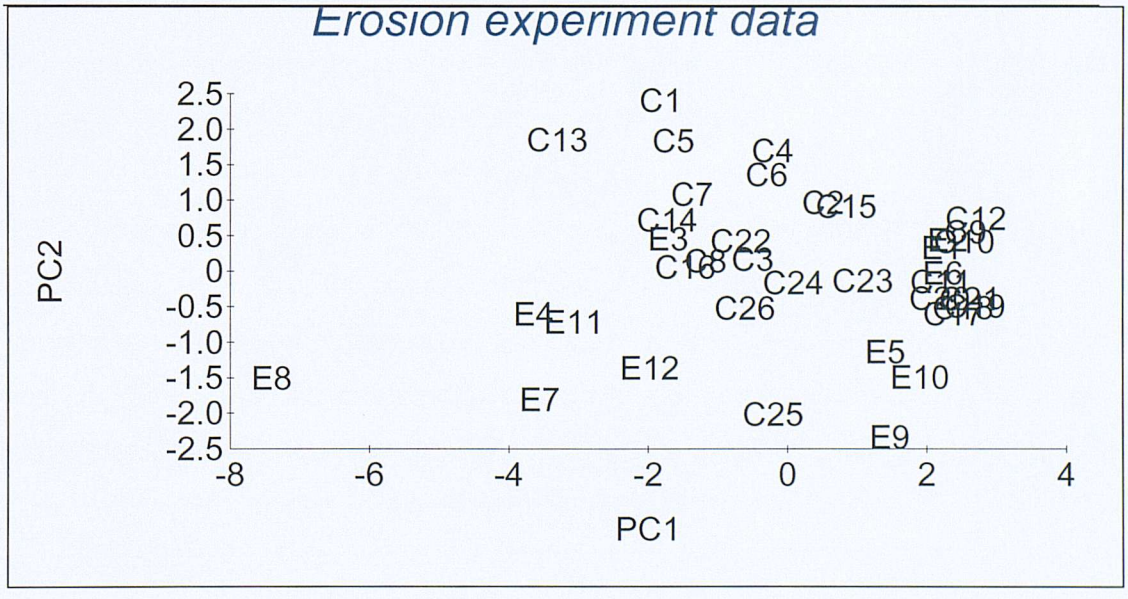


Figure 5.55 Sample plot obtained from the multivariate Principal Components Analysis (using PRIMER 5.0). The 2 axes represent the first 2 principal components identified in the data. The data labels are the core reference numbers. Closeness

between points represents similarity in samples, increased distance is related to disparity.

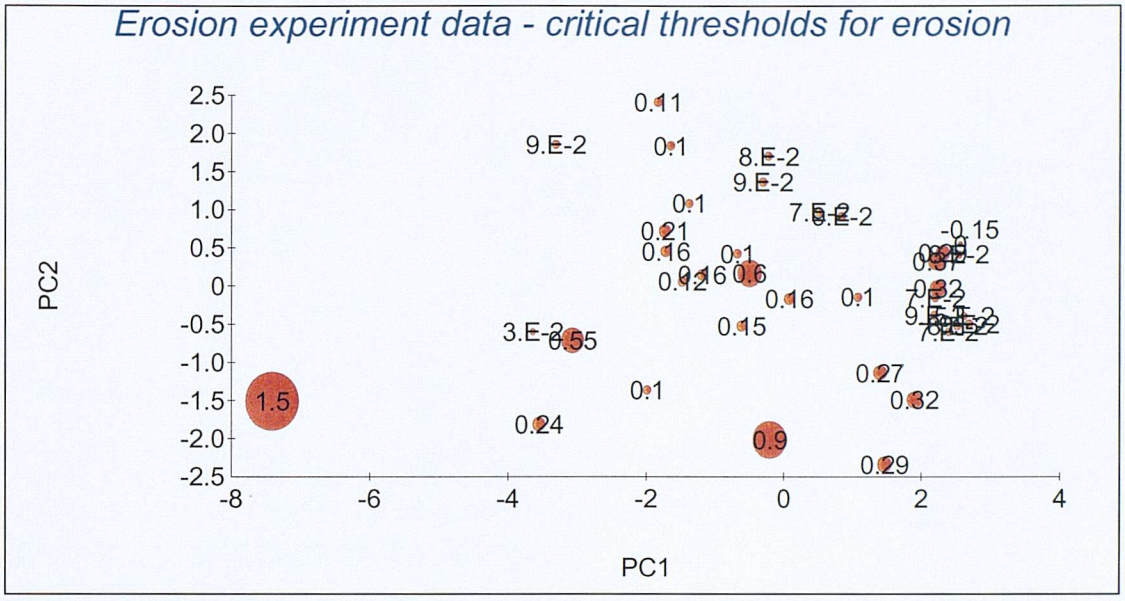


Figure 5.56 Bubble plot showing τ_c data and values superimposed on the previous plot. The position of each sample (individual cores) remains the same. Cluster or grouping of similarly sized bubbles together would indicate clear correlative relationship with one or more factors included in the analysis.

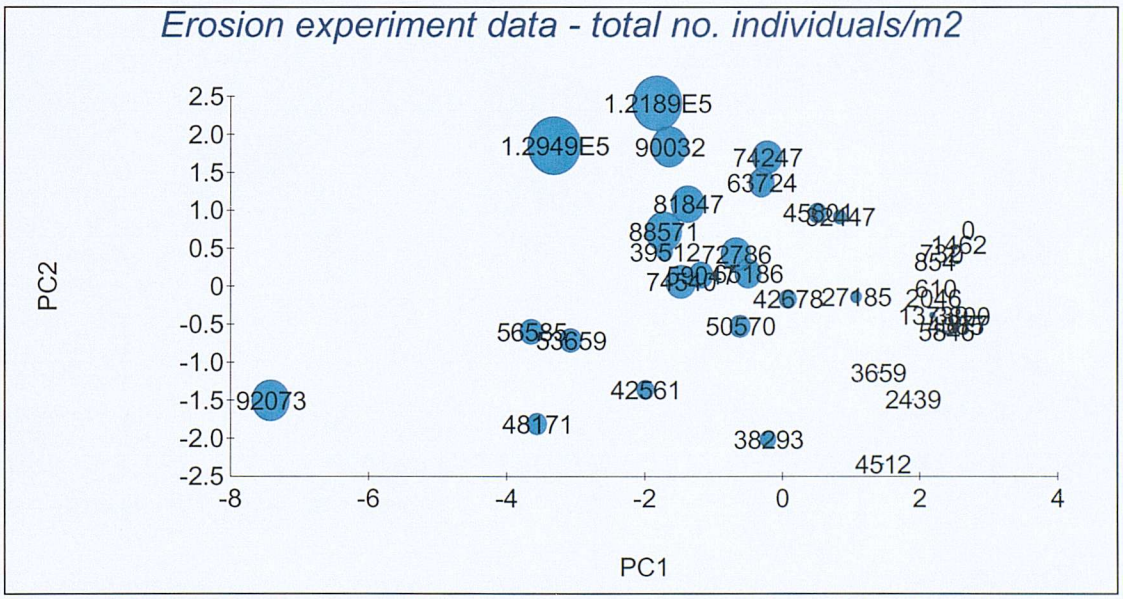


Figure 5.57 Bubble plot from PCA showing superimposed faunal density values. Clusters of similar sized bubbles/values indicate correlation between 2 or more included factors.

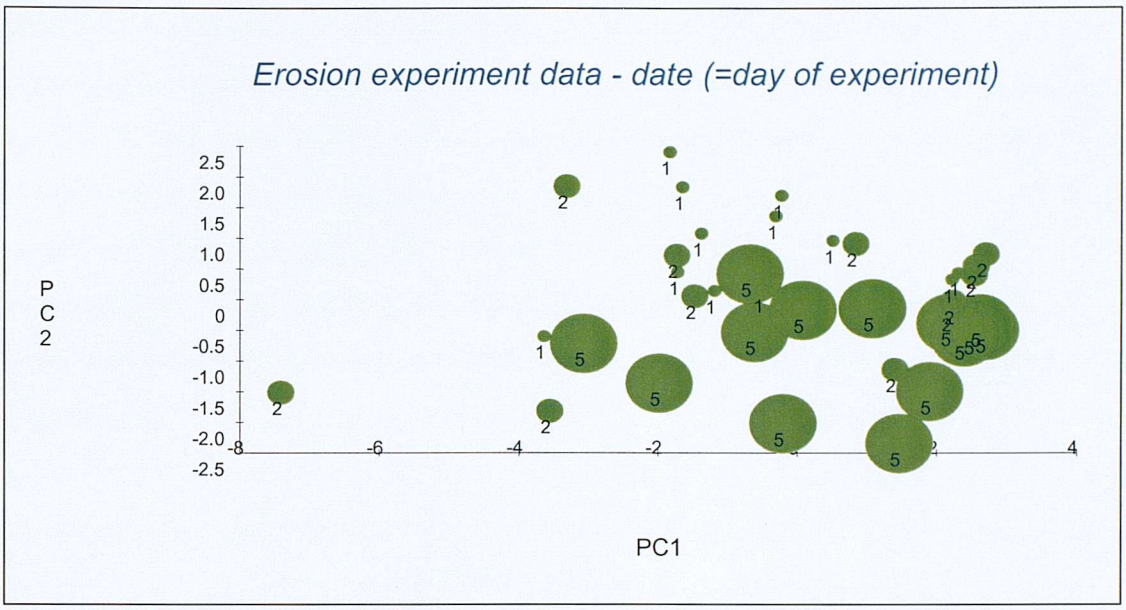


Figure 5.58 Bubble plot with time factor from experiments superimposed. Note the similarity in cluster position between this factor and faunal density in the previous figure that suggests negative correlation.

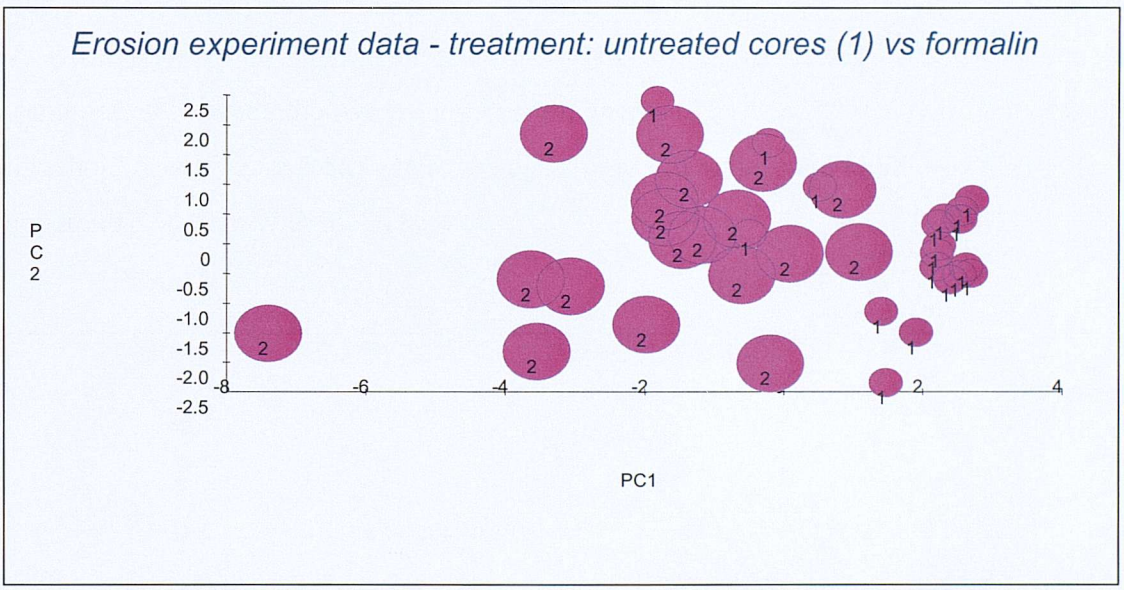


Figure 5.59 Bubble plot showing effects of (formalin) treatment superimposed. 1 = untreated cores, 2 = formalin treated cores. Note the almost complete lack of overlap between the two sets of samples.

5.3.8 Erosion rate data obtained from EROMES

Time series plots of calculated erosion rates from the EROMES experiments are presented in Figures 5.60 to 5.71. The graphs indicate that differences in erosion rates are present between individual cores. Erosion rates were recorded to a maximum of 11 x 10^{-5} $mgm^{-2}s^{-1}$ from core 10. Cores 1, 2, 3, 9, 10 and 12 appeared to display higher rates of erosion compared to the remaining cores. Minimum erosion rates overall were

recorded from core 11. All cores displayed erosion except core 8. Note that where the erosion rates appear to go to zero between approximately 1500 and 2500 seconds in cores 2, 3, 5, 6, 9 and 10; this effect is due to saturation of the OBS output shown previously on corresponding erosion profiles as a 'plateau'. This does *not* mean that no erosion was occurring (or that erosion rates were nil), but that the erosion rate could not be calculated due to differences (increases) in SSC being beyond the detectable limits of the OBS. It is highly probable that erosion continued during this phase of the experiments.

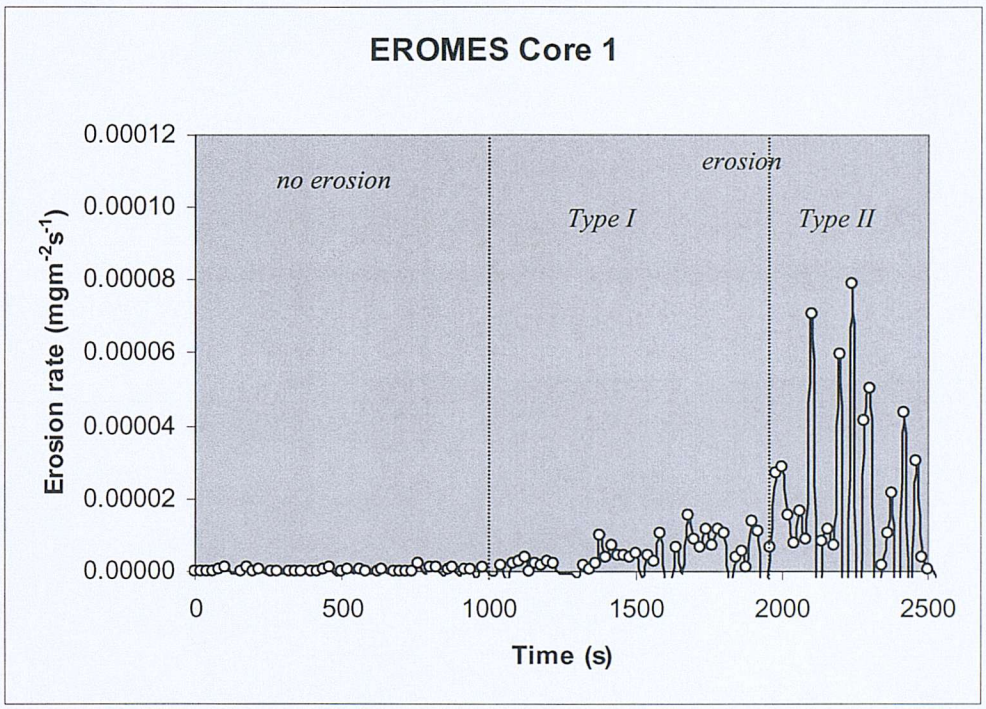


Figure 5.60

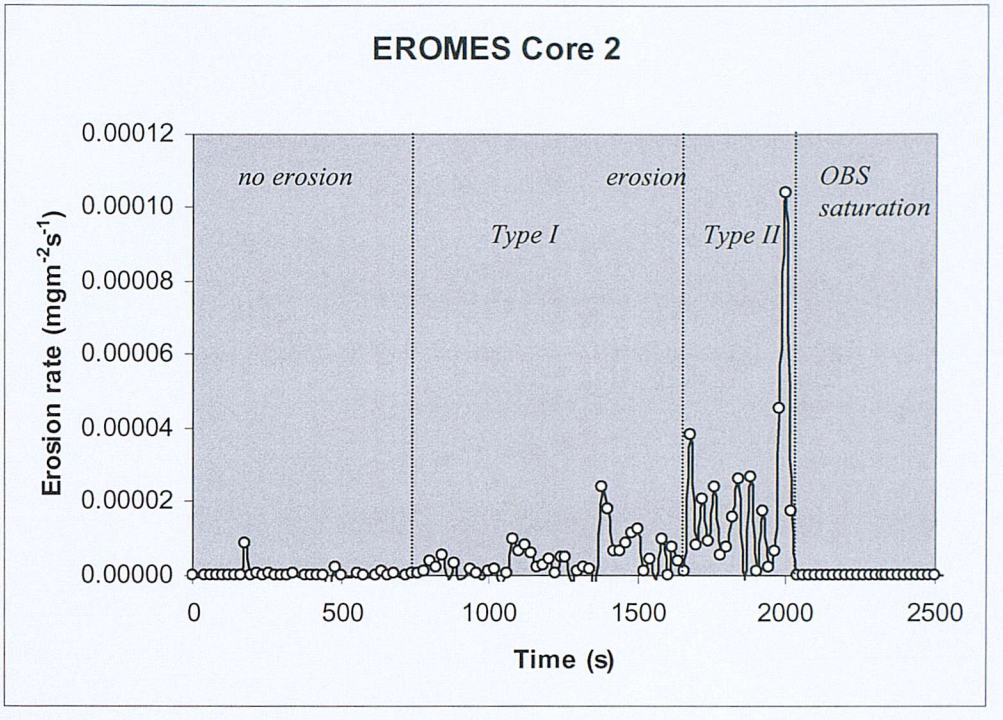


Figure 5.61

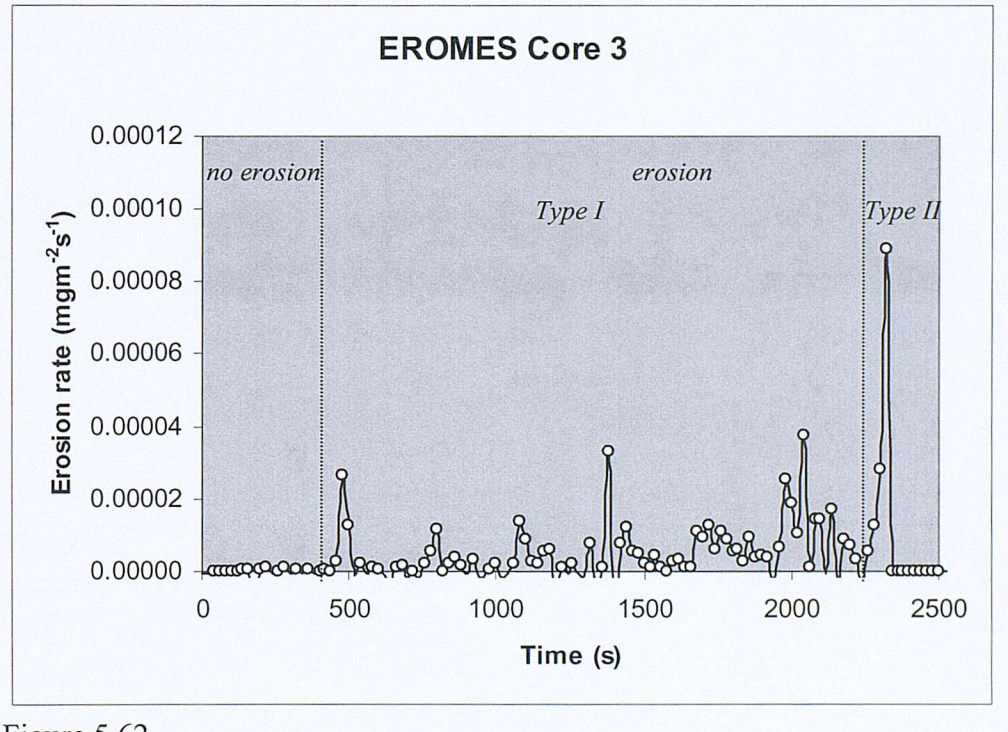


Figure 5.62

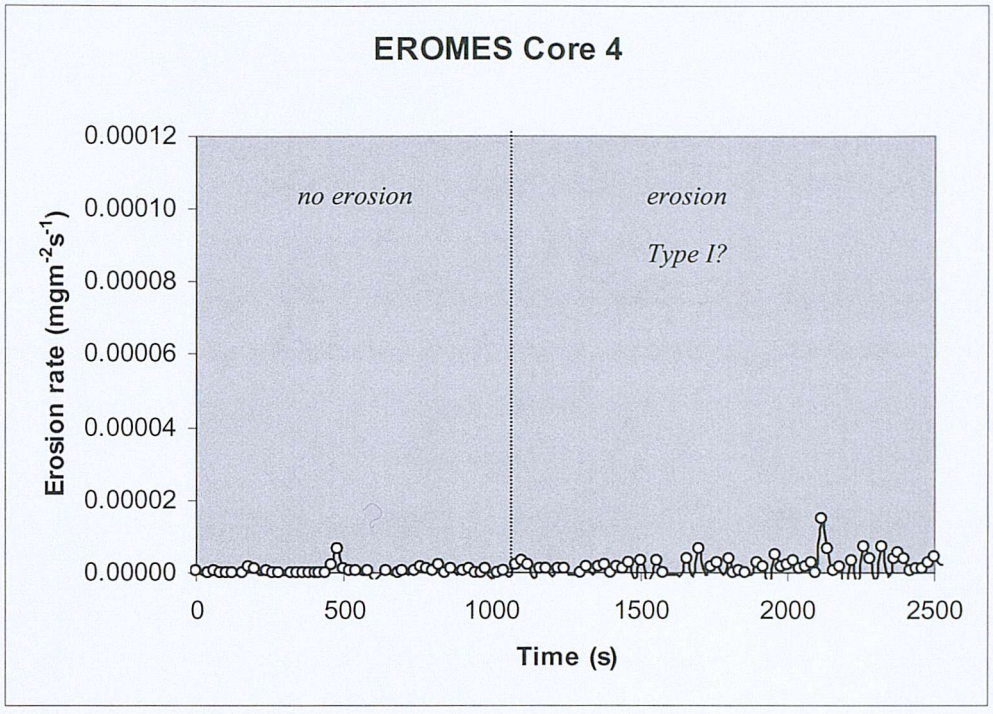


Figure 5.63

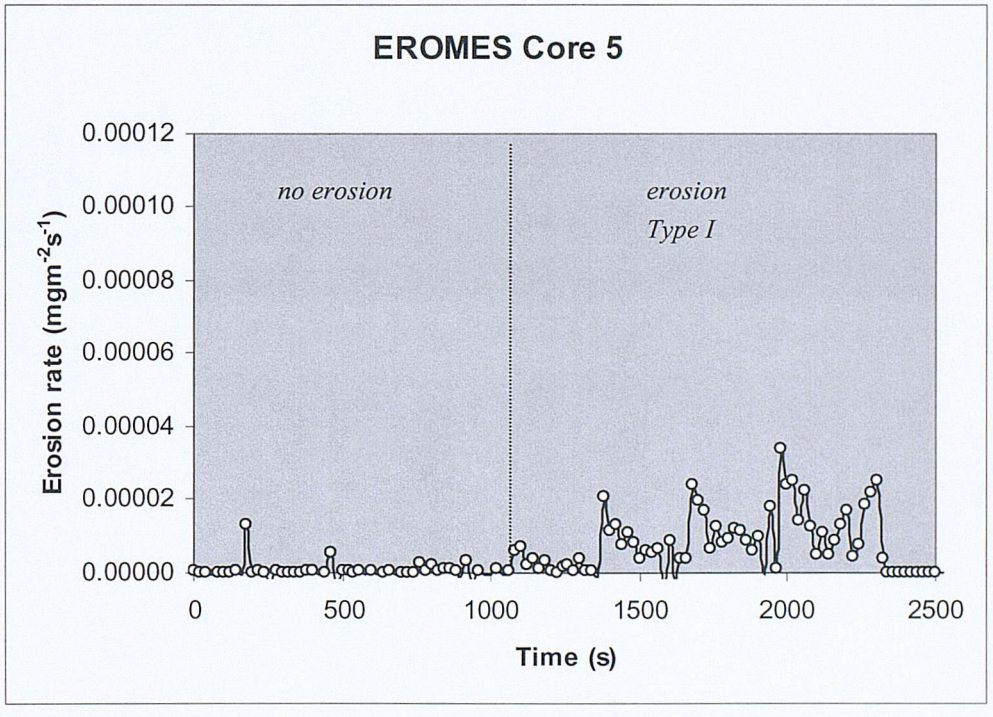


Figure 5.64

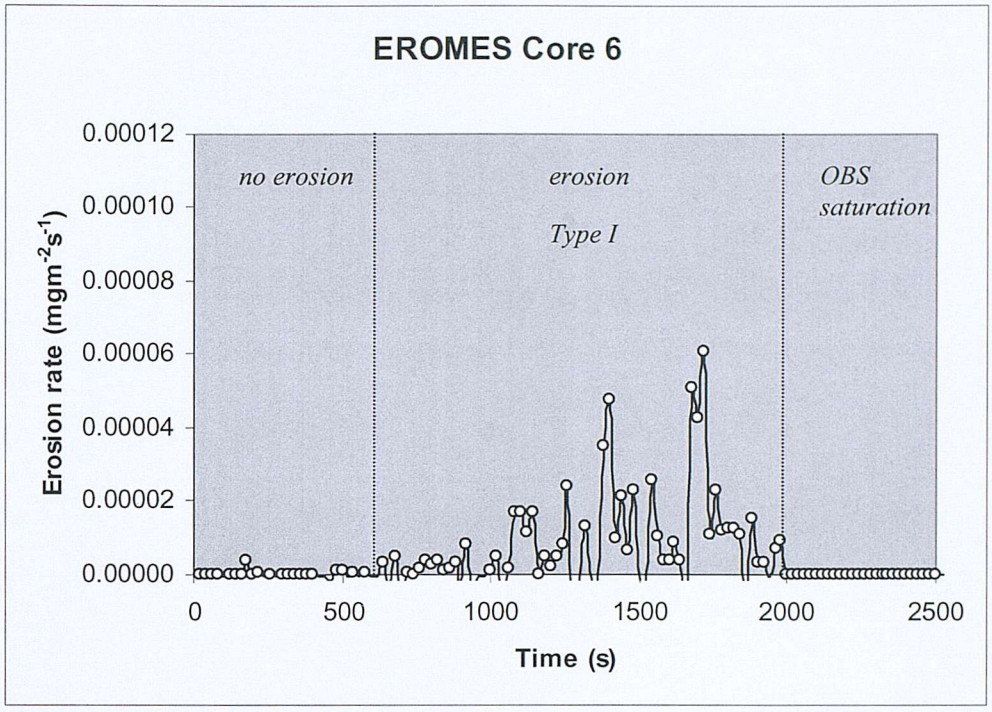


Figure 5.65

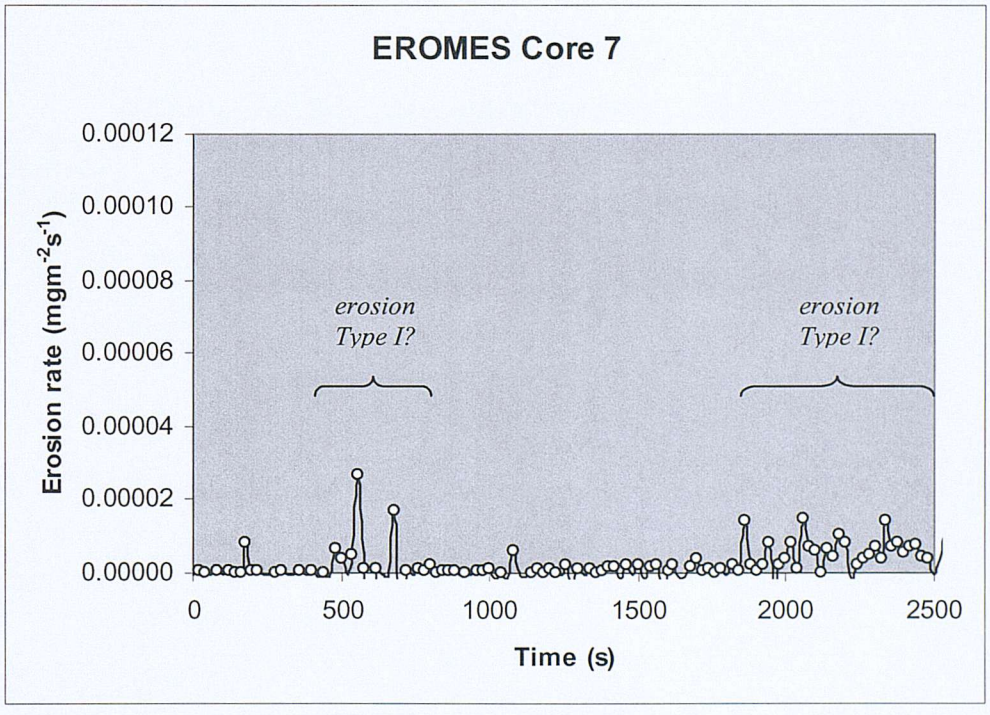


Figure 5.66

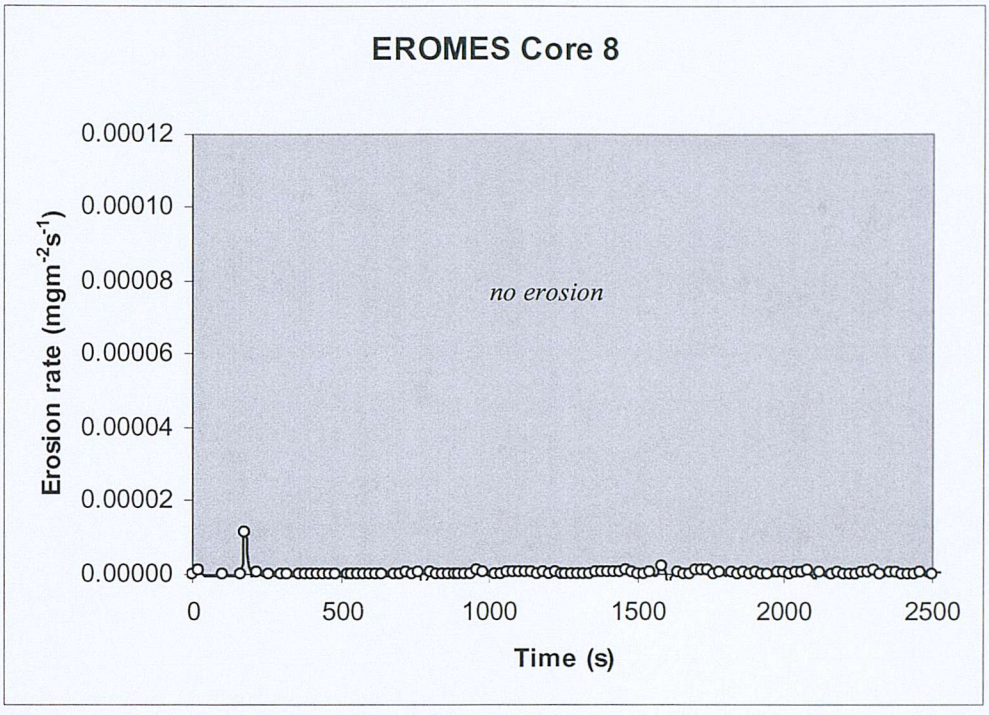


Figure 5.67

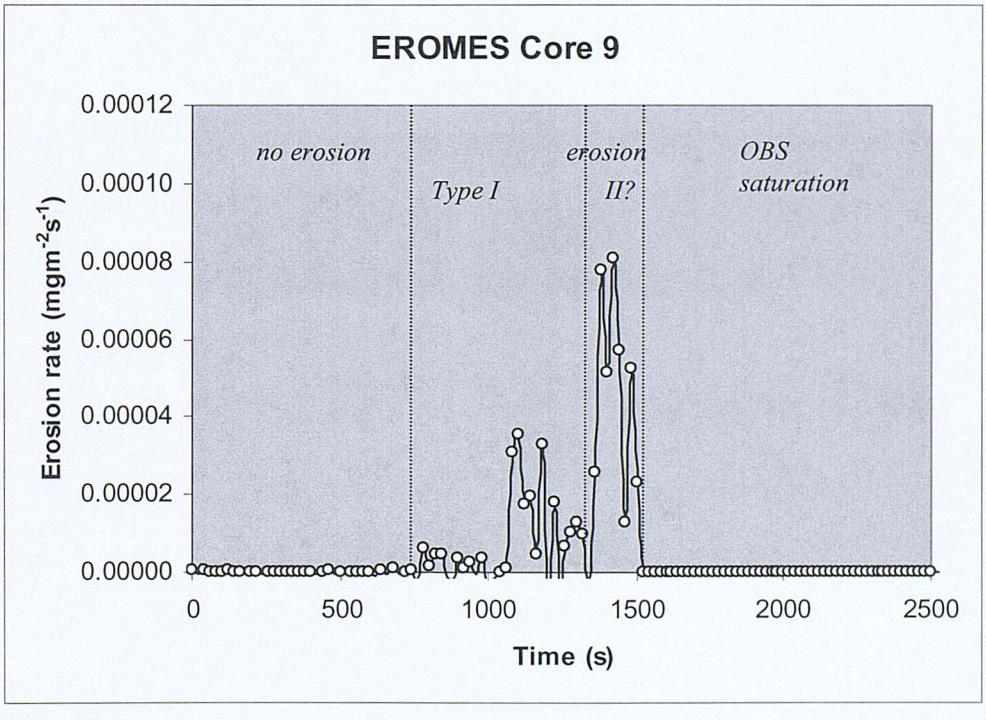


Figure 5.68

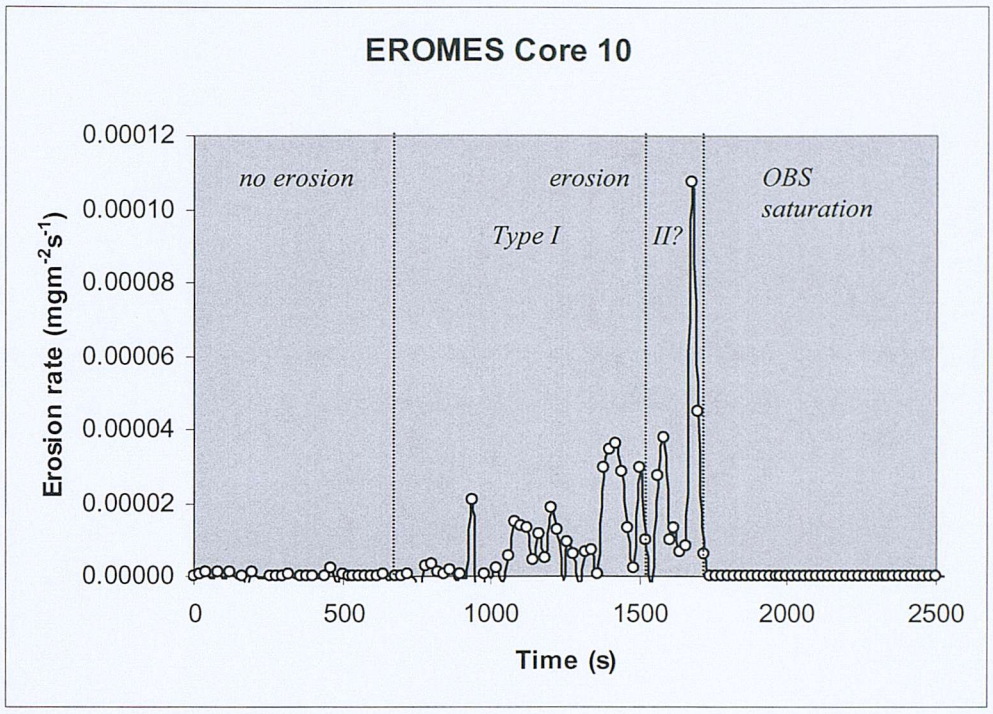


Figure 5.69

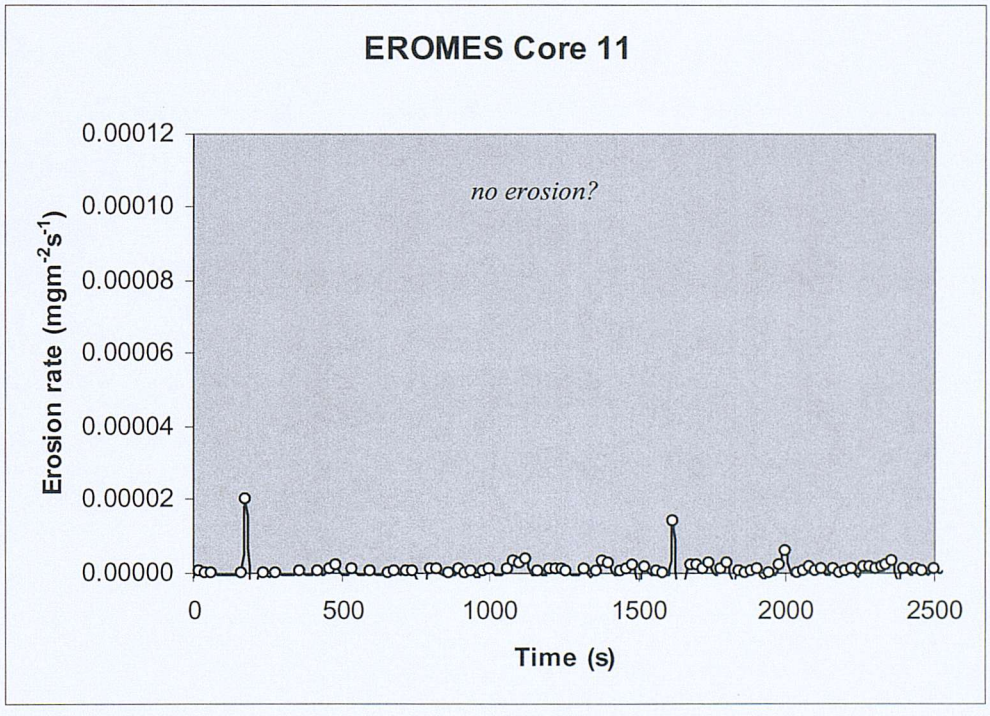


Figure 5.70

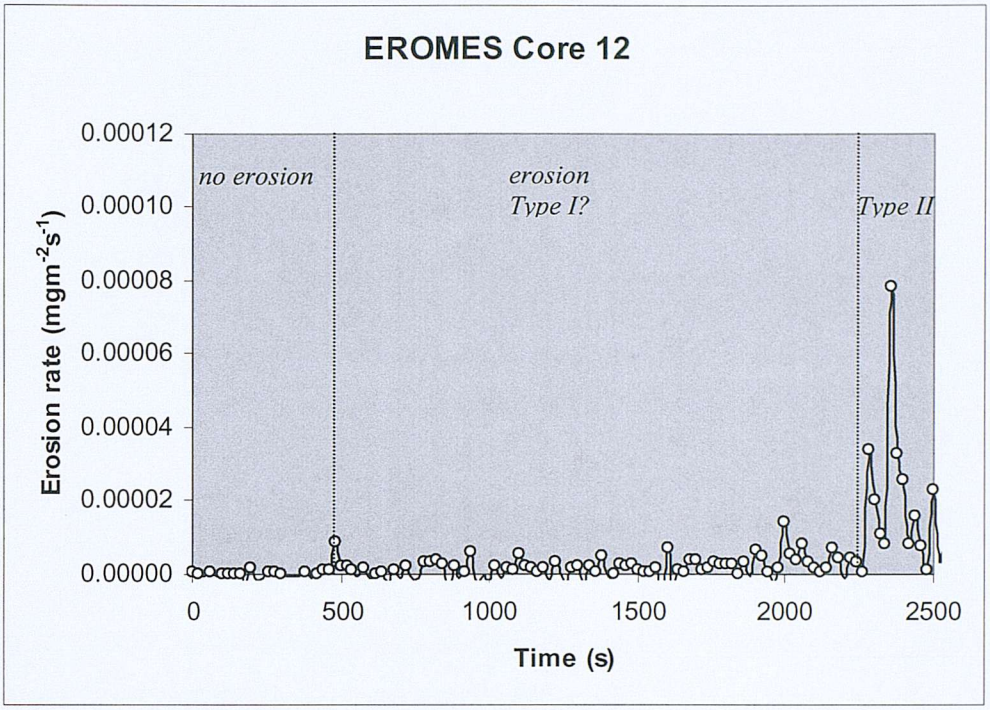


Figure 5.71

Mean erosion rates (E_m) were calculated from the EROMES data for each core. E_m and standard deviation were calculated from differences in suspended sediment concentration from the erosion phase ($\tau_0 > \tau_{cr}$) for each experiment (see methods, section 5.2.5.2 for details). The results are shown in Table 5.7 below; mean calculated values were found to range between 1.52×10^{-7} and 1.56×10^{-5} $mgm^{-2}s^{-1}$. This range of values illustrates the variation present between cores indicated in the time series plots. The relatively large associated standard deviations represent the variation in erosion rate present within a single experiment as a response to increasing shear stress and natural sediment variation.

Core ref.	E_m ($mgm^{-2}s^{-1}$)	Standard deviation
E1	9.14×10^{-7}	1.86×10^{-5}
E2	8.48×10^{-6}	1.53×10^{-5}
E3	5.80×10^{-6}	1.19×10^{-5}
E4	1.25×10^{-6}	2.53×10^{-6}
E5	9.17×10^{-6}	8.16×10^{-6}
E6	8.86×10^{-6}	1.43×10^{-5}
E7	2.14×10^{-6}	6.06×10^{-6}
E8	1.52×10^{-7}	6.14×10^{-7}
E9	1.56×10^{-5}	2.28×10^{-5}
E10	1.20×10^{-5}	1.90×10^{-5}
E11	7.61×10^{-7}	2.68×10^{-6}
E12	4.23×10^{-6}	9.79×10^{-6}

Table 5.7 Mean erosion rates calculated from EROMES data

5.3.9 Regression analysis of mean erosion rates on faunal density data

Figures 5.72 to 5.78 show the results of the regression analyses carried out between biological data (macrofauna densities) and mean erosion rate data (both treated and untreated cores). Exponential regression was found to provide a better fit than linear regression. Goodness of fit is shown on the graphs as R^2, for which values were obtained between 0.0021 and 0.6362 (confidence intervals for regressions are 95 %). The results suggest that a correlative relationship is present between faunal density and mean erosion rates. The highest correlation was found between *Caulleriella* density and mean erosion rate ($R^2 = 0.6362$). Similar R^2 values were obtained between total cirratulid density and mean E_m ($R^2 = 0.6237$), total polychaete density and mean E_m ($R^2 = 0.62$), total faunal density and mean E_m ($R^2 = 0.6182$) and spionid density and mean E_m ($R^2 = 0.6163$). No significant correlation was found between bivalve density and mean erosion rate ($R^2 = 0.0238$), although bivalves were present in the samples in relatively low numbers compared to the other taxa (which accounts for the stacked appearance of the data points). No correlation was found between motile polychaete density (*Nereis diversicolor* and *Nephtys hombergi*) and mean erosion rate ($R^2 = 0.0021$). These two species were also less abundant than other polychaetes. The data points which align to zero on the x-axis of the regression plots correspond to cores with zero counts for that faunal group (see also Table 5.3), possible reasons for this are discussed in section 5.4.8.

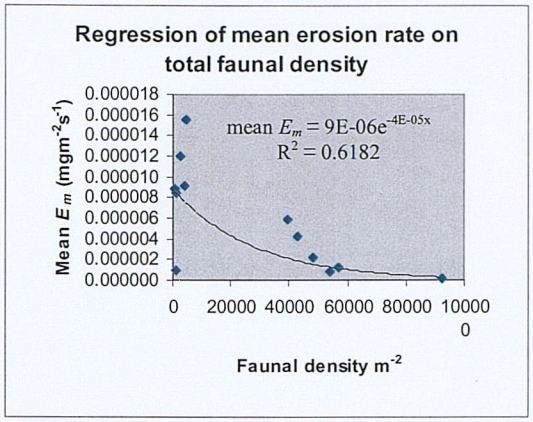


Figure 5.72

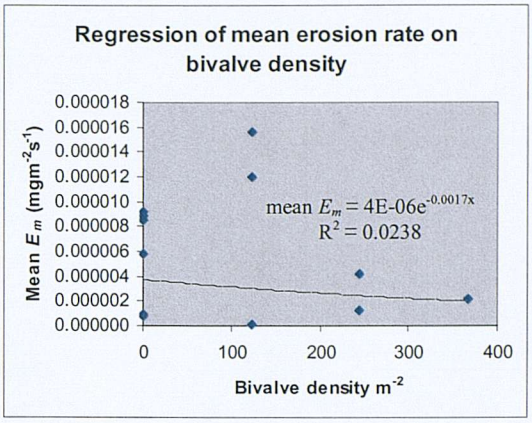


Figure 5.73

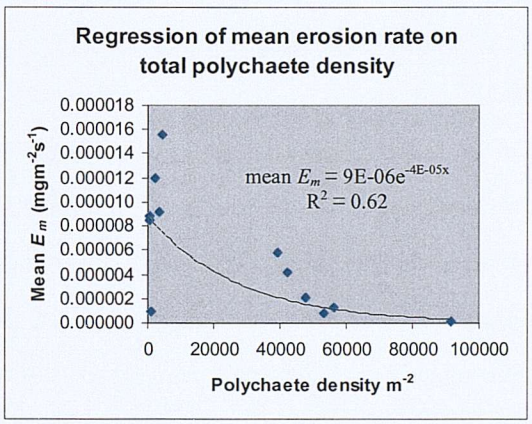


Figure 5.74

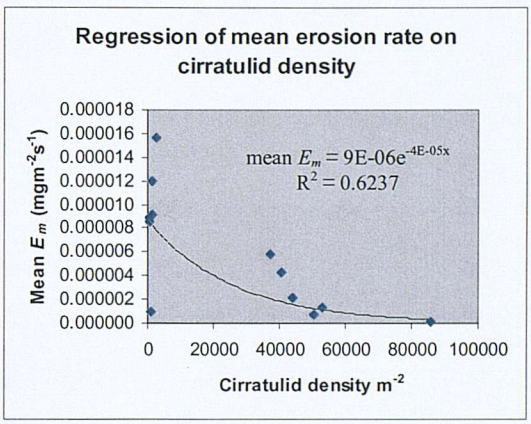


Figure 5.75

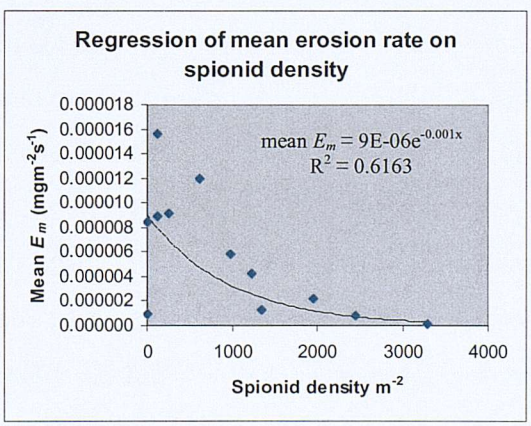


Figure 5.76

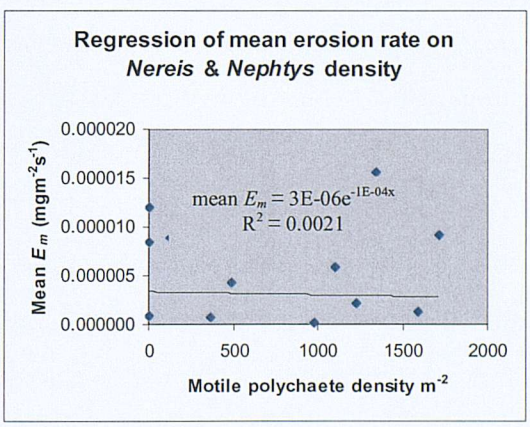


Figure 5.77

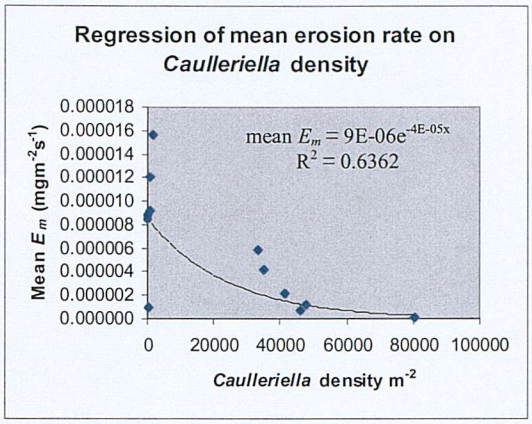


Figure 5.78

5.3.10 Results of Kruskal Wallis tests on mean erosion rates

The results of the statistical analyses of the mean erosion rates are given in Table 5.8. The results of the Kruskal Wallis tests imply that the relationship between faunal density and mean erosion rate is not statistically significant at a confidence level of 95 %. In contrast, the correlation between E_m and the treatment of the cores with formalin is highly statistically significant (P = 0.025). No significant relationship was indicated between the day of the experiment and E_m (P = 0.668).

Response variable	Source	Median	z	*DF*	*H*	*P*
Mean erosion rate ($mgm^{-2}s^{-1}$)	Day of experiment					
	1	$3.53x10^{-6}$	-0.68	2	0.81	0.668
	2	$5.50x10^{-6}$	-0.71			
	5	$8.11x10^{-6}$	0.85			
Mean erosion rate ($mgm^{-2}s^{-1}$)	Treatment					
	untreated	$9.02x10^{-6}$	2.24	1	5.03	0.025
	formalin treated	$1.70x10^{-6}$	-2.24			
Mean erosion rate ($mgm^{-2}s^{-1}$)	Total macrofauna density (individuals m^{-2})					
	610 - 92073	$8.86x10^{-6}$ – $1.52x10^{-7}$	-	11	11.00	0.443
Mean erosion rate ($mgm^{-2}s^{-1}$)	Total polychaete density (individuals m^{-2})					
	366 - 91341	$8.48x10^{-6}$ – $1.52x10^{-7}$	-	11	11.00	0.443
Mean erosion rate ($mgm^{-2}s^{-1}$)	Cirratulid density (individuals m^{-2})					
	366 - 85610	$8.67x10^{-6}$ – $1.52x10^{-7}$	-	10	10.96	0.361
Mean erosion rate ($mgm^{-2}s^{-1}$)	*Caulleriella* density (individuals m^{-2})					
	0 - 80366	$8.86x10^{-6}$ – $1.52x10^{-7}$	-	10	10.96	0.361
Mean erosion rate ($mgm^{-2}s^{-1}$)	Spionid density (individuals m^{-2})					
	0 - 3293	$4.70x10^{-6}$ – $1.52x10^{-7}$	-	9	9.69	0.376
Mean erosion rate ($mgm^{-2}s^{-1}$)	Mobile polychaete density(*Nereis* & *Nephtys*, individuals m^{-2})					
	0 - 1707	$8.48x10^{-6}$ – $9.17x10^{-6}$	-	9	8.49	0.486
Mean erosion rate ($mgm^{-2}s^{-1}$)	Total bivalve density (individuals m^{-2})					
	0 -366	$7.14x10^{-6}$ – $2.14x10^{-6}$	-	3	1.04	0.792

Table 5.8 Summary of results from Kruskal Wallis tests on erosion rate data. z = standard deviations from the mean, DF = degrees of freedom, H = an approximation of chi square distribution (with k-1 degrees of freedom, where k is the number of samples), P = level of significance.

5.3.11 Regression analysis of untreated and treated cores – mean erosion rates on faunal density

Regression analysis of mean erosion rates on faunal densities was also carried out for both untreated and formalin treated cores independently (Figures 5.79 – 5.86). These plots illustrate the nature of the effect of the formalin treatment on erosion rates and faunal densities. The curves obtained indicate that the trend between faunal density

and mean erosion rate was modified following treatment with formalin, with mean erosion rate positively correlated to faunal density in untreated cores, but negatively correlated following the treatment. The scales of the x-axes also vary between the data from treated and untreated cores. This is due to the consistently higher faunal densities recorded from the treated cores, which may reflect deterioration of the untreated cores as is discussed in section 5.4.8. The significant negative correlation between *Caulleriella* density and mean E_m ($R^2 = 0.9157$), and total faunal density and mean E_m ($R^2 = 0.9113$) was only apparent in the cores treated with formalin, corresponding R^2 values, 0.2864 and 0.4798, from the untreated cores were not significant (all confidence limits were 95 %). As before no relationship was found between bivalve density and mean erosion rates, and the results for total polychaete density and cirratulid density were extremely similar to those for total fauna and *Caulleriella* (as would be expected from the faunal density results and previous regression analyses).

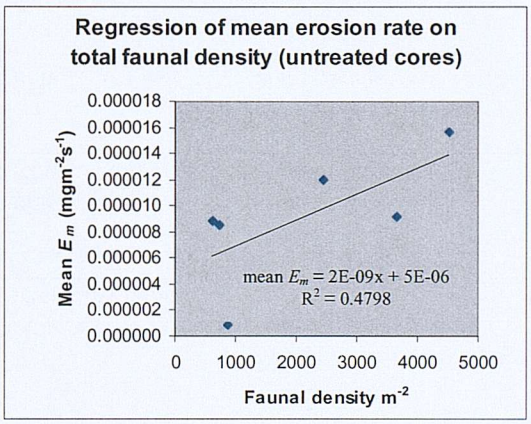


Figure 5.79

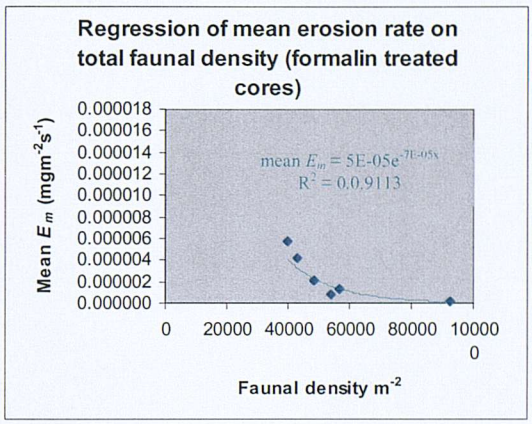


Figure 5.80

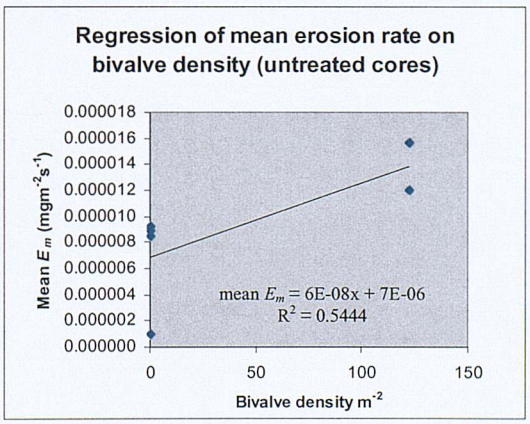


Figure 5.81

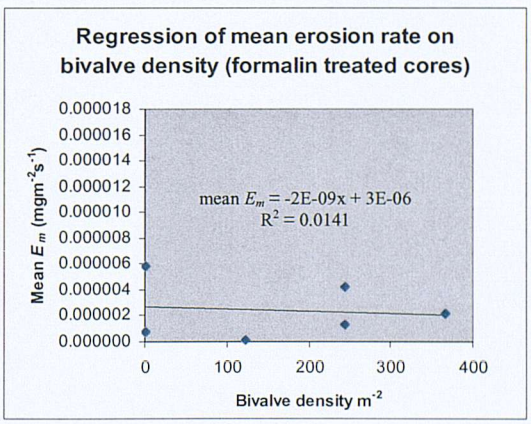


Figure 5.82

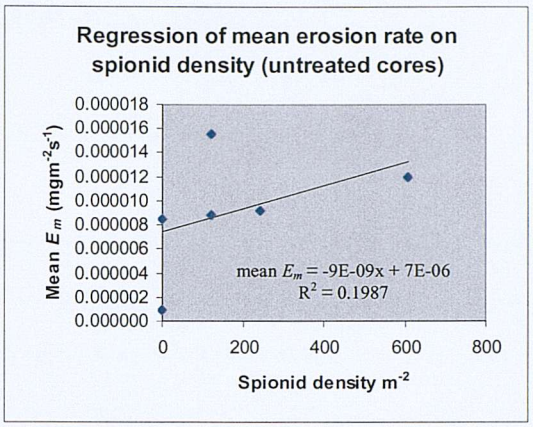


Figure 5.83

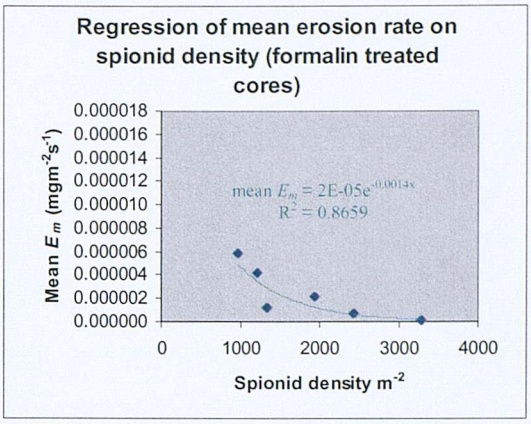


Figure 5.84

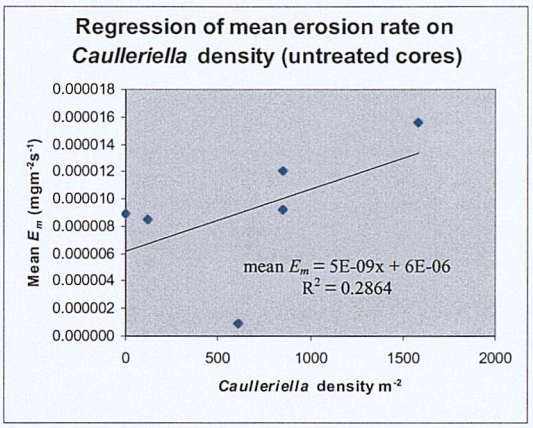


Figure 5.85

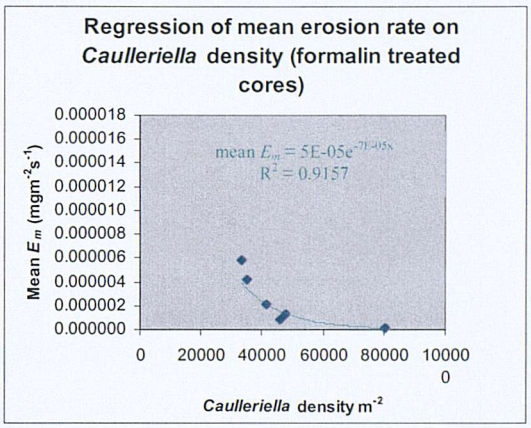


Figure 5.86

5.4 Discussion

5.4.1 Site factors

This study differs from many other previous studies (e.g. Black, 1996) of erosion properties of intertidal cohesive sediments; it is intensive rather than extensive in that it focuses upon one particular area of the intertidal flat. Typically a number of sampling sites are located in shore normal and/or shore parallel directions incorporating a range of elevation, tidal exposure and often significant changes in infauna community which accompany these gradients. These experiments were carried out on cores taken from site 1, Hythe (see Chapter 2 for map and location details), all of which were sampled from a localised part of the lower intertidal zone. This means that variations in the sediment due to changes in elevation, differing regimes of tidal exposure, and cross shore variation on the scale of metres and upwards were minimal. Amos & Mosher (1985) reported that tidal exposure with its associated desiccation by solar radiation and wind exposure can increase the shear strength of intertidal cohesive sediments by up to 80 times. As site 1 tends only to exposed during the lower range of spring tides, the effects of aerial exposure are expected to be minimal. Seasonal variability is also eliminated from this study as all the samples were collected at the same time (at the end of May). The limitations of performing such experiments *ex situ* were understood beforehand, and therefore it was not an aim of this study to define erosion characteristics of the site *a priori*, but to investigate the relative impacts of the infauna.

The lower, intertidal zone of mudflats is rarely studied so that there is little published data to compare the results of these experiments. Previous studies that looked at the effects of the infauna on sediment stability and erodibility in this way are extremely limited (see review, chapter 1), and nothing has been published on the dominant taxa at the site.

The relative lack of tidal exposure at site 1, Hythe also has several implications for sediment properties; primarily a lack of the stabilization processes typically associated with aerial exposure (Anderson & Howell, 1984; Paterson *et al.*, 1990). A second factor of importance is that diatom biofilms were not visible as far downshore as the sampling site. Consequently it is thought that their development is inhibited due to the lack of light available during the short, infrequent exposed periods, and relatively high

turbidity of the water column (Austen *et al.*, 1999). Taxa which are known grazers of microphytobenthos, such as *Hydrobia ulvae* and *Corophium volutator*, have only been recorded seasonally or periodically at low densities at the site or were absent.

5.4.2 Erosion threshold data

The erosion profiles plotted from the data obtained using the CSM and used to estimate τ_c by regression for these cores indicate variation both between values, and in the shape of the profiles. The range of values obtained is relatively close (as may be expected for samples from the same site), although some heterogeneity is implied. This is likely to reflect natural variation in the sediment such as localised heterogeneity in water content, bulk density, biological structures and biological activity. The data are compared to that from other studies below.

The correlation coefficient obtained for the scatter plots for the calibration of the EROMES OBS output to SSC (Figure 5.27) ($R^2 = 0.7845$) indicates departure from a perfect linear relationship. The accuracy of the relationship also appeared to decline with suspended sediment concentrations greater than 400 mgl^{-1} (indicated by increased scatter). Accuracy may be increased for this type of calibration by increasing the frequency of water samples taken (i.e. from each step of the program for every core), although some heterogeneity would be unavoidable.

The times series plots of the EROMES data were annotated to indicate regions of no erosion, erosion and still water settling in relation to the program and stepped increments of increasing bed shear stress. It can be seen that most of the cores displayed both Type I and Type II erosion (Amos *et al.*, 1992). Type I erosion commonly occurs in the upper few millimetres of the bed and corresponds to what is often known as the 'fluff layer' (= the layer of loose unconsolidated material at the surface). This type of erosion is sometimes termed as benign, as it diminishes exponentially over time. The asymptotic curves characteristic of Type I erosion are evident in the graphs, and it can be seen how the next 'onset' of erosion corresponds to the increase in shear stress. Type II erosion is also indicated in the plots where the increase in SSC rises unchecked to peak at the maximum shear stress. In many of these plots this can be seen as a plateau where the upper limit of the OBS was reached.

Type II erosion, which is sometimes termed as 'chronic erosion' is indicative of the erosion threshold being exceeded for deeper, more consolidated parts of the bed.

The erosion profiles plotted from the EROMES data were also used to estimate τ_c values by regression, and to obtain values of the gradient or slope of the profile. The EROMES τ_c values were found to be slightly higher than those obtained from the CSM (Tables 5.1 and 5.2). Possible reasons include differences in small-scale topography and features, such as shell debris, and instrumentation error.

Day (2000) includes measurements of erosion parameters made on cores collected from the mudflat at Hythe using the CSM and an earlier version of EROMES. Values for critical erosion thresholds of 0.27 ($\pm$ 0.20) Pa and 0.49 Pa were obtained for the slightly higher low shore site (around MLWN) from EROMES and the CSM respectively. Compared to the results of this study, the range of EROMES values are very similar (0.10 – 0.55 Pa). The CSM values obtained from these experiments were considerably lower (< 0.21 Pa). This difference may be due to a number of factors including sampling error or instrument calibration. However, it may reflect the localised difference in elevation and therefore tidal exposure times between the two sites. Overall, however, the τ_c values obtained during these experiments fall within ranges previously reported from cohesive intertidal sediments for other locations (e.g. Amos *et al.*, 1999; Austen *et al.*, 1999; Cappucci *et al.*, 2000; Friend *et al.*, 1999; Tolhurst *et al.*, 1999; Tolhurst *et al.*, 2000; Widdows *et al.*, 1998), indicating that they are fairly typical or representative for this type of sediment.

5.4.3 Faunal density and erosion threshold

No correlative relationship between critical thresholds for erosion (τ_c) and faunal densities were indicated in the data, suggesting that such a relationship did not exist. τ_c is a measure of the erosive strength or properties of the sediment surface. It seems likely therefore that the impacts of the infauna on the sediment structure would be more evident sub-surface, particularly as surface 'features' such as shell debris (often in the form or disarticulated and whole *Cerastoderma* shells), stones and small tufts of filamentous algae may dominate the surface characteristics and mask the effects of the infauna at this level. This was also observed by Grant & Daborn (1994) who found

that sub-surface measures of sediment erosion properties such as erosion rate showed better correlation with macrofauna data. The lack of correlation may also partially arise from laboratory effects that are discussed below.

The regression analysis indicated a weak correlation between bivalve density and erosion threshold within a confidence limit of 90 %. Bivalve densities were extremely low and bivalves were absent from many of the samples, therefore it is possible that a significant relationship may exist at higher densities. In a previous study using annular flumes, Widdows *et al.*, (2000) found that dense populations of the clam, *Macoma balthica*, had a destabilising effect on the sediment through bioturbation. They found that low erosion thresholds were associated with high densities of clams, and high erosion thresholds were associated with low densities of clams and highly developed algal biofilms.

Austen *et al.* (1999) found that τ_c was correlated to the abundance of deposit feeders. However, it appeared to be mainly controlled by the relationship between algal biomass (expressed as chlorophyll *a* content) and the abundance of deposit feeders; in this case *Hydrobia ulvae*, a grazer whose main diet consists of diatoms. Additionally they found that the snails' faecal pellets were more easily eroded than the cohesive bed. It is thought that *Hydrobia* occurred too infrequently at the Hythe site to detect such an influence.

Meadows *et al.* (1990) and Meadows & Tait (1989) reported that secretions produced by the amphipod *Corophium volutator*, and the ragworm *Nereis diversicolor* stabilised the sediments. However, that study was carried out on sandy sediments, and it is possible that these effects would not be detectable in mud due to the cohesive nature of the sediment or that the secretions are reduced or unnecessary for animals in such cohesive sediment. It is also likely that any net effect of *Nereis* on τ_c was not detected in these experiments due to the relatively low density of this species at the site.

5.4.4 Friction coefficient (ϕ) and faunal density

The slope or gradient of the erosion profiles (Figures 5.40 – 5.50) is representative of the friction coefficient (ϕ), which is a parameter used to relate to consolidation processes within the bed. Increased values of ϕ (i.e. steeper slopes) indicate increased consolidation and therefore potentially increased shear strength and bulk density. Amos *et al.*, (in press) found that measurements of ϕ that increased in a linear fashion with increasing strength were diagnostic of self-weight consolidation. In contrast, sub-surface layers of the bed where $\phi = 0$ are diagnostic of no consolidation with depth which may reflect the presence of bioturbation.

The results from this study suggest correlation between erosion profile gradient (as a measure of ϕ) and faunal density. Cores E4 and E12 gave high values for faunal density and the lowest values for profile gradient, whereas for cores E6, E9 and E10, the highest gradients corresponded to the lowest faunal densities. This implies that less self-weight consolidation has occurred in the samples containing the most fauna and is indicative of bioturbation. Amos *et al.* (in press) conclude that 'ϕ appears to be an important factor in determining the erosion of a bed, yet is complex in its vertical structure and thus difficult to define'. Further work needs to be undertaken but the potential exists for this parameter to be used as an indicator of relative bioturbation in quantifying the effects of infaunal populations on bed properties.

5.4.5 Erosion rates

The variation in erosion rates between the twelve cores can be attributed to localised heterogeneity within the sediment (on a scale of millimetres to centimetres). This may be due to many factors; physical, sedimentological, hydrodynamic and biological, and it is this non-uniformity which makes prediction of the behaviour and transport of cohesive intertidal and estuarine sediments so difficult. Factors known to affect the stability/erodibility of cohesive beds also include fluxes in sediment deposition rate and consolidation rates including seasonally induced variation, tidal and hydrodynamic variation in sediment transport and those caused by the burrowing and biodepositional activities of fauna.

Differences between the cores may also be an effect of the treatment with formalin solution: With the exception of core 3, the formalin treated cores (cores 4, 7, 8, 11 and 12, see Figures 5.60 – 5.71) were found to have lower erosion rates overall (see also discussion below). The application of non-parametric statistical (Kruskal Wallis) tests confirmed that these differences were statistically significant, and were therefore likely to reflect real differences in the rate of erosion between the treated and untreated cores. The question arises therefore, as to whether this effect of suppression of erosion rates was due to a direct chemical effect on the sediment fabric, or whether another factor was involved. The results of the regression analysis (with the biological data) suggest that the treatment may have affected erosion rates through biological interaction. The implications of this are addressed below.

Type I erosion can be seen in the erosion rates time series plots for cores 2, 3, 4, 5, 6, 7, 8, 11 and 12, characterised by a peak in erosion rate near the beginning of the experiment which then rapidly drops back to zero or thereabouts.
Cohesive sediments are found to exist in the form of laminae (Amos *et al.*, 1992). These may be described as layers of sediment interdispersed with weaker fracture zones or interfaces. This type of structure can account for the series of peaks and troughs produced in the time series plots, as the weaker layers are eroded more rapidly than those in between which exhibit higher shear strength. The shape of the graphs, or series of peaks and troughs forming the erosion rate plots are also due to the form of bed shear stress applied during the experiments. The eroding pressure, and therefore shear stress, was applied stepwise with incremental increases which may be expected to result in concurrently deeper layers of sediment eroding rapidly as the shear stress is increased, followed by a subsequent fall in erosion rates until the τ_{cr} for the next layer is exceeded by the next increase in shear stress.

A limitation of these experiments was the inability of the OBS to detect increases in SSC beyond around 1.5 gl^{-1}. This meant that it was not possible to calculate erosion rates for the latter phase of several experiments once this limit had been reached. This characteristic of the instrument is a reflection of its being designed for measuring the onset of erosion and surface erosion, and the results suggest that it may be unsuitable for recording subsurface erosion, and erosion of deeper layers of a sediment bed. A

suitable alternative instrument, which would overcome this problem, is not known of at this time.

Mean erosion rates were calculated from the data in order to provide a one-point estimate to analyse together with other parameters such as faunal density (see regression analysis). The relatively large standard deviations quoted reflect the wide spread and variance of the data which was indicated in the time series plots and is believed to reflect natural variability in the sediment due to the factors previously discussed.

5.4.6 Erosion rates and faunal density

The results of the regression analyses carried out on the data from all twelve EROMES cores (both formalin treated and untreated), implied a weak correlation between faunal density and erosion rates, with R^2 values of between 0.62 for total faunal density and 0.64 for *Caulleriella* density. The shape of the curve denotes an inverse correlation which suggests that E_m may decrease in response to increasing faunal density. No relationship was found between bivalves and E_m ($R^2 = 0.02$) or between the mobile polychaetes (*Nereis diversicolor.* and *Nephtys hombergi*) and E_m. Taken independently this would imply that the latter two genera of polychaetes and bivalves present have no effect on erosion rates (possibly due to low density), whereas the more numerically dominant, relatively sessile cirratulids may be responsible for a reduction in erosion rate through biostabilisation. However, the non-parametric analysis of variance (Kruskal Wallis tests) carried out on this data (see Table 5.5) indicates that the effect of the treatment of half of the cores with formalin was highly significant as a factor influencing erosion rates ($P = 0.025$), and that faunal density as a factor was statistically weak.

Regression analysis was therefore repeated on the treated and untreated cores independently to identify the effects of the treatment on erosion rates: The results of the regression analysis of the untreated cores indicated little correlation. In contrast with the previous results, the highest R^2 value was obtained from bivalve density ($R^2 = 0.54$), the direction of the trendline suggesting a weak trend of destabilisation. Values for total faunal density and *Caulleriella* were 0.48 and 0.29 respectively with no clear trend suggested. The results from the regression of treated cores were, however, in

complete contrast: strong correlation was indicated with R^2 values up to 0.92 (for *Caulleriella*), which appear to indicate increasing erosion rates with increasing faunal density suggestive of a destabilising effect. No correlation was found between E_m and bivalve density ($R^2 = 0.01$).

It is possible that the lack of relationship indicated between the density of bivalves (largely *Cerastoderma* spp.) and erosion rates may be due to the low population density of these organisms at the site. For example, Widdows *et al.* (1998) concluded that the erodiblity of natural intertidal sediments from the Humber Estuary was related to the population density and bioturbation of *Cerastoderma edule* and the clam *Macoma balthica*. Furthermore, in a subsequent study Widdows *et al.*, (2000) reported that long-term changes in sediment erodibility (also in the Humber Estuary) correlated with the density of *Macoma.*

The implications of the results of the analysis between the cirratulid polychaetes and erosion rates appear therefore to depend upon the nature of the effects of the formalin treatment on the cores. It appears that an effect of the formalin was to act as a preservative on the biota and sediments, and there is some evidence to suggest that the untreated cores underwent deterioration in the laboratory (as was implied by visual observations during processing, see section 5.4.9 for further discussion). Therefore, as it is probable that the lack of a clear trend from the untreated cores may be due to deterioration of the cores, and the clear correlation between faunal density and erosion rates in the treated cores was obtained from dead fauna, no conclusion can be made as to the nature of the relationship between faunal density and erosion rates.

Previous studies indicate that although modification of sediment properties by infaunal organisms is almost universal, accurate prediction of the type/quantification of effect, without site specific experimental evidence is extremely difficult. The same species of infauna has been found to confer either stabilisation or destabilisation depending upon population density, community structure, and available resources and feeding mode. For example, the amphipod, *Corophium volutator*, has been found to both stabilise sediments by increasing their shear strength through secretions which bind the sediment together, and cause destabilisation through the grazing of diatom biofilms (Grant & Daborn, 1994; Meadows *et al.*, 1988). *Nereis diversicolor*, has also been

reported to stabilise sediments by increasing the shear strength of muddy sand through mucoid secretions (Meadows & Tait, 1989; Meadows *et al.*, 1990). However, this effect was found to be linked to population density, and with the relatively low densities of this species recorded from Hythe (see Table 5.3, this chapter, and Chapter 2) it seems unlikely this factor would play an important role at this site. It can only be concluded therefore that the erosion rates (in contrast to erosion thresholds) at the site are affected, and probably modified by the activities of infauna, and that the relationship is likely to be density dependant. It should also be noted that, despite the previous success reported by Faas *et al.* (1993) and Grant & Daborn (1994), the use of formalin as a biocide in studies which aim to determine faunal effects upon bed stability and erodibility is not recommended due to the unclear nature of its effects upon the sediment. This study also highlights the difficulties in undertaking such investigations in the laboratory, and emphasises the desirability of *in situ* field experiments wherever possible.

5.4.7 Instrument comparison

Significant differences in the data were found to have arisen from differences between the two instruments used to measure erosion properties. One major difference between EROMES and CSM is the methods they employ to erode the sediment: whilst EROMES uses a motor driven propeller to apply horizontal shear stress to the bed, the CSM applies pressure in the form of a vertical water jet. Although much effort has been made to equilibrate jet pressure to horizontal bed shear stress through calibration and empirical relationships (see Tolhurst *et al.*, 1999 for full details), there is still some question as to the absolute accuracy and reliability of this method, and it seems likely to be a source for error, particularly when comparing data obtained by the CSM to that from other instruments.

Further differences exist between the two instruments as variation in optical thresholds, and definition of the onset of erosion. EROMES measures turbidity using an OBS, and the onset of erosion is often defined as 'a constant increase in SSC', whilst the CSM data is analysed by taking a user-defined decrease in percentage light transmission (e.g. a 10% reduction) as the onset of erosion. Tolhurst (2000) reported that EROMES gave slightly higher values for τ_c than the CSM at equivalent horizontal

bed shear stresses below 0.5 Pa, and suggests that this was due to differences in the optical thresholds. It is therefore worth noting that 37 of the 38 τ_c values obtained from these experiments corresponded to shear stress values of less than 0.5 Pa. It is therefore possible that this accounts for much of the variation seen in this study, and that if further samples with a higher range of τ_c values (for example, cores taken from higher up the shore where typically higher erosion thresholds have been reported (Day, 2000; Quaresma *et al.*, 2002), above this limit were eroded the results from the two instruments may compare more favourably.

Aside from the differences discussed above, EROMES and the CSM also have distinctly different footprints (0.0082 m^2 cf. 0.00066 m^2 respectively). This means that the bed is eroded and data collected at different spatial scales, and that small scale topology and features such as shell debris which are by necessity avoided by the CSM are incorporated into the experiments carried out with EROMES. Thus the inclusion of areas of increased shear strength such as stones and empty shells may increase the overall estimation of surface erosion threshold compared to the CSM derived values. During these experiments, where higher values of τ_c were obtained from EROMES, visual observations of the core surfaces were made, and it was noted that the cores used for EROMES did in fact include shell debris along with small patches (several millimetres to a cm in diameter) of filamentous algae. Although more difficult to define and impossible to quantify here, this difference of scale may also have implications for biological sediment structures such as localised areas of sediment destabilisation bordering localised compaction of sediment around animal burrows or faecal pellets.

These aspects of scale and topological features may also be expected to affect the roughness length, in particular for the EROMES cores. EROMES had been calibrated for a low roughness, however, the inclusion of features such as shell debris or stones on the sediment surface would have the effect of increasing the roughness, which would not have been compensated for in the calibration (Thompson *et al.*, in press). This therefore represents an additional potential source of instrument inaccuracy in the data.

Instrument calibration more generally may also introduce a source of potential error, particularly where data from two or more instruments with different calibration procedures are involved. The portable version of EROMES used during these experiments was eventually calibrated using abiotic quartz sand and the modified Shields curves (Quaresma *et al.*, 2002). This technique is not ideal when using the instrument for cohesive sediment, but proved to be the only realistic method of calibration available. Slight error apparently arising from this factor was evident during data processing when several values for bed shear stress (at the lower end of the spectrum) were calculated as negative. Day (2000), when using an earlier prototype of EROMES, also failed to find significant correlation between EROMES and CSM data. Interestingly, in contrast to the results from this study the author reported a higher mean value (0.49 Pa) obtained from the CSM than EROMES (0.27 ± 0.20 Pa) for a lower shore site at Hythe. One possible explanation would be the non-random selection of cores from well-drained ridge features necessitated by the CSM (to prevent disturbance to surface sediments during sampling and transport), whereas the larger footprint of EROMES necessitates the inclusion of both ridge and trough sediments within a sample.

It is worth noting here that both the CSM and EROMES were designed for the measurement of surface erosion, in particular the initial onset of erosion, and that the investigation of erosive properties of deeper layers of the bed and subsurface erosion may be more appropriate where the activities of benthic infauna are in question. The OBS of EROMES in particular proved unsuitable for the measurement of sediment concentrations above 1.45 gl^{-1}.

5.4.8 Experimental effects of formalin treatment

The aim of the formalin treatment was to kill all macrofauna in the treated cores and provide a comparison between untreated cores (to approximate normal faunal activity) and treated cores (faunal activity inhibited). This method was previously used with some success in the field and laboratory flume studies by Faas *et al.* (1993) and Grant & Daborn (1994). However, the results of the statistical analyses from this study indicate that the formalin affected the erodibility of the sediment. A significant relationship was found to exist between the formalin treatment and mean erosion rates

(Table 5.6, P = 0.025). However, the results indicated that the treatment of half the cores with formalin did not significantly affect the erosion threshold of the surface sediment either directly by altering sediment properties, or indirectly by its affects on biological activities within the sediment (Table 5.4, P = 0.173).

These results imply that the erosion properties of the subsurface sediments were affected by the formalin, and may be a reflection of the effects of the treatment/non-treatment on the fauna (see section 5.4.9 for further discussion), or an indication of modification of sediment properties/structure chemically. For example, it is well established that EPS (extra polymeric substances) produced by infauna during tube construction and burrowing and microalgae such as diatoms, acts to stabilise cohesive sediments, and it is possible that the formalin affected sediment erodibility through interaction with EPS. As it was beyond the scope of these experiments to directly measure EPS future studies would be desirable to address this issue.

5.4.9 Time and laboratory effects

Results indicate that laboratory effects were present, which were probably caused by keeping the cores in the laboratory during the experiments. The biological count data and appearance of the cores indicated that, whilst the application of formalin would have killed the infauna and prevented further activity (and therefore modification of sediment properties) in the treated cores, it also acted as a preservative. The significantly lower counts taken from many of the untreated cores, especially those eroded at the end of the experiment, and appearance of some of the cores when processed indicated that biological activity also departed from 'normal' conditions in the untreated cores, and that die-off and decomposition of infauna had occurred.

The most likely explanation for the lower faunal counts in the untreated cores is that deterioration of the samples had occurred due to laboratory conditions. Whilst measures were taken to minimise disturbance, and temperature fluctuation, the latter in particular may have proved inadequate as the experiments were carried out at the end of May/beginning of June when air temperatures were warm. The initial temperature of the setup was 17 $^{\circ}$C and although subsequent checks did not show great departure from this, constant monitoring was not possible or practical, and diurnal fluctuations extreme enough to inhibit the fauna may have occurred.

A second possible explanation for low faunal counts could be that individuals were swept out of the bed during the erosion sequence. However, as a still water settling period was included at the end of each experimental run with EROMES, and great care was taken to ensure that nothing was 'lost' from the area immediately above the core surface for processing this seems improbable. As the untreated cores were immediately transferred to sealed containers, containing 10 % formalin at the end of each experiment and disaggregated for storage prior to processing, it is unlikely that deterioration would have occurred at this stage.

It was necessary to carry out the erosion experiments over a period of several days in the laboratory due to the length of time needed to complete each run. A correlation between laboratory incubation time (day of experiment) and faunal density was identified in the PCA. However, incubation time did not appear to have any detectable effect on the erosion threshold of the sediment as was confirmed by the Kruskal Wallis tests ($P = 0.912$). Therefore it is concluded that the main effect of keeping the samples in the laboratory system would have been on the fauna and not directly on sediment properties examined here (potential disturbances to the cores during sampling and transportation are discussed below).

The cores were kept covered by seawater once in the laboratory and only uncovered for the erosion experiments. It is believed that this would have caused relatively little departure from 'normal' water content of the sediment due to the infrequent and limited exposure and drainage at the site. In fact, this may have minimised factors induced by unnatural or prolonged exposure to the advantage of the experiment: Widdows *et al.*, (1998) concluded that, dependent on the organisms present, prolonged exposure may result in increased EPS or a reduction in bioturbation secretion which may positively affect the erosion threshold.

A direct comparison of laboratory /field results is not possible here, as it was not practical to collect *in situ* erosion data. However, this was undertaken by Tolhurst (2000), in a study to compare field and laboratory measurements of erosion properties of intertidal sediments using CSM and another version of EROMES. The author reported that box cores from the Humber Estuary, transported back to the laboratory with no special measures of precaution against disturbance suffered visible disruption.

Their erosion thresholds were found to be considerably increased, either through physical disturbance (vibration, compaction and water loss) or changes in the behaviour of infaunal organisms following excavation (Tolhurst, 2000). However, the author reported that, when disturbance was minimised due to careful transportation and handling, laboratory and *in situ* erosion thresholds were comparable.

The cores used for these experiments were handled and transported following Tolhurst's suggestions to minimise disturbance. As the top surface of the cores were transported without overlying water to prevent this creating an erosive force, it was important to minimise desiccation. In this case, the journey was relatively short and the 'exposure' would not be expected to greatly exceed natural tidal exposure that day. The cores were inundated as soon as they were in the laboratory to represent site conditions as closely as possible, and may have helped to prevent excessive production or accumulation of EPS or a reduction in bioturbation due to prolonged drying (Widdows *et al.* 1998).

Thus it may be concluded that there is still no reliable way of gaining field representative erosion parameters for intertidal cohesive sediments in the laboratory, although errors introduced through sampling and transport can largely be minimised. Laboratory conditions proved to be more difficult to maintain within such narrow limits and it is believed that they inhibited the fauna during this study. It is recommended that, to minimise these effects, similar future experiments should be carried out *in situ* wherever possible, or at least in closely controlled micro- or mesocosm systems where this is not possible.

5.5 Summary and Conclusions

Measurements of sediment erodibility were made on cores using two instruments; EROMES and the CSM. Erosion thresholds (τ_c) and mean erosion rates (E_m) were calculated, and faunal effects were investigated using regression analysis. The conclusions reached from the experiments are summarised below:

Values estimated for τ_c (0 - 0.55 Pa) were within ranges reported for soft cohesive intertidal sediments from *in situ* studies.

No significant relationships were identified between critical erosion threshold data and faunal densities, with the latter expressed as total faunal density, density of the dominant species, and/or density of other taxonomic groups. The reasons for this may be partially due to the limitations of the methodology used. Further studies, preferably *in situ* field measurements would be desirable to confirm or dispute this.

A weak (but significant) relationship was identified between the gradient (as a measure of ϕ) of the erosion profiles and faunal densities. Small gradient values obtained corresponded to higher faunal density and vice versa, indicating bioturbation.

Results indicate that infauna at the site, in particular the numerically dominant cirratulid polychaetes, are likely to have a net effect in modifying erosion rates, which is almost certainly density dependent. However, the nature of this impact was not entirely clear due to effects of the keeping the cores in the laboratory and the formalin treatment.

Significant variation in τ_c values was found between the EROMES and CSM data and was probably caused by inherent differences between the two instruments such as instrument calibration, roughness length and the spatial scale of measurements.

The two instruments used in this study to measure erodibility (EROMES and the CSM) were designed to investigate the initial onset of erosion and surface erosion. The results indicate that the infauna may have more impact in modifying subsurface

erodibility (e.g. erosion rates), therefore instruments designed with this purpose in mind may be more suitable for use in future studies.

The statistical analyses indicate that erosion rates were affected by pre-treatment with formalin. The use of this type of biocide in future studies is therefore not recommended due to its unknown effects on the sediment fabric.

The desirability of *in situ* field studies wherever possible is emphasised, as it was concluded that the untreated cores kept in the laboratory for the duration of the experiment experienced some deterioration (including die-off and decomposition of macrofauna) that was reflected in the results.

6 Investigation of Hythe tidal flat sediment bulk density and biological modification using Computer Tomography

6.1 Introduction

6.1.1 Bulk density

It has been recognised that mass physical properties of sediment play an important role in determining sediment stability and benthic ecology (Black *et al.*, 1998; Meadows & Meadows, 1991). Bulk density, often expressed as wet bulk density, is a parameter commonly measured for investigating the stability and/or erosion properties of cohesive sediments of intertidal and estuarine mudflats. Bulk density (sometimes expressed as percentage water content) is of additional interest to ecologists investigating the effects of the benthos on sediment stability and erosion through the processes of bioturbation, biodeposition and biostabilisation (see Chapter 1 for review).

The macrostructure of a cohesive sediment bed is known to be an important factor in determining erosion thresholds, erosion rates and erosion type (Amos *et al.*, 1996; Faas *et al.*, 1993). Populations of burrowing organisms and tube builders may decrease or increase stability within the bed through bioturbation and biostabilisation respectively (e.g. Meadows *et al.*, 1988; Grant & Daborn, 1994; Widdows *et al.*, 1998). Microbial processes, such as the production of EPS (extra polymeric substances), also play an important role in determining the stability of the uppermost few millimetres of the bed (Paterson, 1994). These biological processes also affect sediment bulk density, which has been found to be an important index of sediment strength (Amos *et al.*, 1996).

6.1.2 Computer Tomography

Computer tomography or CT is a relatively novel method for obtaining high resolution measurements of bulk density within marine sediments, which has several advantages over more traditional methods.

CT is a radiological imaging technique that uses X-ray technology to measure density and atomic composition inside opaque objects (Wellington & Vinegar, 1987). It was first developed as a clinical tool in Great Britain in 1972 by G.N. Hounsfield, who was later awarded the Nobel prize for medicine (Wellington & Vinegar, 1987). Since then

CT has been used in a wider range of applications including ones in petrochemical engineering, forestry, mineralogy and other areas of the geosciences, and more recently, marine sedimentology (see also literature review, Chapter 1).

Like conventional X-ray radiography, CT is based on the passage of X-rays through the sample, where variation in density causes differences in attenuation. A digital two-dimensional matrix of absorption values of the scanned slice is thus created. CT maps the distribution of the linear attenuation coefficient (μ) of the sample over the entire transverse section: The matrix is effectively a numerical map consisting of the values of μ (x,y) corresponding to each three-dimensional pixel known as a voxel (Duliu, 1999).

The resulting digital image, or tomogram, is produced where the HU values are presented as shades of grey; the darker zones representing relatively lower X-ray attenuation and the lighter zones representing higher attenuation (Boespflug *et al.*, 1995). The images are saved in digital format for processing using image analysis software, and tomographic slices can be added lengthwise to produce a 3-D representation of the sample.

CT has a number of advantages over more traditional methods of obtaining density measurements from sediments: CT is a non-destructive method of analysis; the data are obtained without any form of disturbance to the sediment structure or fabric. It is not necessary to extrude the sediment from the cores for analysis, and the cores can be scanned in a frozen state (Wellington & Vinegar, 1987) provided any changes in density due to freezing are accounted for. Freezing the samples in the field avoids disturbance caused by transport.

CT offers increased resolution (up to 0.2 mm vertically) over traditional X-ray radiography, it also eliminates the need for labour-intensive manual sectioning which disrupts the sediment fabric. Whereas alternative methods such as nuclear transmission densitometry and traditional X-ray attenuation methods measure, after calibration, mean bulk density through a selected slice of the sediment sample, CT measures attenuation for each individual voxel (Kenter, 1989). Standard techniques for bulk density analysis in sediment cores such gamma-ray attenuation or geotechnical

sampling do not have the spatial resolution or the discrimination of X-ray attenuation to resolve details of near-surface macrostructure (Amos *et al.*, 1996).

Data can be processed with or without individual, non-sedimentary bed components (e.g. shells), which may skew density values. This either produces data comparable to that obtained from sieving techniques, which exclude large grain-size components, or data comparable to that produced by geotechnical methods such as GRAPE, which include these components (Boespflug *et al.*, 1995).

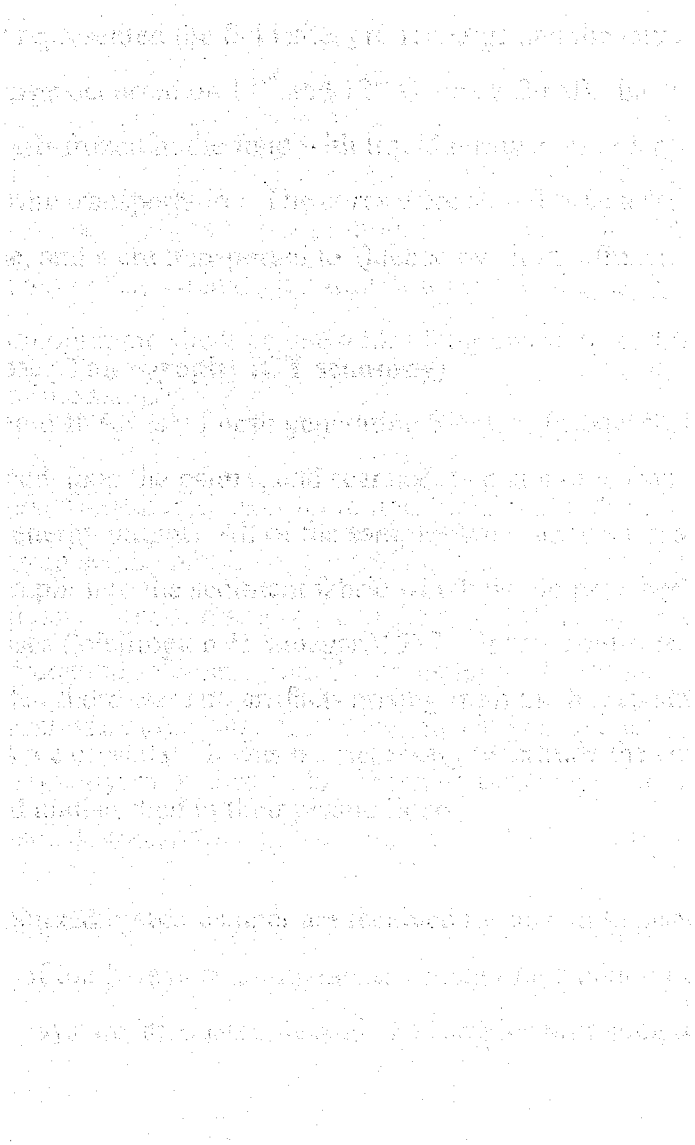
Scan time is rapid; current (fourth generation) CT scanners can produce thousands of images in a few hours, representing a significant saving in labour compared to traditional laboratory analysis involving physical sectioning. Additionally, because the data produced from CT are in digital format, they are far more convenient for quantitative analysis with readily available image processing software (Ketcham & Carlson, 2001).

The main disadvantage of CT is the high initial cost of the equipment. At present most CT scanners are owned by hospitals or the petroleum industry, with only a few devoted to other academic research (B. Long - personal communication). However, if use of existing equipment can be arranged, the method is comparatively cost-effective on a per-sample basis.

6.1.3 Study aims

The main aim of this work was to utilise a CT scanner to obtain high-resolution bulk density profiles at a scale relevant to the macrofauna. CT was used as it would not be possible to achieve this level of resolution by traditional wet sediment analysis techniques. Further aims were: To identify and investigate any indications or evidence of the recharge, such as a fine layer of deposition, too subtle to be picked up by the field study. To identify any recently deposited distinct facies or laminae – including distinct subsurface layers of shell debris, which were commonly observed on the surface of the flats at Hythe. To compare the data obtained by CT to the field data from Chapter 2 and laboratory data from the stability experiments (Chapter 5): The bulk density profiles provide characterisation of the sediment properties and structure at site 1, Hythe, which is complementary to the faunal data, bed elevation data and *in*

situ physico-chemical profiles in Chapter 2. A final aim was to investigate effects of the macrofauna on sediment structure and properties in terms of burrow distribution and structure, and potential bioturbation. Due to time limitations, only a preliminary, qualitative investigation of these aspects was possible. However, this aim directly relates to the stability experiments in Chapter 5, and the data is discussed in relation to the results from this chapter.



6.2 Methods

6.2.1 Sample collection and storage

Twelve core samples (four 25 mm diameter push cores 180 mm in length, and eight 30 mm diameter syringe cores 120 mm in length) were collected from site 1, Hythe at low water (location as for Chapter 2). The cores were transported to Quebec City, Canada to be analysed by the CT scanner owned by Professor Bernard Long, INRS (Institut National de la Recherche Scientifique), University of Quebec.

Two cores collected on each of the following dates were scanned; 6th July, 18th September, 13th, 16th and 18th October 2000, and 23rd February 2001. Therefore, the first four cores represented the field site, pre-recharge and the latter eight were post recharge (recharge occurred on 11th and 12th October, 2000). Each of the cores was immediately flash-frozen in the field with liquid nitrogen in order to prevent disturbance during transportation. The cores were stored frozen (at –20°C) until required for use, and were transported to Quebec by air in a frozen state.

6.2.2 Computer Tomography (CT scanning)

The CT scanner at INRS is a fourth generation Siemens (model SOM 5 SPI). The cores were placed upon the gantry, and scanned, two at a time from top to bottom at 120 kV (beam energy output). All of the samples were scanned in a frozen state to prevent the disruption to the sediment fabric which would have been caused by thawing processes (Wellington & Vinegar, 1987). It was confirmed through inspection of the images that there were no artifacts arising from the frozen state of the cores visible (such as ice crystals). It was not necessary to extrude the cores for scanning, which remained undisturbed in their plastic liners.

The X-rays generated by the scanner are received by several thousand receptors, the helical rotation of the X-rays enabling measurement of attenuation through the sample at 360°. The X-rays are attenuated through the samples following the Beer-Lambert Law:

$$I = I_o e^{-\mu x}$$

where, I_o, is the initial intensity of the X-ray incident beam, I is the measured intensity of the attenuated beam and x is the path length (or sample thickness) respectively. The linear attenuation coefficient, μ, is dependent upon both the atomic number and the density of the investigated object (Boespflug *et al.*, 1994). For relatively homogenous marine sediments μ is expressed in Hounsfield Units (HU):

$$HU = ((\mu_s/\mu_w)-1)*1000$$

where μ_s is the attenuation coefficient of the sample, and μ_w is the attenuation coefficient of pure water (Hounsfield, 1973). Hounsfield Units, otherwise referred to as tomographic density values, are a function of mineralogy, grain size and sediment compaction (Boespflug *et al.*, 1995; Kenter, 1989). Generally, high HU values correspond to high-density sample components.

The process known as 'reconstruction' (i.e. the reconstruction of the distribution of μ within the sample) was carried out and transverse sections with a slice thickness of 2 mm were selected. The (horizontal) pixel resolution of the sections was 0.1 x 0.1 mm.

Analysis of the CT scan data and image processing were carried out using the image analysis software OSIRIS v3.1 developed for the medical industry at the University Hospital of Geneva (Ligier *et al.*, 1994).

6.2.3 Data processing

For each image (i.e. transverse section) the contrast was adjusted for optimal clarity, and features of the sample including core liner, compaction, shells and shell fragments, *Cerastoderma* and burrows were identified.

For each image a ROI (region of interest) was selected to incorporate the entire surface area of the section, minus the zone of compaction (identified as a light coloured band up to several millimetres wide) just inside the core liner (see Figure 6.1). The ROI covered an area of 450 mm^2 for cores collected between 18/10/00 and 23/02/01, and an area of just 275 mm^2 for those collected on 06/07/00 and 18/09/00 (these two cores

were sampled using a push corer which had a slightly reduced diameter compared to the syringe cores).

A frequency distribution chart of the HU values for all pixels (i.e. voxels) within the ROI for each slice was plotted, and used to calculate the mean, standard deviation, minimum and maximum HU values for that slice. These values were recorded together with the image number and slice location and used to calculate bulk density as described below.

For slices containing whole, or large fragments of shells, and those with significant volumes of airspace (e.g. at the top of the core where the surface was uneven), a second ROI was selected incorporating the same area as the first one, but excluding these features. For each of these the statistics described above were recalculated and recorded, resulting in a set of uncorrected (original values) and corrected values for the relevant sections. This process enabled greater accuracy in calculation of the sediment density as it eliminated significantly large or non-sedimentary features, which would otherwise significantly skew the data through the effect of an elevated carbonate content or partial volume effects (see discussion).

The data set (including the corrected values) for each core was entered into a spreadsheet (SigmaPlot v7.0). Depth in millimetres from the sediment surface was calculated for each image using the slice location references. Mean, minimum, maximum and standard deviations of the HU values were converted to CT numbers (after Orsi, 1994):

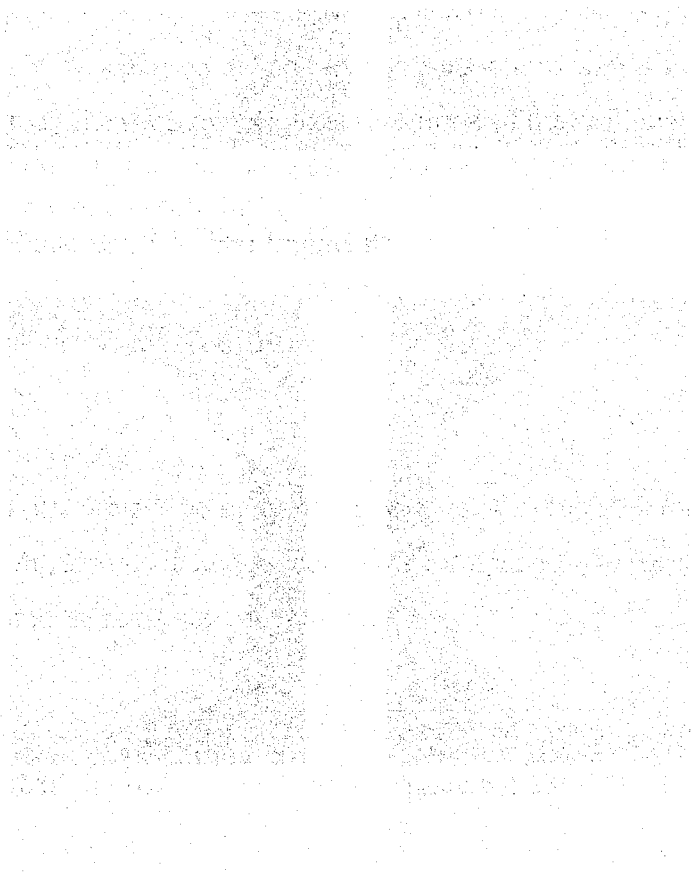
$$CT = (\text{mean HU}/1000)+1$$

Mean, minimum and maximum values for wet bulk density (in kgm^{-3}) were then calculated using a derived calibration for saltwater (as the interstitial salts in marine sediment samples yield different CT values from those in freshwater samples) after Amos *et al.*, (1996):

$$\rho_b = 272 + 694 * CT$$

where ρ_b is bulk density (for fine-grained sediments).

Changes in the density of the samples related to the frozen state of the cores were estimated and accounted for. The estimated density shift was found to be minimal; 0.05 % at 10 psu, increasing to 0.07 % at 20 psu for an estimated *in situ* sediment temperature of 10 °C (which equates to <1 kgm^{-3} for consolidated sediments of 1400 kgm^{-3}). The scatter within the data was calculated from the standard deviations of the HU values for each histogram (Amos *et al.*, 1996). A longitudinal density profile was then plotted for each core.



6.3 Results

6.3.1 CT scan images

CT generated images of five consecutive transverse sections of a core (core1, 23/02/01) from Hythe, are shown in Figures 6.1 to 6.5. The sections shown here represent 2 mm thick slices taken every 2 mm from a depth of 22 mm to 30 mm from the sediment surface. The date, core number and slice reference numbers for each image are included on the corresponding caption (e.g. Figure 6.2 23/02/01_1_15).

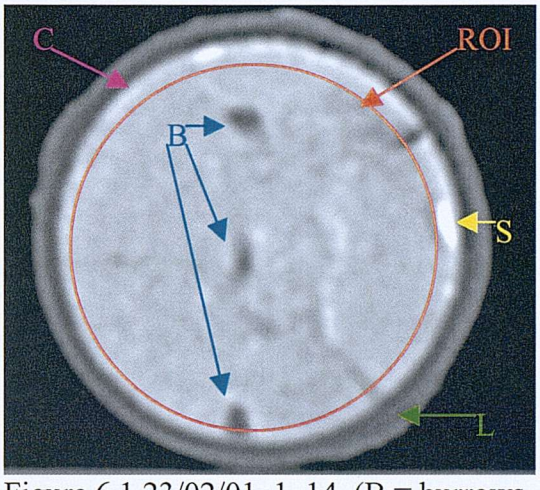


Figure 6.1 23/02/01_1_14 (B = burrows, C = core compaction, L = core liner, ROI = region of interest, S = shell fragment)

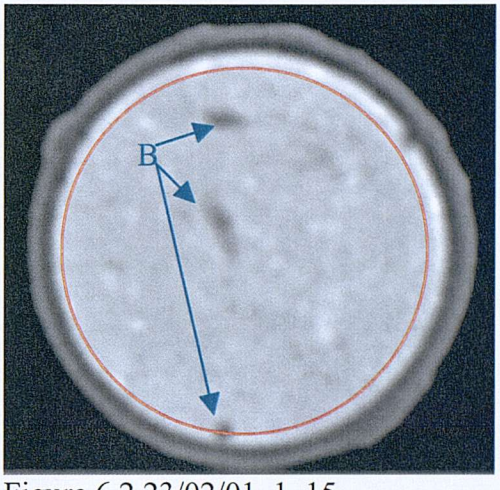


Figure 6.2 23/02/01_1_15

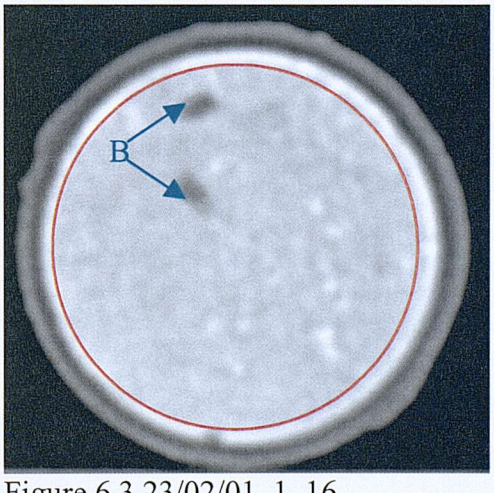


Figure 6.3 23/02/01_1_16

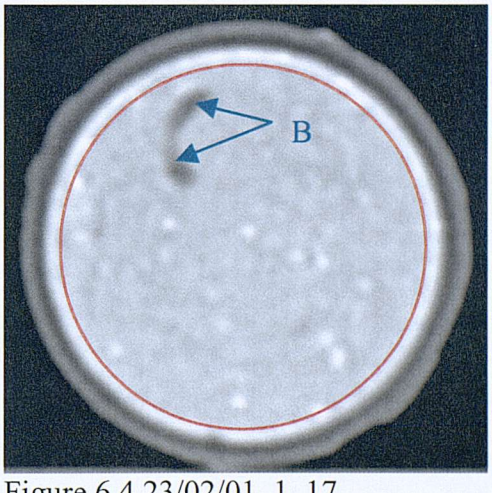


Figure 6.4 23/02/01_1_17

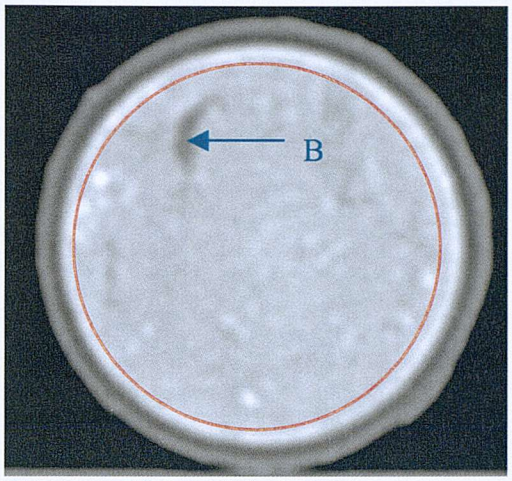


Figure 6.5 23/02/01_1_18

The annotations on Figures 6.1 to 6.5 illustrate features of the sample: The core liner (L) can be seen as the mid-grey band around the perimeter of the images. Immediately inside this is narrow a dark grey or black band which is where the outer sediment of the core meets the plastic liner. A zone of core compaction (C) is visible as a pale grey band, approximately 4 mm thick adjacent to this. The ROI used for density calculations is the area within the red circle (R) and covers an area of 450 mm^2. The white oval-shaped object to the right in Figure 6.1 is a shell fragment (S). The dark oval shaped features within the sediment are macrofauna burrows (B). The upper two burrows appear to be branches of a single burrow (probably constructed by *Nereis diversicolor*) and can be seen to merge together in Figure 6.5.

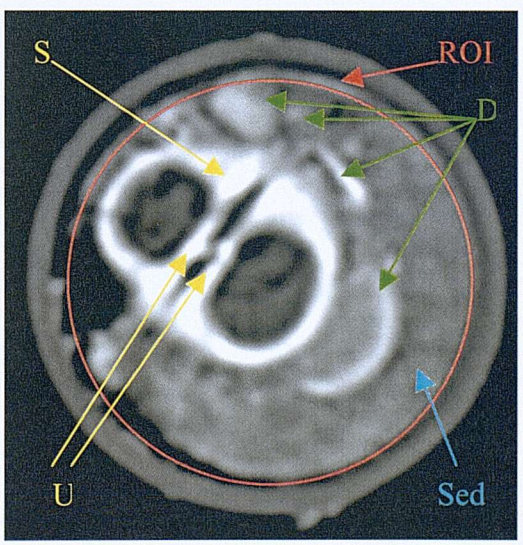


Figure 6.6 23/02/01_2_6
ROI = region of interest, S = cockle shell, Sed = sediment, U = *umbi*, D = shell debris

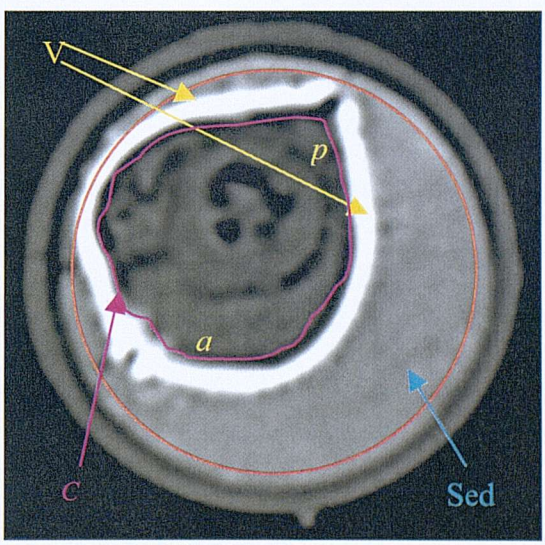


Figure 6.7 23/02/01_2_8
C = *Cerastoderma*, *a* = anterior, *p* = posterior of animal, Sed = sediment, V = valves of shell

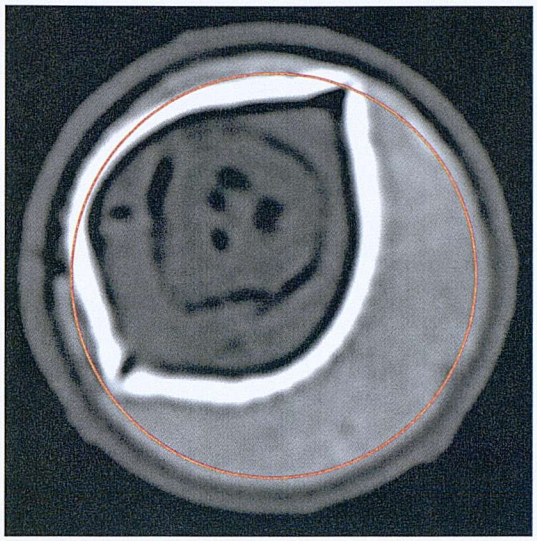


Figure 6.8 23/02/01_2_9

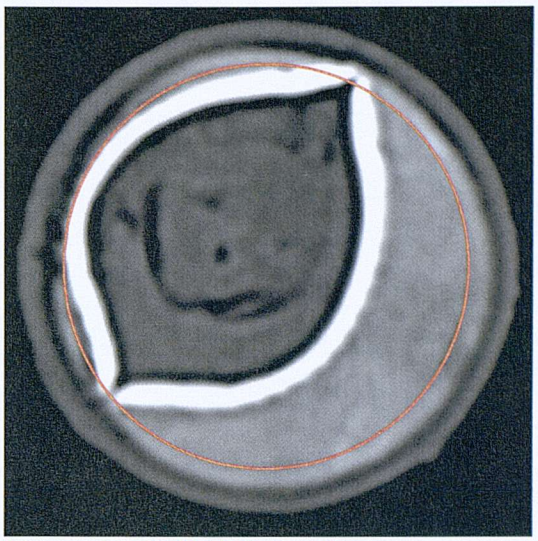


Figure 6.9 23/02/01_2_10

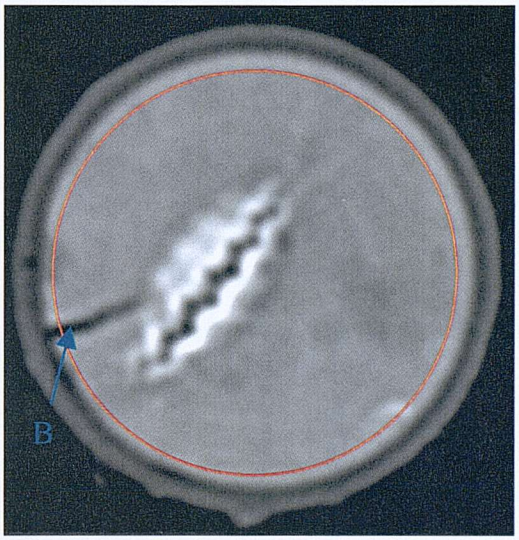


Figure 6.10 23/02/01_2_16

Figures 6.6 to 6.10 show a series of CT images produced from core 2, 23/02/01. The images represent sections at depths of 0 mm (sediment surface, Figure 6.6), 4 mm (Figure 6.7), 6 mm (Figure 6.8), 8 mm (Figure 6.9) and 20 mm (Figure 6.10). Features are labelled as before.

It can be seen from Figure 6.6 that in addition to a large bivalve shell (S) (*Cerastoderma* sp.) small pieces of shell debris (D) were present in the top 2mm section. The top of the bivalve appears orientated towards the sediment surface, the two articulated valves (V) and the two *umbi* (U) are clearly visible. As before, the ROI used for density calculation is shown as a red circle. The sediment can be seen as grey material, with a grainy appearance, surrounding the shells within the ROI.

The tissues of the cockle can be seen inside the shell (*C*) in Figures 6.7 – 6.10 indicating that this specimen was probably live when the core was taken. The anterior (*a*) and posterior (*p*) ends of the animal have been identified. The bottom of the cockle shell can be seen in Figure 6.10. The dark, elongated feature towards the bottom left of image (B) may represent a crack in the sediment, caused by movement of the animal following coring. No debris is visible in these subsurface images.

6.3.2 Density profiles

Bulk density profiles derived from the CT scan data are given in Figures 6.11 to 6.22. The profiles are longitudinally orientated through the sediment cores. Depth from the sediment surface (mm) is plotted against wet bulk density (kgm^{-3}). The error bars represent 1 standard deviation of the HU histogram, which relates to the heterogeneity (of density) in the data. Characteristic layers within the cores have been labelled as follows:

A 'fluff layer' (f), characterised by low densities (1000 –1050 kgm^{-3}) and high organic matter content is indicated for some of the cores. This layer was found at the sediment surface when present (Figures 6.11, 6.12, 6.16, 6.17 and 6.19). The collapse zone, *sensu* Droppo & Amos (2001), is characterised by sharply increasing density of between 1050 and 1200 kgm^{-3}. A collapse zone is indicated in cores 06/07/00_1 (Figure 6.11), 06/07/0_2 (Figure 12), 13/10/00_2 (Figure 6.16), 16/10/00_1 (Figure 6.17), 18/10/00_1 (Figure 6.19), 23/02/01_1 and 2 (Figures 6.21 and 6.22). Core 23/02/01_1 (see Figure 6.21) appears to display two distinct collapse zones, which indicates a layer of recent deposition has taken place (d). This is suggested to a lesser extent for core 06/07/0_1 (Figure 6.11).

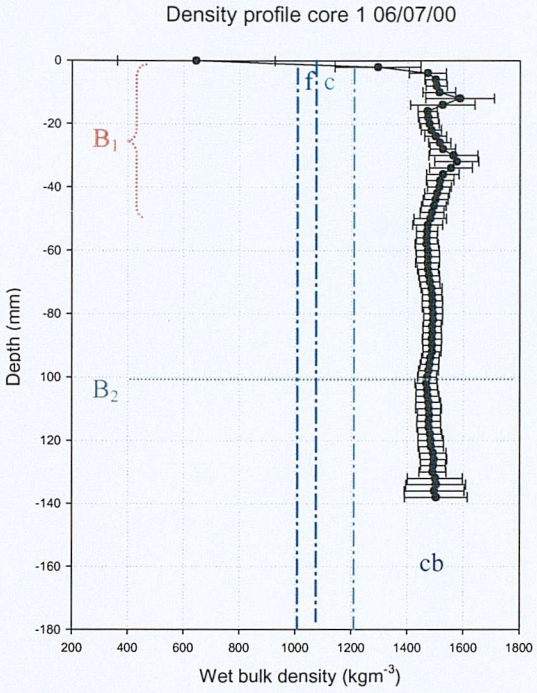


Figure 6.11 f = fluff layer, c = collapse zone, cb = consolidated bed, B_1 = max bioturbation zone, B_2 = limit of infauna

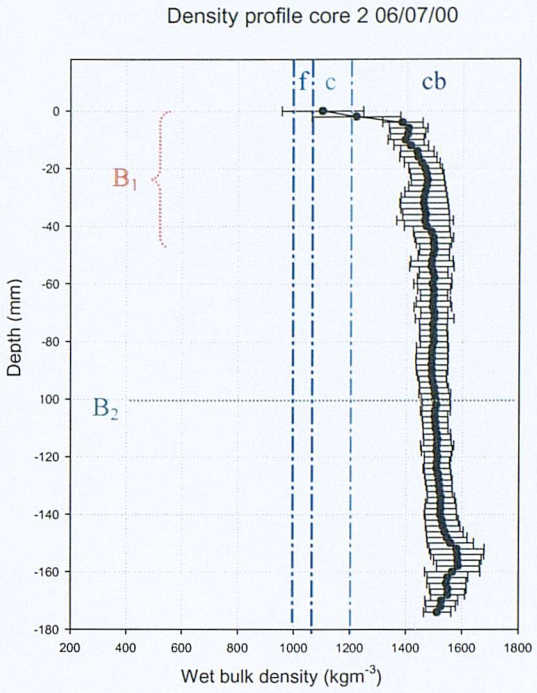


Figure 6.12

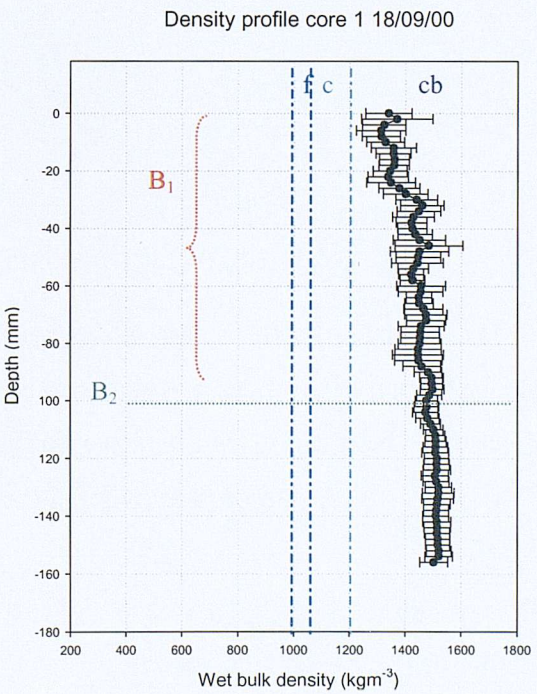


Figure 6.13

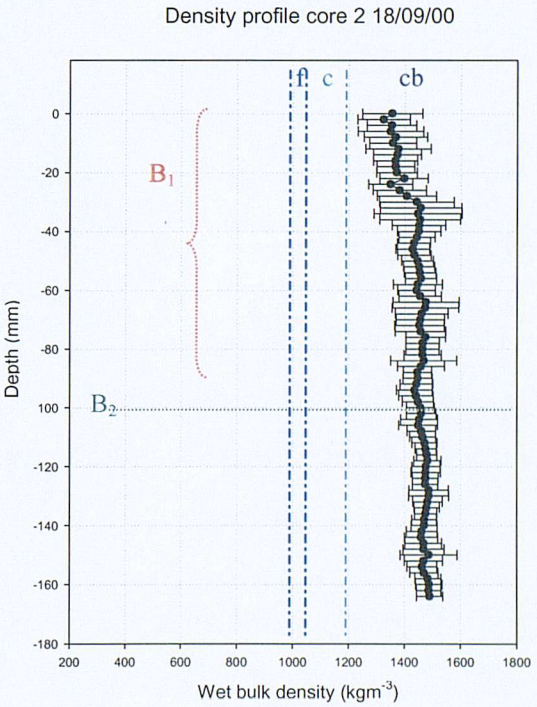


Figure 6.14

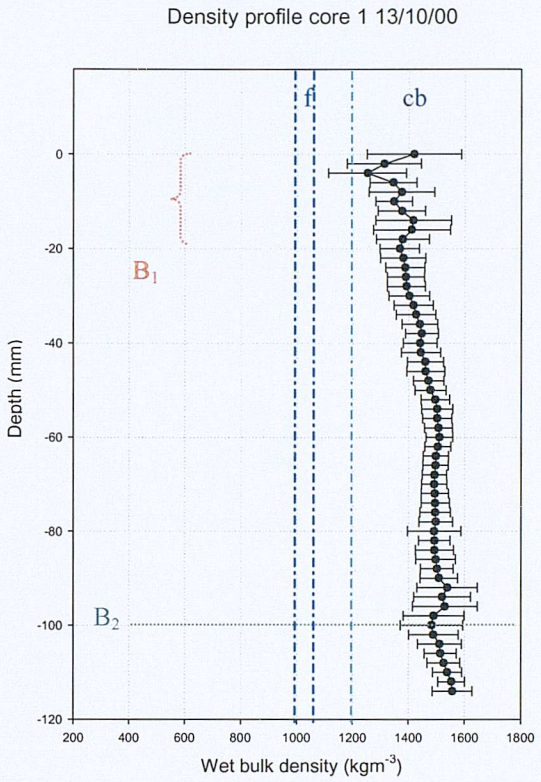


Figure 6.15

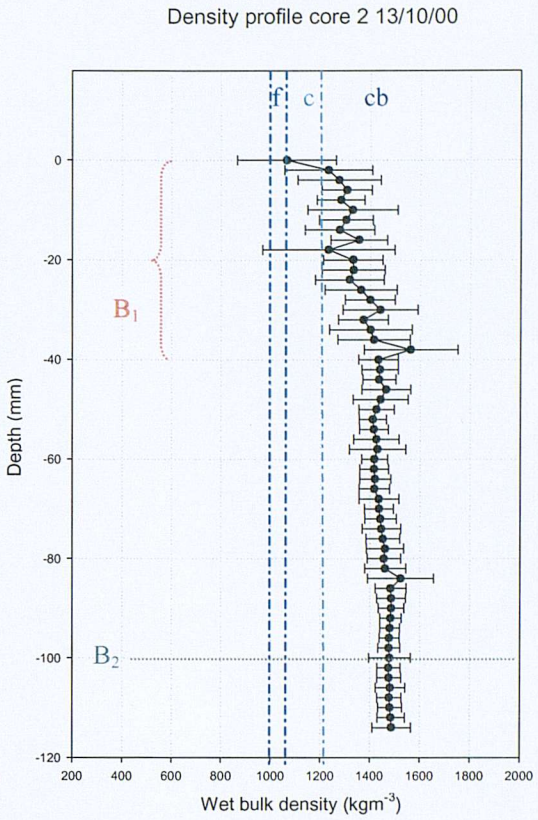


Figure 6.16

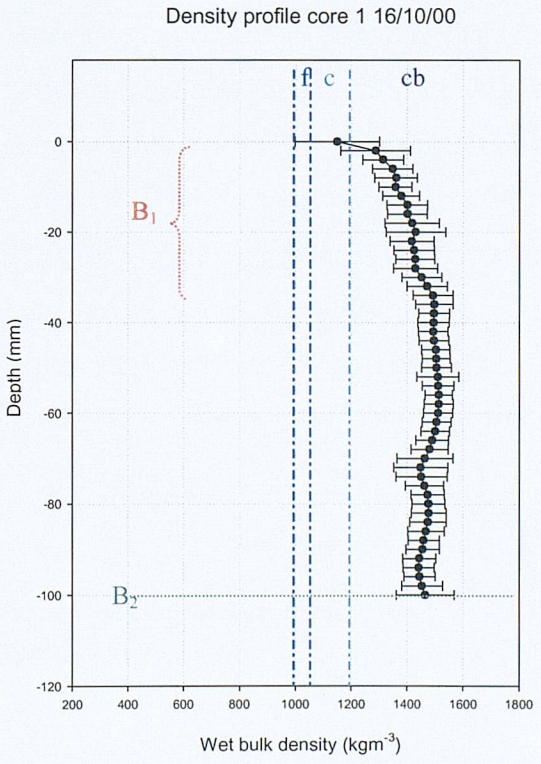


Figure 6.17

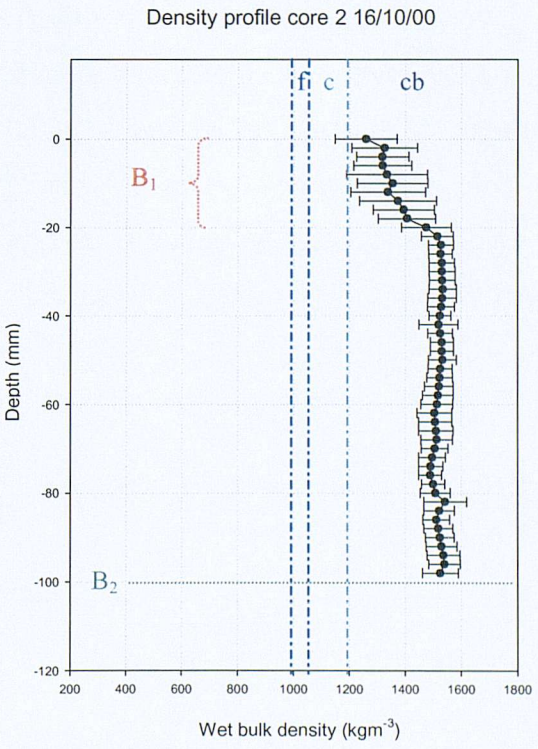


Figure 6.18

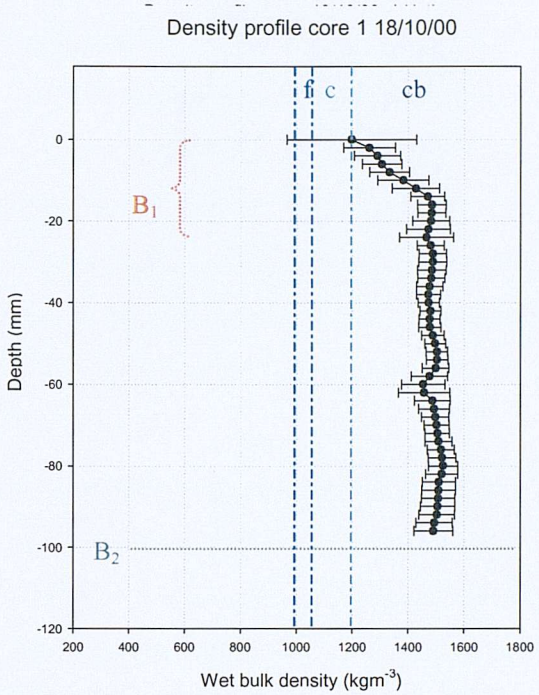


Figure 6.19

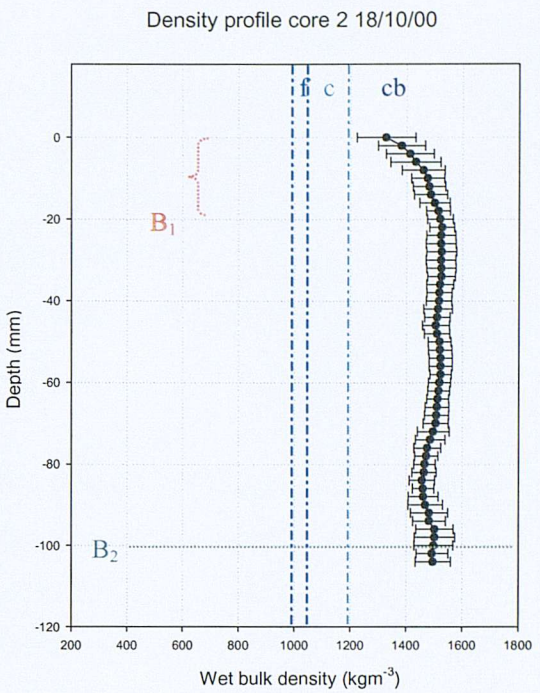


Figure 6.20

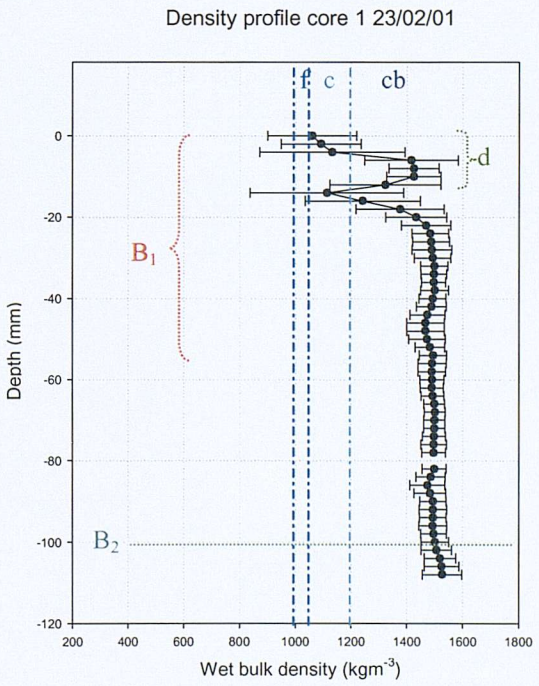


Figure 6.21 d = depositional layer (other labels as before)

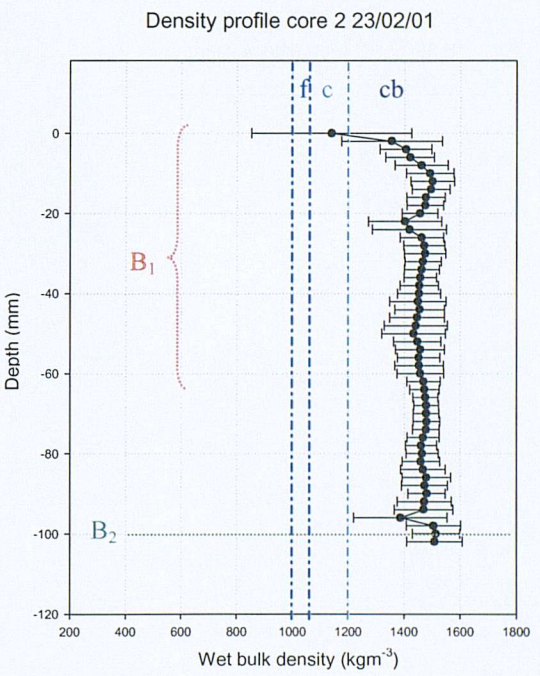


Figure 6.22

Below these surface layers, the density increases downwards to a maximum of 1550 kgm^{-3}. Densities of >1200 kgm^{-3} indicate the presence of a consolidated bed (Van Rijn,

1993). Relatively little variation is present between the cores in this area of the curve (which were all taken at the same site/tidal elevation). The error bars are relatively uniform across the profiles and relatively small suggesting that there was minimal heterogenity in the data. Localised areas of increased variability, which occur further down the cores (see Figure 6.12, 6.15, 6.17, 6.19) indicate individual non-uniform features such as shell fragments or disused animal burrows. The high scatter in the lowest few data points at the bottom of the profile is most likely caused by disruption to the sediment when sampled.

Two further zones have been labelled on the plots: B_1; the zone of maximum macrofauna activity; which corresponds to the maximum heterogeneity in the sediment, and B_2; the lower limit of macrofaunal colonization, below which no macrofauna were found. This was identified/determined by biological core analysis (see chapter 2 for details of methodology). The distribution of the scatter within B_1 is indicative of bioturbation and varies between individual cores.

6.4 Discussion

6.4.1 CT generated images

Biological structures such as polychaete burrows can be clearly seen in the images generated by the CT scanner (Figures 6.1 – 6.5). The images shown are magnified by approximately 400% (actual corer diameter = 30 mm), and the burrows shown in these images are probably produced by one of the larger taxa of polychaete worms found at the field site. The top two burrows are shown to merge (Figure 6.5) indicating that they represent branches of the same Y-shaped burrow and are in all likelihood produced by the same animal, most likely *Nereis*, which is known to build U or Y-shaped burrows (Scaps, 2002). Bivalve shells were also easily identified from the images, with the resolution achieved being sufficient to determine whether the shells were empty or contained an animal at the time of sampling. Shell fragments from bivalves and gastropods, and debris, which were commonly observed on the mudflats at Hythe, were also identifiable from their white colour produced in the CT images due to their relatively high X-ray attenuation. Observation of the 800 or so images produced from the 12 cores, however, indicated that individual burrows of the smaller infauna such as *Caulleriella* and *Tharyx* were less easy to identify. These structures are numerous and extremely small (individual worms are around 250 – 500 μm in diameter), however, it is thought that they can just be seen; giving a grainy appearance to the images (e.g. Figure 6.1). The horizontal pixel resolution of the CT scanner used was 100 μm so it is likely that burrows and structures produced by individuals of this size are within the limits of the resolution. Structures produced by the smaller meiofauna are therefore likely to be beyond the resolution of the scanner. Examination of the images showed that, lower down the cores below the depth colonised by the infauna at the site (50 – 100 mm), this grainy appearance disappears and the sediment is relatively uniform in terms of colour and texture. It is believed that the grainy appearance corresponds to the bioturbated layer of sediment, which is discussed below in relation to the density profiles.

A further observation made from the visual inspection of the images is that the density appears to follow a trend of increasing with depth (as may be expected). This could be seen as a gradual lightening of the sections down the cores. In terms of investigation of

the biological structures produced by the infauna, it is thought that the main limitation to this study was not the resolution of the scanner, but the size (diameter) of the cores used. It was necessary to use relatively small syringe cores as all samples had to be transported frozen by air (therefore resulting in strict limitations on weight). However, although larger burrows could be identified individually, these were generally < 2 per core, presumably due to the patchy distribution of larger species at the site (see Chapter 2). It would therefore be desirable to scan larger diameter cores (100 mm upwards) to facilitate quantitative analysis of these structures in terms of their density, distribution and effects on the sediment structure. As discussed above, structures arising from the smaller taxa appeared numerous within the images, but quantitative analysis was beyond the scope of this study. Vertically, the size of the cores selected for use with the CT scanner was found to be appropriate as results confirmed previous observations that the infauna at this site are restricted to the top few centimetres of the bed.

6.4.2 Bulk density data and profiles obtained by CT

The bulk density data obtained from the CT scanner and subsequent data analysis are given as vertical profiles down through the cores (Figures 6.11 - 6.22). Bulk density is generally a good measure or indicator of consolidation. Several previous field studies have shown that exceptions to this generalisation are a result of biological effects (e.g. Amos *et al.*, 1998; Tolhurst *et al.*, 1999). Thus, trends in bulk density in the sediment can be potentially correlated with biological structures and used to identify processes such as bioturbation.

Several 'layers' of the cores were identifiable from the density data: The surface (upper 2 mm) 'fluff layer' of the sediment, where present, is buoyant, unconsolidated sediment of low bulk density similar to that of seawater (typically 950 – 1050 kgm^{-3}) This layer is usually rich in organic matter, often reported to consist of a biofilm. The fact that this layer appeared present in some cores and not others suggests that it may have been eroded from the latter by ebb currents prior to sampling (which was carried out during spring tides).

The 'collapse zone' was defined by Droppo and Amos (2001). It can be described as a layer of rapidly increasing density near the surface of the bed, typically corresponding to values of 1050 to 1200 kgm^{-3}. This is a region of the bed where the open floc

structure quite literally collapses under the overlying weight of the sediment particles as the bed is formed through deposition. The absence of this layer from a proportion of the cores suggests erosion. This may be due to natural circumstances, caused by tidal currents, or it may be an effect of sampling. The tidal flat at Hythe (site 1) has a markedly uneven sediment surface, consisting of localised ridges or crests, and troughs which do not completely drain of surface water at low tide. It is probable that, with the absence of a developed algal biofilm, the unconsolidated surface sediment is readily re-suspended and eroded by the tidal currents. Random sampling was undertaken to ensure a mixture of 'crest' and 'trough' samples which probably accounts for the variation seen in surface structure of the cores (e.g. Figures 6.11 and 6.12 cf. Figure 6.13 and 6.14).

Marked on each the density profiles is the zone of maximal faunal abundance (B_1). This layer corresponds to relatively greater heterogeneity within the sediment, which indicates the occurrence of bioturbation (see below). The distribution of this layer between the cores is variable, perhaps reflecting patchiness in the distribution and density of the infauna. This layer can be seen to occupy the top 20 to 90 mm of the sediment cores. Earlier extensive core analysis using traditional methods revealed that >95% of the macrofauna at the site were found within a depth of 50 mm (see Chapter 2). Some bioturbation may occur below this level to a maximum depth of 100 mm where the occasional larger organism (e.g. *Cirriformia*, *Nereis* or *Cerastoderma*) was found (Chapter 2). It may be hypothesised that bioturbation (and therefore density variation of this source) would fluctuate seasonally in line with population density fluctuations. No such trends can be seen in the data here, possibly due to the limited number of samples analysed combined with the naturally patchy distribution of the fauna. The area of maximum variability in the top few centimetres of the bed also corresponds to the area of maximum variability in the physico-chemical sediment properties of temperature, salinity, dissolved oxygen and pH (see sediment profiles; Figures 2.5 to 2.8, chapter 2). This is the layer of the bed where the greatest exchanges (e.g. oxygen, ammonia, carbon dioxide) take place, an observation that also confirms that biological reworking of the sediments is present in this region.

Below the bioturbated layer, the density profiles are more uniform in appearance, as shown by the relatively small error bars. This correlates to the uniform, solid grey

appearance of the corresponding CT images which infers that any relict animal burrows or tubes in this layer have collapsed and been filled by the process of consolidation. This layer is also less penetrable to exchanges of oxygen and water (as is reflected in the decreased DO values in the sediment profiles in chapter 2).

The sediments below the collapse zone layer were calculated to possess bulk densities of 1200 to 1550 kgm^{-3}, suggesting the presence of a consolidated bed. The density profiles took the general form of an asymptotic curve, which is monotonic and characterised by increasing density with depth. This section of the core can be classed as having undergone 'self-weight consolidation', and is fairly typical of estuarine muds (e.g. Droppo & Amos, 2001).

No clear trends were present in the sediment profiles that could be attributed to the recharge. This supports earlier observations that the material was not deposited at this site. Neither were there any clear signs of seasonal variation in sediment density such as summer – winter variation. The lack of seasonal patterns may be in part due to the limited number of cores analysed. The depositional event indicated in Figure 6.21 (and to a lesser extent 6.11), is difficult to associate with a distinct seasonal trend, as is the surface erosion indicated in many of the cores (e.g. Figure 6.13). The events of deposition and surface erosion identified do, however, appear to correlate with the bed elevation data for the site presented in Chapter 2 (Figures 2.2 and 2.4). This suggests that the surface of the flat is dynamic, and that long-term monitoring over a period of several years may be necessary to identify seasonal or long-term trends of deposition or erosion.

The density profiles suggest that wet bulk densities for the site are in the region of between 1050 to 1550 kgm^{-3}. These values fall well within the ranges found by other authors in previous studies using traditional techniques for calculating bulk density. For example, Day (2000) obtained a mean (wet) bulk density of 1271 ± 51 kgm^{-3} for the lower shore on the same mudflat at Hythe. Tolhurst *et al.* (2000) quote values for wet bulk density of 1430 kgm^{-3} and 1340 kgm^{-3} for two sampling stations for muddy intertidal flats on the Sylt-Rømø Bight. Values for intertidal cohesive mudflats in the Humber Estuary were found to range between 1240 ± 30 to 1560 ± 20 kgm^{-3} (Widdows

et al., 2000), and values were quoted for intertidal mudflats in the Danish Wadden Sea between 1130 and 1480 kgm^{-3} (Austen *et al.* (1999).

The values measured (this study) for the surface sediments of 1050 to 1200 kgm^{-3} fall within the ranges usually defined as fluid muds possessing no yield strength (1000 and 1100 kgm^{-3}, Van Rijn, 1993), and 'dense fluid muds' (1250 – 1400 kgm^{-3}, Amos *et al.*, in press). This matches *in situ* observations, where, during the short exposure period, the surface sediments were repeatedly observed as poorly drained, and of a 'thick, liquid-like' consistency suggestive of high water content, rather than a solid texture. The values recorded for the top 2 mm of the sediment (below the collapse zone) are also in agreement with Dyer (1994), who states that the typical surface density of typical estuarine muds is from 1250 to 1350 kgm^{-3}. Amos *et al.* (in press) define the 'solid region' of intertidal muds as having a density of greater than 1400 kgm^{-3}, whereas Van Rijn (1993), defines material of >1400 kgm^{-3} as being 'hard mud' that typically takes 100 years or more to form in estuaries.

The calculation of bulk density was corrected for the presence of large shell fragments and other high-density debris such as stones as described in the methods section. These corrections were made so that the values obtained reflected the density of the sediment itself and were not positively skewed by the inclusion of these materials. This was particularly important in the case of shell fragments, as carbonate rich materials have been found to significantly increase CT numbers due to increased attenuation via photo-electric phenomena, which is caused by the higher atomic number of the carbonate samples (Orsi & Anderson, 1999). The discrepancy was found to increase as bulk density increased, resulting in differences of approximately 12 % at 1300 kgm^{-3} and over 20 % at sediment densities of 2000 kgm^{-3}, implying that, were the corrections not applied during this study, values with an inherent error of around 15 % would have been incorporated in the density profiles.

6.4.3 Implications of results and appraisal of CT technique

Successful recharge may be expected to lead to the deposition of a layer of material with relatively low density. The CT profiles showed no evidence for such a layer, thus supporting the conclusions reached in Chapter 2. The CT data showed that the shell debris observed at the site (Chapter 2) was present within the subsurface sediment, but

it appeared to be randomly distributed throughout the cores as opposed to being deposited in distinct lamina. This suggests that this material is frequently transported and deposited by waves and tidal currents and is not restricted to occasional events such as storm surges.

In terms of biological modification of sediment properties, these observations indicate that the subsurface sediment structure and properties such as density are modified by the infauna within the B_1 layer and the relatively decreased density values calculated for this layer (excepting the unconsolidated fluff layer and collapse zone at the surface present in some cores) suggests that bioturbation is present. The CT images confirm earlier observations that the infauna at the site is mostly limited to the top 50 mm of the bed and that the sediment was devoid of macrofauna below a depth of 100 mm. The data suggest that bioturbation is largely restricted to the top 50 mm of the sediment, and indicate that a minor destabilising effect may be present at the site due to the infauna. Decreased density in the top few mm of B_1 is clearly indicated in some of the profiles and this may reflect faunal destabilisation (the smaller, more numerous polychaetes such as *Caulleriella* were found to occupy this depth), although consolidation processes need to be taken into account. The fact that the amount of bioturbation indicated on several of the cores appeared to be limited in extent suggests that the infauna may not play a major role in determining the properties of the bed and erosive behaviour at the site, which supports the results from the erosion experiments in Chapter 5.

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The observed variance in bulk density recorded from the B_1 layer (reflected in the relatively wider span of the error bars) suggests that faunal modification of sediment density results in a mosaic of small features (on a spatial scale of mm) of higher and lower density (which causes the 'grainy' appearance of the sediment seen in the images). This probably reflects the effect of the species assemblage present: Some of the species recorded from the site (such as the spionids *Polydora* and *Pygospio)* are known to build reinforced tubes that may result in localised increases in bulk density and lead to stabilisation. Other species found at the site (such as the free burrowing *Nereis* and the cirratulids), do not build reinforced structures and are more likely to decrease sediment density and stability through bioturbation. The end result may

therefore be no clear trend of either stabilisation or destabilisation overall, which is in agreement with the results from Chapter 5.

The results obtained confirm that CT is a satisfactory technique for obtaining fine-scale bulk density measurements from sediment cores from intertidal mudflats. The increased resolution achievable with CT compared to traditional laboratory density analysis gave measurements on a scale that would be difficult if not impossible to achieve through wet sediment analysis alone.

The resolution should be appropriate to investigation of biological structures on the scale of macrofauna (although it may be insufficient for investigating the effects of smaller individuals and meiofauna as features towards the limit of resolution were more difficult to distinguish if densities were similar to the adjacent sample). The main limitation of the technique seems to be access to CT equipment.

The main limitation to this study was the small size and number of cores that could be preserved and transported. More extensive results may have provided information on seasonal sedimentation trends, and quantitative seasonal effects of the infauna on sediment structure through biological effects such as bioturbation. Recent investigative studies of this type using CT technology have been carried out at the University of Quebec to investigate the colonisation and recovery of a benthic community following catastrophic flooding (de Montety, 2002; de Montety *et al.*, 2000; Michaud *et al.*, in press). Results obtained so far suggest that CT has great advantages as an alternative to existing techniques for investigating sediment structure such as X-ray analysis and Sediment Profile Imaging (Keegan *et al.*, 2001).

6.5 Summary and Conclusions

A CT scanner was used to investigate high-resolution distribution and biological modification of bulk density in sediment cores from Hythe. Several distinct layers of the bed were identified from the bulk density data, including a collapse zone (*sensu* Droppo & Amos, 2001) and a sub-surface zone of self-weight consolidation. Biological modification of the sediment structure in the form of bioturbation and individual burrows were also indicated. No clear seasonal trends of erosion or deposition were identified, nor were there any signs of recharge, thus supporting earlier conclusions. Overall, CT was deemed to be an eminently suitable technique for the investigation of fine-scale bulk density and sediment structure. The conclusions derived from this study are presented below:

Bulk density values of the sediment were found to be 'typical' of intertidal and estuarine cohesive sediments, and were in agreement with results published by other authors.

Some evidence of surface erosion and deposition was present, but this could not be tied to a distinct seasonal or long-term trend.

A bioturbated layer was identified; mostly restricted to the top 50 mm of the sediment column. This corresponded to sediment profiles of physico-chemical properties in Chapter 2, which also indicate that this region of the bed is the most biologically and chemically active and the most variable. Below this layer the bed appeared to consist of relatively uniform, stable, consolidated sediment within which density was relatively constant.

No evidence of recharge was seen, supporting earlier conclusions that the disposal material was not deposited or did not remain at this site.

It is thought that CT would also be an excellent tool for investigating depositional and consolidation processes in the case of recharge in future studies. There is also believed to be much scope for the quantitative analysis of biological structures such as burrows, and faunal modification of sediment properties using this technique.

7 Summary and conclusions

7.1 Summary and Overview of Research

The research described in this thesis was based on the lower intertidal region of a mudflat on the western side of Southampton Water at Hythe, Hampshire, UK. The study was an interdisciplinary research project, incorporating aspects of benthic ecology and cohesive sediment dynamics of intertidal mudflats (Figure 1.1). The primary purpose of the study was to monitor the impacts of an intertidal recharge experiment planned and carried out by ABP (ABP, 2001). Components of the research included a field monitoring programme, designed to assess the impacts of the recharge experiment on the macrobenthic community at the site. A second aim of the field programme was to collect baseline data on environmental sediment conditions, sediment properties, localised bed elevation dynamics and the macrofauna community at the site prior to the recharge. A 1000 l capacity experimental microcosm system was designed and built in the laboratory to provide a facility in which to carry out controlled experiments on 'live' sediment cores collected from the field site. A series of five manipulative smothering experiments were carried out in the above system, both for individual species and natural mixed species assemblages from Hythe. The purpose of these experiments was to investigate the effects of burial under several pre-defined levels of sediment on the benthos, as may occur during recharge. Impacts of smothering were assessed in terms of survival and migration ability, and tolerance thresholds were identified for individual species. A set of erosion experiments was carried out on natural sediment cores (from Hythe) with the aim of investigating the influence of the benthic macrofauna present on sediment erodibility and stability. Erosion experiments were conducted on the cores using two instruments; the CSM (Paterson, 1989), and a newly developed version of EROMES (Quaresma *et al.*, 2002; Schunemann & Kuhl, 1991). Erosion thresholds (τ_c) and mean erosion rates (E_m) were calculated, and an instrument comparison was included. Finally, sediment cores from the field site, collected both prior to and following the recharge were analysed using a CT scanner at the University of Quebec (Canada). The images generated by CT were analysed to calculate and investigate fine-scale bulk density, and high resolution vertical density profiles were drawn up. The profiles provided information on wet bulk density at the site and a further investigation into the effects of the recharge.

Impacts of the macrofauna on sediment structure and bulk density were also identified and analysed to compliment and tie in with the erosion and bed stability work carried out in Chapter 5.

Several problems and setbacks were encountered during the project. The most significant of these was lack of deposition at the site following the recharge experiment when the disposal material was transported downstream and offshore (ABP, 2001). This meant that there were no (long-term) effects to investigate, which therefore necessitated significant modifications to the research programme more than one year into the project. The original field monitoring programme did, however, enable characterisation of the site which provided important background information on the benthic community and conditions present which puts the subsequent laboratory investigations into context. A further problem occurred when, during the fifth smothering experiment, there was a general power cut in the laboratory over the weekend that resulted in the loss of 75 % of samples from the experiment.

Overall, however, the research succeeded in fulfilling (at least in part) most of the aims and objectives outlined in chapter 1, and provided significant information that has important implications for future recharge activities at the site (see summary of conclusions below). The results of the experimental studies also provided further relevant information on the ecology and animal-sediment interactions of the macrobenthos at the site, in particular on the site dominant, the cirratulid polychaete *Caulleriella caputesocis*, which is very poorly understood. The results obtained from the smothering experiments for the cockles; *Cerastoderma edule* and *Cerastoderma glaucum*, and the clam *Tapes philippinarum* may also have implications for active fisheries of these species; for example Poole Harbour, Dorset.

7.2 Summary of Conclusions

A summary of the main conclusions relating to the aims and hypotheses in Chapter 1 is given below:

7.2.1 Baseline information on the benthic macrofauna community at Hythe

The benthic macrofauna community of the lower intertidal zone on the mudflat at Hythe was found to be an impoverished community typically found in physically and anthropogenically stressed environments. It was characterized by low diversity, and corresponds to a modified *Macoma balthica* community *op cit.*, after Barnes (1973). The ecological observations from this study were comparable to results from previous studies (e.g. Barnes, 1973; Houston *et al.*, 1983) carried out at the site, which indicate that the community is of a stable nature in the long-term. The benthic fauna was dominated by deposit feeding, tubiculous, cirratulid polychaetes of the species *Caulleriella caputesocis* and, to a lesser extent, *Tharyx marioni. Cirriformia tentaculata*, *Nereis diversicolor*, *Nepthys hombergi*, *Pygospio elegans*, and *Polydora ligni*, also occur at lower densities along with the bivalves *Cerastoderma edule* and *Cerastoderma glaucum*. Less common species found at the site included the bivalve *Abra abra*, the polychaetes *Ampharete ampharete*, *Melinna palmata*, and *Capitella capitata*, the oligochaetes *Tubificoides benedi* and *Tubificoides pseudogaster*, and the amphipod *Grandidierella japonica*. The distribution of the benthos was found to be inherently patchy (this has often been observed in soft sediments and is believed to be a naturally forced phenomenon (e.g. Gray, 1981; Hall *et al.*, 1992; Heip, 1992)).

7.2.2 Hythe intertidal recharge experiment

The results of the field monitoring in Chapter 2 support the conclusions of ABP (ABP, 2001) that the disposal material was transported downstream away from the site, and was not deposited upon the tidal flats at Hythe. No evidence of recharge was identified by the CT analyses, in support of the conclusion that a significant amount of the disposal material was not deposited, or did not remain at the site. The lack of deposition at the site following recharge means that the hypotheses outlined in Chapter 1 (section 1.3.1, 1) were not supported or rejected by the field experiment.

7.2.3 Impacts of smothering experiments on Hythe macrofauna

The results of the smothering experiments support the hypothesis (section 1.3.1, 1) that the long-term survival of burial in macrofauna depends on the ability of individuals to escape and reach the upper layer through vertical migration. The results obtained confirm that migratory ability is species dependent, closely related to individual morphology and life habits, and that highly motile, active burrowers such as nereid and nephtyd polychaetes are far more likely to survive burial without negative effects than relatively less mobile, weaker burrowers such as near-surface living suspension feeding bivalves or small tubiculous species.

Tolerance thresholds were also identified for individual species. Cockles (*Cerastoderma* spp.) appeared to be highly sensitive to smothering. Results indicated that mass mortality would be expected to occur if animals were buried under more than 2 cm of sediment, and that burial of 10 cm or less was capable of wiping out entire populations. The manila clam (*Tapes philippinarum*) was found to be reasonably tolerant to short-term burial. However, as this species displayed relatively limited migratory ability it is believed that long-term survival of these animals would be unlikely if burial were extended for prolonged periods. Results indicated that in the case of indefinite burial, more than 2 cm of deposited sediment could have deleterious effects on this species. *Nereis diversicolor* was found to be highly tolerant of burial under fine sediment, and no negative effects were observed when individuals of this species were buried under half a metre of mud, which gave rise to a 100 % survival rate. This was attributed to the relatively high motility and burrowing ability of this species. The results from this experiment suggest that ragworms would not be severely impacted by recharge if deposition were less than half a metre (notwithstanding significant differences in sediment type or chemical effects from increased levels of sediment pollutants). The cirratulid polychaetes *Caulleriella caputesocis*, *Tharyx marioni* and *Cirriformia tentaculata* displayed some tolerance to burial (also under native sediment). However, tolerance threshold for these species appears to be limited to a few centimetres. This has significant implications for future recharge at the site as, from the results obtained here it is predicted that populations would incur severe losses at a burial depth of 5 to 10 cm. It is suggested therefore that, as the site is dominated by these species, future recharge incurring in excess of 5 cm of deposition would (at least on a temporary basis) cause significant changes to the

community at Hythe. Mud snails, *Hydrobia ulvae*, appeared to be extremely sensitive to deposition. Results suggest that significant deleterious effects on populations can be predicted in the event of recharge with as little as 2 centimetres of sedimentation. It is believed that the sensitivity of this species is due to it being largely surface dwelling. This is in agreement with previous research that has indicated that epifauna or surface dwellers are relatively more sensitive to smothering than infaunal species (e.g. Chandrasekara & Frid, 1998). The results obtained from these experiments support the hypothesis (section 1.3.1 1) that increased burial depth increases deleterious effects on fauna.

7.2.4 Implications of smothering experiment results for recharge at Hythe

The results from these experiments have important implications for future recharge operations, and suggest that intertidal recharge at Hythe would be likely to significantly diminish macrofauna populations. It is predicted that a resulting layer of sediment deposition in excess of ten centimetres thick would essentially defaunate the area affected. As previous research indicates that patch-size (of the area disturbed) is an important factor in the recolonisation of soft sediments by macrofauna following disturbance (e.g. Zajac & Whitlatch, 1982), in the event of a large area being affected it is likely to cause significant and perhaps long-term changes in faunal abundance. It follows therefore that, if long-term changes occur, a bottom-up effect may be felt on the surrounding ecosystem. This is of particular concern at Hythe as part of the neighbouring shoreline and saltmarsh is a designated nature reserve for birds (many of which are likely to depend on the macrofauna as a local food source). The results of the smothering experiments suggest that negative impacts on the macrofauna Hythe from recharge could be reduced if the layers of sediment deposited did not exceed 2 centimetres in thickness at a time, and intermittent periods for recovery and repositioning of fauna were allowed. Many of the species tested during these experiments may potentially survive (although it is expected that numbers would be significantly depleted) a layer of deposition 5 cm thick, although the experimental results indicate that this would be particularly deleterious to cockles.

7.2.5 Effects of Hythe macrofauna on sediment stability and erosion

It was hypothesised (section 1.3.1 2) that the infauna and their activities may bring about modification of the erosive properties of the bed, either causing net stabilisation

or destabilisation through bioturbation, and that this effect may be density dependent. No significant correlative relationships were identified between faunal densities and critical erosion threshold values obtained from the erosion experiments although this may be due to limitations of the methodology used. The results suggest that the infauna present at this site may have a greater influence on the sub-surface sediments than the surface shear strength. A potential relationship was indicated between the gradient (as a proxy for internal friction coefficient or ϕ) of the erosion profiles and faunal densities. Small gradient values obtained from the experiments corresponded to increased faunal density and vice versa, i.e. that the gradient decreased with increasing faunal density. As ϕ is a measure of consolidation, it is believed that this indicates the presence of bioturbation. Results indicate that infauna at the site, in particular the numerically dominant cirratulid polychaetes, may modify erosion rates. The nature and extent of this impact was not entirely clear due to laboratory effects, however, it appeared to be density dependent. The relationship identified from the calculated gradient values (and the results of the CT analysis, see below), suggests that increased faunal density results in increased bioturbation and erosion rates.

The statistics indicated that the critical erosion threshold values obtained were significantly affected by the formalin treatment (used on half of the cores), i.e. that the treatment caused an experimental effect. It is possible that this was a preservative effect of the formalin solution, which prevented deterioration and decomposition of the treated cores, as significantly higher faunal densities were recorded from treated cores at the end of the experiments. The erosion rate data were also shown to be significantly affected by the formalin treatment. Despite reported success using formalin for this purpose in previous studies (Faas *et al.*, 1993; Grant & Daborn, 1994), the use of this type of biocide is therefore not recommended in future studies due to its unknown potential effects upon the sediment fabric. However, despite observed differences between the two instruments used for the erosion experiments, and the reported laboratory and experimental effects, values obtained for τ_c were comparable to results quoted for soft cohesive intertidal sediments in past studies undertaken in the field, including Hythe (e.g. Day, 2000). This suggests that the impact of the experimental effects may have been greater upon the macrofauna than the properties or structure of the sediment itself.

The results of the statistical tests indicated that the significant variation detected in τ_c values arose from differences between the two instruments used. These differences probably arose from inherent variance in factors such as instrument calibration, and roughness length and spatial scale of measurements between EROMES and the CSM. These results highlight the difficulty in comparing data obtained from different instruments. The instruments used in this study to measure erodibility (EROMES and the CSM) were both designed to investigate the initial onset of erosion and surface erosion. As the results of the study indicate that the macrofauna may have more impact in modifying subsurface or subsequent erosion properties, such as E and ϕ, instruments designed with this purpose in mind may be more suitable for use in future studies with similar aims.

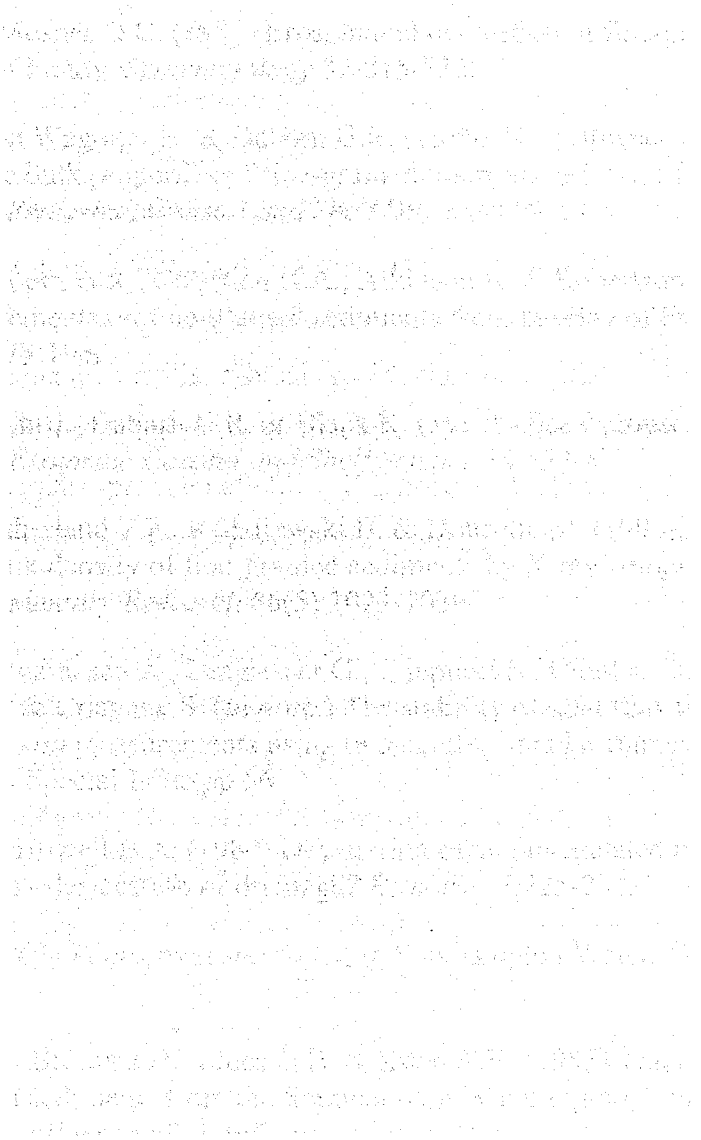
7.2.6 Characterisation of high resolution sediment bulk density by Computer Tomography

Wet bulk density values of sediments obtained by CT for the site at Hythe were within 'standard' ranges of values previously defined for distinct layers of a sediment bed (Amos *et al.*, in press; Dyer, 1986). This indicates that the density and structure of the bed at the site is 'typical' of intertidal and estuarine cohesive sediments, and suggests that, where available, Computer Tomography is a good alternative method to more traditional techniques for obtaining bulk density profiles from sediment cores (e.g. methods involving time-consuming physical sectioning). The technique was found to be highly suitable for obtaining accurate and high-resolution measurements for the investigation of fine-scale sediment density. The CT data confirmed field observations that the disposal material was not deposited at site 1, Hythe following the recharge.

7.2.7 Investigation of faunal modification of sediment structure and properties using Computer Tomography

The use of CT enabled non-destructive investigation of biologically mediated sediments on a spatial scale that was appropriate to the study of macrofauna activities. Analysis of the CT images and data supported the hypothesis (section 1.3.1, 3) that a bioturbated layer was present. Bioturbation was mostly restricted to the top 50 mm of the sediment column. This corresponds to the faunal distribution analysis, and the sediment profiles of physico-chemical properties (chapter 2), which also indicated that this region of the bed is the most biologically and chemically active and the most variable. The density profiles and CT images showed that, below this layer, the bed

appeared to consist of relatively uniform, consolidated sediment within which density gradually increased with depth. There is believed to be much scope for the development of quantitative analysis methods of biological structures such as burrows, and faunal modification of sediment properties using this technique.



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