Impacts of Sea-Level Rise on an Estuarine Ecosystem: A Case Study of the Humber Estuary, UK

by

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Abstract

In many estuaries, extensive areas of intertidal habitats could be lost in the future by rising sea levels squeezing beaches and tidal flats against both established and newly constructed sea defences. Furthermore, an increase in sea levels will cause saline intrusion upstream and, in addition to a straightforward loss of intertidal area, the remaining area is likely to become steeper and composed of coarser sediment particles, particularly around the outer region of estuaries. This thesis examines the potential impacts of sea-level rise on estuarine ecosystems with emphasis on the abundance and biomass of benthic invertebrate assemblages, and hence the consumers they support, especially fish, shrimps and shorebirds.

Field surveys were conducted in September 2003 and 2004 in order to provide parameters for statistical models aimed at predicting changes in macrobenthic biomass in response to environmental gradients in the Humber estuary, U.K. The dominant species were *Cerastoderma edule*, *Macoma balthica* and *Nereis diversicolor*. Multiple regression analysis indicated that macrobenthic biomass was significantly explained by key environmental variables such as salinity, sediment characteristics and morphological elements, consistent with the general picture claimed for estuaries elsewhere. Field observations also confirmed that beaches experience steepening in response to coastal squeeze, and model simulations using these statistical models revealed that if sea level rises by 0.3 m, between 3.9 % and 22.8 % of macrobenthic biomass could be removed from intertidal habitats, depending on how such key environmental variables actually respond to sea-level change.

Analysis of shorebird distributions on the Humber revealed good associations with the amount and distribution of the prey at the beginning of wintering period, and the latter are the major determinants of the density and distribution of benthivorous shorebirds. This suggests that population of intertidal dependent birds could be described as a function of the quality (mean macrobenthic biomass) and quantity (area) of intertidal habitats. A study of fish diets confirmed that intertidal habitats are also important as feeding grounds for these species. Any loss of intertidal habitats for fish species may be particularly significant in the Humber estuary because of the substantial difference in macrobenthic biomass in subtidal and intertidal areas.

This thesis has confirmed that sea-level rise is a significant threat for estuarine ecosystems and there is a need to find appropriate coastal and estuarine management approaches in order to sustain both nature conservation interests and socio-economic needs. Managed realignment is increasingly seen as a key approach in dealing with sea-level rise, but more appropriate ecological objectives and success criteria need to be identified for such schemes in estuarine environments.

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Author's Declaration

A part of Chapter 1 of this thesis was presented with Prof. Dave Raffaelli in an international conference "5th ICEF Environmental Future of Aquatic Ecosystems", held in Zurich, Switzerland, on 23 – 27 March 2003. An abstract for Chapter 3 of this thesis has been submitted to "Estuarine & Coastal Science Association (ECSA) 41st International Conference", being held in Venice, Italy, on 15 - 20 October 2006. With these exceptions, I declare that the work contained in this thesis is my own and has not admitted for any other degree or award.

Toyonobu Fujii

Dedication

I dedicate this thesis to my family, Mike & Seiko

- without you, this could not have been possible

Chapter 1

Literature review

1.1 Introduction

Estuaries are ecologically important coastal environments situated between freshwater rivers and the sea, characterised by highly varying physicochemical, morphological and hydrological conditions (Carter, 1988; Ysebaert et al., 2002) and providing some of the most biologically productive habitats on earth (Kennish, 2002; McLusky & Elliott, 2004). These unique habitats have long been known as important nurseries for species of fish and crustaceans (McLusky, 1989; McLusky et al., 1992; Marshall & Elliott, 1996; Elliott & Hemingway, 2002), and estuarine intertidal sand- and mudflats are of great conservation value because they serve as vital feeding grounds for many shorebirds (Prater, 1981; Wolff, 1987; Watkinson et al., 2004). Many of the world's estuaries are, however, already significantly affected by anthropogenic activities, by virtue of their long history of usage by humans for settlement, for agricultural and industrial development, for navigation, trade and transportation, and for biological exploitation. Estuaries have also been used as repositories for the effluent of industrial processes and domestic waste (McLusky, 1989). They are also prime sites for reclamation for industrial or agricultural lands (McLusky, 1989; McLusky et al., 1992). Estuaries are also natural sinks for contaminants, such as agricultural biocides and nutrients, originating from the catchment hinterland (O'Riordan et al., 2000; Kennish, 2002). Estuaries and their fringing wetlands or low-lying hinterlands are important habitats for wildlife, but at the same time under ever increasing pressure of anthropogenic activities, and they are likely to continue to experience a diverse range of environmental stressors such as habitat loss, habitat alteration, eutrophication, overfishing, freshwater diversion.

Furthermore a recent report from the Intergovernmental Panel on Climate Change (2001) argues that global warming is expected to become more pronounced in the future and this is likely to cause the present rate of sea-level rise to increase, posing additional conservation and management concerns for coastal and estuarine environments. Among the most serious concerns of the estuarine zone relating to climate change are increased flood risks with sea-level rise, threats to the maintenance of coastal hard defences through increased wave and tidal energy, and the process of coastal squeeze by which the area of intertidal habitats is reduced as the beach is prevented from moving inshore due to hard coastal defences. Sea-level rise will therefore affect not only the area of ecologically important habitats but also the vulnerability of wildlife and people around estuarine environments, so that management decisions will have major implications for the health of both estuarine ecology and society.

In this chapter, the implications of sea level-rise over the medium to long term for estuarine and coastal management in the UK are considered, with particular reference to estuarine intertidal ecology. First, a brief account of the most likely sea-level rise scenarios for coastal areas and for estuaries is provided, followed by an account of the likely impacts on the physical elements of estuarine habitats. The utilisation of estuaries by shorebirds, fish and crustaceans is reviewed, and factors affecting the distribution, abundance and biomass of estuarine macrobenthos that is an important food source for the estuarine predators are further considered. A simple conceptual model of the likely impacts of sea-level rise on the physical and biological elements of estuarine habitats is then provided, with particularly emphasis on macrobenthic biomass in relation to changes in sediment particle size, beach morphology and salinity regime. Discussion is also made of how changes in nitrogen loading and sediment supply from riverine inputs and increased sea surface temperatures may interact with the effects of sea-level rise to shape the structure and function of invertebrate assemblages and their dependent higher trophic levels. After the consideration of the adaptive strategies likely to be adopted to cope with sea-level rise and how these are likely to impinge on the estuarine environment, the aims, objectives and overview of the thesis are presented.

1.2 Climate change and sea-level rise scenarios

The Earth's climate has warmed by 0.6 ± 0.2 ℃ during the last century and research in recent years has demonstrated that human activities, mainly through the emission of green house gasses or aerosol particles, are likely to be responsible for the observed increases in the mean global surface temperature in the latter part of the 20th century (IPCC 2001). Climate observation also shows that the rate of warming from 1976 onwards is greater than any other time witnessed in the past 1,000 years (Walther et al., 2002), and evidence for warming is now seen for other physical and ecological indicators, such as retreating glaciers, thinning of Arctic sea-ice (IPCC 2001; Watkinson et al., 2004), change in the phenological behaviour of some bird and butterfly species, shifts in the ranges of plant species (Walther et al., 2002).

Sea-level changes reflect climate change mainly through thermal expansion of the upper layer of the oceans and release of water from glacier or ice sheet melt (Watkinson et al., 2004). Sea-level change is a natural process at the geological time scale, showing a marked fluctuation pattern over a range of 120 m during the last 140,000 years (Jones, 1994). A continuous rise of sea levels has been observed since the last de-glaciation approximately 19,000 years before present (Jones, 1994) and it appears the observed rise of sea level by 10–25 cm during the last century is due primarily to the concurrent global warming (IPCC 2001). The rate of sea-level rise relative to land will vary depending on geological location, due to a constant isostatic readjustment of the height of land masses caused by changing relationships between the volumes of water in the oceans and various factors in the earth's crust. During the last ice age, the ice depressed the earth's crust, and segments of the crust (land) have now been adjusting their levels according to their thickness and density, or any disturbance such as erosion, sedimentation and change in the amount of land-based ice (Jones, 1994).

However, the adjusted net rates recorded from a number of tide-gauge stations in the UK, for instance, provide convincing support for the view that anthropogenic forcing is accelerating the sea-level rise, with an annual rate on the Humber of 2.0 mm at Blacktoft and 2.5 mm at Immingham (Winn et al., 2003).

At the global scale, models predict that the annual global mean surface air temperature will rise 0.5 - 1.7 ℃ by 2040 and 1.4 - 5.8 ℃ by 2100, based on increases of green house gasses, and sea levels are predicted to rise 7–36 cm by 2050 and 9–88 cm by 2100 mainly through expansion of the warmer upper ocean and up to 20% from melting land-ice (IPCC 2001). The wide range of estimates reflects the full set of the emissions scenarios analysed by the IPCC (2001) based on expectations of changes in economic growth and human population as well as degree of mitigation and emission of greenhouse gasses. Therefore the amount by which sea levels will rise is dependent upon the scenarios, and the relationships diverge with time in a non-linear manner with the central estimate of 48 cm by 2100.

However, more detailed modelling efforts predict that there are likely to be marked regional and local-scale variations relating to sea-level rise. With respect to regional variations, seven out of nine models indicate a maximum sea level rise in the Arctic Ocean and a minimum in the circumpolar Southern Ocean, whilst the rise may be relatively less south of the Gulf Stream compared to the area to the north (IPCC 2001). At the local level, consolidation of sediment by groundwater extraction as well as aforementioned isostatic land re-adjustment will exacerbate or reduce relative sea-level rise according to location.

In addition to global warming and increased sea level, there are predicted changes in the frequency of severe droughts, excessive precipitation and extreme events together with changes in seasonal and diurnal temperatures at local and regional scales (IPCC 2001; Hulme et al., 2002). Although such aspects of climate change are less certain than those of increases in temperature and sea levels, all these components of climate change undoubtedly have significant implications for estuarine and coastal management. For example, coasts are likely to experience a higher frequency of surges and greater wave action in response to rising sea levels. Because the frequency of surges scales logarithmically with surge height (Hulme et al., 2002), a relatively modest rise in sea level increases the frequency of surges and thus the risk of coastal flooding markedly. For example, in the UK, a "once-in-50-year" surge will occur every 10 years under the high emissions scenario by 2080 (Hulme et al., 2002). Similarly, the energy of waves reaching intertidal flats is a function of both local water depth (Crooks, 2004) and the height of the waves offshore (Carter, 1988). For coastlines where an increase in average wind-speed and wind extremes is predicted, waves will be higher and the wave climate over intertidal habitats will be more energetic. If sea-level rise results in deeper water locally, then this will exacerbate the problem, because less wave energy will be dissipated prior to the wave breaking on the shore. Therefore, in addition to a straightforward loss of intertidal habitats, sea-level rise is likely to undermine the infrastructure of coastal defences or require the increase in maintenance costs of the structures.

1.3 Implications for estuarine physical elements

There is considerable uncertainty involved in predicting how climate change and sea-level rise will impact upon estuaries, but several physical processes that are likely to occur in response to the rise are of great relevance for the management of estuarine ecosystems.

Rising sea level will result in changes in coastal and estuarine geomorphology as a consequence of increased water depth and enhanced wave and tidal energy (Crooks, 2004). Such physical changes will be manifested through the landward progression or redistribution of landforms found in an estuary such as subtidal bedforms, intertidal flats, saltmarshes, shingle banks, sand dunes, cliffs and low-lying hinterland (Pethick & Crooks, 2000). Thus, the extensive linear sand-banking, which is formed by dominant marine processes and often seen in the lower estuary, for example, is expected to migrate upstream in response to rising sea-level. Similarly, intertidal mudflats that are found in less exposed environments will also migrate landwards to a lower energy level as coastal wave energy increases (Pethick, 1996). If the estuary is allowed to migrate inland, then it could re-establish its original structure further upstream. However, for many estuaries this will not be permitted due to existing flood defences protecting agricultural and residential areas and because of canalisation, especially further upstream. This inability for the estuary to move inland and upstream will lead to substantial losses of intertidal habitats due to coastal squeeze, a process by which saltmarshes and mudflats are eroded away as they become trapped between rising sea-levels and fixed sea defences. A reduction in spatial coverage of intertidal habitats would decrease the abundance and biomass of the benthic intertidal invertebrates on which larger consumers depend, notably shorebirds, fish and epibenthic crustaceans. If land is made available above the current high water level for the replacement of important estuarine intertidal habitats by setting back current flood defences (managed realignment), this will reduce the environmental impacts in estuarine ecosystems. However, managed realignment may not be a viable option for most estuaries because of the economic value of the bordering land.

In addition to the loss of habitat due to coastal squeeze, change in sedimentary process and associated morphological change over intertidal flats in response to sea-level rise will have significant implications for the estuarine ecosystem. Estuarine intertidal mudflats are characterised by fine muddy sediment with a rich organic matter content. Such habitats typically support high numbers of benthic invertebrates, which in turn plays a central role in supporting estuarine food webs. Through natural sedimentary processes in response to sea-level rise, mudflats will migrate landwards and could be replaced by sand beaches, which have similarly migrated from more exposed coastal environments (Pethick, 1996), shifting the entire sedimentary distribution further upstream along estuarine longitudinal gradient. However, existing flood defences are likely to prevent the re-establishment of mudflats at higher shore, whilst increased water depths and a more energetic wave climate may lead to changes in the beach's morphodynamic state, moving from a dissipative beach characterised by flatter, low energy conditions with finer sediment, to a reflective one with a steeper slope, higher energy conditions with coarser sediment. Such shifts may be more pronounced in areas around the outer reach of an estuary where wave action and extreme climatic events are dominating forces in the process of coastal land formation. For instance, Taylor et al. (2004) have investigated changes in 1084 coastal profiles throughout England and Wales, and found that 61 % of the coastlines studied had experienced steepening since in the middle of 19th century, generally due to foreshore erosion and the use of sea walls and embankments. These changes in the sediment and morphology of intertidal flats are likely to have significant impacts on benthic intertidal fauna, and thus trophic interactions within the system.

Finally, there will be effects of sea-level rise on the estuarine salinity gradient from freshwater to marine conditions. The structure of this gradient varies according to estuarine morphology, freshwater run-off and turbulence or mixing (Carter, 1988; Raffaelli & Hawkins, 1996). Many of world's estuaries will experience a widening and deepening in estuarine water volume resulting from rising sea levels and a concurrent increase in tidal prism and tidal range (Kennish, 2002), leading to a greater salt intrusion further upstream. This upstream shift in salinity gradient may affect the vegetation communities fringing the estuary through penetration of salt water into the fresh ground water table, leading to salinisation of habitats currently characterised by freshwater (Jones, 1994). In addition, changes in salinity distributions will result in changes in the species composition of benthic communities (see later sections), and will change the region of turbidity maximum, where vigorous mixing of fresh and marine water and intensive particle deposition occur, further upstream. This would increase local silt accretion rates further upstream, but reducing silting processes and increasing the accumulation of coarser sediment in the outer reaches of the estuary.

These general processes will be modified by local geology, the land use of drainage basin, the size, shape and human usage of the estuary (Jones, 1994), and it is therefore important to carefully examine the physical settings of each estuary to predict the impacts of sea-level rise on estuarine ecosystems.

1.4 Estuarine benthic macrofauna and their predators

In coastal and estuarine ecosystems, birds are often regarded as top predators and fish occupy intermediate trophic levels, both of which are supported by benthic macrofaunal prey. Changes in such benhtic prey, due to sea-level rise effects on estuarine physical elements, are therefore likely to impact on these higher trophic levels. The high benthic biomass is a reflection of the presence of extensive intertidal flats and fine sedimentary deposits created through processes such as tidal asymmetry and flocculation (Raffaelli & Hawkins, 1996). In addition, there are coarser deposits on relatively open areas and finer particles in sheltered areas as well as in the upper reaches of estuaries, depending on the local hydrographic regime and chemical processes. Fine muds and silts provide a huge surface area for the accumulation of organic matter and microbial processes, and allow intertidal mudflats to support a high invertebrate biomass, especially with deposit- and filter-feeding taxa (Heip et al., 1995; Raffaelli & Hawkins, 1996; Herman et al., 1999). This benthic invertebrate biomass provides food for higher trophic levels, epibenthic crustaceans, fishes and shorebirds, which use the flats as a nursery area for juvenile stages and/or as adult feeding grounds (McLusky & Elliott, 2004).

Much of the interest from conservation and management agencies in evaluating the effects of sea-level rise on estuaries lies in the potential impact on these higher trophic levels, especially shorebirds (Davidson et al., 1991; Rehfisch et al., 2003). In NW Europe, particularly dependent avian species are brent geese, shelduck, pintail, oystercatcher, ringed plover, grey plover, bar-tailed and black-tailed godwits, curlew, redshank, knot, dunlin and sanderling, whilst grey geese and whooper swan may utilise this habitat for roosting (Davidson et al., 1991; Elliott et al., 1998). The species prey base is quite restricted (Table 1.1), with well over 90 % of the total benthic invertebrate macrofauna comprising a relatively small range of species (Packham & Willis, 1997; Prater 1981). Except for some herbivorous and omnivorous species such as brent goose, mallard or golden plover, many of shorebirds feed mainly on intertidal organisms when the flats are exposed, and they can be highly selective both as to where they feed and the size of prey, in order to maximise their efficiency (Barnes, 1994). Prey selection is also associated with bill size (Table 1.2).

Table 1.1. Characteristic species of the macrofauna of British estuaries known to be important food sources for shorebirds (after Prater, 1981)

Type	Species
ANNELIDS	*Nereis spp.*
	Nephtys hombergii
	Scoloplos armiger
	Arenicola marina
	Lanice conchilega
	Peloscolex benedini
MOLLUSCS	*Hydrobia ulvae*
	Littorina spp.
	Mytilus edulis
	Cerastoderma spp.
	Macoma balthica
	Scrobicularia plana
CRUSTACEANS	*Balanus spp.*
	Corophium spp.
	Crangon vulgaris
	Carcinus maenas

Table 1.2. Principal prey items (in column) of the main wading birds in the Wash, the UK, with reference to Goss-custard et al. (1977)

		GP	DL	KN	RS	TS	OC	BG	CL
Mollusca	*Hydrobia*		*	*	*				
	Macoma	*		*	*		*	*	
	Cardium	*		*		*	*		*
	Scrobicularia								*
	Mytilus					*	*		
Crustacea	*Crangon spp.*				*				
	Carcinus	*			*				*
Polychaeta	*Scoloplos*							*	
	Nereis	*	*		*			*	
	Nephtys spp.		*						
	Lanice	*						*	*
	Arenicola								*

Key: GP, grey plover; DL, dunlin; KN, knot; RS, redshank; TS, turnstone; OC, oystercatcher; BG, bar-tailed godwit; CL, curlew.

Thus, plovers have the shortest bill amongst the estuarine waders and tend to feed mainly by surface pecking, consuming *Hydrobia* and other small organisms found on the surface, while only the longer-billed birds such as curlews and bar-tailed godwits can cope with the deep burrowing larger prey such as the lugworm *Arenicola* and the ragworm *Nereis* (Prater, 1981). Further, Brown and O'Connor (1974) showed that predation by oystercatchers on cockles in Strangford Lough, Northern Ireland falls more on second year and older animals, but small first-year cockles are not taken. Knots are also specialised in bivalves and known to show their own restricted range of prey sizes (O'Conner and Brown, 1977). Prey size selectivity is an important consideration when assessing the effects on sea-level rise on shorebirds via changes in their invertebrate prey (Table 1.2). Differences in food availability account for a large proportion of the variation in densities of bird feeding in intertidal habitats (Yates et al., 1993; Rehfisch et al., 2000), and shorebirds tend to concentrate where prey density and availability are relatively high and the energy expenditure required to feed is relatively low (Goss-Custard et al., 1977). Intertidal flats within an estuary exhibit significant spatial variations in macrofaunal species composition, density and biomass, and this is highly correlated with substratum type or sediment particle size. Thus, Yates et al. (1993) were able to show that quite broad sediment characteristics can be used to predict the densities of shorebirds, allowing bird distributions to be predicted from sediment maps derived from remote sensing data (Packham & Willis, 1997). If the size of feeding ground and the distribution of sediment types change with sea-level rise, there is likely to be a response in bird distribution and abundance. This has indeed been the case where change in substratum type or loss of intertidal habitat has occurred. For instance, the spread of the cordgrass, *Spartina anglica*, over the mud flats in southern England resulted in reductions in the numbers of Dunlin using affected areas (Goss-Custard & Moser, 1988). Also, on the Oosterschelde estuary in the Netherlands, a 30% reduction in the area of intertidal feeding habitat resulted in a reduction in numbers of European oystercatchers (Meire, 1991). Similarly, due to the impact of habitat loss as a result of land-claim on the intertidal area of the Forth estuary, Scotland, significant declines were noted for overwintering populations of dunlin and bar-tailed godwit (McLusky et al., 1992). Given the scale of sea-level rise, which will affect estuarine environments at the global scale, a reduction in the spatial extent of shorebird feeding habitats could have a devastating impact on shorebird populations (Galbraith et al., 2002), apart from any impacts via changes in macrofaunal prey abundance.

Table 1.3. Fish commonly found in estuaries around NW Europe and their feeding type (adapted from Elliott & Hemingway, 2002).

Family	Species (common name)	Feeding guild
Anguillidae	*Anguilla anguilla* (eel)	Dpisc
Gadidae	*Merlangius merlangus* (whiting)	Dpisc
	Gadus morhua (cod)	Dpisc
Gobiidae	*Gobius minutus* (goby)	Db
	Pomatoschistus minutus (sand goby)	Db
Moronidae	*Dicentrarchus labrax* (sea bass)	Dpisc
Mugilidae	*Liza aurata* (mullet)	Pplank
	Mugil cephalus (mullet)	Pplank
Pleuronectidae	*Pleuronectes platessa* (plaice)	Db
	Platichthys flesus (flounder)	Db
	Solea solea (sole)	Db
	Limanda limanda (dab)	Db
Salmonidae	*Salmo salar* (salmon)	Dpisc

Key: Db, demersal benthivore; Dpisc, demersal piscivore; Pplank, pelagic planktivore; (feeding guild adapted from Buchan, 1997)

In contrast to most shorebirds, many fish and crustaceans found in estuaries move up the flats with the flood tide to feed (McLusky, 1989; Elliott et al., 1998). A diversity of fish is found in many estuaries (Elliott & Hemingway, 2002) and the majority are bottom feeders (Table 1.3). Fish have the option of feeding intertidally or subtidally, but there has been an increasing evidence for the importance of intertidal feeding for many of fish species either throughout life or in part of their life cycle in estuarine environments (Elliott & Taylor, 1989; Elliott et al., 1990; Costa & Elliott, 1991; Marshall & Elliott, 1996). In summer in Europe, for example, large numbers of flatfish (*Playtchthys flesus*, *Pleuronectes platessa*), gobies (*Pomatoschistus* spp), crabs (*Carcinus maenas*) and shrimp (*Crangon* spp), move onto the flats to feed on mobile epifauna and sedentary infauna (Elliott et al., 1998). They also crop parts of prey, such as the tail ends of *Arenicola* and *Heteromastus*, the feeding tentacular crowns of fanworms and siphons of bivalve molluscs (Vlas, 1979; Barnes, 1994). Even piscivorous species such as cod *Gadus morhua* or whiting *Merlangius merlangus* are heavily dependent upon estuarine macrobenthos through direct feeding on benthic infauna, or through feeding on other fish species, crab *C. maenas*, shrimp *C. crangon* which have themselves fed on the benthic invertebrates (Buchan, 1997). Migratory species, such as salmon and eels, can also be found in these areas on passage to other wetlands, although they appear to have no requirement for mud and sand intertidal flats (Elliott et al., 1998). With respect to prey preference, many demersal fish are opportunistic predators within estuarine environments and the choice tends to reflect the infaunal species distribution of the area (Costa & Elliott, 1991). In tropical and subtropical areas, the juveniles of many species of commercially important crustaceans, particularly penaeid prawns, utilise estuaries, lagoons and mangroves for feeding grounds before migrating offshore to spawn (Raffaelli & Hawkins, 1996). In view of the large number of juvenile fish and crustaceans found in estuarine habitats, estuaries are potentially important in the maintenance of commercial offshore fisheries. For instance, it has been argued that five out of the six most important commercial fishery species in the USA may be dependent upon estuaries (Smith et al., 1966) and extensive estuaries on the east coast of the USA may be responsible for at least half the commercial landings each year (Day et al., 1989). Some lessons may be learned from coastal and estuarine systems in Nigeria where intensive human activities such as land-claims and heavy pollution have severely affected the breeding and nursery grounds of commercial fish species, and landings from capture fisheries have declined from 500,000 t in the late 1980s to 300,000 t in the early 1990s (Ezenwa & Ayinla, 1994).

Predation pressure on benthic macrofauna from fish, crustaceans and shorebirds can be high. For instance, redshank are estimated to remove 16-38 % of *Corophium voltator* on the Ythan estuary, Scotland (Goss-Custard, 1969), bar-tailed godwits 25 % of *Arenicola marina* (Smith, 1975), and oystercatchers 14 % of the mussels, cockles and other molluscs on which they prey (Goss-Custard, 1977), although the impact of birds on intertidal invertebrates varies greatly from site to site (Table 1.4). Food consumption of fish that move in to estuaries at various seasons also has an impact roughly equal to or even higher than that of the birds. For instance, plaice and flounder consume up to 15 g ash-free dry weight of benthos m^{-2} yr^{-1}, which accounts for 30 % of the estimates of total benthic production, in the Oosterschelde in the Netherlands (McLusky, 1989). In the case of Ythan estuary in Scotland, the fish are estimated to consume three times the amount of food consumed by the birds (Table 1.5).

Table 1.4 Consumption of invertebrate production by shorebirds in various estuaries in the UK (after Baird et al., 1985)

Site	% Production consumed
Gravelingen estuary	6
Waddensee intertidal	17
Langebaan lagoon	20
Ythan estuary	36
Tees estuary	44

Table. 1.5 The total food consumption of fishes and birds in the Ythan estuary in Scotland (after McLusky, 1989).

Predator	Total food consumption (kcal $m^{-2}yr^{-1}$)
Fish	
Flounder (*Platichthys flesus*)	58.1
Goby (*Pomatoschistus minutus*)	8
Birds	
Oystercatcher (*Haematopus ostralegus*)	23.9
Dunlin (*Calidris alpina*)	
Redshank (*Tringa totanus*)	
Shelduck (*Tadorna tadorna*)	

However, despite the apparent large amounts of benthic prey consumed, the production rates of the benthos are so high in comparison with the absolute amounts consumed by predators (Raffaelli and Milne 1987; Kalejta, 1993; Heip et al., 1995; Little, 2000) that the effects of predation on prey populations may be insignificant overall. The corollary of this is that shorebird densities, and possibly those of fish and crustaceans, are probably limited by invertebrate biomasses on intertidal flats (Little, 2000), and it is likely that even a slight change in invertebrate production due to environmental change could have a significant impact on the carrying capacity of estuaries for shorebirds and fish. ‘Carrying capacity’ implies the maximum population of given organism that a particular environment can sustain. Although defining carrying capacity in absolute terms in ecological disciplines still remains vague and elusive (Little, 2000), the concept of carrying capacity often incorporates factors such as physical space, food and/or nutrient availability, biological interactions, mortality, time or temporal considerations (Dhondt, 1988; del Monte-Luna et al., 2004). For overwintering shorebirds in estuaries, West et al. (2005) suggest that a minimum food : bird ratio above which mortality does not vary and below which mortality increase steadily, be incorporated into management in order to maintain the ability of the site to support shorebirds at their present fitness level, and hence maintain the current carrying capacity of the site. From this point of view, change in the biomass and production of estuarine benthic invertebrates is an important component in maintenance of predator populations, and this will be especially so for the more productive intertidal, as compared to subtidal areas. An understanding of the likely responses of invertebrate assemblages to changes in key environmental elements associated with sea-level rise is therefore important.

1.5 Estuarine benthic macrofauna in relation to sediments and to intertidal morphology

1.5.1 Exposed estuarine shores and sandy intertidal flats

Fortunately, there is a long history of studies in which the distribution and the abundance of intertidal benthic assemblages have been related to water depth/tidal height and sediment particle size. The overall morphology and dynamics of the beach, captured by the dissipative and reflective spectrum (Table 1.6), is the predominant factor in controlling intertidal faunal assemblages on exposed sandy intertidal flats (Brown & McLachlan, 1990), often located at the outer region of an estuary. For instance, on the north and west coasts of Scotland, McIntyre (1970) found that on exposed sandy flats, isopods, such as *Eurydice pulchra*, dominate, but with moderating exposure, the proportion of polychaetes increases. He found the highest biomass on sheltered beaches dominated by molluscs, mainly the bivalve *Tellina tennuis*. Similarly, in his review of Australian, South African and north-east Pacific USA beaches, McLachlan (1990) found that species diversity showed a linear increase from reflective to dissipative beach state and from steep to flat slopes with abundance increasing logarithmically.

Table 1.6. General features of different beach types according to the beach morphodynamic state (after Brown & McLachlan, 1990)

	MODAL BEACH TYPE		
	DISSIPATIVE	INTERMEDIATE	REFLECTIVE
Energy source	Infragravity, standing waves and bores	Gravity, and infragravity waves, rips	Gravity and edge waves
Morphology	Flat, with multiple bars	Variable bars	Deep water inshore
Sand storage	Shores in surf zone	Shifts between surf zone and beach	Stores on beach
Dunes	Usually large	Intermediate	Usually small
Filtered volume*	Small	Intermediate	Large
Residence time*	About 24 h	6 to 24 h	About 6 h
Surf circulation	Vertical, bores on surface, undertow below	Horizontal cells	No surf zone. Mini - circulation within cusps
Surf-zone diatoms	Rich	Variable	None
Intertidal fauna	Rich	Variable	Poor

* Filtered volume is the volume of sea water flushed daily through the intertidal sand; residence time is the time it takes to percolate.

However, biomass was best correlated with wave energy rather than beach morphodynamic state, whereas individual biomass increased with exposure, suggesting the total faunal biomass on exposed and semi-exposed beaches may not differ as much as expected (Bally, 1981). In northwest Europe, mussel beds (*Mytilus edulis*) can form extensive biogenic reefs with substantial biomass on at lower shore levels, particularly at the mouths of meso- and macro-tidal estuaries where there is favourable tidal flow (Widdows & Brinsley, 2002). Apart from this patchy high biomass, the assemblages found in exposed estuarine sandy areas are generally characterised by a high diversity of types of organisms but neither a high biomass nor a high productivity. With decreasing exposure and increasing sediment stability, species richness, abundance and total biomass increase to reach a maximum on muddy sheltered shores.

1.5.2 Sheltered estuarine shores and mud flats

Sheltered shores are found in areas of low energy and they have poorly sorted sediments with high levels of organic matter and an high silt content (Dyer, 1979).

Fine particles tend to accumulate at higher shore levels and particle sizes become sandier towards lower shore levels to form coarse mobile sands in the subtidal river bed due to scour. The levels of biomass and production for estuarine benthic macrofauna within the mudflats are typically much higher than in subtidal areas (McLusky, 1989) and rates of annual production may differ by a factor of two (Heip et al., 1995). This high biomass and production is characteristically attributable to the large number of deposit- and filter-feeding invertebrates. Thus polychaete worms such as *Nerine* and *Ophelia* become more frequent in area of moderate disturbance, and with more sheltered conditions, burrowing bivalve molluscs, such as tellinids and large clams, and epibenthic mussels become abundant (Raffaelli & Hawkins, 1996). In addition, several tidal migrants occur including mysids, amphipods and decapods or drifting species associated with algal growths (e.g. *Neomysis integer*, *Melita obtusata*, *Dexamine spinosa*, *Stenothoe marina*, *Idotea* spp.).

The nature of the substratum can markedly affect the distribution patterns of benthic macrofauna within an intertidal area (McLusky, 1989). In Britain, for instance, Eltringham (1971) observed a three-zone pattern on muddy shores: an upper zone characterized by either a complete absence of fauna or by the polychaete *Nereis diversicolor*; a mid-tide zone of the bivalve molluscs *Scrobicularia plana*, *Cerastoderma edule* and *Macoma balthica*, and where some sand is present, the lugworm *Arenicola marina*; a low-shore zone with the bivalves *Mya arenaria* and possibly *Tellina fabula*, depending on the sediment particle size, along with a number of polychaete species.

The relationships between tidal level and abundance or biomass of macrofauna tend to show similar patterns between tidal flats in different locations (Beukema, 2002). For instance, in different parts of the Dutch Wadden Sea, Beukema (2002) showed that numerical abundance and biomass of macrofauna increased from values close to 0 at high water level to maximum values around mean-tide level or halfway between this level and low-tide level, then declined to low values towards the low water level, whereas mean biomass per individual increased from high- to low-water level. A similar trend was shown by Key (1983, in Jones, 1988) for mudflats on the Spurn Bight of the Humber estuary, UK. Here biomass increased from 26.8 g m^{-2} at the highest zone to the maximum biomass value of 32.7 g m^{-2} in the muddy midshore then decreased to 5.97 g m^{-2} in the sandy low shore.

It is clear that tidal height and sediment type can play an important role in shaping the distribution and abundance of the benthic macrofaunal assemblages in estuarine intertidal flats.

1.6 Effects of salinity on estuarine intertidal benthic macrofauna

In contrast to studies of macrofaunal biomass and production, many publications on the distribution range of species and the diversity of benthic assemblages in estuaries have focused on salinity (Boesch, 1977; Heip et al., 1995; Mees et al., 1995). Diversity generally declines on shores affected by low salinity (e.g. Dankers et al., 1981; Hardwick-Witman, 1983; Jones, 1988; Ysebaert et al., 2003), and in the case of Severn estuary UK, the body size of the lugworm *Arenicola marina* also diminishes with distance from the sea (Mettam, 1980). It has been suggested that the fauna can be divided into a series of communities with distance from the mouth up-estuary (Elliott & Kingston, 1987), and Little (2000) points out that many of the communities in the outer regions are related to sediment types, hydrology and depth, but in the inner estuarine areas, the very low diversities correlate with low salinity. Furthermore, in those studies where biomass over the entire estuarine area has been described, there is a trend from lower biomass in the upper estuarine regions to higher biomass in the more downstream (seaward) parts (e.g. Dankers et al., 1981; Meire et al., 1991; Dauer, 1993; Ysebaert et al., 2003). With respect to productivity, Edgar and Barrett (2002) showed that faunal biomass and estimated productivity were highly correlated with salinity at the low-tide and shallow subtidal level for Tasmanian estuaries. Salinity seems therefore to be a key determinant of macrofaunal abundance and distribution patterns in estuarine systems.

1.7 Effects of temperature on estuarine intertidal benthic macrofauna

Annual variability in climatic conditions will have strong influence on ecology of estuarine macrobenthic assemblages through changes in individual growth rates, fecundity and recruitment success. In their long-term study in the Wadden Sea in the Netherlands, Beukema et al. (1993) noted that year-to-year fluctuations were marked for almost all benthic species, which were known to be important food sources for higher trophic levels, than in total macrozoobenthic biomass. In the Balgzand area, 12 out of 29 species showed higher rates of mortality during cold than during mild winters, and several species such as *Cerastoderma edule*, *Nephtys hombergii*, and *Lanice conchilega*, showed minimal biomass values due to low over-winter survival during severe winters (Beukema et al., 1993). In addition, bivalve species such as *Macoma balthica* show a significant negative correlation between winter temperatures and subsequent recruitment, with higher *M. balthica* densities in the year following a cold winter, probably due to an increase in fecundity (Beukema et al., 1993; Beukema et al., 1998; Widdows & Brinsley 2002). Predatory juvenile fish and crustaceans can also be affected by the severe winters, and it is notable that recruitment of the cockle *Cerastoderma edule* is unusually high in the summers following widespread mortality of the consumers of their spat (Jensen & Jensen, 1985). The effects of climate change on intertidal macrobenthos are clearly complex at species level because these will involve a range of cause-and-effect relationships through their ecological interaction, but the possibility of milder winters and warmer summers in the future implies that the overall biomass of estuarine benthic assemblages may be expected to increase.

Changes in ambient temperature will also alter individual growth rates and other physiological functions of benthic organisms in estuarine environments. For example, Moens & Vincx (2000) investigated respiration and food assimilation of two estuarine nematodes at a range of temperatures, salinities and food densities, and found that temperature had a large effect on metabolic rate and production, showing increase with higher temperature up to 25 ℃. Secondary production in benthic invertebrates can be estimated directly, but such estimates are usually taxa-dependent, time-consuming and labour-intensive (González-Oreja & Saiz-Salinas, 1999). Indirect estimates based on empirical relations are available (Tumbiolo & Dowing 1994) and the latter reveals a strong effect of temperature on secondary production, suggesting the importance of incorporating changes in mean annual temperature into the estimates of biomass and production in order to predict the impact of climate change on the ecology of estuarine benthic assemblages. As yet it has not been possible to determine the degree to which any loss of invertebrate biomass may be compensated by these factors.

1.8 Effects of food supply on estuarine intertidal benthic macrofauna

There is increasing evidence that food supply regulates benthic biomass and secondary production by affecting individual growth and fecundity of adult invertebrates at large scales (Olafsson et al., 1994; Heip et al., 1995; Herman et al., 1999). For instance, Heip et al. (1995) examined the relationships between pelagic primary production, advection of organic matter, sedimentation of organic carbon, and benthic biomass and productivity, and they found a significant dependence of total system biomass of commercial benthic suspension feeders on the residence time of the water in the system. The model assumed that residence was an inverse measure of food exchange with the coastal sea, and that system productivity was the basic limiting factor for suspension feeder biomass which is often the dominant component of estuarine benthic assemblages. In the case of Balgzand area of the Dutch Wadden Sea, Beukema and Cadée (1997) showed that a substantial increase of pelagic primary production between 1970s and the 1980s was followed by a nearly proportional increase of system-averaged benthic biomass. Similarly, in a comparison of 15 different estuarine systems throughout the world, Herman et al. (1999) found that system-averaged macrofaunal biomass was significantly correlated with the system primary production. However, primary production itself is controlled by such factors as light, temperature (Barranguet et al., 1998), nutrients (Heip et al. 1995), and anthropogenic activities (e.g. discharge, dredging) (de Jonge, 2000). It is therefore important to understand how in the long term sea-level rise will affect any of these factors, which in turn regulate primary productivity and thus benthic macrofaunal biomass within an estuary at the whole system scale.

1.9 Implications of sea-level rise for estuarine ecology and shoreline management

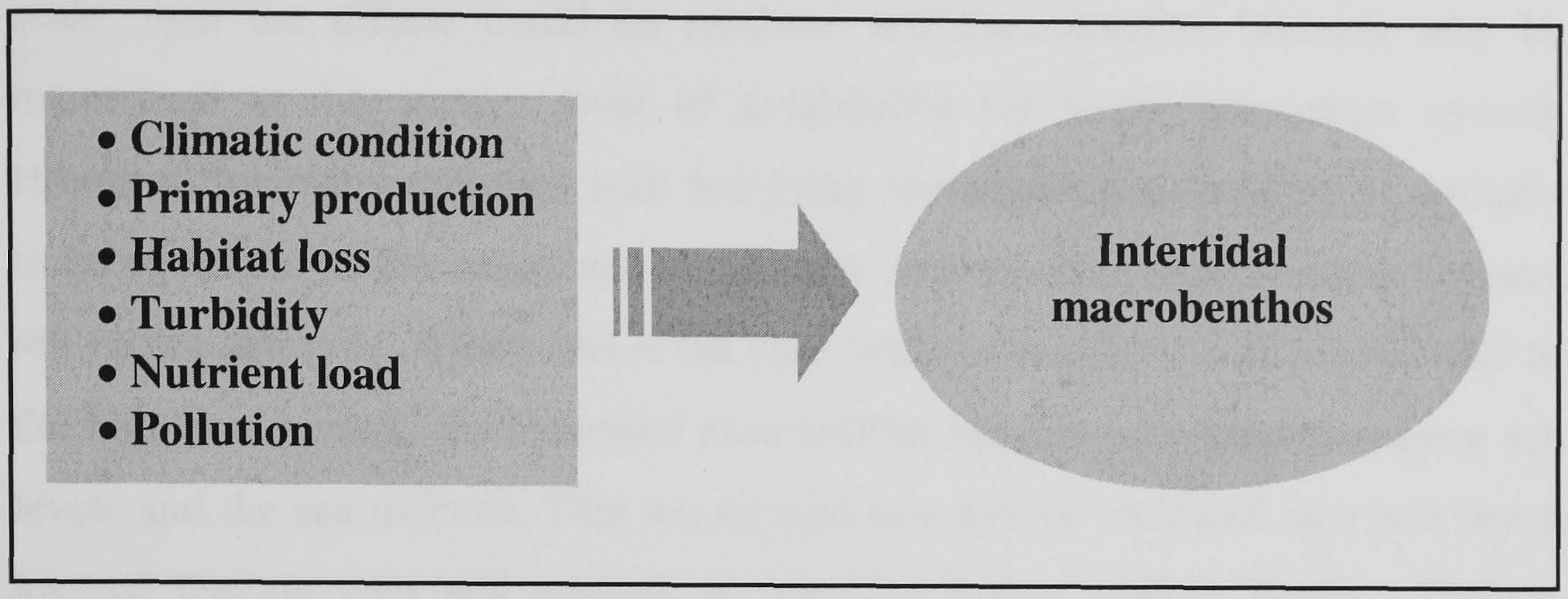
The above review indicates that benthic macrofauna plays a pivotal role in estuarine trophic interactions, and the biomass of the macrobenthos is likely to be affected by a number of environmental variables. Figure 1.1 shows conceptual models of relations between environmental conditions and macrobenthic biomass within a typical estuary. At the whole estuary scale, the total macrobenthic biomass available for higher trophic levels is affected by such factors as climatic conditions, level of nutrient supply and turbidity (Fig. 1.1.A). At this scale, sea-level rise will affect the amount of estuarine macrobenthos through coastal squeeze, leading to a reduction of intertidal habitats.

Within an estuary, along the longitudinal gradient, macrobenthic biomass per unit area is controlled mainly by salinity and sediment particle size (Fig. 1.1.B). Here, biomass is negatively affected by low salinity towards fresh water and by the coarser sediment particle sizes which tends to accumulate near the mouth of the estuary. At this scale, sea-level rise will affect macrobenthic biomass through changes in salinity and particle size distributions.

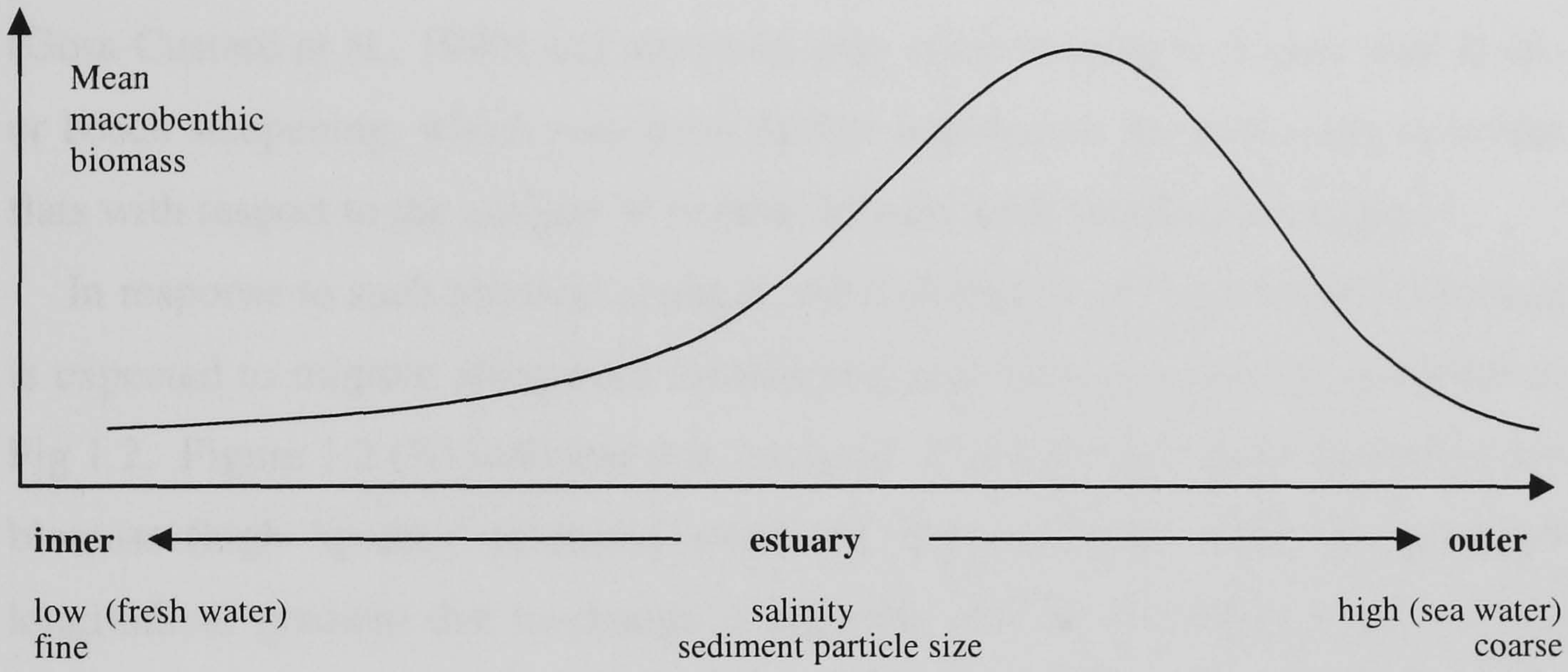
Finally, along the intertidal vertical gradient from mean high to mean low water level, macrobenthic biomass per unit area is affected by the tidal height (Fig. 1.1.C) at any point of the longitudinal gradient. At this scale, sea-level rise will affect macrobenthic biomass through changes in beach profile or morphology resulting from increased wave energy and associated sediment redistribution.

These models indicate that there is a nested relationship between the three conceptual models from whole system through longitudinal to local vertical scales. Thus the amount of biomass available for higher trophic levels can be viewed differently depending on what scale is used in coastal management.

(A) at estuarine system scale



(B) along estuarine longitudinal gradient



(C) along estuarine intertidal vertical gradient

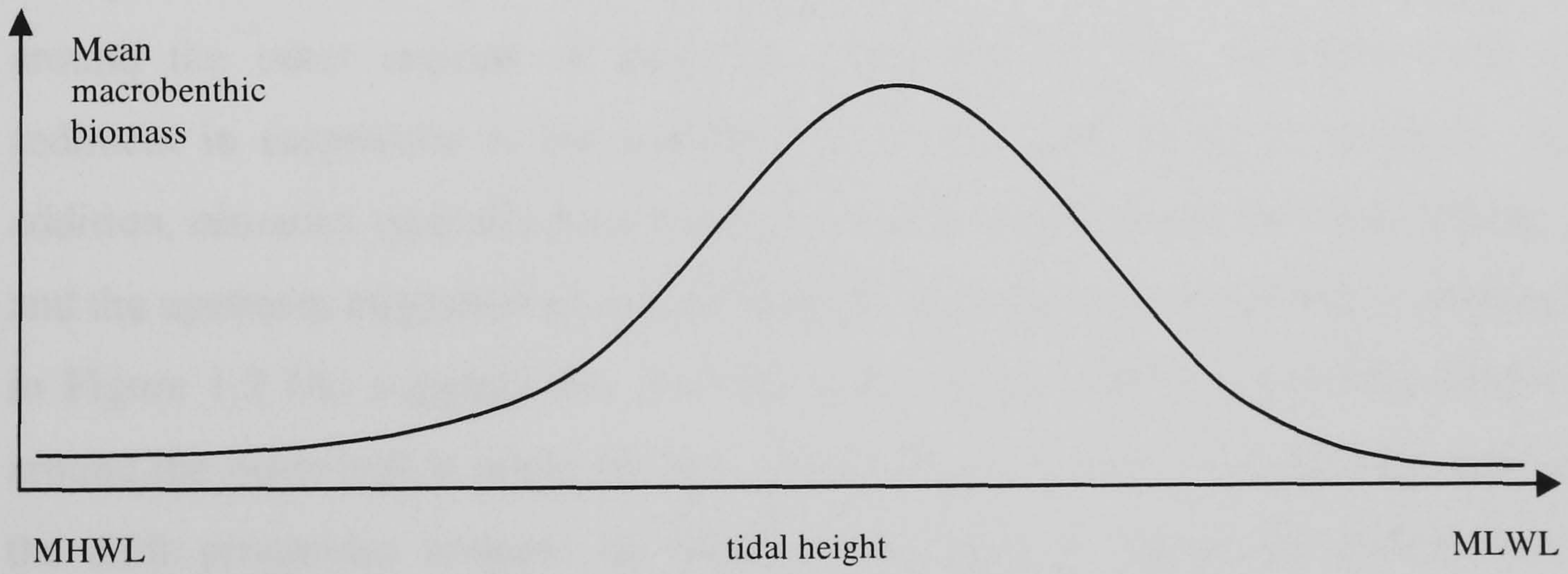
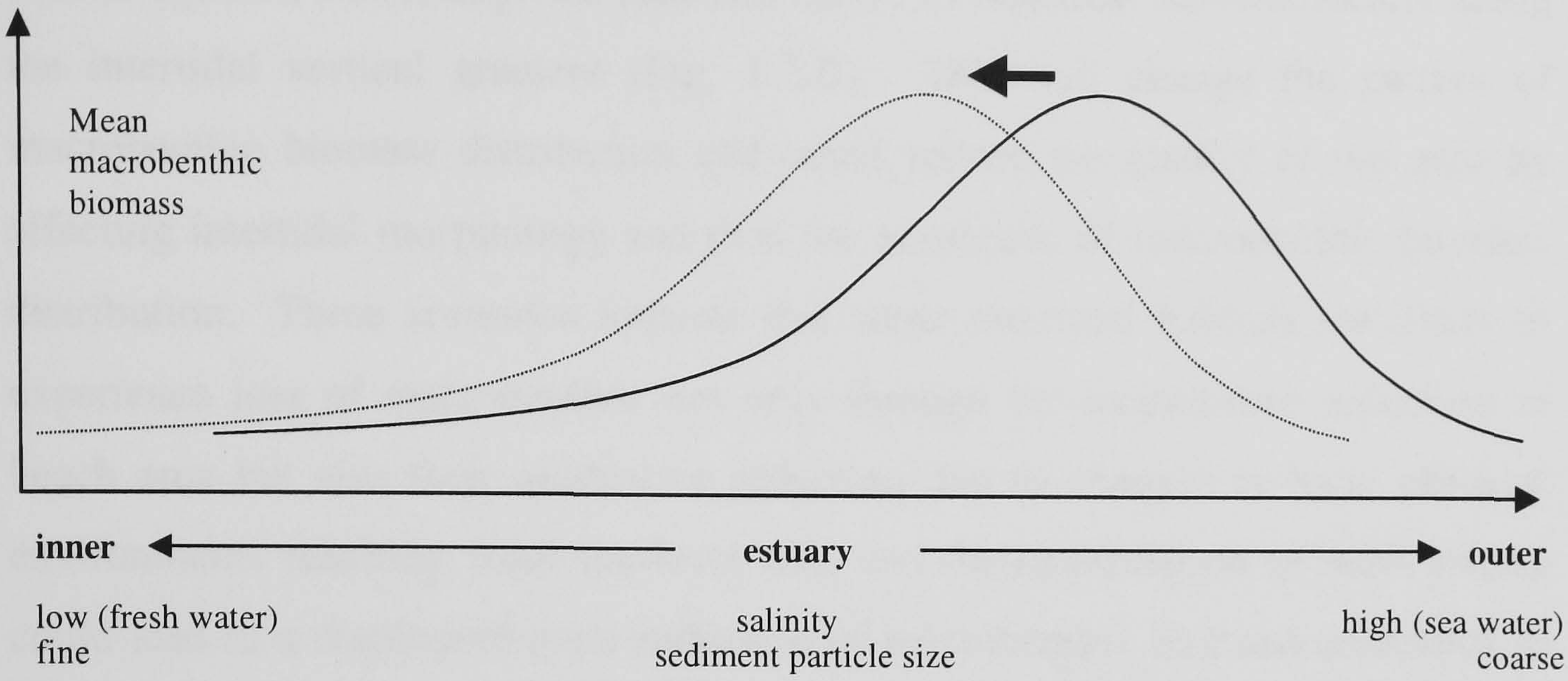


Fig. 1.1. Conceptual representation of relations between estuarine intertidal macrobenthic biomass and various environmental components: (A) at estuarine system scale; (B) along longitudinal gradient within an estuary; (C) along intertidal vertical gradient at any point of an estuary. MHWL and MLWL represent mean high water level and mean low water level, respectively.

For example, if the estuary is allowed to transgress inland at the whole system scale, then the effects could be minimal and macrobenthic biomass may be maintained at the current level of availability for larger consumers overall. However, for many estuaries with low-lying hinterland, transgression is unlikely to be permitted as the adaptive management strategy may take the form of more robust sea defences. Where this is the case, and the sea defence is located near to the high water mark, the intertidal area will be squeezed between the rising sea levels and the sea defence. This would lead to a loss of intertidal area and hence loss of feeding area and amount of prey for higher trophic levels. Further, sediment erosion may increase mainly through a more energetic wave climate, and the fine material suspended would then be removed to low energy sinks (Goss-Custard et al., 1990) and accretion may occur leading to higher tidal levels or beach steepening, which may even further impoverish the remaining intertidal flats with respect to the amount of benthic infauna such beaches can support.

In response to such physical changes, the distribution of macrobenthic biomass is expected to migrate along both longitudinal and vertical gradients as shown in Fig 1.2. Figure 1.2 (A) indicates that intertidal area with high mean macrobenthic biomass (high 'quality' intertidal area) will shift upstream along the estuarine longitudinal gradient due to change in sediment particle distribution and salinity regime. Here, given the relationship between sediment organic content and particle size, the abundance of deposit feeders is likely to decline with the loss of fine particles, and suspension feeders may increase in abundance, particularly around the outer regions of estuaries, assuming that any increased load of sediment in suspension is not sufficient to reduce filter feeding efficiency. In addition, estuaries typically have larger intertidal areas towards the outer regions, and the upstream migration of overall macrobenthic biomass distribution indicated in Figure 1.2 (A) suggests that the high quality area which is currently located around the outer region could become constrained to a narrower intertidal area as the shift progresses towards up estuary, leading to a significant reduction of overall macrobenthic biomass at the scale of the whole estuarine system.

(A) along estuarine longitudinal gradient



(B) along estuarine intertidal vertical gradient

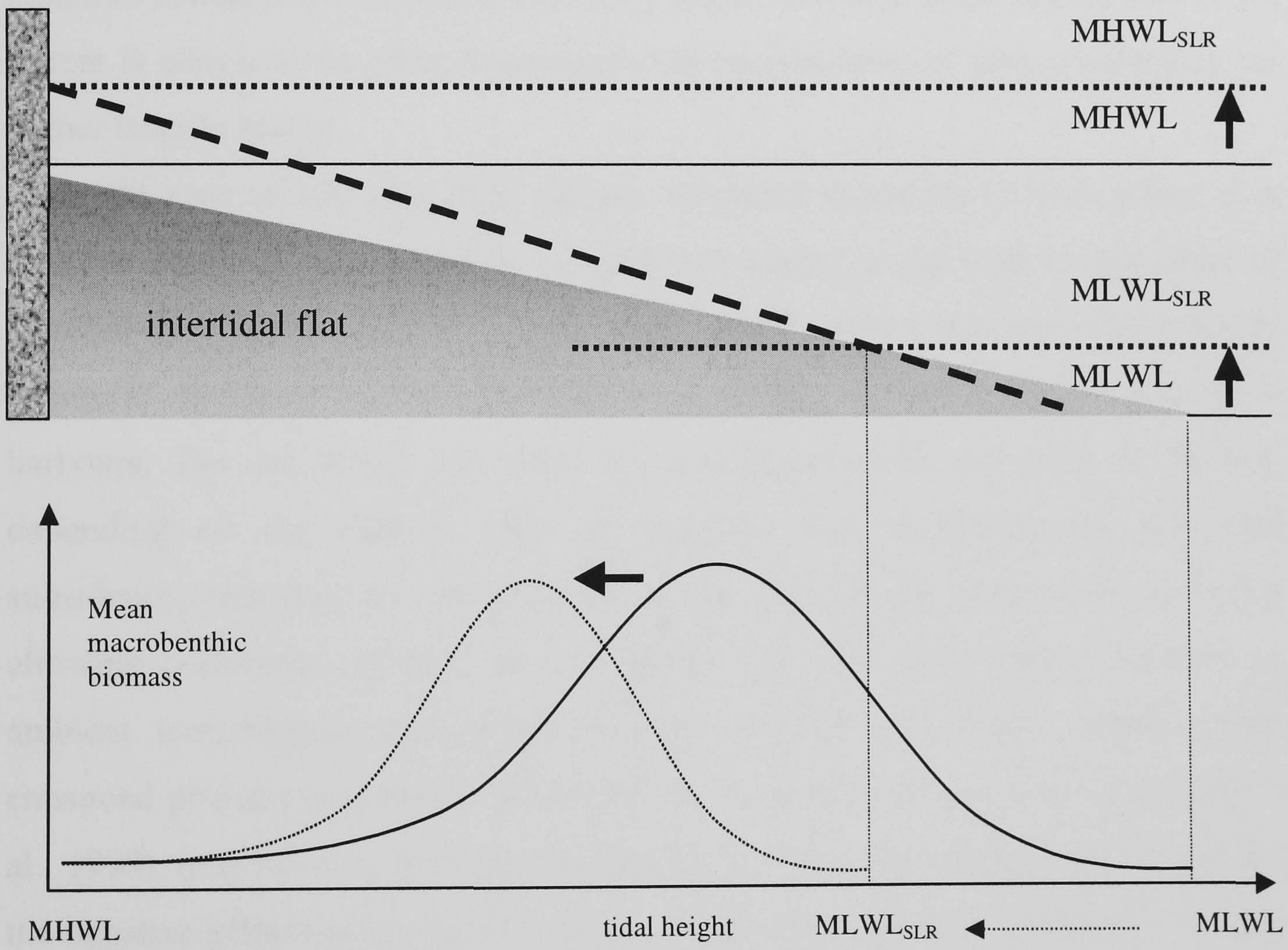


Fig. 1.2. Two conceptual models illustrating possible shifts of mean macrobenthic biomass distributions resulting from sea-level rise: (A) along estuarine longitudinal gradient due to salinity intrusion up river and increased particle sizes towards mouth; (B) along estuarine vertical gradient over intertidal flat due to change in intertidal profile. In diagram (B), the horizontal solid and dotted lines represent current and newly established mean high and low water levels in response to sea-level rise, respectively. The dashed line represents newly established intertidal profile. The vertical patterned strip behind the intertidal flat represents sea wall defence.

Further, intertidal steepening and associated sediment redistribution caused by coastal squeeze will change the area and nature of intertidal habitats locally along the intertidal vertical gradient (Fig. 1.2.B). This will change the pattern of macrobenthic biomass distribution and could reduce the quality of the area by affecting intertidal morphology and thus the amplitude of macrobenthic biomass distribution. These scenarios indicate that some intertidal habitats are likely to experience loss of macrobenthos not only through the quantitative reduction in beach area but also their qualitative reduction due to changes in local physical environments resulting from sea-level rise, and the combination of such effects could lead to a disproportionate reduction of macrobenthic biomass compared to an amount straightforward loss of intertidal area would suggest in an estuarine ecosystem. In the context of shoreline management, it would therefore be essential to consider carefully and exactly where to retreat to as well as how much retreat is necessary in order to maintain the current level of prey availability for higher trophic levels.

In the case of estuaries with narrow, restricted entrances or with a bar, it is possible that a general increase in sediment supply could lead to accretion of material on intertidal flats as the suspended matter enters a less hydrodynamically energetic environment (Goss-Custard et al 1990), as seen in many man-made harbours. The net effect here could be an increase in the elevation of the flat, depending on the relative rates of sea-level rise, sedimentation and land subsidence, reducing the productivity of the flats in the long term and even allowing conversion of flats to salt marsh. On the other hand, increases in ambient temperatures and milder winters, at least in Europe, together with enhanced primary production stimulated by increased nitrogen run off (Smith et al., 1999), may serve to increase invertebrate biomass and productivity and offset the negative effects of changes in beach morphodynamic state, although it is very difficult to quantify such possibilities at present. However, from what has been described thus far, it seems clear that the abundance and biomass of invertebrate assemblages are strongly associated with physical factors within an estuarine ecosystem when such long-term factors (increase in ambient temperatures or nutrient inputs) are held constant, and the conceptual models created in this chapter provide a good starting point to cope with the threat of sea-level rise from which policy makers can begin to accommodate a concept of ecological sustainability in estuarine systems. The models can also provide a useful framework from which testable hypothesis can be derived and from which much can be learned for future integrated coastal and estuarine management.

1.10 Aims and overview of the thesis

Many estuarine intertidal habitats have high conservation value because of their utilisation of shorebirds and fish, and it is essential to protect such habitats to maintain the availability of macrobenthic prey, which should be built into the process of decision making in coastal and estuarine management. Intertidal habitats are dynamic features which depend on the interactions between wind direction, wave energy, tidal movements and sediment transportation and over time, these habitats and features change and migrate land-wards as sea-level rises. Thus the effective conservation of such areas involves providing sufficient space to allow them to move and evolve in response to the changing environmental conditions (Jones, 1994). On the other hand, rising sea levels are a serious threat to the human communities living on estuaries as this will inevitably increase the risk of flooding and inundation. Therefore, where urban, industrial or high-grade agricultural development exists, the most likely option for coastal management will be the reinforcement of the existing sea defence walls ("hold the line" approach) to reduce the risk of flooding, even though this may be unsustainable both economically and physically because of increased storminess and altered wave energy patterns. For nature conservation, however, the best management option will be achieved by setting back flood defences in the estuary ("managed realignment") because this will allow space for intertidal habitats to evolve and migrate inland, and will create natural intertidal flats and salt marshes that also effectively reduce wave energy and therefore reduce the maintenance cost of flood defences (Winn et al., 2003). Clearly, there is a need to find appropriate ways to manage the estuarine and coastal environments that sustain both nature conservation interest and socio-economic needs. Taking these issues into account, two relevant research questions arise as follows:

- What will be the consequences for the biomass of macrobenthos and populations of shorebirds, fish and shrimps if society maintains the "holding the line" policy on the basis of different sea-level rise scenarios over the next 50 years?
- If managers are to create new intertidal habitats, how can they identify the locations and amount of suitable sites around the estuary where "managed realignment" will help maintain the estuary's ecological integrity and meet long-term flood defence needs?

The aims of the thesis are thus as follows:

- to investigate how the spatial patterns of the intertidal macrobenthos is related to key environmental variables in the Humber estuary, UK;
- to investigate the likely changes in key environmental variables in response to sea-level rise and hence their impacts on macrobenthic biomass;
- to consider how the changes in the macrobenthic biomass may affect the trophic interactions in an estuarine ecosystem;
- to consider how to maximise environmental benefits in the long term in the context of future shoreline management in estuarine environments.

The Humber estuary has been selected because the estuary:

- is one of the largest estuaries in the UK;
- has significant societal infrastructure (ports, urban and agricultural areas, etc);
- is vulnerable to flooding from an accelerating rate of sea-level rise;
- is of high conservation status for shorebirds.

To fulfil the research aims, the basic hypothesis to be examined in this thesis are:

- The observed spatial variation in macrobenthic biomass in an estuarine intertidal area can be explained by key environmental variables such as salinity, sediment particle size and beach morphology;
- Coastal squeeze affects key environmental variables and thus macrobenthic biomass in estuarine intertidal habitats;
- The amount and distribution of the prey (macrobenthic biomass) at the beginning of wintering period are the major determinants of the density and distribution of intertidal dependant birds;
- Intertidal habitats are important feeding grounds for fish species that utilise the estuarine environment.

To answer these questions, a research programme was structured as reflected in the individual chapters of this thesis.

Chapter 2 quantifies the key environmental and biological variables based on an extensive field survey in the Humber estuary in order to test the conceptual model described in Chapter 1 (Fig. 1.1), and examines the role of the key environmental variables in explaining the observed variation of intertidal macrobenthic biomass.

Chapter 3 investigates how physical and biological variables are likely to change in response to coastal squeeze (Chapter 1, Fig. 1.2), based on a field survey comparing an intertidal flat that has already been squeezed due to historic land-claims with the adjacent natural (non-squeezed) intertidal habitat. This chapter provides parameters for statistical models aimed at predicting changes in intertidal macrobenthic biomass and allows a simulation of how much of intertidal macrobenthic biomass will change in 50 years time as a result of sea-level rise in the Humber estuary.

Chapter 4 investigates how the Humber estuary is utilised by shorebirds and fish and how the potential change in their prey items in response to sea-level rise may impact on these populations.

Chapter 5 describes the current status of shoreline management in the Humber estuary and considers what sort of issues or constraints affects the development and decision making processes of shoreline management. This chapter also makes recommendations for effective and sustainable estuarine coastal management in order to maximise environmental benefits.

Chapter 6 concludes this thesis, assessing how the research objectives have been fulfilled and makes suggestions for further study and future work.

Chapter 2

Spatial patterns of benthic macrofauna in relation to environmental variables in an intertidal habitat in the Humber estuary, UK

2.1 Introduction

Intertidal flats within an estuary exhibit significant variability in benthic macrofaunal species composition, density and biomass, and there is a long history of investigations in which this variability has been related to such environmental variables as salinity, sediment types and tidal level (McIntyre, 1970; Jones, 1988; McLusky, 1989; Meire et al., 1991; Dauer, 1993; Elliott et al. 1998; Beukema, 2002; Ysebaert & Herman, 2002; Ysebaert et al., 2003). However, there have been few attempts to model the relations between benthic invertebrate biomass and these environmental variables at a whole system scale in order to predict the likely responses to accelerated sea-level rise. In the case of the Humber estuary in the UK, the rate of sea-level rise relative to land has been between 2 and 2.5mm per year over the last 100 years (Winn et al., 2003), but the Ministry of Agriculture, Fisheries and Food (MAFF, 1999) has recommended that for planning purposes an average rate of 6 mm per year should be assumed for the next 50 years, implying that sea levels may rise by a total of 0.3 m over that period. As shown in Chapter 1, this accelerating rise of sea level poses serious threats to the estuarine intertidal habitats, in particular, a reduction in the spatial coverage of these habitats (Jones, 1994; Galbraith et al., 2002), salt intrusion which would shift the salinity distribution and turbidity maximum further upstream (Jones, 1994; Scavia et al., 2002), and changes in particle size composition of sediment through the shift of morphodynamic state of the flats from dissipative with fine particles to reflective with coarser particle conditions (Goss-Custard et al., 1990; Raffaelli & Hawkins, 1996). In addition, the frequency of extreme events, such as storm surges, is expected to increase. These predicted physical changes will have significant implications for intertidal macro benthos, and thus their predators.

In the face of such problems relating to sea-level rise, there is increasing interest from conservation and management agencies for reliable predictive tools for planning the sustainable use of coastal systems. One approach is to create a model which can make quantitative predictions of how intertidal habitats and their macrobenthic biomass may change in response to changes in key environmental variables induced by sea-level rise. Such a model can be used to decide how much land needs to be set aside for re-creation of intertidal habitats and where such schemes would be most effective.

The aim of this chapter is to develop a method for predicting the response of intertidal macrobenthic species to sea-level rise with particular emphasis on their total biomass. First, patterns in the distribution, abundance and biomass of benthic macrofauna in the Humber estuary are described, and this is used to investigate the role of environmental variables in explaining the observed variability within the system using correlation and multiple regressing analyses over a variety of spatial scales based on a hierarchically scaled field study. The rules which best link values of key environmental variables with macrobenthic biomass at the most relevant spatial scales were then used to establish a preliminary model for predicting how the amount of macrobenthos in the system would be affected by selected key environmental variables. Implications of this approach are then discussed.

2.2 Material and methods

2.2.1 Study site

The sites sampled were estuarine intertidal flats situated along the Humber on the east coast of England which forms the boundary between Yorkshire and Lincolnshire and flows into the North Sea (Fig. 2.1). The mean tidal range is approximately 5 m and maximum spring tide range can attain over 7 m, being one of the largest macro-tidal estuaries in the UK. A marked increase in width towards the sea results in an approximately 8 km-wide mouth at Spurn Head which decreases to less than 0.5 km at both the river Trent and river Ouse. The Humber has a catchment area of 24,472 km^2, a fifth of the area of England, and the tidal waters have a length of 317 km (Winn et al., 2003), the Humber itself stretching approximately 60 km from the confluence of the Trent and the Ouse to the mouth at Spurn. In this study, the mouth of the Humber is defined as a midpoint between Spurn Head on the north bank and Tetney Haven on the south (OS grid ref. 538000, 408000). The system supports a large area of intertidal habitat particularly towards its mouth, estimated at 120 km^2, 90 % of which comprises mudflat and sandflat (Winn et al., 2003). However, in areas with extensive sea defence walls and commercial development such as around Hull and Grimsby, tidal flats are narrow or absent because of truncation by sea defence walls (Coombes et al., 2004).

The study site can be divided into 4 large sections (upper, middle, lower and outer region) defined by Barr et al. (1990) (solid lines on Fig 2.1.b), and sediment types vary considerably from relatively sandy particles with some fringing fine mud in the upper region, through fine mud in the middle and lower regions, to coarse sand towards the outer region. Sampling sites were thus carefully chosen to cover the range of environmental gradients. I selected 14 transects (7 transects on the north and 7 transects on the south side of the Humber) so that they were evenly distributed along the longitudinal gradient over the four estuarine regions within the system (Fig. 2.1.b).

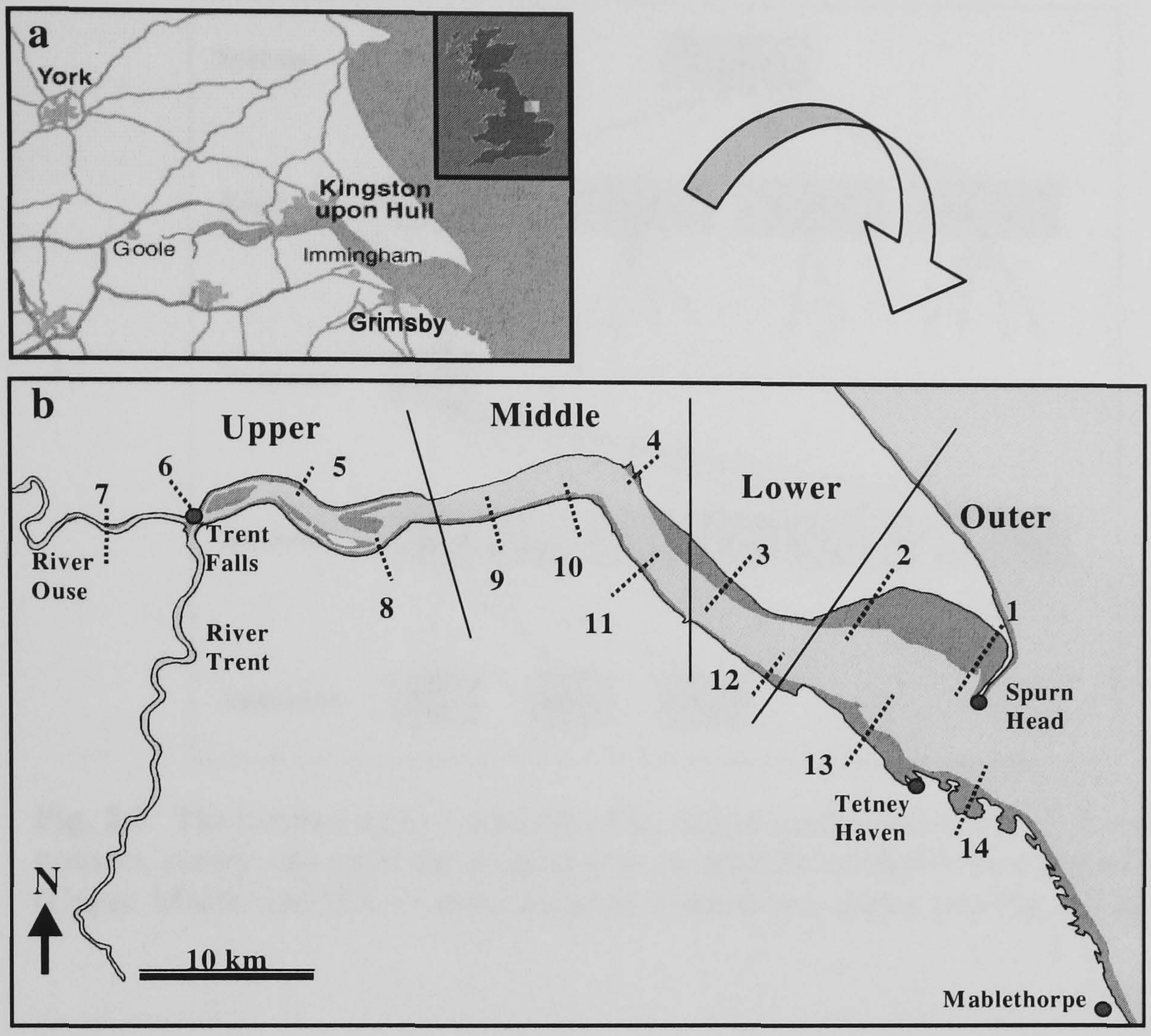


Fig. 2.1. (a) The Humber estuary, and (b) location of transects (numbers with dotted line) within the study site at low tide. Dark shaded area represents intertidal area and long solid line represents a boundary between sections (see text).

Each transect was established so that it ran from MHWL (mean high water level) to MLWL (mean low water level) aligned along the direction of tidal ebb. Both MHWL and MLWL were determined by consulting Ordnance Survey 1:25,000 scale map revised in 2000, and 7 sampling stations were selected along the transect at equal intervals over the intertidal profile. Thus, total of 98 stations (14 transects × 7 stations) were chosen throughout the study site, and sampling was replicated 3 times at each station.

This sampling design was hierarchically scaled, covering 5 different spatial scales: system (10^5 m), region (10^5-10^4m), transect (10^2-10^3m), station (10^1-10^2m), and replicate (10^0-10^1m) (Fig. 2.2). All the field sampling and survey were carried out from 8th to 27th of September 2003, except for one transect at Patrington (transect 2 on Fig. 2.1.b) where the field work was conducted on 25th October 2003.

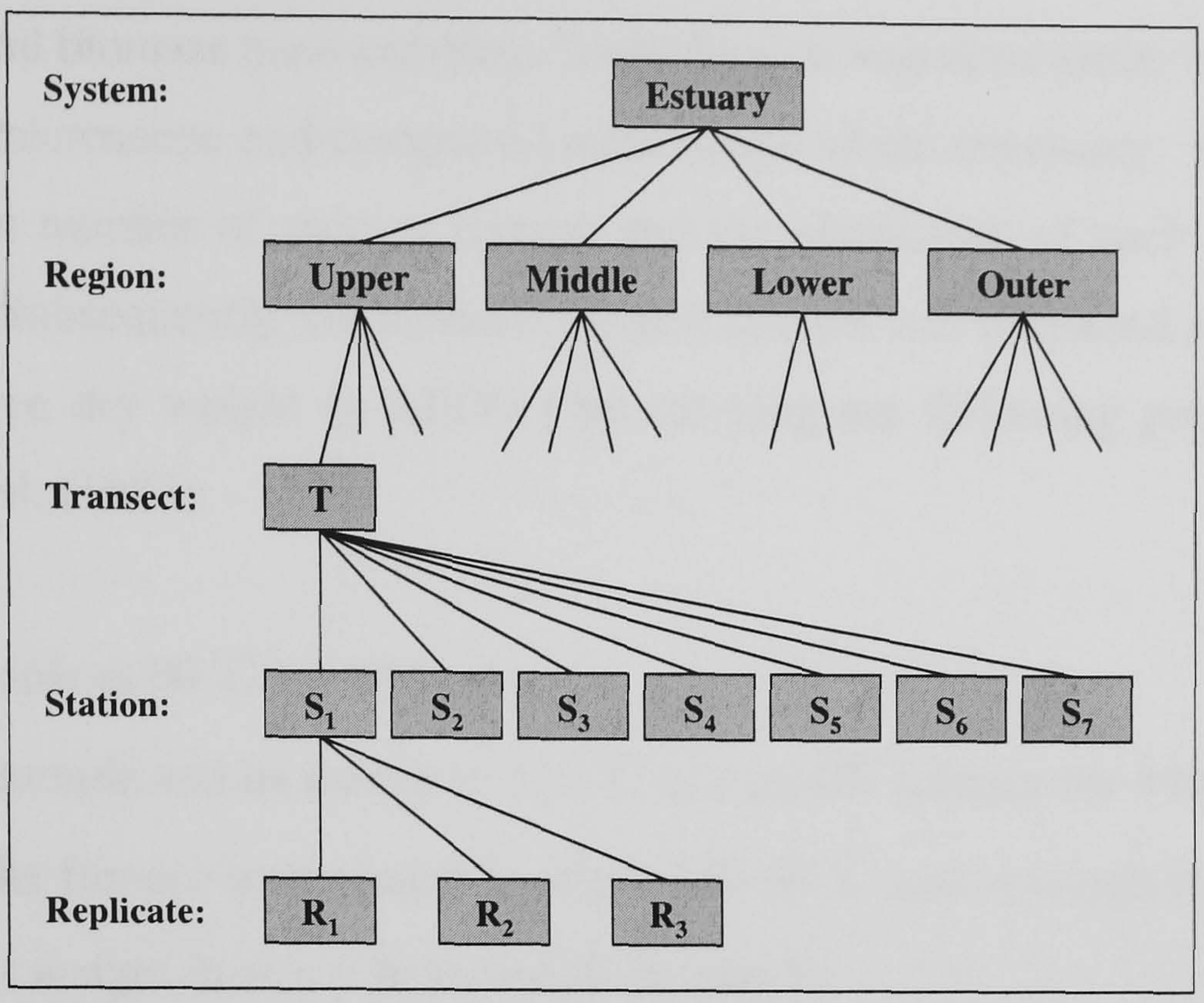


Fig. 2.2. The hierarchically scaled sampling design used in this study. T, S and R indicate transect, station and replicate, respectively. 4 intertidal transects were placed in 3 regions (Upper, Middle and Outer), and 2 transects in the Lower region (see Fig. 2.1.b).

However, only 5 out of 7 stations were sampled along two of the transects due to difficulty in working on extremely deep mud in the lower part of Kilnsea (transect 1 on Fig.2.1.b), and due to much shorter length of transect than expected from a consulted map at South Ferriby (transect 8 on Fig. 2.1.b). All the details of each transect can be found in Appendix 1.

2.2.2 Biological measurements and sampling

At each sampling station, a cylindrical corer (10 cm in diameter) was pushed into the sediment to the depth of 15cm on a randomly chosen surface to sample the benthic macrofauna. This material was sieved on a 0.86 mm mesh with filtered sea water (with a 0.063 mm mesh) on site. (The lower size limit of macrofauna is usually regarded as 0.5 or 1.0 mm, and therefore meshes that are 0.5 and 1.0 mm are widely used in tidal flat literature (Hartley et al., 1987). However, the 0.86 mm mesh was used in this study due to a matter of availability.) This procedure was replicated three times and, on return to laboratory, the organisms collected were preserved in 70% ethanol for subsequent sorting, species identification, counting and biomass measurements. Identification was done either by eye or by a binocular microscope and compound microscope where necessary. For each core sample, the number of species present and the abundance of each species were recorded. Subsequently, the biomass of each species was measured and expressed in g ash-free dry weight (g AFDW) by carrying out following procedure, after Hartley et al. (1987):

1) dry sample at 90 ℃ for 48 hours;
2) weigh sample and incinerate at 550 ℃ in a muffle furnace for 3 hours;
3) allow the furnace temperature to drop to 80-90 ℃ and re-weigh the sample;
4) subtract weight in step 3 from weight in step 1.

2.2.3 Physical measurements and sampling

2.2.3.1 Longitudinal elements

Monthly measurements over at 40 monitoring locations along the Humber estuary were used for estimates of salinity value at each transect. The data were derived from the coastal (C) component of the Rivers-Atmosphere-Coast Study, RACS(C), of NERC's Land-Ocean Interaction Study (LOIS) programme, and, in the programme, the locations of the monitoring stations were chosen to cover the salinity range from fresh to coastal and to take into account any possible lateral heterogeneity in the entire Humber system (Uncles et al., 1998). Salinity was measured at a depth of 1 m at each of the monitoring stations, and surveys were undertaken during spring tides, except for May and December 1994 and March 1995 when surveys were conducted during neap tides (Uncles et al., 1998). Salinity values used in subsequent analysis were the average salinity of the period of 12 months between March 1994 and March 1995 at monitoring stations located over each of the transects established in this study. This was the most recent detailed salinity data set available for the Humber estuary, and thus differences in monthly average salinity between the above period and year 2003 (survey year in this study) could not be examined. Wave exposure for each transect was defined as a simple open angle of the shore (midpoint of each transect) to, or subtended by, the open sea horizon, expressed in radians (after Baker & Crothers, 1987).

2.2.3.2 Sedimentary elements

Three replicate core samples (30 mm diameter) were taken from the top 5 cm of the sediment at each sampling station for sediment analysis at the same time as the biological samples were collected. Organic matter content of these samples was measured according to the following procedure, after Sutherland (1996):

1) grind a dried sediment sample of approximately 10 g with a pestle and mortar;
2) transfer the sample into a crucible which has been weighed beforehand and oven-dry the sample at 90 ℃ until constant weight, then weigh the sample;
3) incinerate the sample at 375 ℃ in a muffle furnace for 16 hours;
4) allow the furnace temperature to drop to 80-90℃ and re-weigh the sample.

The loss on weight by ignition is the organic matter content of the sample.

Particle size composition was determined by both wet sieving (for particle sizes smaller than 0.063 mm) and dry sieving (for particle sizes larger than 0.063 mm). Wet sieving was carried out by washing a weighed amount of sediment on a 0.063 mm mesh sieve placed in a basin until no further material is seen to pass through. The sediment retained was then dried and re-weighed to provide the silt content fraction. The dried material remaining on the 0.063 mm sieve was then used for dry sieving by passing it through a tower of successive sieves with larger mesh sizes. Finally, the median particle size expressed in the Wentworth scale (phi) was derived graphically as defined in Holme & McIntyre (1971). Here, the median particle size was determined from the cumulative curve by reading the phi value which corresponds to the point where the 50 per cent line crosses the cumulative curve. However, the fraction of particle sizes < 0.031 mm (> 5 phi) could not be examined in this study and therefore median particle sizes which fell < 0.063 mm (> 4 phi) may be subject to slight change depending on the percentage fraction of particles < 0.031 mm (the more fraction of particles < 0.031

mm, the smaller median particle sizes (phi), when they are > 4 phi).

2.2.3.3 Morphological elements

The tidal depth (elevation) of each station in relation to mean high water level (MHWL) was measured by a theodolite and a staff. The length of the staff was 4 m and the MHWL at each site was determined by the height of the point where a marked line of algal growth or entangling dried organic matter (drift line) was uniformly found on sea defence walls, beaches, or fringing saltmarshes. Because tidal range varies along the longitudinal gradient, tidal depth for each station was standardised by taking a percentage of the depth measurement in relation to the local mean tidal range observed at each transect. The local mean tidal range was calculated as:

Mean tidal range = (mean maximum tidal amplitude + mean minimum tidal amplitude) / 2

and tidal depth thus can be expressed 0 % and 100 % if the station is located at MHWL and at MLWL, respectively. In addition, the median depth of each transect was determined from the intertidal profile by reading the tidal depth which corresponds to the point where the horizontal distance from the MHWL reaches 50 % of the total width of the beach. The median depth indicates that if the value is larger than 50 %, the beach profile is concave, but if the value is smaller than 50 %, then the shape of the beach is convex. Beach width was measured as the distance between mean high water level (MHWL) and mean low water level (MLWL) aligned along the direction of tidal ebb taken from the Ordnance Survey 1:25000 scale map revised in 2000. Because MLWL does not emerge during neap tides, field sampling was carried out only around the period of spring tides when the pre-established transects were fully exposed at ebb tide. In this study, measures of beach slope were obtained at transect scale (transect slope) and at station scale (station slope). Transect slope was measured differently depending on where MHWL was located in relation to the highest level of beach:

Transect slope = $-\text{Log}_{10}$ (mean tidal range / beach width)

where MHWL is located on the beach (sedimentary part), or

Transect slope = $-\text{Log}_{10}$ (height between top of the shore and MLWL / beach width)

where MHWL or the drift line is found up on the sea defence wall.

Secondly, the slope at each station was calculated as follows:

Station slope$_i$ = $-\text{Log}_{10}$ (relative vertical height ($S_{i-1} - S_{i+1}$) / relative width ($S_{i+1} - S_{i-1}$))

where S denotes station and i represents an arbitrary station number. The slope values calculated in this study typically fall in a range between 1 and 5, higher values indicating shallower slopes.

2.2.4 Statistical analysis

Macrobenthic abundance and biomass data were expressed in numbers m^{-2} (ind. m^{-2}) and g Ash Free Dry Weight (g AFDW m^{-2}), respectively. The general trends in species richness, abundance and biomass of macrobenthos and physical characteristics were examined mainly in relation to the longitudinal and intertidal vertical gradients of the Humber estuary using univariate analyses over various spatial scales. Because the relation between tidal depth and macrobenthic biomass distribution was expected to be non-linear (Chapter 1, Fig. 1.1.C), special attention was paid to find the depth of tidal level at which peak biomass can be found over the intertidal profile. Subsequently, a new variable, "depth index", was established to express how the value of tidal depth at any station deviates from the maximum biomass level (see section 2.3.2.3). The relationships between physical variables and macrobenthic biomass were first investigated by Spearman rank correlation. Multiple regressions were then used to identify the role of the measured environmental variables in explaining the observed spatial variability in macrobenthic biomass at the station and transect scales. For these statistical processes, the biomass data were log(1000x + 1) transformed prior to analysis due to their non-normality and heterogeneity of variance in most cases. The 1000x scalar was used because this generated the best normal distribution of the data. Data for salinity were normalised by natural log transformation. The environmental variables were divided into three physical components: longitudinal (salinity and exposure), sedimentary (median particle size, silt content, organic matter content) and morphological (beach width, station slope, transect slope, tidal depth, median tidal depth, and depth index). A forward step procedure was used to determine the subset of environmental variables that best explained the observed variation in macrobenthic biomass. All graphics were produced on Excel or SPSS for Windows and all statistical analyses were performed with SPSS for Windows. From the statistical analyses described above, key environmental variables at the most relevant spatial scale were chosen to develop a preliminary model to predict the amount of total macrobenthic biomass in the Humber estuary.

2.3 Results

2.3.1 Physical characteristics of the Humber estuary

2.3.1.1 Salinity and tidal range

Mean salinity values over 12 months for 14 transects in relation to distance from mouth are shown in Fig. 2.3. Salinity decreased steadily from 30 psu at the mouth to 2.6 psu in the upper most region. Within the Humber estuary, mean salinity over the year can be expressed as a quadratic equation as shown in Fig. 2.3, and the apparent low salinity around the mouth (28.38 when $x = 0$) may be a reflection of winter months when the Humber has significantly higher freshwater inflow (Uncles et al., 1998).

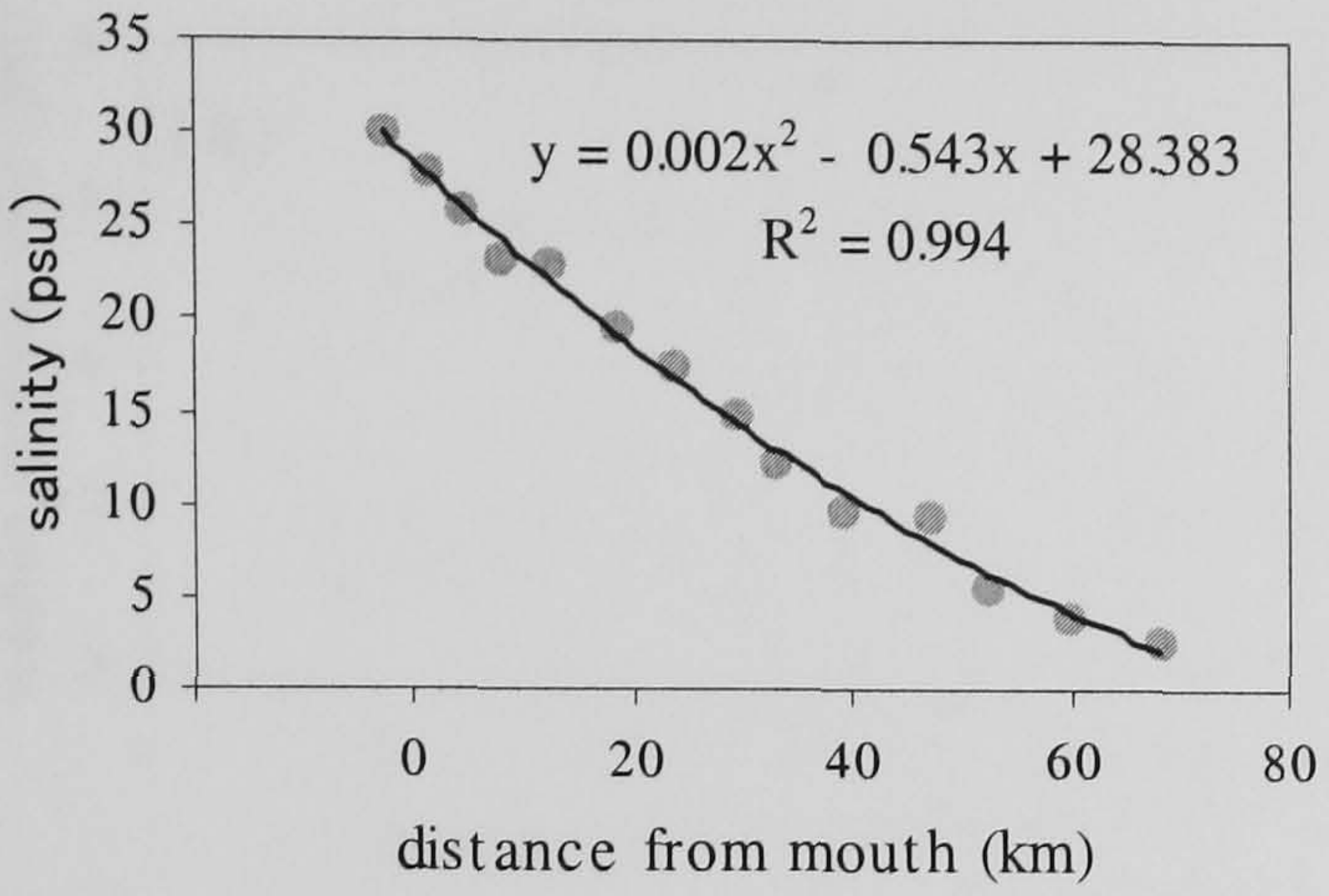


Fig. 2.3. Relationship between mean monthly salinity (psu) for each transect and the distance from mouth. (km) with quadratic equation and R^2 value.

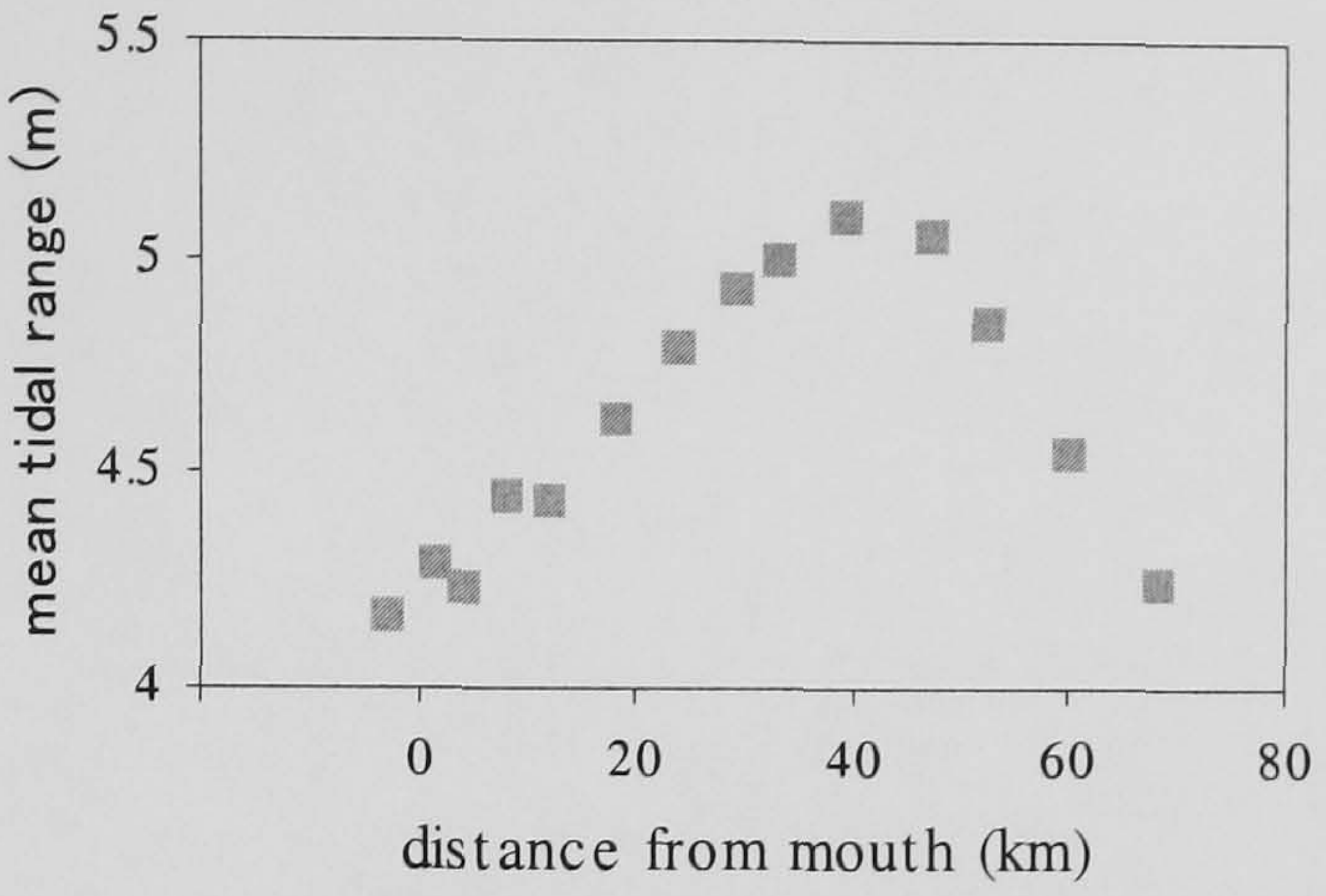


Fig. 2.4. Relationship between local mean tidal range (m) and distance from mouth (km).

Figure 2.4 shows the relationship between mean tidal range and distance from the mouth. Mean tidal range (defined in 2.2.3.3) was 4.2 m at transect 14 located at the mouth rising to a maximum of 5.1m at transect 9 around the middle region of the Humber, then decreasing again to 4.2m at the upper most transect (Fig. 2.4).

2.3.1.2 Sedimentary elements

Average median particle size for each transect was 2.5 phi at the outer most transect, characterised by fine sand, with the highest value of 4.9 phi at transect 12 situated near the boundary between lower and outer regions, characterised by highly muddy sediment (Fig. 2.5.a). Uniform muddy sediments occur over the 40 km along the middle and lower regions of the estuary, but there is an abrupt change from fine to coarse sediment at the outer region towards the mouth, and more gradual change in the upper region (Fig. 2.5.a). Similar trends were also observed for silt content (Fig 2.5.b) and organic matter content (Fig 2.5.c). Sediment characteristics observed around upper region could be attributable to the high average freshwater inflow and therefore stronger scouring (Jickells et al., 2000). All the transects situated in middle and lower regions of the estuary had > 80 % sediment silt content and a high organic matter content of around 3-4 %, values decreasing towards both outer marine and upper freshwater sections.

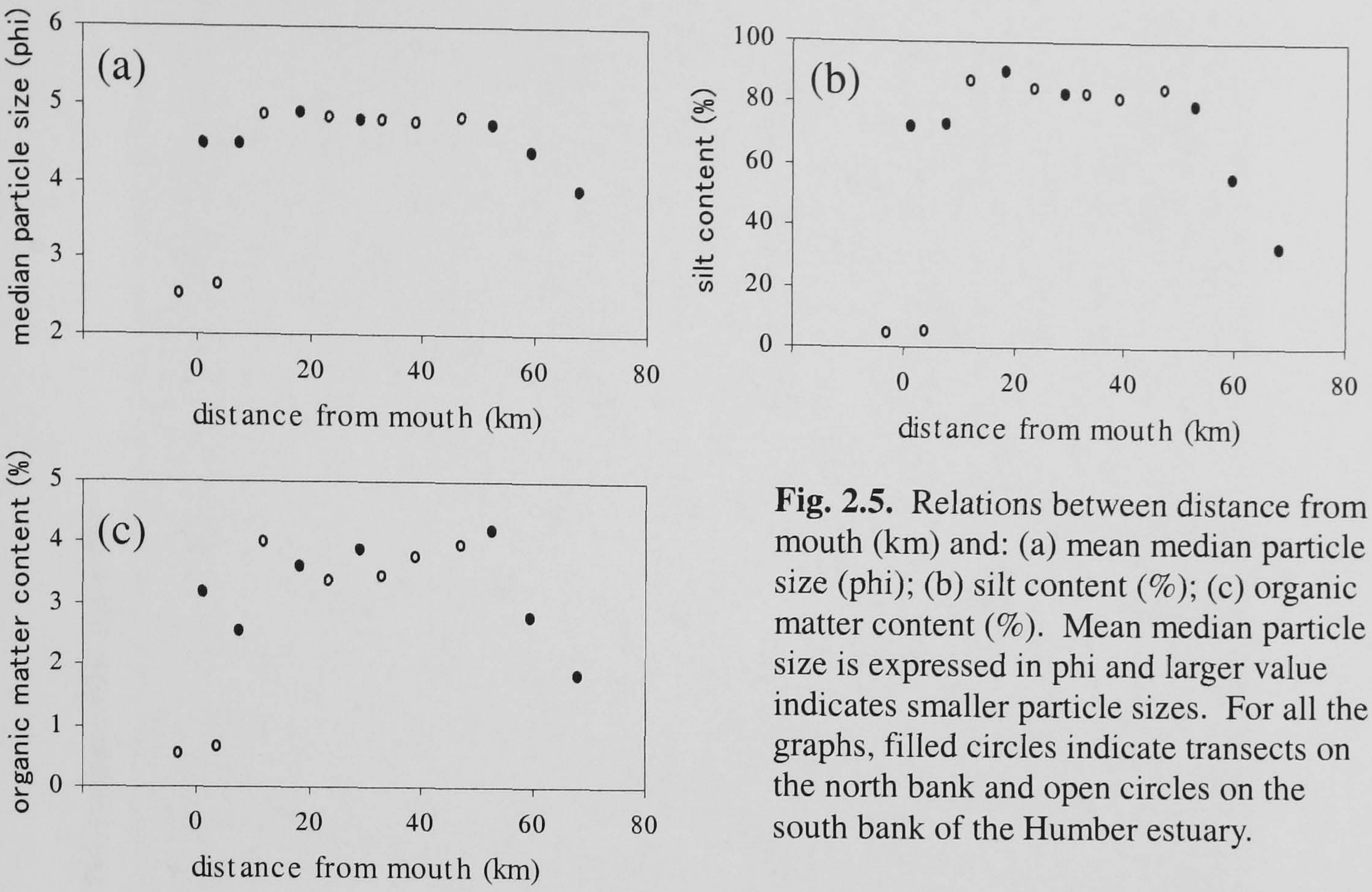


Fig. 2.5. Relations between distance from mouth (km) and: (a) mean median particle size (phi); (b) silt content (%); (c) organic matter content (%). Mean median particle size is expressed in phi and larger value indicates smaller particle sizes. For all the graphs, filled circles indicate transects on the north bank and open circles on the south bank of the Humber estuary.

2.3.1.3 Morphological elements

The transect profiles are shown in Fig. 2.6. For each transect, percentage tidal depth of each station in relation to mean tidal range is plotted against distance from MHWL. The slopes varied between 1/14 at transect 5, and 1/589 at transect 1. Low median depth values (%) were found at transect 5 (9.2 %) and transect 14 (12.2 %), indicating that their slopes are shallow over the upper and middle shore, but steeply shelving at lower shore showing concave shape over the intertidal profile (Fig. 2.6). In contrast, transect 6 and transect 10 had a high percentage median depth of 85.7 % and 88.6 %, respectively, indicating that they shelve steeply at high and middle shore levels, but becoming shallower towards the lower shore level showing concave beach face morphology (Fig. 2.6). Generally, the width of the transects became exponentially longer and the steepness of the profiles became shallower towards the mouth, but there was no clear trends between beach morphology and estuarine longitudinal gradient.

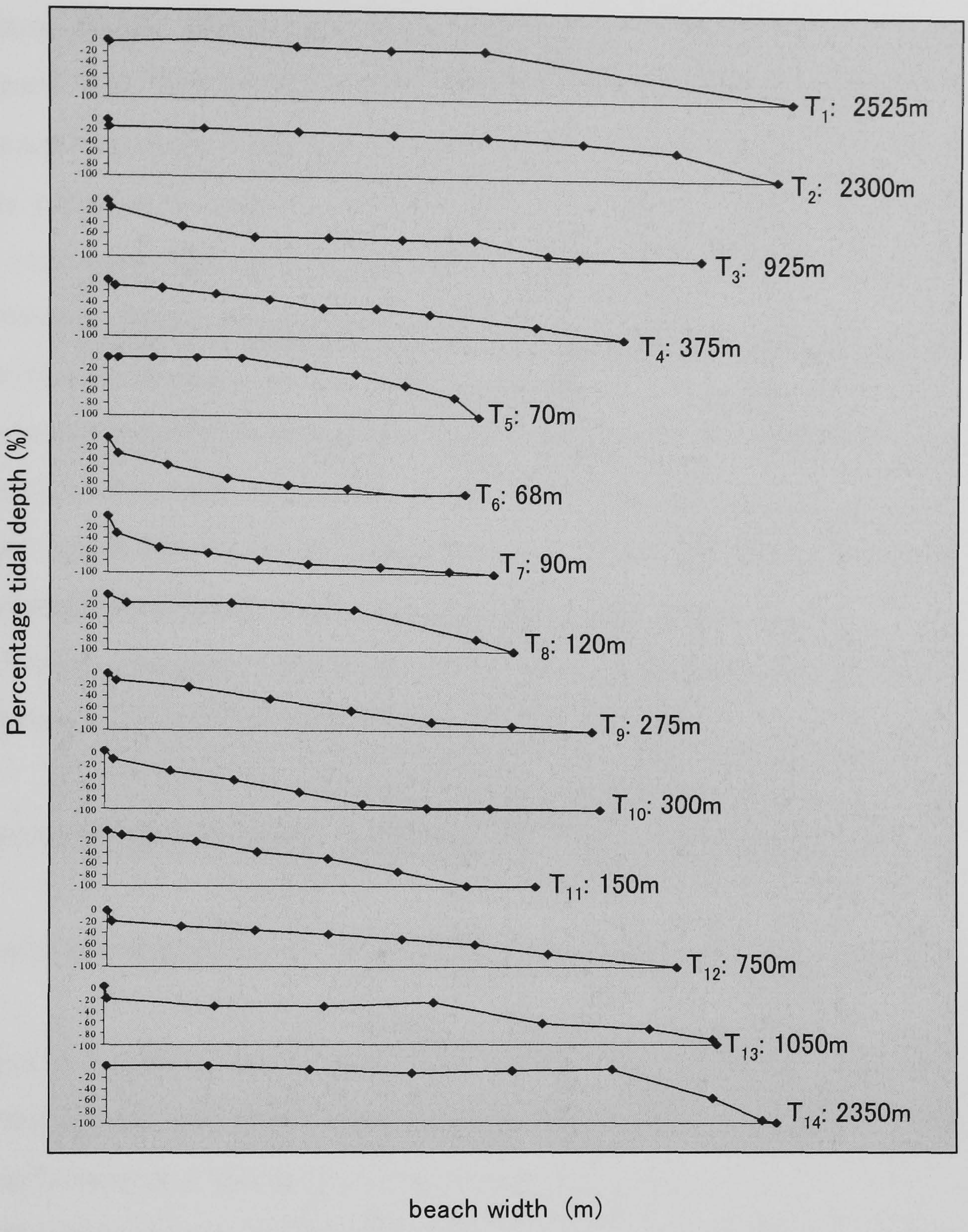


Fig. 2.6. Intertidal face profiles at the 14 transects sampled in this study showing percentage tidal depth (%) for each sampling station plotted against beach width (m). Tidal depth is expressed in percentage because this allows different transects with different mean tidal range to be comparable. The beach width axis was log-scaled to make the varying sizes of transects also comparable. T denotes transect and the subscript number indicates transect number.

2.3.2 Macrobenthos

2.3.2.1 Macrobenthic individual species

A total of 42 macrobenthic species were recorded from the sites: 9 oligochaetes, 17 polychaetes, 8 crustaceans, 6 molluscs and 2 others. Amongst the polychaetes worms, *Nereis diversicolor* was found throughout the system, and *Pygospio elegans* was the second most numerous polychaete in the estuary. Other characteristic polychaete species, such as *Nephtys hombergii* and *Eteone longa*, were common towards the outer region, and *Nephtys cirrosa* was only found on the very sandy flats in the outer region. The oligochaete *Paranais litoralis* was commonly found around upper region, but this was gradually replaced by *Tubificoides benedeni* when moving towards the lower and outer regions. The crustacean amphipod *Corophium voltator* was widely distributed, but abundance was markedly higher around the middle region. *Macoma balthica* was distributed from the middle region and most of the molluscs, such as *Hydrobia ulvae* or *Cerastoderma edule* became common towards the outer estuary.

System averaged abundance (ind. m^{-2}) and biomass (g AFDW m^{-2}) for dominant macrobenthic species were calculated as follows:

$$\text{System averaged abundance} = \sum (D(T_n)*\text{Area}(T_n)) \,/\, (\text{total area}) \quad (n = 1, 2, \ldots, 14)$$

$$\text{System averaged biomass} = \sum (B(T_n)*\text{Area}(T_n)) \,/\, (\text{total area}) \quad (n = 1, 2, \ldots, 14)$$

where $D(T_n)$, $B(T_n)$ and $\text{Area}(T_n)$ denote mean density, mean biomass for transect T_n and spatial area (km^2) represented by transect T_n, respectively. Details for $\text{Area}(T_n)$ and total area can be seen in Chapter 3 (Table 3.4).

Pygospio elegans attained the highest system averaged density accounting for 26.4 % of the total abundance, followed by *T. benedeni* (22.7 %) and *M. balthica* (18.3 %) (Table 2.1). An averaged > 7400 macrobenthic individuals m^{-2} were found in the intertidal area in the Humber, and 11 species accounted for over 95 % of the total mean abundance.

System averaged biomass was dominated by three species: *C. edule*, *M. balthica* and *N. diversicolor*, which accounted for 51.7 %, 25.0 % and 12.1 % of the total biomass in the study site, respectively (Table 2.2). The mean total biomass was 8.6 g AFDW m^{-2} and 15 species were found to account for over 99 % of total biomass in the study site, most of which are known to be important food sources for higher trophic levels.

Table 2.1. System-averaged abundance of macrobenthic species in the Humber. The species are listed in a decreasing rank order of mean density m^{-2}.

Species	Mean density (ind. m^{-2})	Percentage (%)	Cumulative percentage (%)
Pygospio elegans	1956.8	26.4	26.4
Tubificoides benedeni	1684.9	22.7	49.1
Macoma balthica	1358.0	18.3	67.3
Nereis diversicolor	559.4	7.5	74.9
Corophium volutator	470.8	6.3	81.2
Paranais litoralis	333.0	4.5	85.7
Tharyx spp.	225.9	3.0	88.7
Cerastoderma edule	219.4	3.0	91.7
Nephtys hombergii	141.5	1.9	93.6
Eteone longa	76.6	1.0	94.6
Hydrobia ulvae	71.1	1.0	95.6
Others	326.6	4.4	100.0
TOTAL	**7424.0**	100.0	

Table 2.2. System-averaged biomass of macrobenthic species in the Humber. The species are listed in a decreasing rank order of mean biomass g AFDW m^{-2}.

Species	Mean biomass (g AFDW m^{-2})	Percentage (%)	Cumulative percentage (%)
Cerastoderma edule	4.47	51.7	51.7
Macoma balthica	2.16	25.0	76.7
Nereis diversicolor	1.05	12.1	88.9
Nephtys hombergii	0.30	3.5	92.3
Pygospio elegans	0.16	1.8	94.2
Tubificoides benedeni	0.11	1.3	95.5
Scrobicularia plana	0.07	0.8	96.3
Ampharetidae	0.07	0.8	97.1
Corophium volutator	0.05	0.6	97.7
Nephtys cirrosa	0.03	0.3	98.0
Eteone longa	0.02	0.2	98.3
Hydrobia ulvae	0.02	0.2	98.5
Tharyx spp.	0.02	0.2	98.7
Paranais litoralis	0.02	0.2	98.9
Spiophanes bombyx	0.02	0.2	99.1
Others	0.08	0.9	100.0
TOTAL	**8.65**	100.0	

2.3.2.2 Trends in species richness at the transect and station scales

Figure 2.7.a shows the relationship between total species richness per transect and the distance from the mouth in the Humber estuary. The maximum number of 26

was found at transect 14, the outer most region, whilst only 1 species was recorded at transect 7, at the upper most site. The exponential increase from the upper to the outer region (Fig. 2.7.a), suggests that salinity have marked influence on species richness. However, the width of intertidal profile also increased exponentially from upper to outer region (Fig. 2.7.b) and a longer intertidal profile often accommodates more heterogeneous habitats and therefore a greater species richness. Figure 2.7.c shows the relationship between mean species number per station and the distance from the mouth. The mean number of species did not increase exponentially with distance, but changed in a quadratic manner up to the region around the mouth, then began to tail off towards the outer open coastal region (Fig. 2.7.c). This trend indicates that there are another factors affecting species richness other than salinity along the longitudinal gradient.

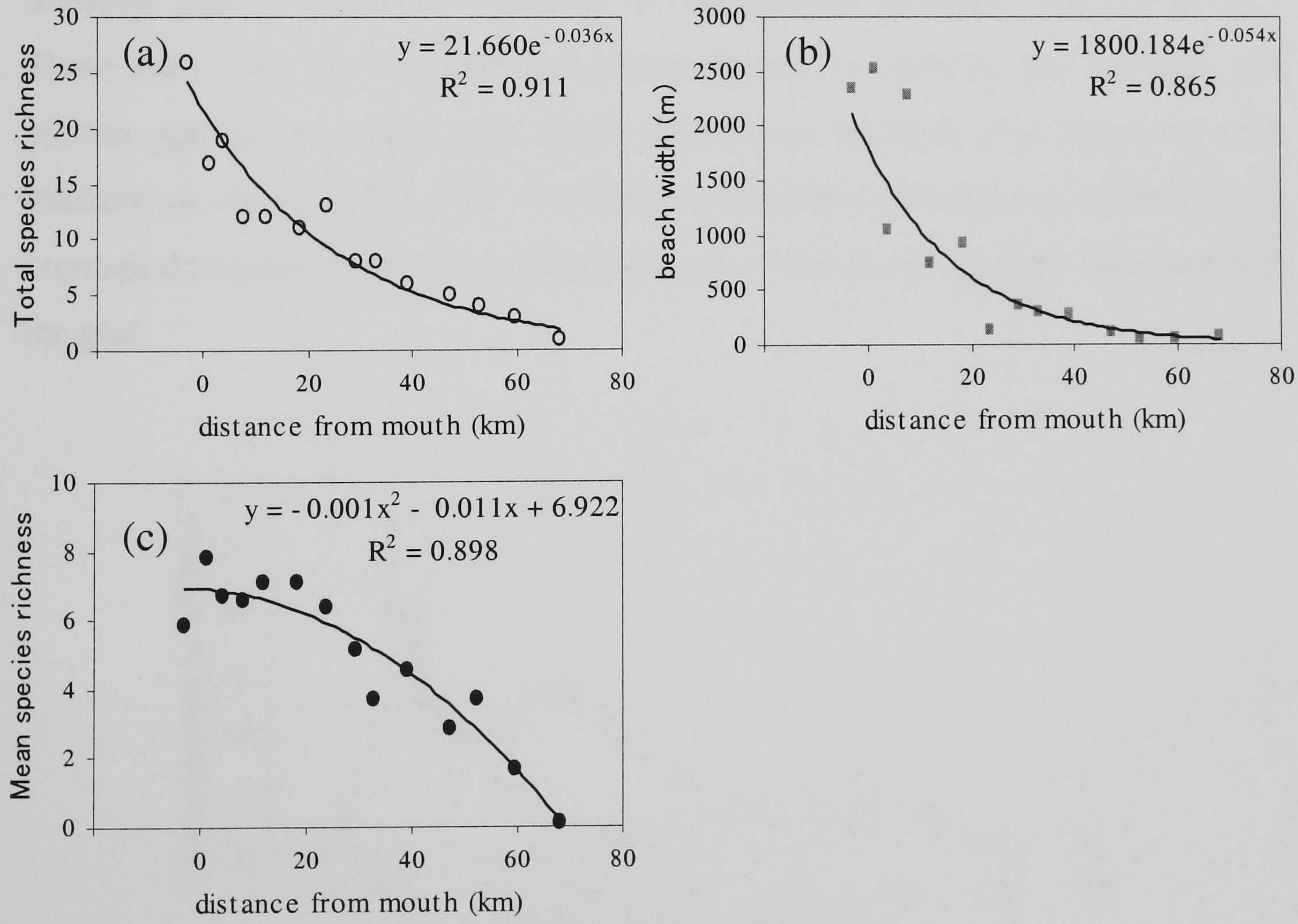


Fig. 2.7. Relations between distance from mouth (km) and: (a) total species richness per transect; (b) width of transect (m); (c) mean species richness per station, with equations and R^2 values.

2.3.2.3 Trends in biomass along longitudinal and vertical gradients

Figure 2.8 shows the relationship between mean macrobenthic biomass per station along each transect (g AFDW m^{-2}) and the distance from mouth. The highest mean biomass was recorded on transect 1 (mean value of 22.5 g AFDW m^{-2}) and the lowest on transect 7 (mean value of 0.00036 g AFDW m^{-2}). Generally, transects on the north bank had higher mean biomass than those on the south bank. However, the mean biomass on both banks showed a similar increase from the upper through the middle to the lower region of the estuary, although the south bank showed a steep decrease in biomass where the transect was situated outside the mouth (Fig. 2.8).

To explore the relationship between biomass and tidal depth (intertidal vertical gradient) across all transects, the biomass for each station was standardised as a percentage so that the sum of the new values within each transect was 100 %. This was done in order to make biomass distributions comparable between transects and to avoid the influence of longitudinal gradients such as salinity. These data were further natural log-transformed to stabilise the variance and plotted against percentage tidal depth (in relation to mean tidal range for each transect) as shown in Fig. 2.9. To avoid the influence of sediment variation, only biomass data taken from similar sediment types within each transect were used for the plot.

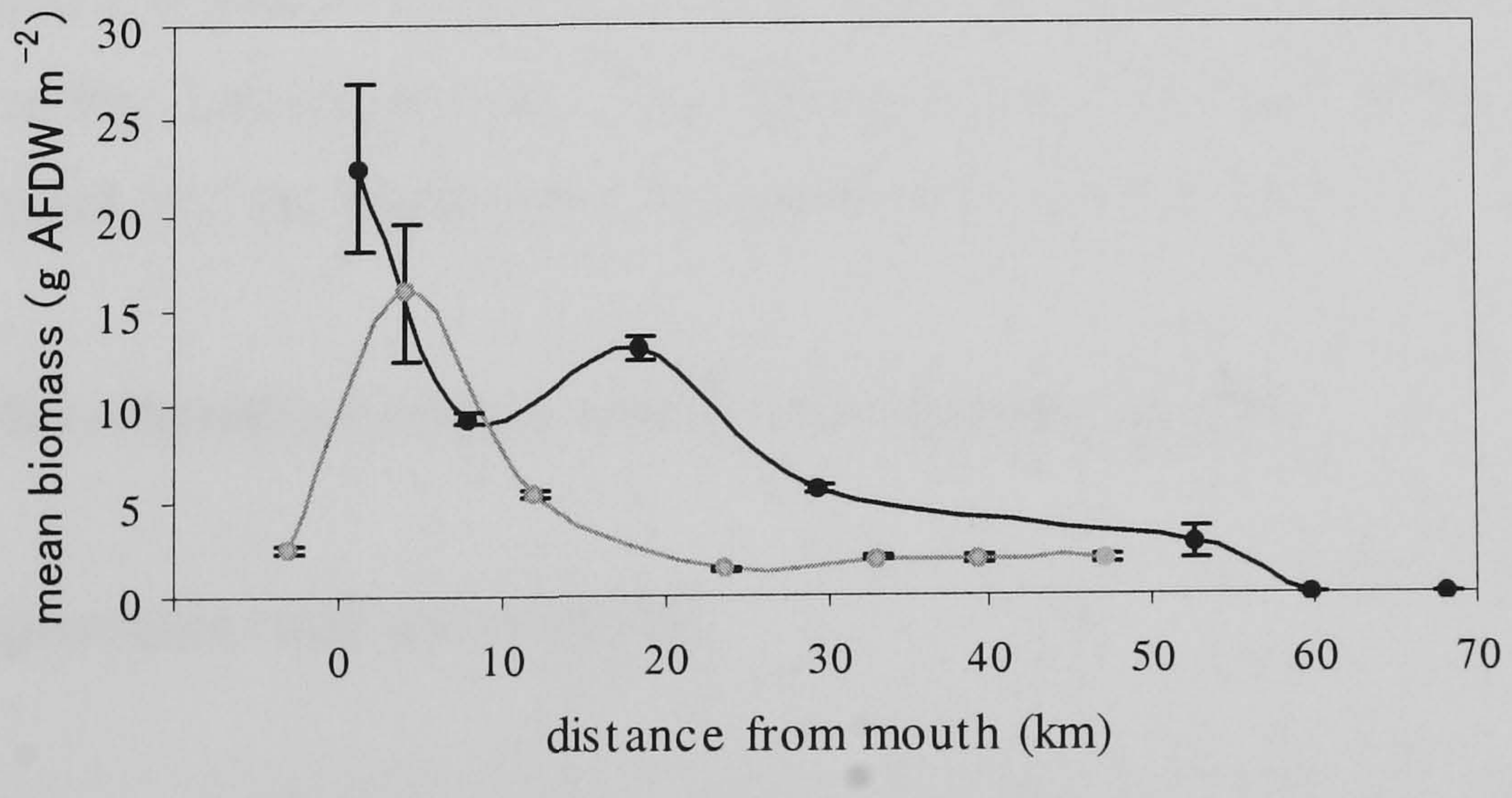


Fig. 2.8. Relations between mean biomass for 14 transects (g AFDW m^{-2}) and distance from mouth (km). Dark markers and light markers represent transects on the north and south bank, respectively, and vertical bars show ±SD.

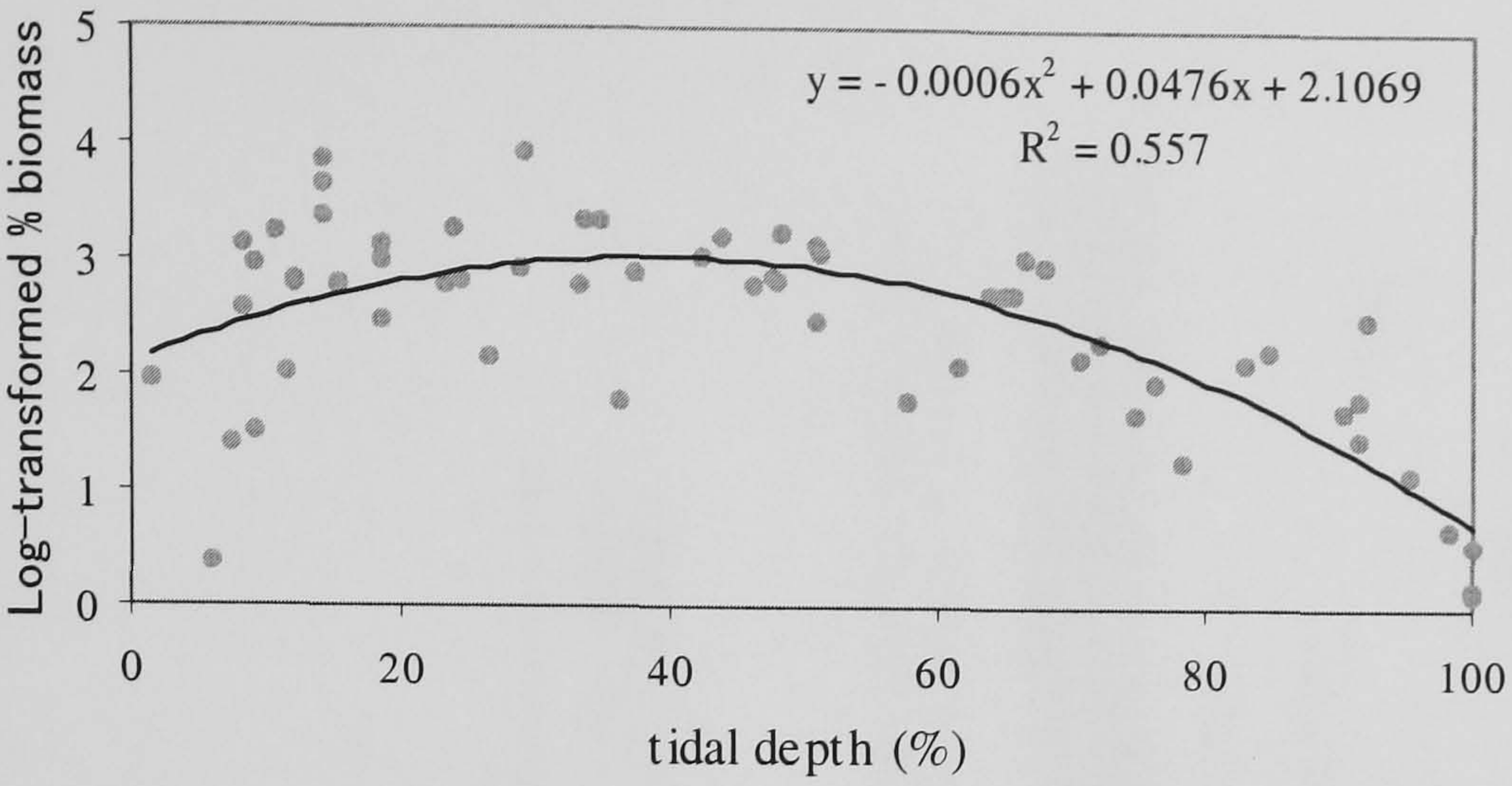


Fig. 2.9. Relations between percentage biomass and tidal depth across all transects. The percentage biomass was natural log-transformed and percentage tidal depth was expressed in relation to the local mean tidal range. The quadratic equation and R^2 value are shown in the plot and the biomass attains the highest when the tidal depth is 39.7 % lower than mean high water level.

There was a quadratic relationship between the two variables (Fig. 2.9), indicating that when longitudinal and sediment gradients are held constant, macrobenthic biomass tends to have a peak value at the point where the tidal depth was around 40 % of mean tidal range lower from MHWL within each transect ($y'=0$ when $x = 39.7$ in the quadratic equation in Fig. 2.9). The new environmental variable "depth index" was thus established for the subsequent analysis to express how the value of tidal depth for each station deviates from the value 39.7 % as follows:

depth index = $f(x) / f(39.7)$

where x and $f(x)$ denote tidal depth (%) for each station and the quadratic equation y shown in Fig. 2.9, respectively. This indicates that if the tidal depth for a station deviates from 39.7 %, depth index deviates from 1 and becomes < 1.

2.3.3 Macrobenthic biomass and environmental variables

2.3.3.1 Spearman rank correlations

Table 2.3 shows Spearman rank correlation coefficients between the biological and environmental components measured at the station scale.

Table 2.3. Spearman rank correlation coefficients between macrobenthic biomass and physical measurements from three environmental components.

	BIO	LONGITUDINAL		SEDIMENTARY			MORPHOLOGICAL					
		SAL	EXP	MD	ORG	SIL	WID	T-SLO	S-SLO	DEP	M-DEP	DEP-I
SAL	(0.55)***											
EXP	(0.21)*	(0.68)***										
MD	(0.41)***	n.s.	(-0.25)*									
ORG	(0.29)**	(-0.31)**	(-0.52)***	(0.61)***								
SIL	(0.43)***	n.s.	(-0.23)*	(0.99)***	(0.61)***							
WID	(0.58)***	(0.94)***	(0.57)***	n.s.	(-0.31)**	n.s.						
T-SLO	(0.57)***	(0.94)***	(0.57)***	n.s.	(-0.32)**	n.s.	(0.99)***					
S-SLO	(0.59)***	(0.68)***	(0.35)**	n.s.	n.s.	n.s.	(0.71)***	(0.70)***				
DEP	(-0.42)***	(-0.30)**	n.s.	n.s.	(-0.27)**	n.s.	(-0.25)*	(-0.23)*	(-0.40)***			
M-DEP	(-0.31)**	(-0.47)***	(-0.25)*	(0.21)*	n.s.	n.s.	(-0.36)***	(-0.33)**	(-0.45)***	(0.57)***		
DEP-I	(0.26)**	n.s.	n.s.	(0.22)*	n.s.	(0.23)*	n.s.	n.s.	n.s.	n.s.	n.s.	

When correlations are significant Spearman rank coefficients are given between brackets. BIO = mean total biomass. ***P<0.001; **P<0.01; *P<0.05; n.s. = not significant. Environmental variables: SAL: salinity; EXP: exposure; MD: median particle size; ORG: organic matter content; SIL: silt content; WID: beach width; T-SLO: transect slope; S-SLO; station slope; DEP; percentage tidal depth; M-DEP: median tidal depth; DEP-I: depth index, for each sampling station.

Table 2.4. Multiple regression analysis of the four macrobenthic biomass categories against physical variables from three environmental components over two different spatial scales. The data on station scale consisted of the average values of replicates (n=94 stations), and whereas the data on transect scale consisted of the average of stations (n=14 transects).

	R^2	P	LONGITUDINAL		SEDIMENTARY			MORPHOLOGICAL					
			SAL	EXP	MD	ORG	SIL	WID	T-SLO	S-SLO	DEP	M-DEP	DEP-I
Station scale (n=94)													
M. balthica	0.65	<0.0001		$(-0.21)_4$*	$(0.32)_2$**				$(0.77)_1$***		$(0.30)_3$**		
C. edule	0.34	<0.0001					$(-0.23)_3$*	$(0.27)_1$**		$(0.24)_2$*			
Other macrobenthos	0.75	<0.0001	$(0.79)_1$***			$(0.71)_2$***					$(-0.38)_3$***		
Total biomass	0.77	<0.0001	$(076)_1$***			$(0.65)_2$***				$(0.47)_3$***			$(0.40)_4$***
Transect scale (n=14)													
M. balthica	0.92	<0.0001	$(0.94)_1$***							$(0.72)_3$**		$(0.85)_2$***	
C. edule	0.98	<0.0001	$(0.93)_4$***	$(-0.95)_3$***	$(-0.97)_2$***			$(0.93)_1$***	$(-0.79)_5$**				
Other macrobenthos	0.86	<0.0001	$(0.91)_1$***			$(0.88)_2$***							
Total biomass	0.81	<0.0001	$(0.91)_1$***			$(0.61)_2$*							

Overall model R^2 and significance P are presented, along with partial correlation coefficients between brackets. See Table 2.3 for * and abbreviations for environmental variables. The subscript number indicates the rank order in which environmental variables were retained in the stepwise procedure. Details of the results (e.g. model coefficients) can be found in Appendix 2.

Macrobenthic biomass was significantly correlated with all the environmental variables measured in the analysis. All showed positive correlations except for tidal depth and median tidal depth which were negatively correlated. Many of the measured variables were inter-correlated. For example, mean median particle size, silt content and organic matter content were positively and significantly correlated with one another, indicating that sediments characterised by higher median particle values (therefore finer particles) tend to have higher silt contents and higher organic matter contents. Similarly, salinity, beach width, and transect slope were significantly positively correlated with one another, suggesting that intertidal flats in the Humber estuary tend to become narrower and steeper with increasing distance from mouth since salinity also changes steadily with distance from mouth (Fig. 2.3). Such a high degree of inter-correlations suggests that smaller number of environmental variables that retain maximum information with minimum redundancy can be selected and used for modelling the spatial variation of macrobenthic biomass in relation to environmental variables.

2.3.3.2 Multiple regression analysis

Multiple regression analysis (linear regression) was used to assess the explanatory role of the environmental variables for macrobenthic biomass over two different spatial scales (station and transect). Because the biomass of two of the bivalve species, *M. balthica* and *C. edule* accounted for approximately 25 % and 52 % of the total biomass, respectively (Table 2.2), these species were analysed separately along with biomass for the remaining species and for total biomass. Two different spatial scales were examined in this analysis – for the station scale, mean data for each sampling station were used (n = 94), and for the transect scale, average of station data within each transect were used (n = 14). Table 2.4 shows the results of this analysis. Models explained between 34 % and 77 % of the variance in biomass of *M. balthica*, *C. edule*, other remaining species and total biomass at the station scale, and between 81 % and 98 % at the transect scale. For the different categories, biomass was largely explained by salinity, organic content and various morphological components.

At the station scale, biomass of *M. balthica* was negative related to exposure, and positively related with finer particles, whereas *C. edule* was negatively related to the silt content. At the same spatial scale, the sum of the remaining species and total biomass showed positive relations with salinity and organic matter content, suggesting that overall macrobenthic biomass tends to be high where salinity and organic matter content are high. The former biomass was negatively related to tidal depth and the latter was positively related to depth index, suggesting that overall biomass also tends to be high where tidal depth is high or close to the mid-point of the local tidal range. Such relations between overall biomass and environmental variables are consistent with the trends found elsewhere in literature (Chapter 1).

At the transect scale, the degree of variability explained by environmental variables was higher than at the station scale. For total biomass and other macrobenthic biomass (excluding *C. edule* and *M. balthica*), only longitudinal components (salinity) and sedimentary components (organic matter content) explained > 80 % of the variation in biomass. However, for the biomass of the two bivalve species on transect scale, morphological components, such as beach width or median depth of the beach, significantly explained biomass together with salinity.

These relations provide a means for describing quantitatively how macrobenthic biomass varies in relation to key environmental variables on the Humber. Ideally, outputs of these models should be compared with observations from different sampling locations within the same system or from similar geographical regions. In this chapter, however, due to the high degree of variability explained at the transect scale, which would also be a relevant spatial scale in the context of estuarine management, these relations were used to graphically compare the observed and modelled variability in macrobenthic biomass. Total macrobenthic biomass is a sum of the biomasses of *M. balthica*, *C. edule* and other remaining species. Fig. 2.10 (a), (b), (c), show the observed and modelled biomasses for these three categories, and the total sum from the model are then compared with the observed total macrobenthic biomass in Fig. 2.10 (d).

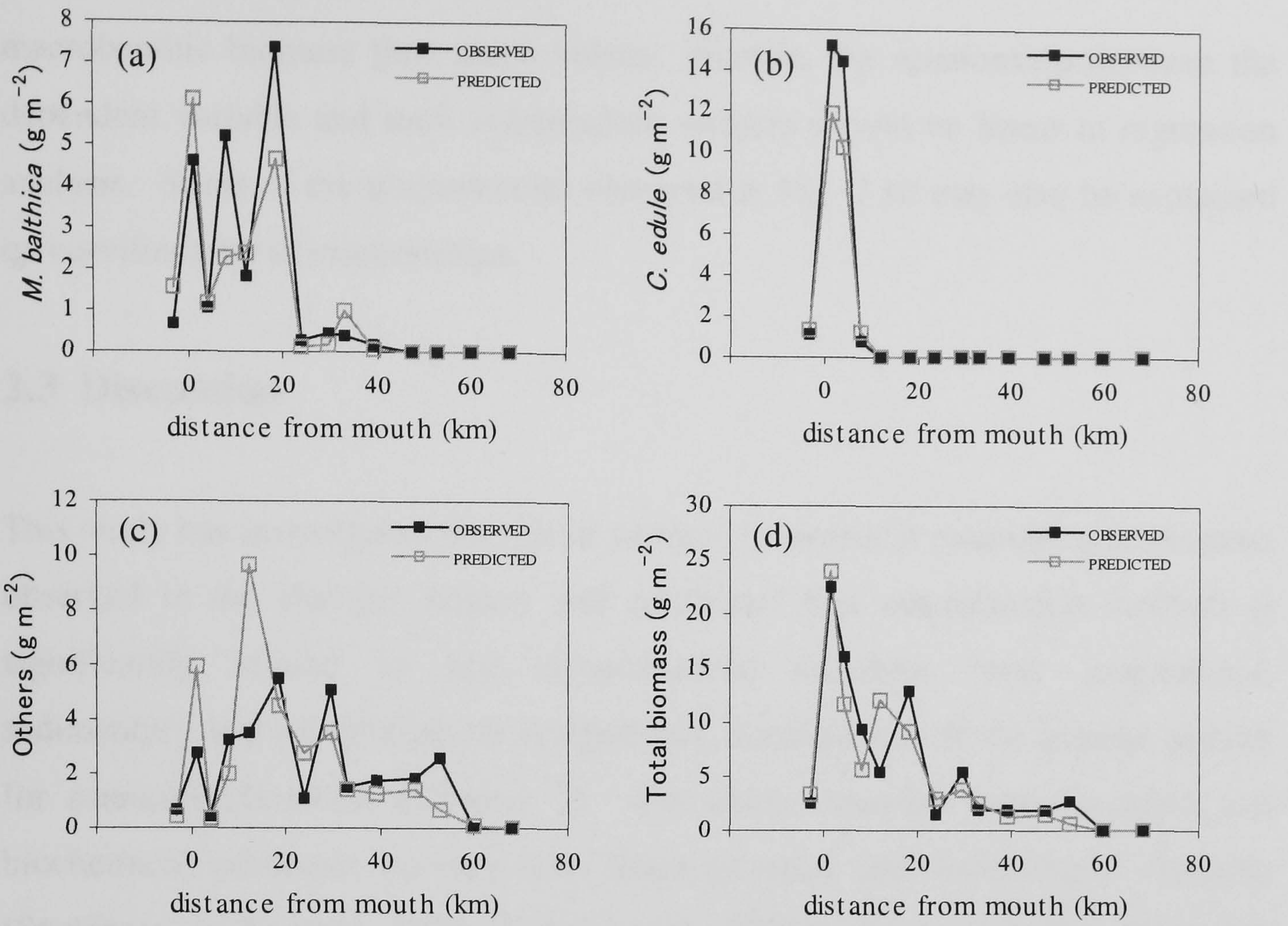


Fig. 2.10. Observed and model predicted biomass (g AFDW m^{-2}) for: (a) *M. balthica*; (b) *C. edule*; (c) total of other remaining macrobenthic species; and (d) total biomass, in relation to distance from mouth (km). For (d), predicted total biomasses are expressed as a sum of model (a), (b) and (c). Open squares with light line indicate model prediction and filled squares with dark line show observed biomass.

The good fit seen in Fig. 2.10 suggests that the models give a reasonable description of how the macrobenthic biomass in the Humber estuary can be predicted from key environmental variables. However, some large discrepancies observed in Fig. 2.10 (a) and (c), for example, could be related to the absence of biological independent variables, such as predation, which could not be incorporated in this study. There are also problems with summarising salinity values into a single variable, in this case the monthly mean value over a spring tidal cycle throughout a year. In estuarine environments, salinity varies considerably over time scales such as the tidal cycle, the spring-neap cycle or seasonally (Little, 2000). It may therefore be more appropriate to summarise salinity as a range or as minimum or maximum values over a defined period, since these values may have more significant influences on the spatial patterns of macrobenthic biomass than mean values. Further, the relationship between the dependent variable and each independent variable should be linear in regression analysis. Some of the discrepancies observed in Fig. 2.10 may also be explained by curvilinearity of relationships.

2.3 Discussion

This study has investigated the spatial patterns in intertidal macrobenthic biomass observed in the Humber estuary and confirmed that macrobenthic biomass is significantly related to key environmental variables from longitudinal, sedimentary and morphological components, consistent with the general picture for estuaries elsewhere (Chapter 1). For many estuaries, oceanographic and biochemical processes co-vary with other physical and sedimentary elements (Snelgrove & Butman, 1994; Thrush et al., 2003), and the resident organisms themselves further modify the sedimentary components, influencing local physical and chemical characteristics (Paterson & Black, 1999; Widdows & Brinsley, 2002; Wood & Widdows, 2002). Such relations can be further influenced by higher trophic levels such as migratory shorebirds that arrive in large numbers and affect the behaviour of prey organisms and thus erodibility of estuarine sedimentary shores (Daborn et al., 1993). Given the strong feedback effects between biological and environmental components, construction of deterministic models based on cause-and-effect relationships to precisely describe future changes in macrobenthic biomass will be difficult in a large-scale estuarine environment. The statistical (empirical) modelling approach used in this study could be a useful tool in the context of coastal and estuarine management, particularly when there is an urgent need to know how macrobenthic biomass is likely to change in response to long-term changes in their physical environment, including changes caused by global warming and sea-level rise.

In the Humber estuary, two bivalve species, *C. edule* and *M. balthica* were found to account for over 75 % of the total macrobenthic biomass, and between 37 % and 99 % of the variance in their biomasses were explained by the key environmental variables, depending on the species and spatial scale at which the multiple regression analysis was carried out. For total macrobenthic biomass, 77 % of the variance was explained by four environmental variables, namely, salinity, organic matter content, station slope and depth index at the station scale (10^1 – 10^2 m). This suggests that the quality of intertidal area, or availability of food items for higher trophic levels, is positively associated with higher salinity, muddier sediments, shallower beach slope and tidal depth that is closer to mid-shore level. In the context of estuarine shoreline management, the Humber flood defences protect nearly 90,000 ha of land (Winn et al., 2003), and there is little undefended land throughout the system. This indicates that unless the accretion of the intertidal areas keeps pace with the rate of sea-level rise, or appropriate areas are made available for flooding, the intertidal habitats will be squeezed between the rising sea and the defence walls, which will inevitably change the benthic invertebrate assemblages through the loss of their habitats and changes in their physical environment. The models developed here also indicate that the biomass of intertidal macrobenthos is likely to be affected by sea-level rise through its effects on the salinity gradient, the width and steepness of the flats and the particle size composition of sediments. This study also showed that biologically productive areas are currently situated around the outer region of the estuary where extensive shallow muddy intertidal areas can be found. Such areas will also be the most subject to the impacts of sea-level rise and increased wave energy due to their outer location and the shallowness of the beach.

The statistical models developed here provide a basis for predicting how the biomass of intertidal macrobenthos is likely to change in response to sea-level rise and concomitant changes in the physical environments. However, such modelling of species-habitat relationships requires large amounts of data from a number of locations over a wide range of habitats (Thrush et al., 2003), and in this respect, it is clear that more data from other areas are required in order to make quantitative predictions with any confidence.

The reliability of the model is also dependent upon how accurately we can predict the change of each environmental variable in response to sea-level rise. For example, there are problems in predicting how the detailed characteristics of the sediments and beach face profiles will change after sea-level has risen by 30 cm in 50 years time as indicated by MAFF (1999). Due to the necessary scales of time and space, such predictions are generally not amenable to manipulative experiments. However, it is possible to identify intertidal habitats in the Humber and elsewhere which have already been artificially squeezed between the low water line and expanded defence walls resulting from historic land-claims. Investigation of such areas and how they have adapted to land-claim could provide insight into how estuarine intertidal area will respond to the coastal squeeze resulting from sea-level rise and allow a formal test of the model. Such an investigation is the subject of Chapter 3.

There are other uncertainties involved in present model. Firstly, the model does not include the impact of temporal components, such as increased ambient temperature on the biomass of macrobenthic assemblages. Annual variability in climatic conditions is known to strongly influence the ecology of estuarine benthic invertebrate assemblages through changes in fecundity and individual growth (Beukema et al., 1993; Beukema et al., 1997; Widdows & Brinsley, 2002) or through changes in predation patterns (Jansen & Jansen, 1985). Increases in ambient temperature and milder winters may lead to increases in invertebrate biomass and productivity, yet the degree to which this may offset biomass loss due to sea-level rise is not clear at present. Secondly, enhanced primary productivity through increased nitrogen run off, as well as increased temperatures may also increase benthic biomass, because there is increasing evidence that primary production is one of the key factors regulating benthic biomass and secondary production by affecting individual growth rate and fecundity (Olafsson et al., 1994; Heip et al., 1995; Herman et al., 1999). Finally, stochastic events such as the expected increased occurrence of storms and surges is another dimension to consider since they could potentially have catastrophic impacts on the ecology of benthic macrofauna in estuarine intertidal habitats.

Notwithstanding the above, the construction of simple models for identifying the role of readily measurable environmental variables in explaining the spatial pattern of macrobenthic biomass is a significant advance for predicting the potential impacts of sea-level rise on the future estuarine environments.

Chapter 3

Modelling the potential changes in macrobenthic biomass of the Humber estuary intertidal flats in response to environmental changes resulting from sea-level rise

3.1 Introduction

Estuarine intertidal habitats can be described as physically dynamic environments that harbour an immense abundance and biomass of benthic invertebrates, attributable to the rich nutrient supplies that estuaries receive. Richness in stocks of invertebrate in turn provides essential food sources for higher trophic levels of epibenthic crustaceans, fishes and shorebirds (Wolff, 1987; McLusky, 1989; Goss-Custard et al., 1990; Lawrence & Soame, 2004), and this is one of the reasons why estuarine intertidal flats are of such high conservation value. Intertidal sand and mudflats within an estuary exhibit significant variations in benthic macrofaunal species composition, density and biomass, and Chapter 2 has explored these variations in relation to salinity, sediments and beach morphology (McIntyre, 1970; Jones, 1988; McLusky, 1989; Meire et al., 1991; Dauer, 1993; Elliott et al. 1998; Beukema, 2002; Ysebaert & Herman, 2002). Sea-level rise can impact on these through coastal squeeze.

Data from a number of locations around the UK coastline indicate rising sea levels, ranging from 0.7 mm to 3.0 mm per year over the recent past (Woodworth et al., 1999). In the case of the Humber, the rate of sea-level rise relative to land has been 2.0-2.5 mm per year over the last century (Edwards & Winn, 2006), but the average rate of this century is predicted to be 6.0 mm per year (MAFF, 1999), implying that sea level could rise by between 0.1 and 0.3 m over the next 50 years.

The accelerating rise of sea level poses serious threats to the estuarine intertidal habitats because this will inevitably reduce the area of intertidal habitats through the process of coastal squeeze in which saltmarshes and mudflats are eroded as they become trapped between rising sea levels and fixed sea defences (Jones, 1994; Galbraith et al., 2002). In addition, they may also be a steepening of the shore and consequent reduction of the habitat quality as a result of the change in sediment regime (Goss-Custard et al., 1990; Raffaelli & Hawkins, 1996; Crooks, 2004; Taylor et al., 2004). Brown and McLachlan (1990) have described the significant correlations between macroinfaunal community parameters and particle size, beach face slope and beach types for open sandy beaches. In such beach habitats, species richness as well as abundance and biomass tend to decrease from wider beaches that have finer sands and flatter slopes, to narrow beaches that are characterised by coarse sands and steep slopes, as in the transition of morphodynamic type of beaches from dissipative to reflective states (Brazeiro, 2001). Where open coastal flats are situated around outer region as in a large estuary, there may be significant biological impact through the coastal steepening and removal of finer sediments as a result of enhanced wave and tidal energy due to sea-level rise.

The primary aim of this chapter is to investigate how physical environments in estuarine intertidal areas are likely to change in response to sea-level rise, and how this will affect macrobenthic biomass. Because of the necessary scales of time and space required, it would be very difficult to verify the behaviour of physical elements within an estuary by experimentation. However, it is possible to examine intertidal habitats which have already been artificially squeezed between the low water line and expanded sea defence walls resulting from historic land-claims. These would manifest similar effects of coastal squeeze caused by sea level rise. Coastal squeeze caused by land-claim may differ from that due to sea-level rise in that land-claim is a rapid artificial process, whereas sea-level rise is a slow natural process. However, sites that have been "squeezed" for more than 100 years may have mimicked the physical and biological processes that would take place in response to coastal squeeze due to sea-level rise over the mid- to long term. Comparison between such squeezed beaches and natural, or "unsqueezed", beaches that are located within a relatively restricted area (Fig. 3.1) may provide a basis for predicting how coastal squeeze due to sea-level rise will affect the physical elements of estuarine intertidal habitats.

This chapter specifically aims to complete the modelling approach established in Chapter 2, and explore how such models can be used to predict changes in macrobenthic biomass in response to sea-level rise. First physical and biological elements of intertidal habitats identified in front of a land-claimed area at Grimsby, in the Humber, were described. These data are then used to validate the modelling approach established in Chapter 2. Next, all the physical data are assessed to derive the likely course of changes in key environmental variables as a result of sea-level rise. Finally, model simulations are conducted to explore how macrobenthic biomass may change under a range of likely future scenarios. The implications of model results are then discussed in the context of intertidal ecology and estuarine and coastal management.

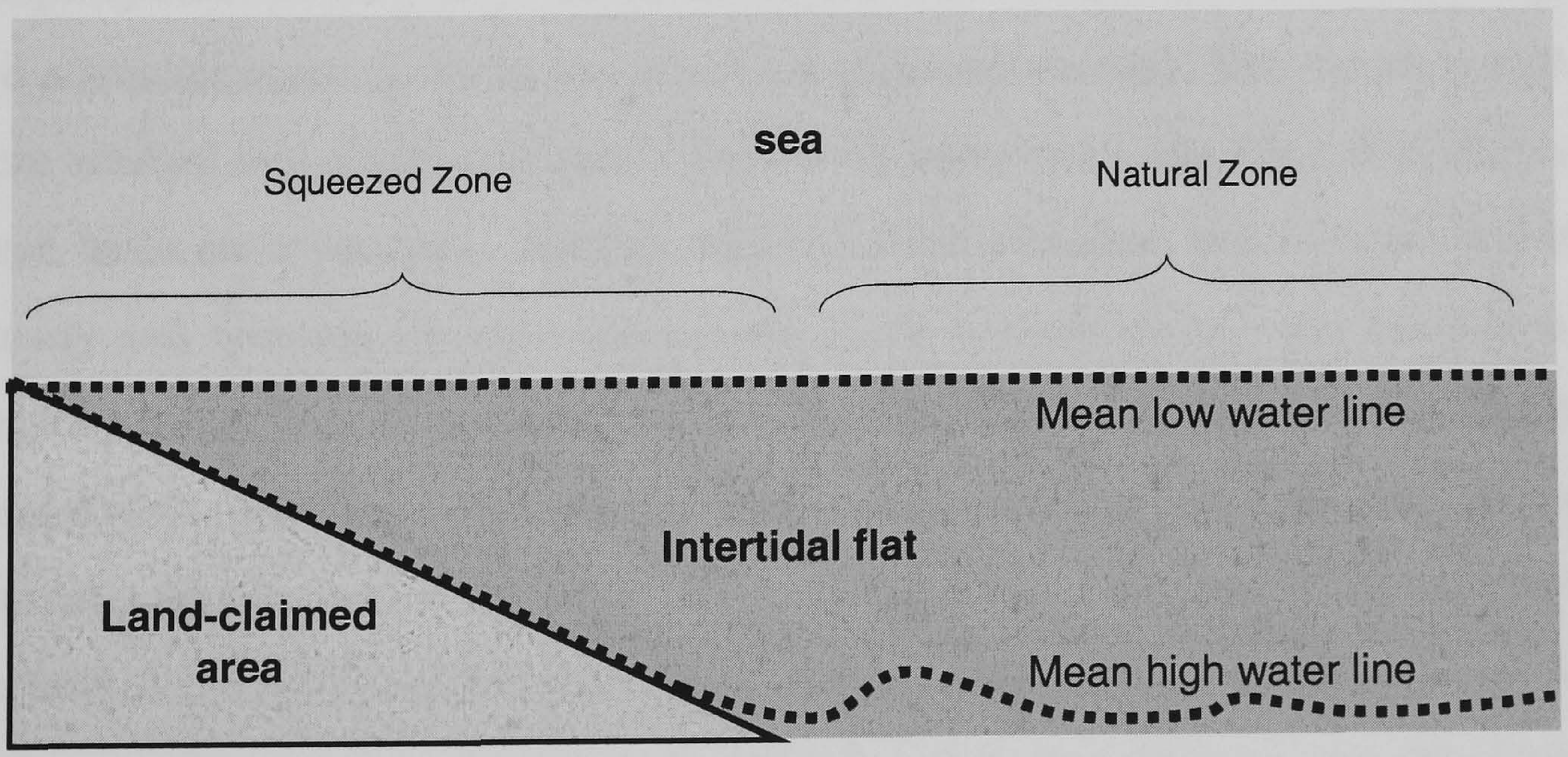


Fig. 3.1. A diagram showing two contrasting zones with respect to beach width ("squeezed" on the left, "natural" on the right) situated within relatively small area. Two dotted lines represent mean low and high water lines. Note that the mean high water level is on the beach in natural zone, but constrained on the sea defence wall in squeezed zone.

3.2 Material and methods

3.2.1 Study site and sampling

The study site was an estuarine sandy intertidal flat situated around outer region of the Humber located along approximately 4 km of the coast between Grimsby and Cleethorpes (from 53°33' to 53°35' N, and from 0°00' to 0°03' W) (Fig. 3.2). Tides in this area are macrotidal with the mean tidal range close to 4.55 m. This intertidal flat was selected because of the marked gradient in beach width from the south-eastern end to the north-western end (Fig. 3.2.b), due to the extensive land-reclamation that took place in front of Grimsby in the middle of the 19th century (Murby, 2001). The study site was progressively squeezed towards the western end, showing distinctive zones of intertidal habitats between "squeezed" to the northwest and "unsqueezed" (natural) to the southeast.

Nine transects were established to cover the whole range of beach width gradients, which run from mean high water (MHWL) to mean low water (MLWL), aligned along the direction of tidal flow (Fig. 3.2.b). The locations of MHWL and MLWL were determined by consulting a 1: 25,000 scale Ordnance Survey map revised 2001. Along each transect, 9 equally spaced shore levels were chosen to establish sampling stations. Three replicate samples were collected from each sampling station using a cylindrical corer (10 cm in diameter) for faunal analysis and a smaller corer (3 cm in diameter) for sediment analysis. The former samples were washed into a 0.86 mm mesh brass sieve using sea water filtered (0.063 mm mesh brass sieve) on site. Species retained were identified and counted, and the density and biomass for each species are given in numbers m^{-2} and ash free dry weight (AFDW) g m^{-2}, respectively. Physical measurements for longitudinal components (salinity, exposure), sediment components (median particle size, silt content, organic matter content), and morphological components (beach width, transect slope, station slope, tidal depth, median tidal depth) were also derived for each sampling station. Detailed accounts of the methodology for the faunal analysis and physical measurements can be found in Chapter 2. Data from 81 sampling stations (9 transects × 9 stations) were thus obtained for subsequent analysis. All field sampling was carried out from 14th to 28th of September 2004. All the details of each transect can be found in Appendix 3.

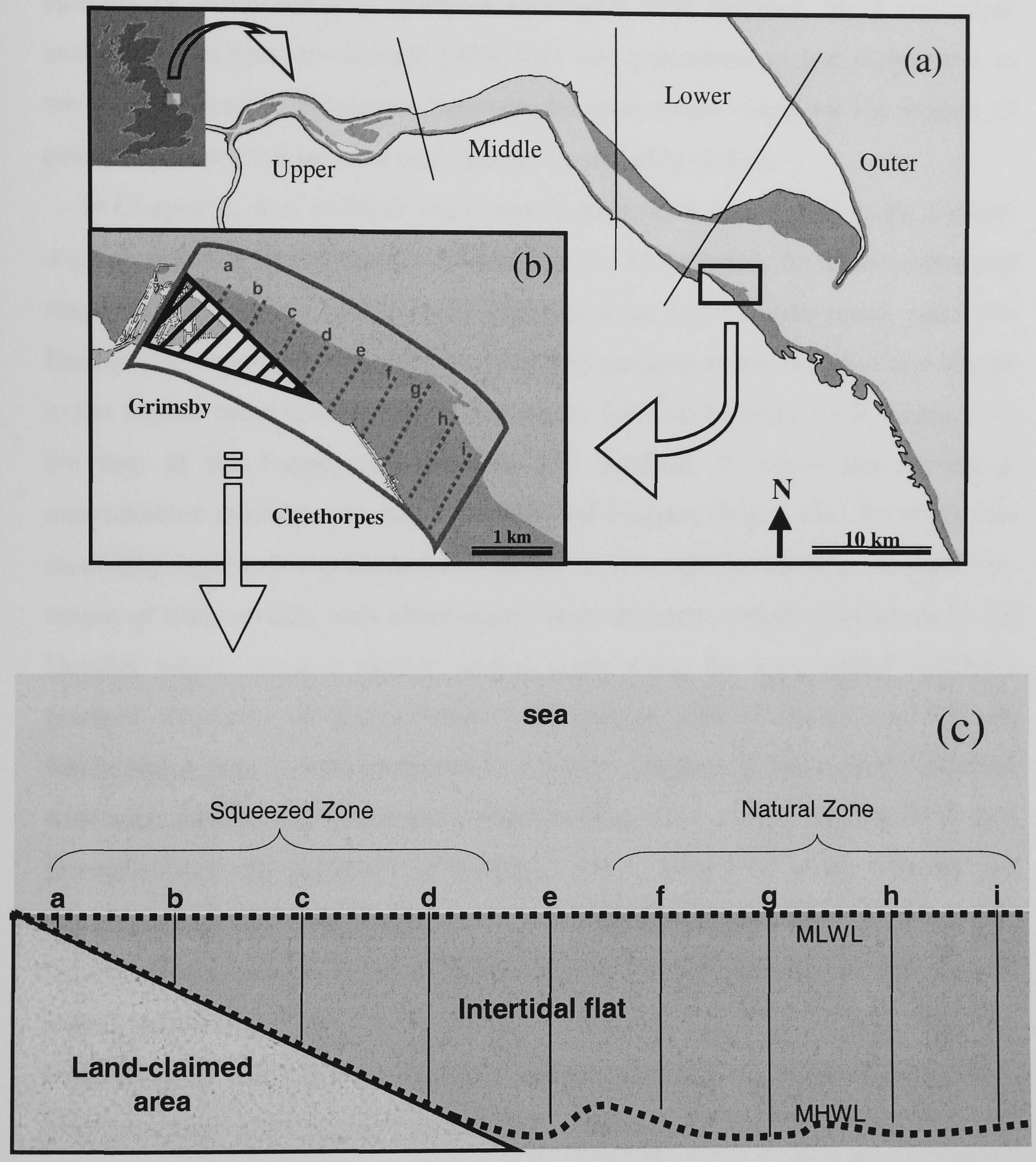


Fig. 3.2. (a) Map of the Humber estuary. The dark shading area indicates the intertidal area. (b) Location of the study site at Grimsby and Cleethorpes (within the slightly distorted rectangle), showing 9 transects from *a* to *i* (the dotted lines). The triangular hatched area is the reclaimed site that used to be a part of intertidal flat before the middle of the 19th century. (c) Simplified diagram of the study site showing three arbitrary zones with respect to state of coastal squeeze: squeezed and natural (unsqueezed). Each zone contains 4 transects. The solid lines correspond to the transects shown in (b) and two dotted lines indicate mean high water level (MHWL) and mean low water level (MLWL).

3.2.2 Statistical analysis

General trends in physical and biological (species richness, abundance and biomass of macrobenthos) characteristics were first assessed using univariate analyses. Analysis of variance (ANOVA) was performed to test differences in mean macrobenthic biomasses between the two zones based on the degree of coastal squeeze (natural zone and squeezed zone) (Fig. 3.2.c).

In Chapter 2, four multiple regression models were established at the transect scale to describe the biomasses distribution for *M. balthica*, *C. edule*, remaining macrobenthic species and for total biomass from key environmental variables. Because the degree of variability explained by environmental variables was higher in the former three models than in the model for total biomass itself (Table 2.4), the sum of the former three models (*M. balthica*, *C. edule* and remaining macrobenthic species) was used to derive total biomass (Fig. 2.10). To verify this modelling approach established in Chapter 2, it would be ideal to compare the output of these models with observations from different sampling locations in the Humber which cover a similar spatial scale along the longitudinal (salinity) gradient. However, the transect data in this chapter were derived from a local site which had a very limited longitudinal gradient, suggesting that model validation with such data is less meaningful. Because the transect data in Chapter 2 deal primarily with the longitudinal gradient at a system-wide scale, whereas the transect data in this chapter deal mainly with beach width gradients operating over local site scale, these data were, instead, combined to establish more detailed and robust macrobenthic models. Data for biomass and salinity were normalised by log(1000x+1), and log-transformation, respectively, prior to subsequent analysis. Multiple regression analysis with a forward step procedure was then conducted for the biomasses of *Macoma balthica*, *Cerastoderma edule* and other remaining benthic macrofauna to re-assess the key environmental variables that best explain the observed spatial variability in each biomass group. A model for total biomass was not created for the same reason as described above (less variability was explained by the key environmental variables than others). The observed variation in total biomass was therefore compared with the sum of the three model predicted values to assess the extent to which key environmental variable can explain the observed spatial variation both at the system (estuary) and local (present study site) scale. Model validations were then made by assessing how the graphical fit between predicted and observed total biomass could capture the biological trends over both local and system scales.

Subsequently, the models were used to simulate how macrobenthic biomass will vary in response to environmental changes resulting from sea-level rise. For simulation purposes, sea level was assumed to rise by 0.1, 0.3 and 0.5 m in the future, and simple assumptions were made as to how the key environmental variables will change in response to such sea-level rises based on field observations and scenarios derived from the literature review in Chapter 1 (Fig. 1.2). First, formal mathematical equations were formulated for longitudinal and sedimentary variables so that they can be expressed as a function of distance from mouth (x km). For morphological variables, their mathematical equations were expressed as a function of the degree of sea-level rise (0.1, 0.3 and 0.5 m). Changes in environmental variables in response to sea-level rise can then be derived by changing the value of x km or the degree of sea-level rise (see section 3.3.4 for details), and substituting the new environmental values for the three models provides changes in macrobenthic biomasses in response to sea-level rise. The model outputs from the survey carried out in 2003 were used to provide a baseline against which the simulation outputs for the different sea-level rise scenarios (0.1, 0.3 and 0.5 m) with the effects of environmental changes were compared.

All graphics were created in Excel and SPSS for Windows, and all statistical analyses were performed with SPSS for Windows.

3.3 Results

3.3.1 Physical characteristics

3.3.1.1 Salinity, tidal range and exposure

Because the study site was located within a relatively small part of the Humber estuary, mean annual salinity varied little between 23.9 psu at transect *a* and 25.8 psu at transect *i*. Local tidal range at the study site was 4.55 m, and exposure increased from 0.68 at transect *a* to 0.98 at transect *i*, indicating transects in the natural zone were more exposed to the open coastal environment than those in the squeezed zone, although this factor is likely to be more related to geographical location than the squeezed nature of the beach.

3.3.1.2 Sedimentary elements

Sediment at the study site consisted of the range of sizes between fine sand and silt, and average median grain size (phi) per transect varied from 2.56 (fine sand) to 3.75 (silt) (Fig. 3.3.a).

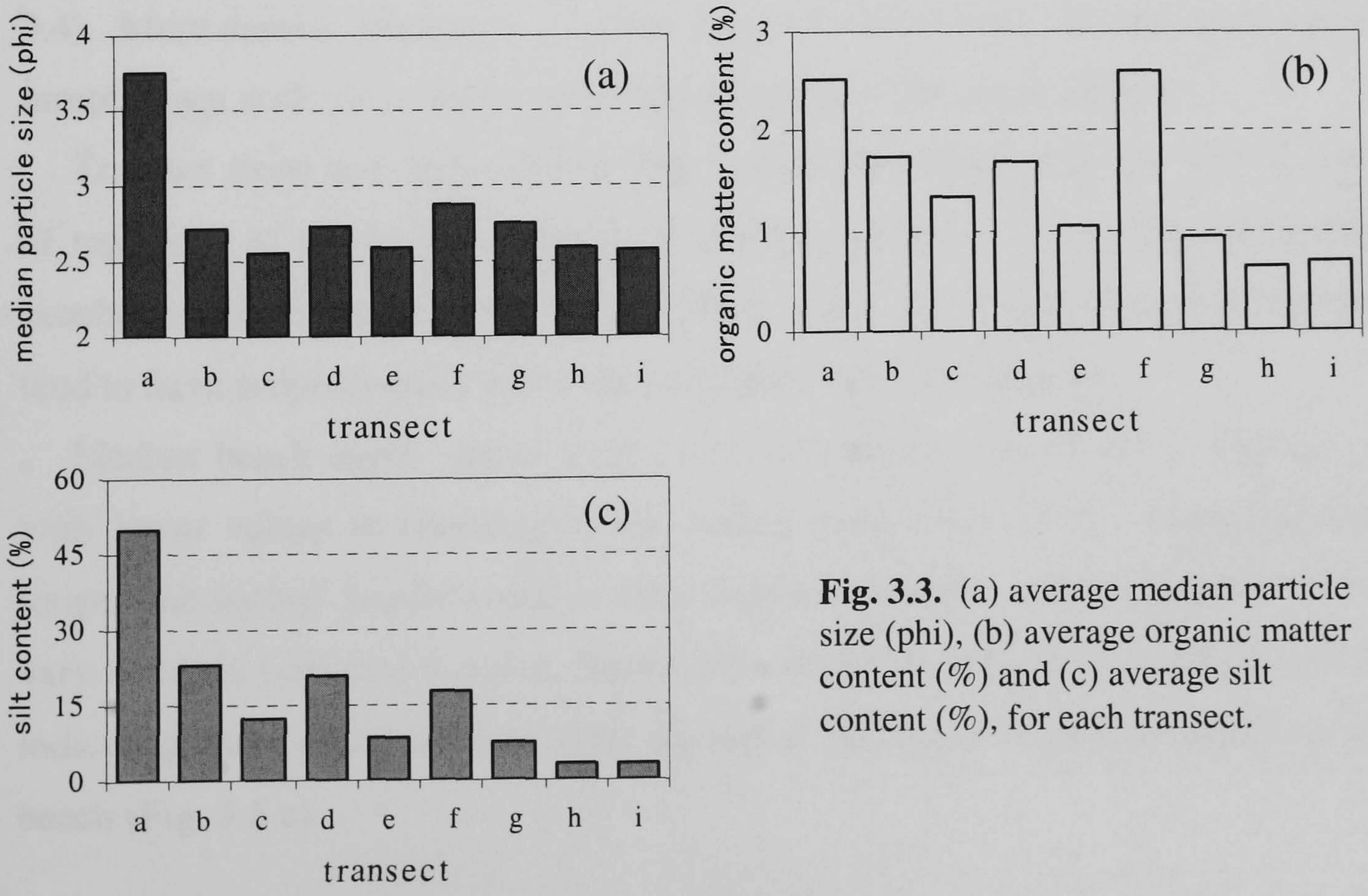


Fig. 3.3. (a) average median particle size (phi), (b) average organic matter content (%) and (c) average silt content (%), for each transect.

There was no obvious trend in average median sizes across the study site, except for transect *a* where sediments were much finer than any other transect. In contrast, silt content and organic matter content showed clearer trends from the natural zone to the squeezed zone (Fig. 3.3.b, c). Silt content varied from 3.3 % at transect *i* to 49.7 % at transect *a*, whereas organic matter content varied from 0.7 % at transect *h* to 2.5 % at transect *a*. The high organic matter content observed at transect *f* was attributable to the presence of a patchy sediment area with a large amount of plant debris at higher shore levels, which increased average value of organic matter content for the transect. Sediment generally become muddier with more organic matter and a high silt content from the most natural beach through to most squeezed beach.

3.3.1.3 Morphological elements

The width of beach length varied from 195 m to 1230 m. The beach face profile of each transect is shown in Fig. 3.4. Transects *a*, *b*, *c* and *d* are situated in front of the land-claimed area in Grimsby (squeezed), and transects beyond *e* towards the outer coastal region are natural beaches. The squeezed beaches (*a* – *d*) are characterised by a short, steep beach width (< 800m), and the beach face profiles became homogenously lower from mean high water to mean low water level (Fig. 3.4). More natural beaches (*e* – *i*) were longer (> 800m) and showed more varied morphology, with some flatter and deepened areas within each transect.

Transect slope and mean station slope values decreased from the natural zone of transect *i* to the squeezed beach of transect *a* (Fig. 3.5.a, b) suggesting that beaches become progressively steeper when squeezed and that natural beaches tend to have proportionally more flat areas than squeezed beaches.

Median beach depth varied from 21.1% at transect *i* to 89.0% at transect *g*, with lower values in transects in the natural zone (Fig. 3.5.c), suggesting that longer and natural beaches tend to have concave profiles. Median depth did not vary between transects *a* and *e*, despite the observed differences in beach width, indicating that coastal squeeze does not affect the value of median depth of the beach (Fig. 3.5.c).

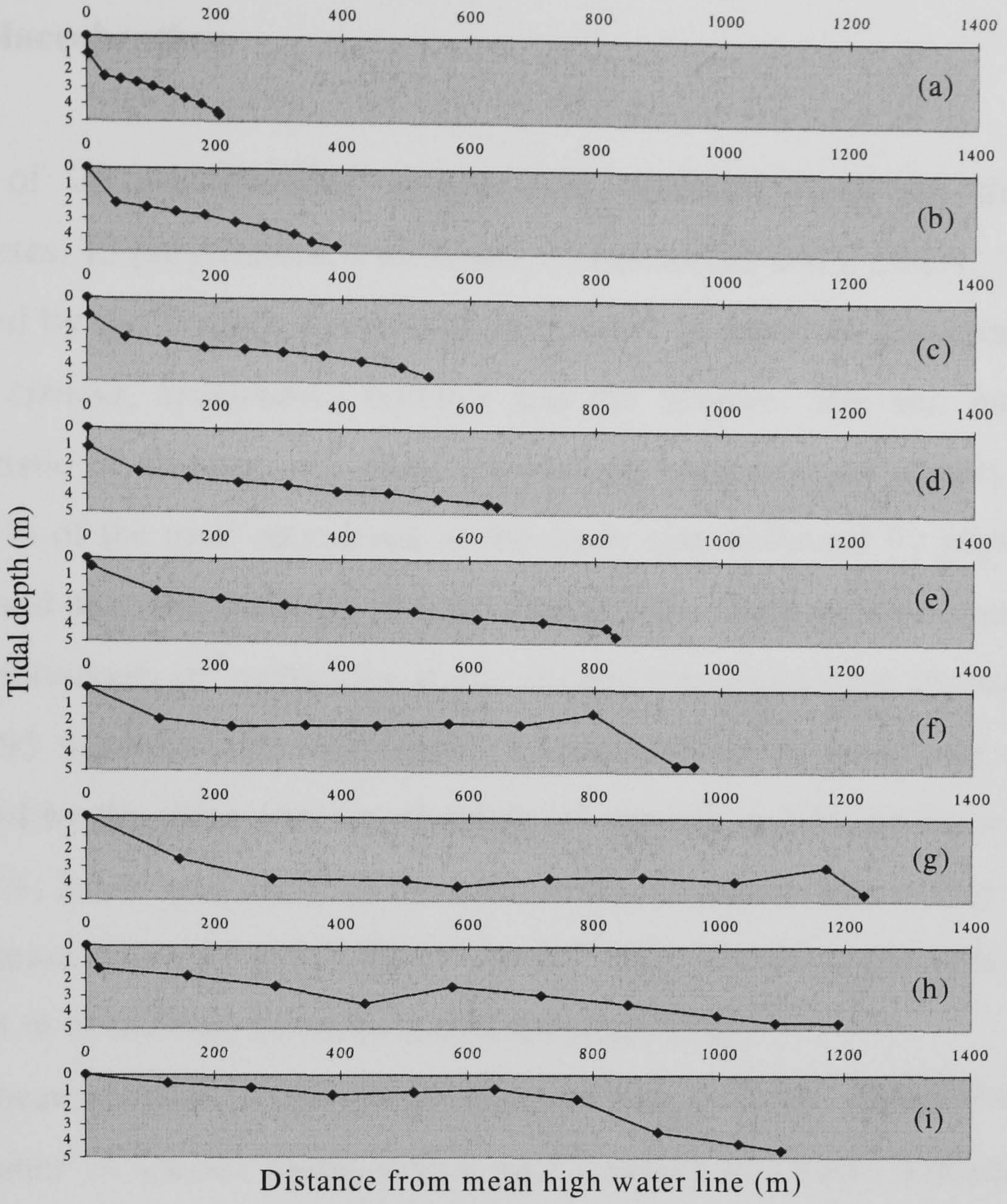


Fig. 3.4. Intertidal profiles at 9 transects sampled in this study. Because the mean tidal range can be assumed the same within the study site, tidal depth is not expressed in percentage in this diagram (cf. Fig. 2.9).

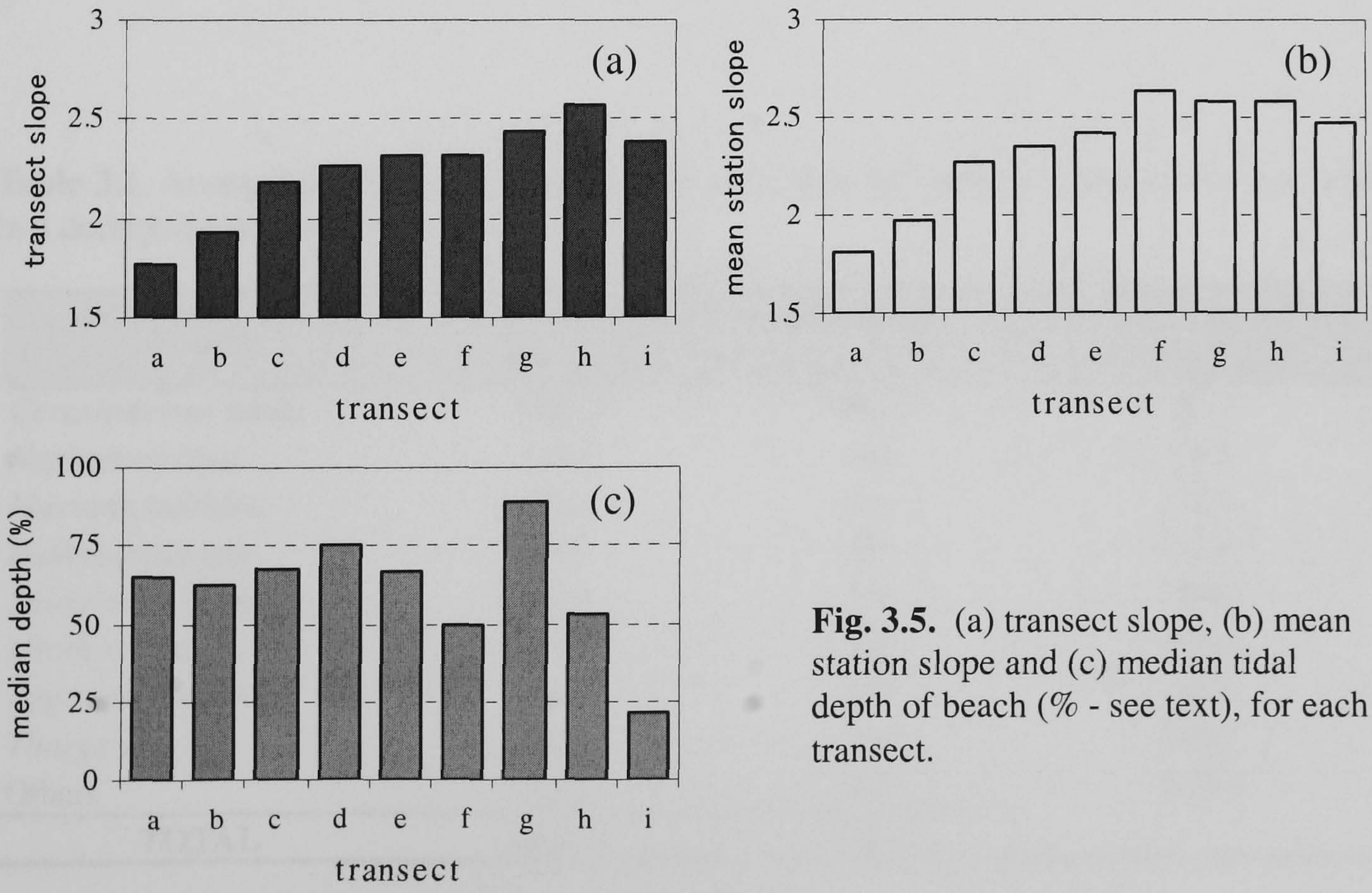


Fig. 3.5. (a) transect slope, (b) mean station slope and (c) median tidal depth of beach (% - see text), for each transect.

3.3.2 Macrobenthos

A total of 24 macrobenthic species was recorded from the study site: 2 oligochaetes, 13 polychaetes, 4 molluscs, 3 crustaceans and 2 others. The site was dominated by the bivalve, *Cerastoderma edule.* In addition, polychaetes such as *Nephtys cirrosa, Spiophanes bombyx* and the bivalve, *Macoma balthica* were characteristic of all sites. *C. edule* showed the highest mean density accounting for 69.2 % of the total abundance at the study site, followed by *Nephtys cirrosa* (9.0 %) and *Macoma balthica* (5.1 %) (Table 3.1). Average total abundance was 1562.9 individuals m^{-2} within the study site and 8 macrobenthic species accounted for over 95 % of the total abundance. Trends in mean biomass were also similar, dominated by the three species, *C. edule, N. cirrosa* and *M. balthica*, accounting for 91.2 %, 3.1 % and 2.4 % of the total biomass, respectively (Table 3.2). Mean total biomass was 6.33 g AFDW m^{-2}, and 13 macrobenthic species accounted for over 99.9 % of the total biomass within the study site.

The mean number of species per transect was 11.0, and Figure 3.6 shows the total number of species found within each transect and mean number of species per station. Transect *f* showed the highest species richness of 14 and transect *h* showed the lowest of 8, whereas mean species richness was more uniform with the highest value of 4.8 at transect *a* (Fig 3.6).

Table 3.1. Average abundance of macrobenthic species in the study site. The species are listed in a decreasing rank order of mean density m^{-2}.

Species	Mean density (ind. m^{-2})	Percentage (%)	Cumulative percentage (%)
Cerastoderma edule	1082.3	69.2	69.2
Nephtys cirrosa	140.6	9.0	78.2
Macoma balthica	79.4	5.1	83.3
Bathyporeia spp.	63.3	4.0	87.4
Spiophanes bombyx	46.9	3.0	90.4
Urothoe spp.	35.1	2.2	92.6
Pygospio elegans	29.4	1.9	94.5
Tharyx spp.	22.3	1.4	95.9
Others	63.6	4.1	100.0
TOTAL	**1562.9**	100.0	

Table 3.2. Average biomass of macrobenthic species in the study site. The species are listed in a decreasing rank order of mean biomass g AFDW m^{-2}.

Species	Mean biomass (g AFDW m^{-2})	Percentage (%)	Cumulative percentage (%)
Cerastoderma edule	5.77	91.2	91.2
Nephtys cirrosa	0.20	3.1	94.3
Macoma balthica	0.15	2.4	96.7
Spiophanes bombyx	0.13	2.0	98.6
Urothoe spp.	0.03	0.4	99.1
Petricola pholadiformis	0.01	0.2	99.3
Bathyporeia spp.	0.01	0.2	99.4
Ampharetidae	0.01	0.1	99.6
Dipteran larvae	0.01	0.1	99.6
Anaities maculata	0.005	0.07	99.7
Nephtys hombergii	0.004	0.07	99.8
Scoloplos armiger	0.004	0.06	99.8
Retusa obtusa	0.002	0.04	99.9
Pygospio elegans	0.002	0.04	99.9
Tharyx spp.	0.002	0.03	99.9
Others	0.003	0.05	100.0
TOTAL	**6.33**	100.0	

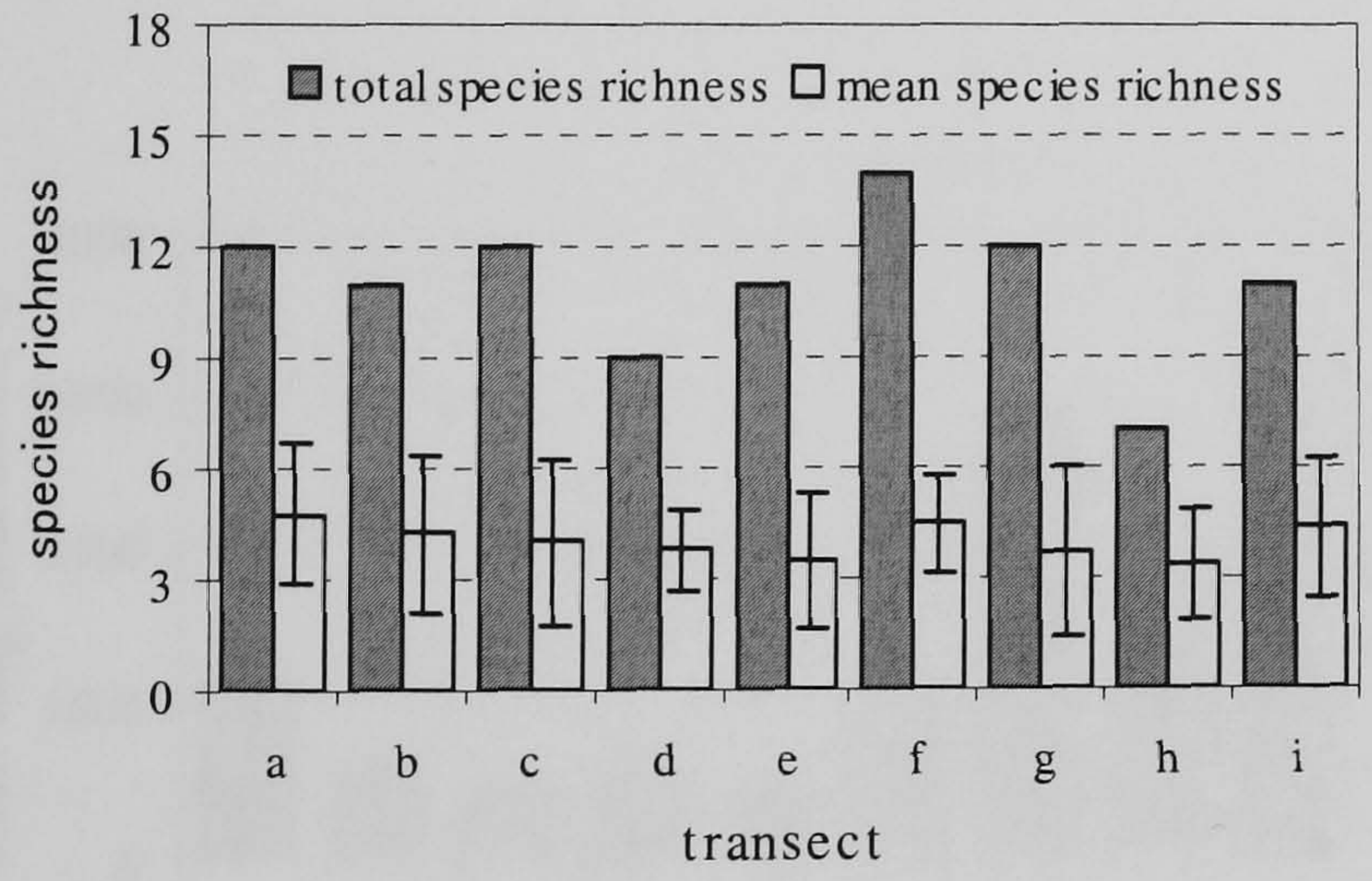


Fig. 3.6. Species richness of intertidal macrobenthos for each transect (filled bars) and mean species number per station (open bars). Vertical bars show ± SD.

There were no clear trends in species richness in relation to the beach width gradient, from the natural beach to the squeezed beach (from transect *i* to *a* in Fig. 3.6), suggesting that coastal squeeze does not have an influence on species richness in estuarine intertidal area.

Figure 3.7 shows mean total abundance for each transect, which ranged from 354 to 2994 individuals m^{-2}. Transect *i*, *g* and *h* had the most individuals and these were located at the natural side of the study site. Total abundance was lower on the remaining transects, located towards the squeezed end, except for transect *a* that had a slightly higher value than any other transect in the squeezed area.

A similar overall trend can be seen for mean total biomass at the transect scale (Fig. 3.8). Smaller biomasses were found in the squeezed area between transect *a* and transect *e*, than transects located in un-squeezed area which had a higher biomass with the highest value of 14.9 g AFDW m^{-2} at transect *g* (Fig.3.8).

The transects were grouped into two zones (squeezed zone and natural zone) (Fig. 3.2.c), with transects *a* to *d* in the "squeezed zone" and transects *f* to *i* in the "natural zone" (4 replicating transects per zone). Biomass data for each sampling station were log(x+1)-transformed to capture the trends in smaller biomass values observed in squeezed zone. For each zone, the transformed data were then re-grouped according to station number from 1 to 9, which represent similar tidal depth across transects from top of the shore to mean low water level (MLWL) (Fig. 3.9). Both zones showed peak of the biomass around station 2, 3 and 4, but the values decreased close to 0 towards both station 1 and station 9. In addition, the amplitude of biomass distribution in squeezed zone was much lower than in natural zone (Fig 3.9).

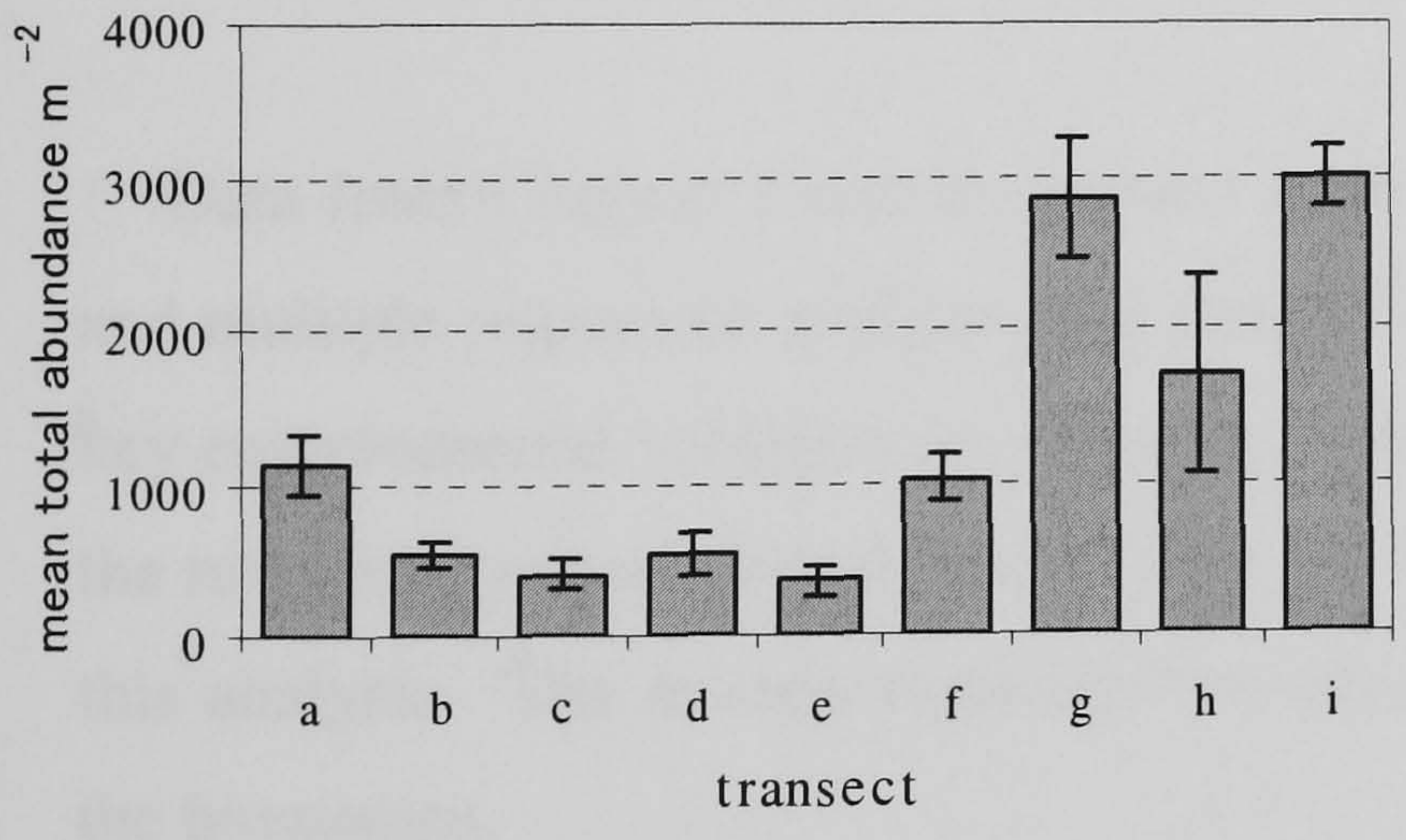


Fig. 3.7. Mean total abundance of intertidal macrobenthos for each transect (ind. m^{-2}). Vertical bars show ± SD.

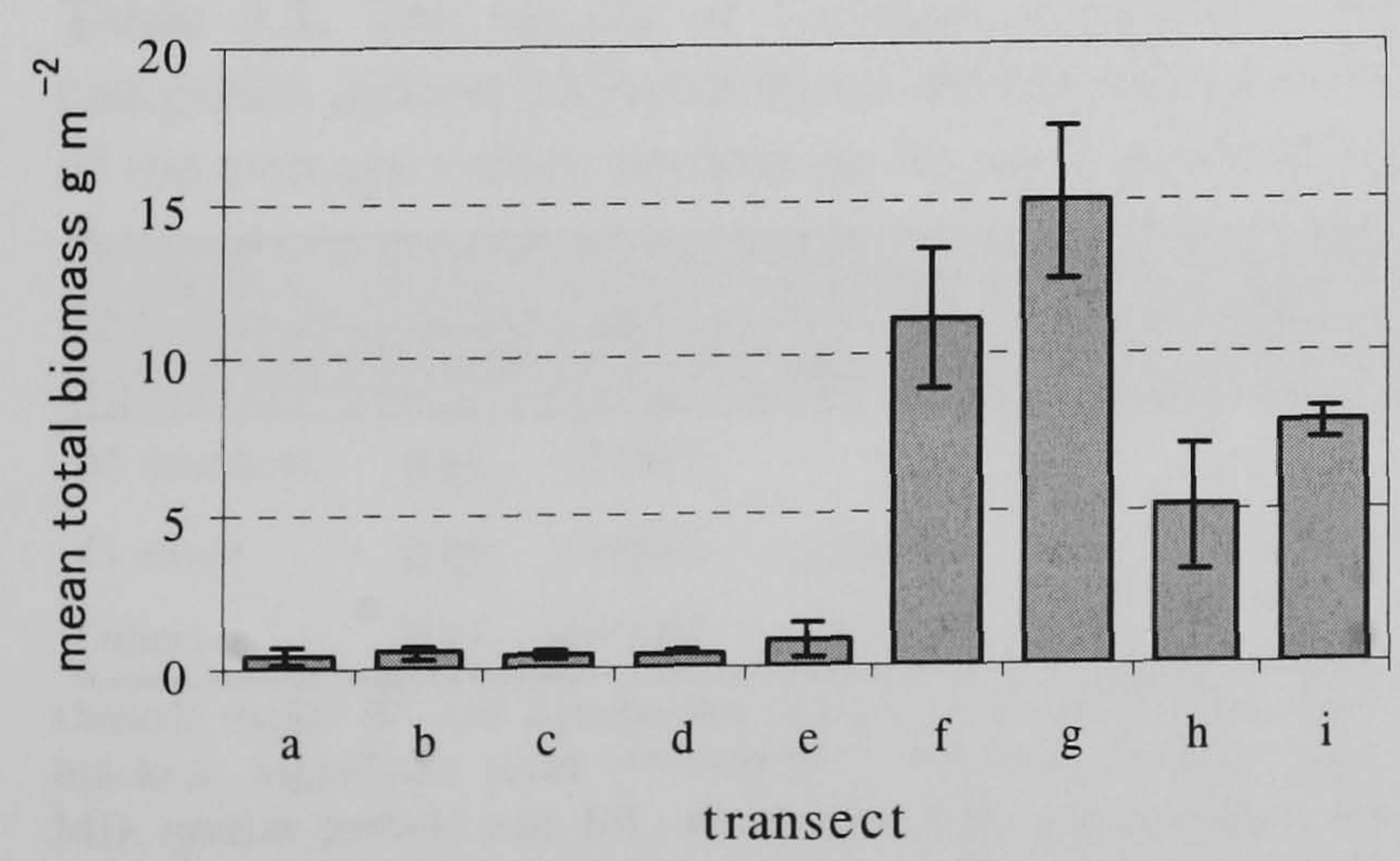


Fig. 3.8. Mean total biomass of intertidal macrobenthos for each transect (g AFDW m^{-2}). Vertical bars show ± SD.

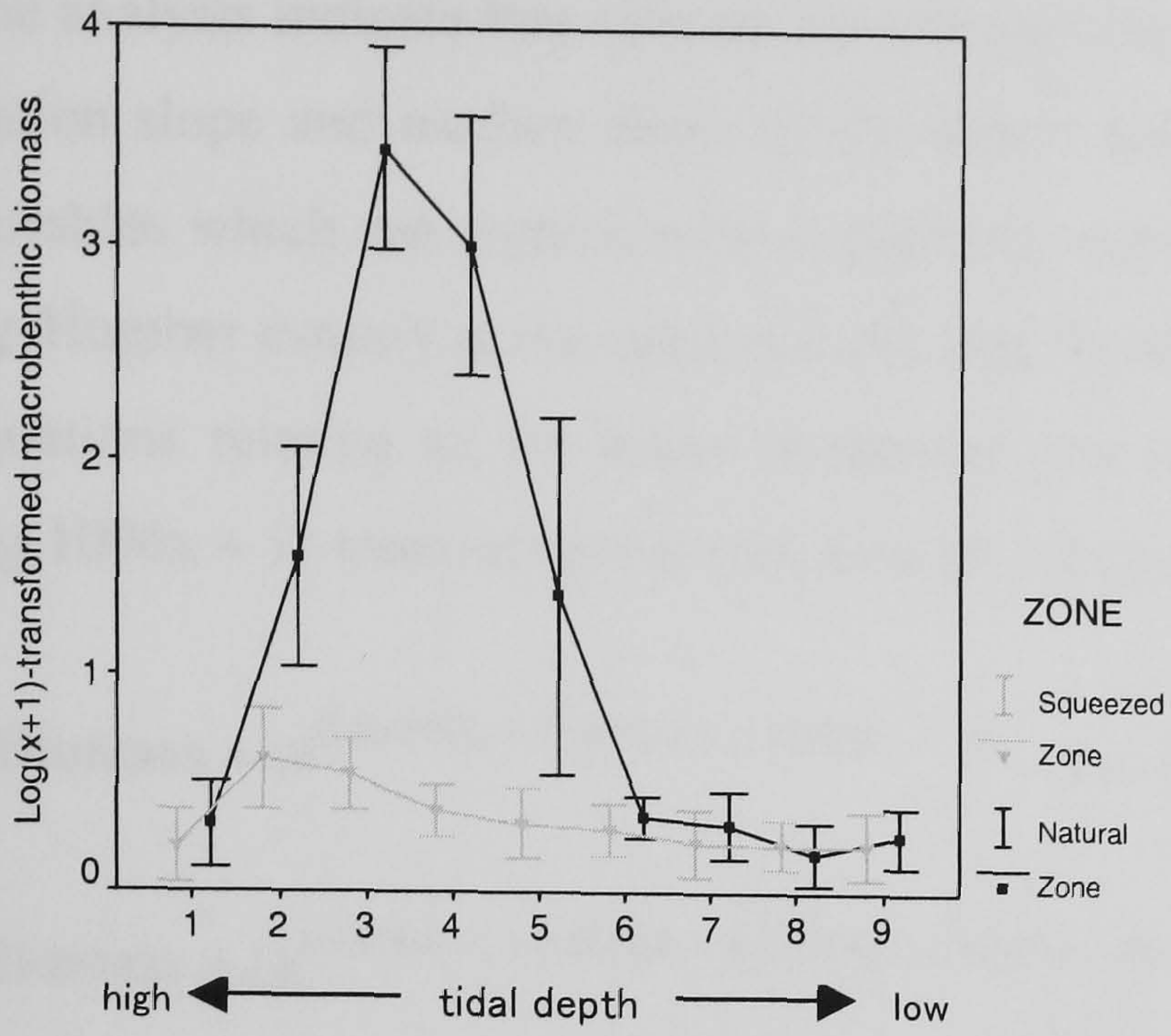


Fig. 3.9. Relationship between biomass and shore level for squeezed and natural zones. Vertical bars show 95% confidence interval.

These differences in mean biomass between two zones were also statistically significant (n=216, F=40.0, p<0.00001), suggesting that coastal squeeze had negative impacts on the biomass of benthic macrofauna in estuarine intertidal habitats.

3.3.3 Validating the macrobenthic models

Data from Chapter 2 and the present study were combined (see section 3.2.2) and multiple regression analysis was used to re-assess the explanatory role of the key environmental variables for biomass of *M. balthica*, *C. edule* and biomass for the remaining species (others) at the transect scale. Table 3.3 shows the results of this analysis. The models explained between 80 % and 85 % of the variance in the biomasses.

Table 3.3. The results of multiple regression analysis of the three macrobenthic biomass categories against physical variables from three environmental components. The data consisted of the average values per station for each transect from the 2003 and 2004 survey (n = 23).

	R^2	P	LONGITUDINAL	SEDIMENTARY		MORPHOLOGICAL		
			SAL	MD	SIL	TSL	SSL	MDP
M. balthica	0.80	<0.0001			$(0.77)_2$***	$(0.86)_1$***	$(-0.51)_3$*	
C. edule	0.85	<0.0001	$(0.66)_3$**	$(0.73)_2$***	$(-0.77)_1$***		$(0.70)_4$***	
Others	0.81	<0.0001	$(0.83)_2$***		$(0.87)_1$***			$(-0.52)_3$*

Overall model R^2 and significance are presented in the first two columns, along with partial correlation coefficients in brackets. Significant level: ***P<0.001; **P<0.01; *P<0.05. Environmental variables: SAL: salinity (log-transformed); MD: median particle size; SIL: silt content; TSL: transect slope; SSL; station slope: MDP: median depth of the beach. The subscript number indicates the rank order in which environmental variables were retained in the stepwise procedure. Details of the results (e.g. model coefficients) can be found in Appendix 4.

The analysis indicate that salinity, median particle size, silt content, transect slope, station slope and median depth of the beach were the six of key environmental variables which can significantly explain the variation in macrobentic biomass in the Humber estuary at the estuary scale and the local scale. From the results, the equations relating to the mean biomasses that are also back transformed from log(1000x + 1)-transformation (see section 3.2.2) can be expressed as:

$$\text{mBiomass} = (e^{0.044*\text{SIL} + 7.38*\text{TSL} - 2.19*\text{SSL} - 7.41} - 1) / 1000 \qquad \text{(equation 1)}$$

$$\text{cBiomass} = (e^{3.78*\text{SAL} + 13.34*\text{MD} - 0.41*\text{SIL} + 3.38*\text{SSL} - 45.57} - 1) / 1000 \qquad \text{(equation 2)}$$

$$\text{oBiomass} = (e^{1.58*\text{SAL} + 0.036*\text{SIL} - 0.016*\text{MDP} + 1.20} - 1) / 1000 \qquad \text{(equation 3)}$$

where mBiomass, cBiomass and oBiomass represent mean biomass for *M. balthica, C. edule* and the other remaining macrobenthos (g AFDW m^{-2}) at the transect scale, respectively, and SAL, MD, SIL, TSL, SSL and MDP denote values for salinity (psu) (log-transformed), median particle size (phi), silt content (%), transect slope, station slope and median depth of the beach (%). The model for predicted total biomass can now be expressed as:

$$\text{Total Biomass (g AFDW m}^{-2}\text{)} = \text{mBiomass} + \text{cBiomass} + \text{oBiomass} \qquad \text{(equation 4)}$$

Based on equations 1 to 4, Figure 3.10 and Figure 3.11 show the model predicted biomasses against observed biomasses at the system scale (estuary) and local scale (present study site), respectively. The models effectively captured the variability in observed biomasses at both scales. In Chapter 2, the system-averaged biomass was estimated as 8.65 g AFDW m^{-2} and total macrobenthic biomass in the system as 1037.5 t AFDW. The model predictions for the two values were 8.29 g AFDW m^{-2} and 995.4 t AFDW, respectively, which fall within ± 5 % of precision from the observed values. These figures and the good fit generated by the equations indicate that the models provide a reasonable description of how macrobentic biomass in the Humber can be predicted from the six key environmental variables.

Estuarine system scale

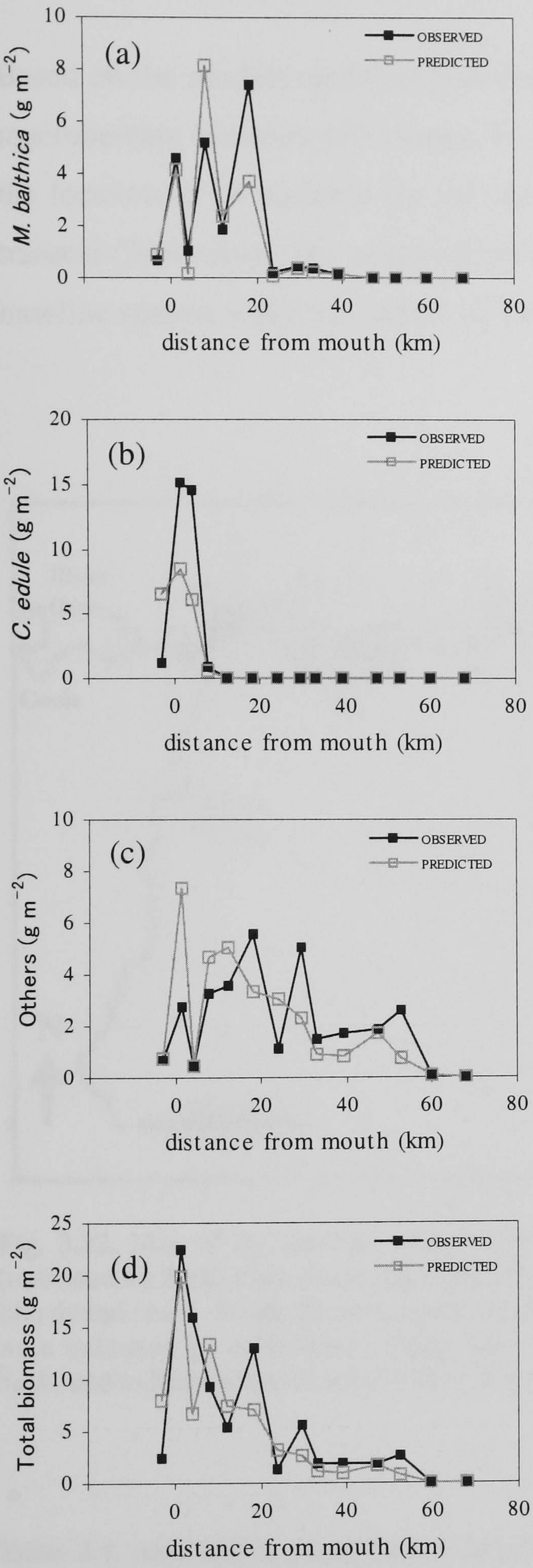


Fig. 3.10. Observed and predicted biomass (g AFDW m^{-2}) at the system scale for: (a) *M. balthica*; (b) *C. edule*; (c) other macrobenthos; (d) total macrobenthos.

Local site scale

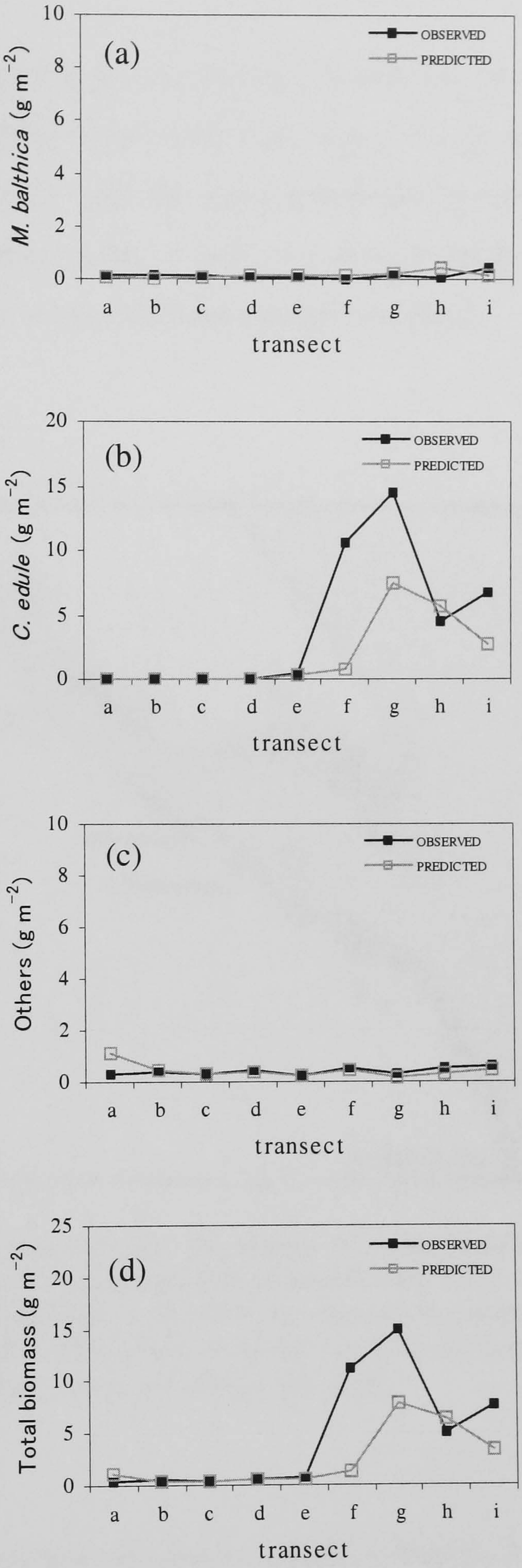


Fig. 3.11. Observed and predicted biomass (g AFDW m^{-2}) at the local site scale in Grimsby for: (a) *M. balthica*; (b) *C. edule*; (c) other macrobenthos; (d) total macrobenthos.

3.3.4. Model simulations

Based on the models established in this study, it is now possible to simulate how macrobenthic biomass will change in response to sea-level rise. Fig. 3.12 shows the location of 14 transects for the simulations, and the area represented by each transect (Table 3.4). The model outputs shown in Fig. 3.10 were used to provide a baseline against which the effects of environmental changes can be compared.

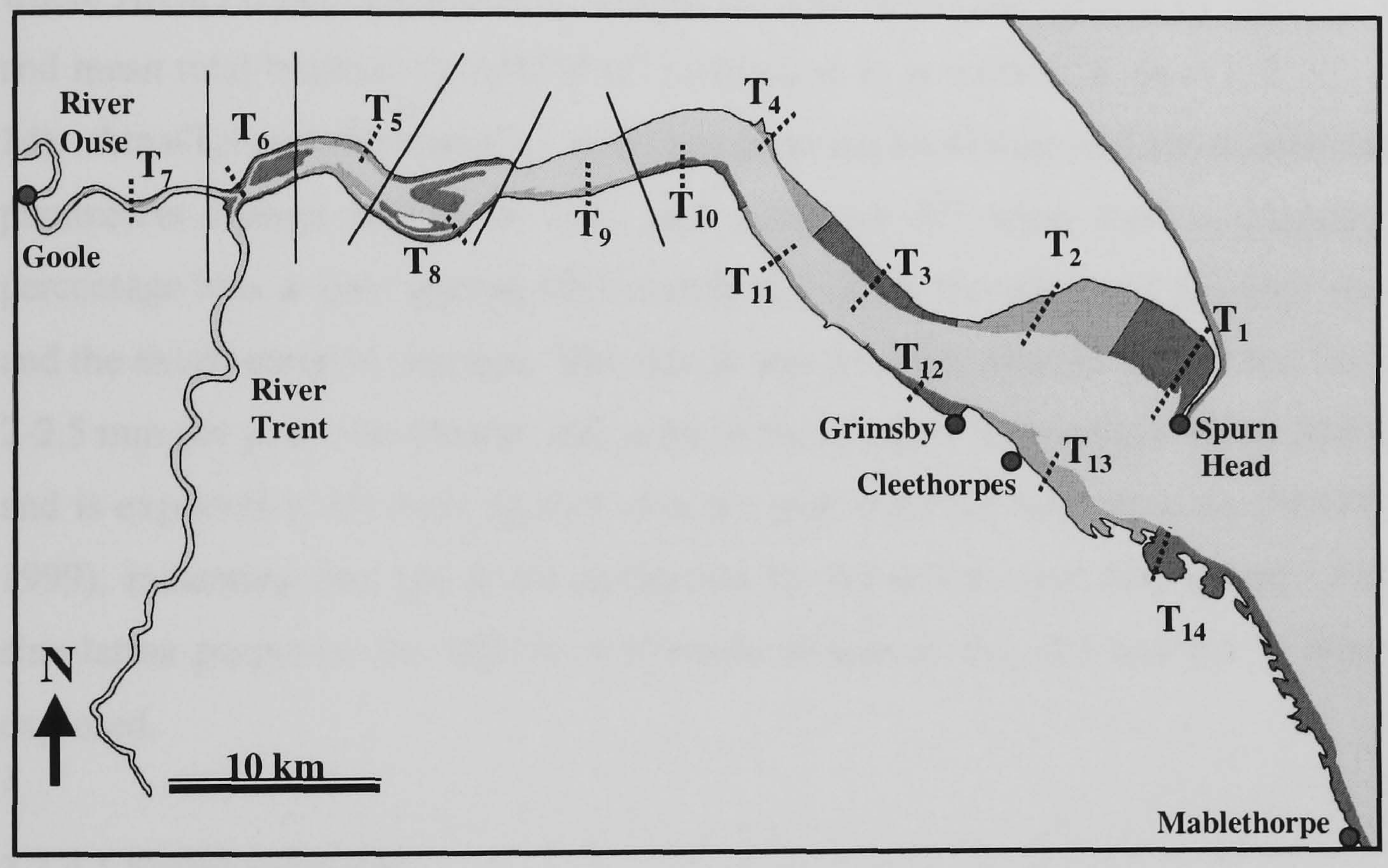


Fig. 3.12. Map of the Humber estuary showing locations and boundaries of intertidal areas (indicated by dark or plain colour) represented by their corresponding transects from T_1 to T_{14} (the dotted lines) When the transects represented intertidal area across the channel, boundaries were indicated by solid lines. Study area covered from Goole to Spurn Head on the north bank, and to Mablethorpe on the South. River Trent was not included in this study.

Table 3.4. Area and its percentage of intertidal habitats represented by transects from T_1 to T_{14}.

Transect	T_1	T_2	T_3	T_4	T_5	T_6	T_7	T_8	T_9	T_{10}	T_{11}	T_{12}	T_{13}	T_{14}
Area(T_n) (km^2)	19.6	13.4	7.6	3.7	4.8	5.3	0.8	7.3	5.2	6.7	1.8	3.5	13.6	26.5
Percentage (%)	16.4	11.2	6.3	3.1	4.0	4.4	0.7	6.1	4.4	5.6	1.5	2.9	11.4	22.1

System average biomass (g AFDW m^{-2}) and system total biomass (t) can be expressed as:

System total intertidal area = $\sum \mathrm{Area}(T_n)$ (equation 5)

System average biomass = $\sum(\mathrm{tBiomass}(T_n) \times \mathrm{Area}(T_n)) / \sum \mathrm{Area}(T_n)$ (equation 6)

System total biomass = $\sum(\mathrm{tBiomass}(T_n) \times \mathrm{Area}(T_n))$ (equation 7)

where $\mathrm{Area}(T_n)$ and $\mathrm{tBiomass}(T_n)$ denote the area (km^2) represented by transect *n* and mean total biomass (g AFDW m^{-2}) (equation 4) at transect *n* ($n = 1, 2, 3, ..., 14$). $\mathrm{Area}(T_n)$ and $\mathrm{tBiomass}(T_n)$ will change as sea level rises and environmental parameters change (equations 1-4), and equations 5-7 were used to calculate percentage loss or gain against the baseline to express the effects of sea-level rise and the environmental changes. The rate of sea-level rise relative to land has been 2-2.5 mm per year over the last 100 years in the Humber (Edwards & Winn, 2006), and is expected to increase up to 6 mm per year over the next 50 years (MAFF, 1999), indicating that sea levels could rise by 0.1-0.3 m over that period. For simulation purposes, the effects of a sea-level rise of 0.1, 0.3 and 0.5 m were explored.

3.3.4.1 Saline intrusion

When sea level rises, water depth increases and this is likely to cause intrusion of saline water up the estuary (Jones, 1994). Simple mathematical calculations were carried out in order to simulate how salinity intrusion occurs in response to sea-level rise. The Humber estuary has a depth of < 5 m around Trent Falls, increasing to almost 20 m at Spurn Point relative to ordnance datum (Jickells et al., 2000). Because the distance between Trent Falls and Spurn Point is approximately 60 km, the water body of the Humber can be expressed as a right triangle as shown in Fig. 3.13.

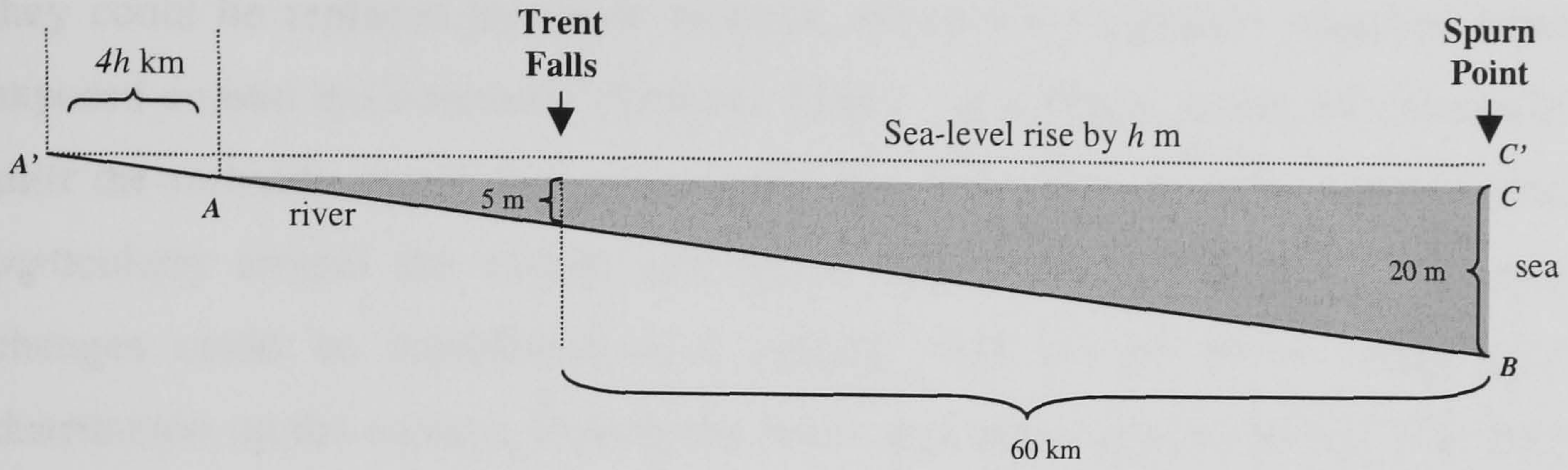


Fig. 3.13. A simplified diagram of the Humber estuary showing the depth of Trent Falls and Spurn Point. Shaded triangle *ABC* represents the current water body in the Humber estuary and dotted line *A'C'* represents the sea level risen by *h* m. Horizontal distance *A'A* indicates the distance the salinity has intruded up river, which can be calculated as *4h* km.

When sea level rose by *h* m, horizontal distance between *A* and *A'* would be the distance the salinity distribution has intruded up river (Fig. 3.13). Because triangle *ABC* is analogous to triangle *A'BC'*, when sea level rises by *h* m, the horizontal distance between *A'* and *A* can be mathematically derived as *4h* km. Therefore, salinity distributions are assumed to migrate up river by 0.4, 1.2 and 2.0 km in response to sea-level rise of 0.1, 0.3 and 0.5 m. Salinity is also a function of distance from mouth (Chapter 2, Fig. 2.3, $R^2 = 0.99$), the equation is expressed as:

$$\mathrm{SAL}(x) = 0.0023\,x^2 - 0.54\,x + 28.38 \qquad \text{(equation 8)}$$

where $\mathrm{SAL}(x)$ denotes salinity value at a location x km from mouth of the Humber estuary. If the saline intrusion causes a shift of salinity distribution by x_1 km up the estuary, a new salinity value for a fixed transect position can be expressed as $\mathrm{SAL}\,(x-x_1)$. In this study, values of x_1 of 0.4, 1.2 and 2.0 km were thus used.

3.3.4.2 Sediment changes

Increased water depth will result in enhanced wave and tidal energy, which in turn will affect sedimentary processes in estuarine environments (Crooks, 2004). Mudflats will migrate landwards and upstream in response to sea-level rise, and they could be replaced by sandy beaches, which have similarly migrated from exposed coastal environments (Pethick, 1996). In addition, saline intrusion will shift the turbidity maximum up river and this will intensify local siltation rates particularly around the middle and upper regions of the Humber. All these changes could be manifested as a gradual shift in the entire sedimentary distribution up the estuary, leaving the lower and outer regions sandier and inner estuary muddier. In this study, the extent of the sediment migration were determined in the same way as for saline intrusion, assuming that the entire sedimentary distribution will shift by 0.4, 1.2 and 2.0 km in response to a sea-level rise of 0.1, 0.3 and 0.5 m. Because the spatial patterns of sedimentary distribution were markedly different between the north and south bank of the Humber estuary around the outer region (Fig. 3.14), mathematical equations were derived separately for both banks so that sediment properties can be expressed as a function of distance from mouth. Figure 3.15 shows silt content values on the south bank in relation to distance from mouth.

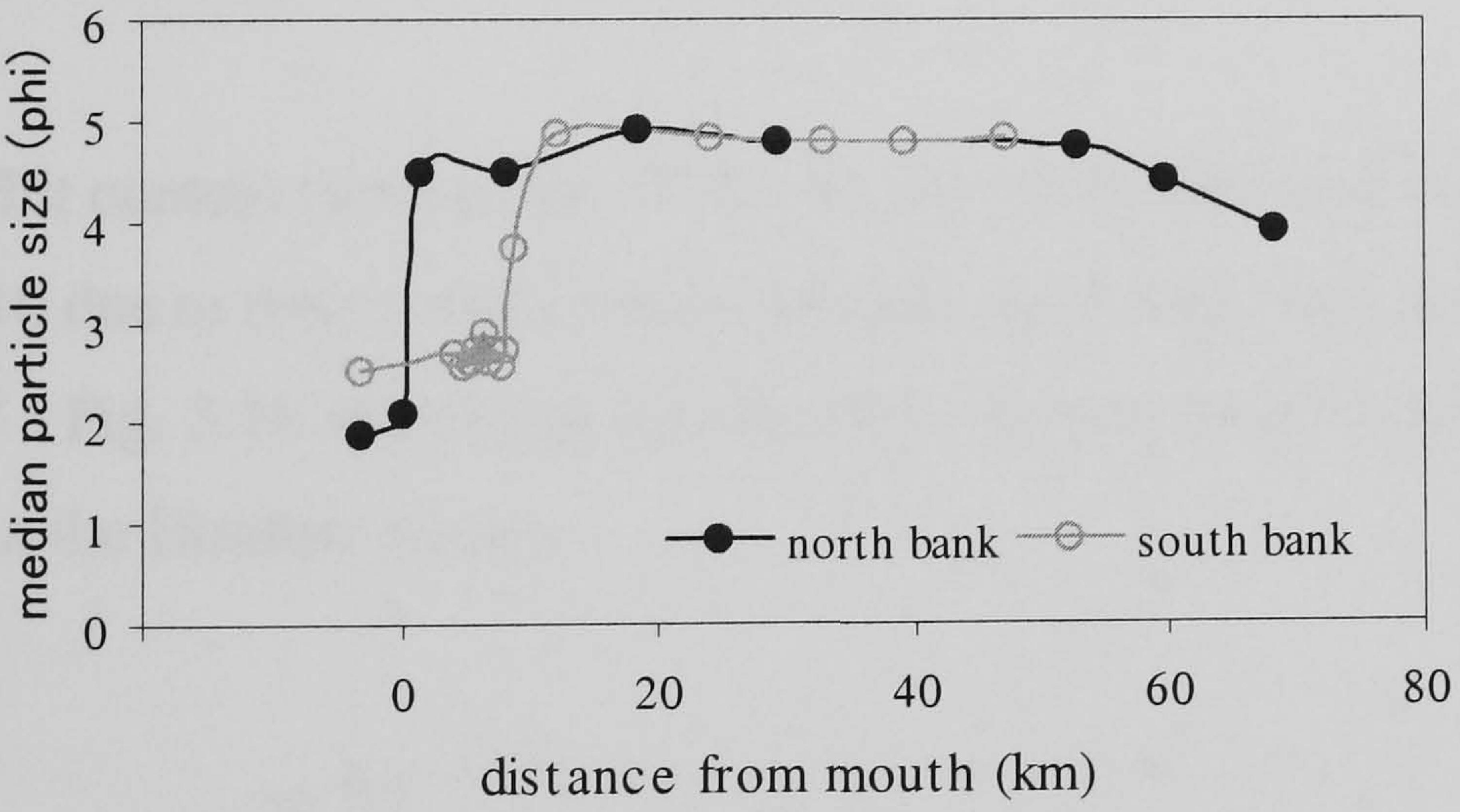


Fig. 3.14. Relations between average median particle size (phi) and distance from mouth (km) on the north (filled circles) and south banks (open circles) of the Humber estuary.

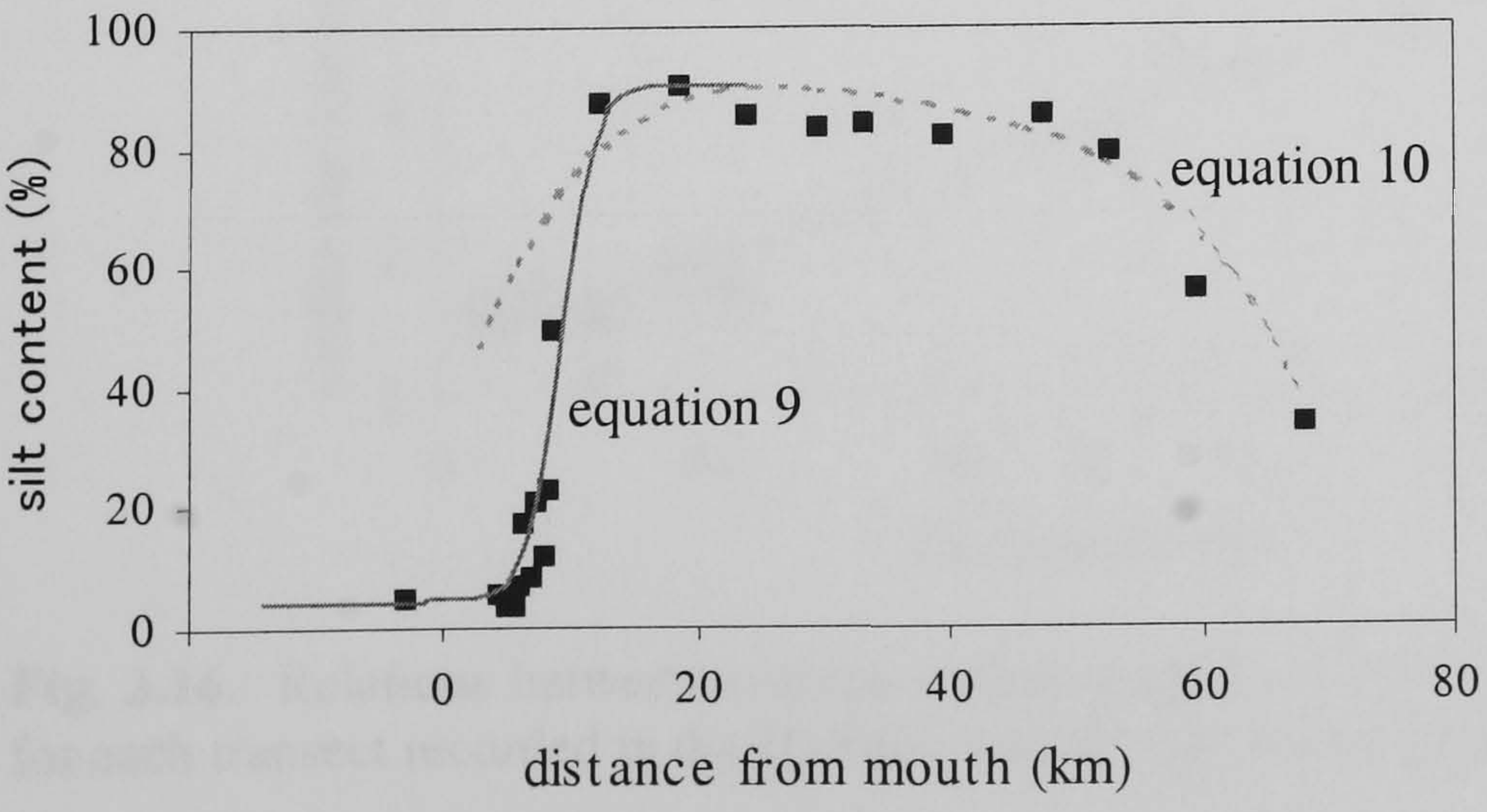


Fig. 3.15. Relations between mean silt content (%) and distance from mouth (km) on the south bank of the Humber estuary showing two fitted curves (equations 9 and 10).

As shown on the graph, the relations between silt content and the distance can be expressed by two separate mathematical equations (one as a logistic curve up to approximately 20 km, and the other as a 5th order polynomial further than 20 km from mouth) as follows:

$SIL_{south}(x) = (83.0 / (1 + e^{8.5 + x})) + 3.0$ (equation 9)
$(x < 20\ km)$

$SIL_{south}(x) = 6.6*10^{-8} x^5 - 2.0*10^{-5} x^4 + 3.9*10^{-3} x^3 - 0.24 x^2 + 6.26 x + 30.25$
$(x > 20\ km)$ (equation 10)

where $SIL_{south}(x)$ denotes the silt content at a location x km from the mouth on the south bank of the estuary.

Similarly, the equation for silt content on the north bank was derived as follows:

$SIL_{north}(x) = (84.7 / (1 + e^{-1.0 + 0.7x})) + 1.3$
$(x < 20\ km)$ (equation 11)

Silt content further than 20 km on the north bank was expressed as in the equation 10 due to their similar trends between north and south banks.

Fig. 3.16 shows the relationship between median particle size and silt content in the Humber estuary.

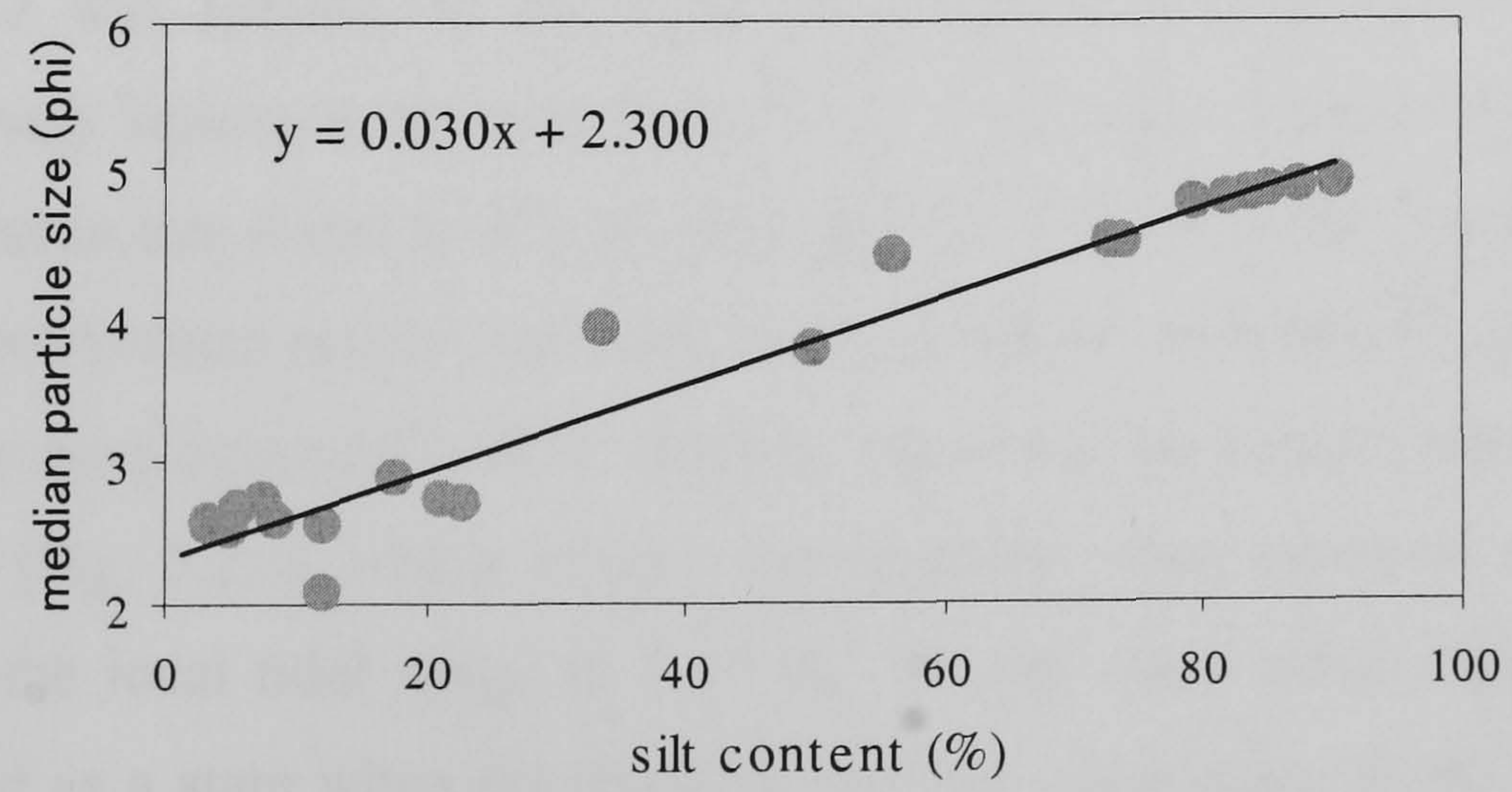


Fig. 3.16. Relations between average median particle size (phi) and mean silt content (%) for each transect recorded in the Humber estuary with linear equation.

Because of the strong linear relationship between the two variables ($R^2 = 0.95$), median particle size can be expressed as a function of distance from mouth by using equation 9 – 11 and the linear equation shown in Fig 3.16:

$$MD_{south}(x) = 0.030 * SIL_{south}(x) + 2.3 \qquad \text{(equation 12)}$$

$$MD_{north}(x) = 0.030 * SIL_{north}(x) + 2.3 \qquad \text{(equation 13)}$$

where $MD_{south}(x)$ and $MD_{north}(x)$ represent median particle size at a location x km from the mouth of the Humber estuary. Shifts in sediment properties can now be expressed as a function of distance by changing the x value.

3.3.4.3 Morphological variables

Intertidal area loss (km^2) at transect n can be expressed as:

$$\text{Intertidal loss } (T_n) = (SLR / TR(T_n)) \times Area(T_n) \qquad \text{(equation 14)}$$

where SLR and $TR(T_n)$ denote sea-level rise (m) and local mean tidal range (m) at transect n, respectively. $\sum(\text{Intertidal loss}(T_n))$ was compared with the current intertidal area to calculate the percentage total loss of intertidal area in the Humber.

From the field observations at Grimsby, both transect slope and station slope were found to become steeper when beaches became squeezed (Fig. 3.5.a, b). Transect *e* was located on the edge of natural zone and beach widths were progressively squeezed towards transect *a*. If an area of land-claim extends to MLWL (mean low water level), no intertidal flat is left in front of sea defence wall and tidal movement occurs vertically between MLWL and MHWL on the defence wall. The most squeezed end of Grimsby intertidal flat beyond transect *a* falls in this state (Fig. 3.2.c), which mimics the situation when sea-level rise has taken place by the local tidal range of 4.55 m. On the other hand, transect *e* can be interpreted as a state when no sea-level rise has taken place (0 m) since MHWL starts to be squeezed by the sea defence wall beyond this transect towards transect *a*. Therefore the beach width gradient from transect *e* (833 m) through *a* to the squeezed end (0 m) corresponds to the extent of sea-level rise from 0 to 4.55 m. Using the five transects between *a* and *e*, transect slope values were plotted against beach width (Fig. 3.17). If sea level rose by 4.55 m, the beach width would become 0 m and the transect would become a vertical wall (transect slope → -∞). Therefore the relationship should be expressed as a logarithmic curve ($R^2 = 0.95$) as shown in Fig. 3.17:

$$\text{Transect slope (WID)} = 0.42*\text{Ln(WID)} - 0.52 \qquad \text{(equation 15)}$$

where WID and Ln(WID) denotes beach width (m) and natural log-transformation of the beach width, respectively. Using equation 15, it is now possible to predict how transect slope will change in response to sea-level rise. Transect *e* has a beach width of 833 m. Because transect slope is a function of beach width (equation 15), a change in transect slope (%) due to decrease in beach width as a result of sea-level rise can be expressed as:

$$\text{Change in transect slope} = 100*(1 - (\text{transect slope (WID)} / \text{transect slope (833)}) \qquad \text{(equation 16)}$$

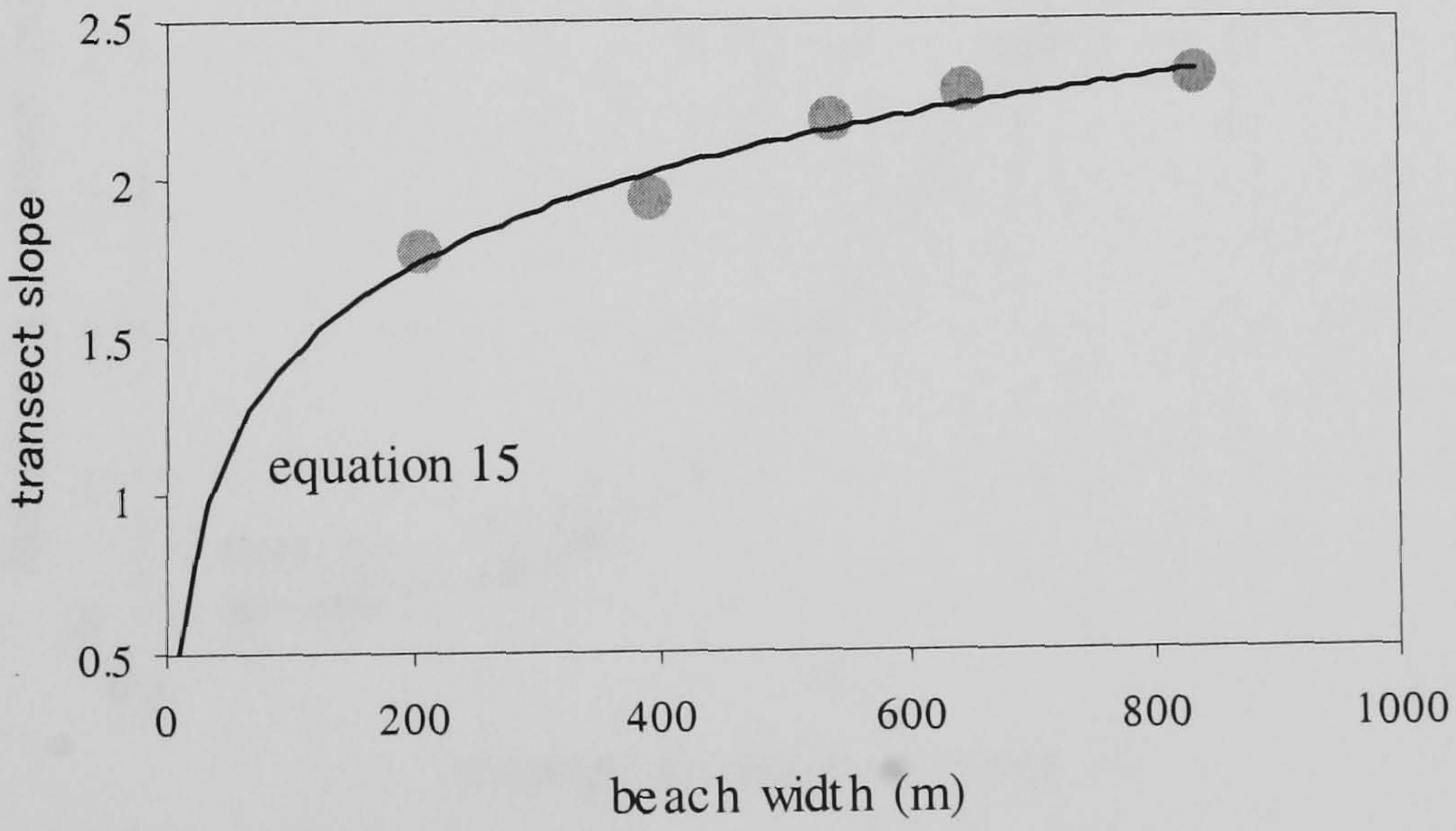


Fig. 3.17. Relations between transect slope and beach width (m) for five transects situated in front of Grimsby.

Further, the beach width of transect *e* (833 m) and the squeezed end (0 m) correspond to sea-level rises of 0 m and 4.55 m, respectively. Because the squeeze in beach width mimics the extent of sea-level rise, the increase in sea level can be expressed as a function of beach width (WID) as follows:

$$\text{increase in sea level} = 4.55 \times (1 - (\text{WID} / 833)) \qquad \text{(equation 17)}$$

Both equations 16 and 17 are a function of beach width (WID), and the relationships between increase in sea level (m) and change in transect slope (%) are thus mathematically deduced from equation 15-17 as shown in Fig 3.18, suggesting that transect slope becomes steeper by 0.40, 1.24 and 2.12 % when sea level rises by 0.1, 0.3 and 0.5 m, respectively. Station slope was also assumed to become steeper by the same rate as transect slope in response to sea-level rise (Fig. 3.5.a, b).

Finally, field observations at Grimsby suggested that median depth of beach does not change regardless the extent of coastal squeeze (Fig. 3.5.c). Therefore, median depth was assumed to remain unchanged in response to sea-level rise for the simulation.

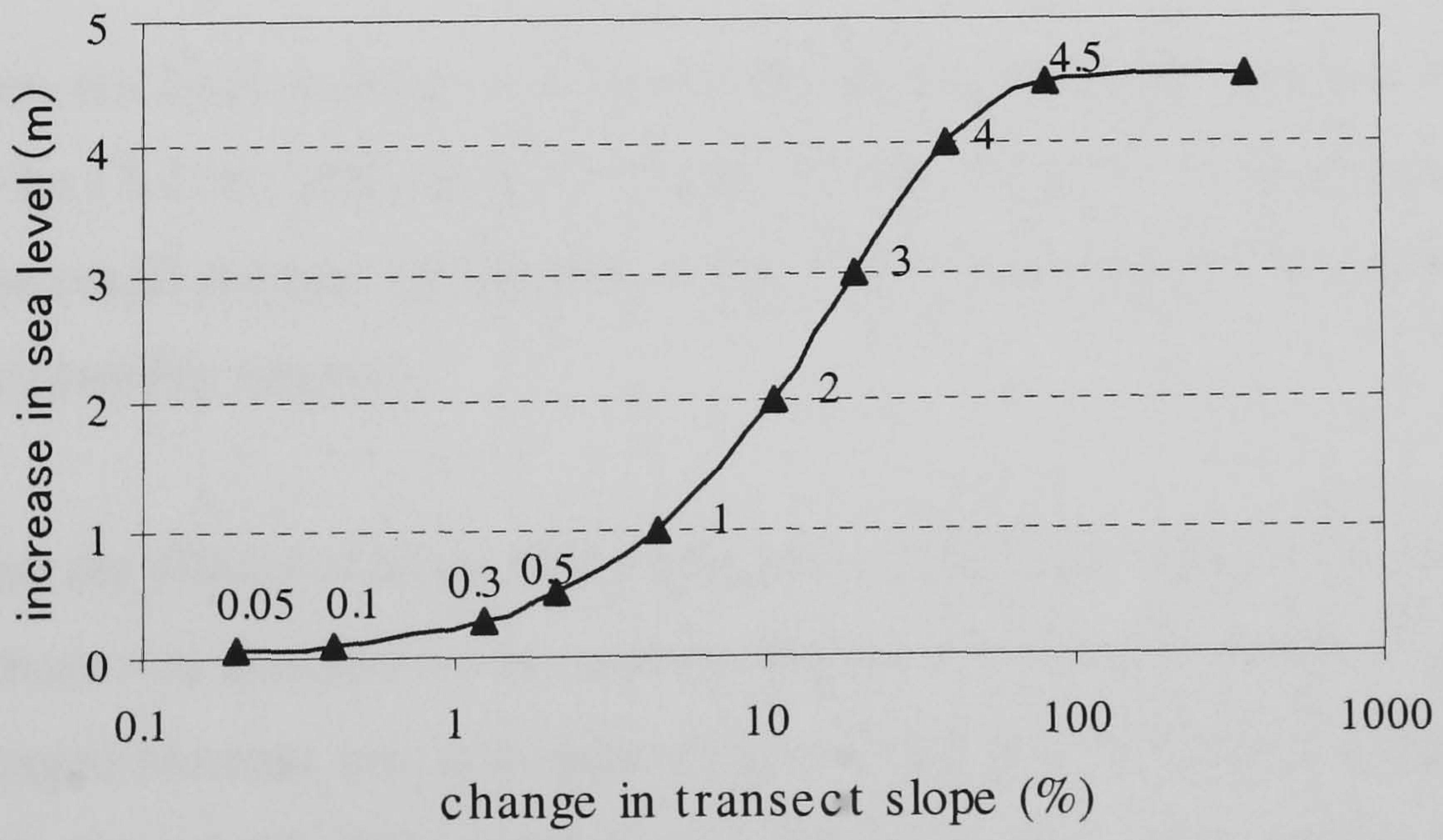


Fig. 3.18. Relations between increase in sea level and expected change in transect slope expressed as a percentage (%). Numbers show sea-level rise expressed in metres.

3.3.5 Simulation results

Using all the equations described above, model simulations were performed. Sea level was assumed to rise by 0.1, 0.3 and 0.5 m in future scenarios, and changes in environmental variables in response to sea-level rise were derived by changing the value of x km for salinity and sedimentary variables (equation 8-13) or by changing the degree of sea-level rise for morphological variables (equation 14-17). Substituting these new environmental values into equations 1-4 provides changes in macrobenthic biomasses at the transect scale in response to sea-level rise, and equations 5-7 allow predictions of changes in: (1) total area of intertidal flat, (2) system average biomass and (3) system total biomass. The effects of expected increase in sea levels, saline intrusion, expected change in slopes and sedimentary shift were combined or added against the baseline model to explore the consequences of different sea-level rise scenarios. Table 3.5 shows the simulation results under various combinations of environmental effects indicated as dark blocks in each column. All the percentage values were derived from the model simulations against the baseline (the model outputs from the survey 2003), and expected values for each category were calculated based on current observed values multiplied by the percentage change from the model simulations. Range of features can be noted from the simulation results:

- ❖ When sea level rose by 0.1, 0.3 and 0.5 m, the intertidal area was reduced by 269 ha (2.2 %), 807 ha (6.7 %) and 1134 ha (11.2 %), respectively, and this alone could account for the loss of 2.3 %, 6.9 % and 11.5 % respective of total macrobenthic biomass.

- ❖ When the effects of slope steepening were added, the extent of the loss in total biomass was doubled by the amount sea-level rise alone could cause. System averaged biomass was also reduced by 3.0, 8.9 and 14.4 % in response to sea-level rise of 01, 0.3 and 0.5 m, indicating that beach steepening could markedly reduce the quality of intertidal area.

Table 3.5. Simulation results showing expected change in intertidal area, change in system-average biomass and change in system total biomass under various combinations of effects of environmental changes from sea-level rise, slope steepening, saline intrusion and sedimentary shift. Element of environmental changes in effect is indicated by dark shaded block in each column. The first column indicates the current state under no environmental change.

Sea-level rise	0.1 m		▨	▨	▨	▨	▨	▨												
	0.3 m								▨	▨	▨	▨	▨	▨						
	0.5 m														▨	▨	▨	▨	▨	▨
Slope steepening	0.40 (%)			▨			▨	▨												
	1.24 (%)									▨			▨	▨						
	2.12 (%)															▨			▨	▨
Saline intrusion	0.4 km				▨		▨	▨												
	1.2 km										▨		▨	▨						
	2.0 km																▨		▨	▨
Sedimentary shift	0.4 km					▨		▨												
	1.2 km											▨		▨						
	2.0 km																	▨		▨
Intertidal area	ha	0	-269						-807						-1344					
	(%)	0	-2.2						-6.7						-11.2					
Average biomass	g AFDW m^{-2}	8.65	8.64	8.38	8.73	8.07	8.47	7.91	8.63	7.88	8.91	7.64	8.13	7.15	8.62	7.40	9.08	7.62	7.80	6.78
	(%)	0	**-0.1**	**-3.0**	**+1.0**	**-6.7**	**-2.0**	**-8.6**	**-0.2**	**-8.9**	**+3.0**	**-11.6**	**-5.9**	**-17.3**	**-0.3**	**-14.4**	**+5.0**	**-11.9**	**-9.8**	**-21.6**
Total biomass	t AFDW system^{-1}	1038	1014	983	1025	947	994	927	966	882	997	855	910	800	919	789	967	812	831	722
	(%)	0	**-2.3**	**-5.2**	**-1.3**	**-8.8**	**-4.2**	**-10.6**	**-6.9**	**-15.0**	**-3.9**	**-17.6**	**-12.3**	**-22.8**	**-11.5**	**-24.0**	**-6.8**	**-21.8**	**-19.9**	**-30.4**

- When the effect of saline intrusion was combined with sea-level rise, system averaged biomass increased by 1.0, 2.0 and 3.0 % under the three sea-level rise scenarios. The negative impacts of sea-level rise on total amount of macrobenthic biomass were almost halved despite the loss of intertidal area. This suggests that saline intrusion can create favourable conditions for macrobenthic invertebrates and could increase the quality of intertidal area, potentially counteracting the loss of intertidal habitats resulting from sea-level rise.

- Sedimentary shift had the strongest negative impacts on the total macrobenthic biomass when its effect was added to sea-level rise, reducing the system total biomass between 8.8 % and 21.8 %. This physical change also reduced the system averaged biomass by 6.7, 11.6 and 11.9 % under the three sea-level rise scenarios indicating that such environmental change could cause significant reduction of macrobenthic biomass in the Humber estuary.

- The strong effect of sedimentary shift was also reflected when all the environmental effects were combined together. For instance, when sea-level rose by 0.3 m, overall environmental effects reduced the total biomass by 22.8 %, but when the effect of sedimentary shift was removed, the negative impact was almost halved to 12.3 % .

- Because slope steepening and saline intrusion are likely to occur, but little is known about the sedimentary shift in response to sea-level rise, it may be assumed that the total macrobenthic biomass is likely to decrease by between 3.9 % and 12.3 % if sea levels increase by 0.3 m as expected over the next 50 years in the Humber estuary. However, sedimentary change could also double the negative impacts on macrobenthic biomass if such extent of sedimentary migration simulated in this study occurred.

3.4 Discussion

This study has revealed a number of implications of sea-level rise on estuarine macrobenthic ecology. At the local scale, the intertidal area in front of Grimsby was made steeper by coastal squeeze and the macrobenthic biomass was progressively reduced with the decrease in beach width. The incorporation of the data set from Chapter 2 with these analyses allowed more detailed models to be constructed to assess how macrobenthic biomass is likely to change in response to sea-level rise at the estuarine system scale. These analyses showed that intertidal area and macrobenthic biomass will be reduced by 6.7 % and 6.9 %, respectively, if sea level rises by 0.3 m. The analysis also indicated that saline intrusion could compensate for the loss of macrobenthic biomass. However, other possible environmental changes, such as slope steepening and sedimentary shift could have much larger negative impacts on the quality of intertidal habitats with a potential loss of macrobenthic biomass of up to 22.8 %, depending on the extent of expected environmental changes. Thus the impacts of sea-level rise on the intertidal macrobenthos will very much depend on precisely how the key environmental variables change in response to sea-level rise.

With respect to morphological change, the extent of coastal squeeze was represented by variation in beach width in Grimsby, and was logarithmically associated with transect slope (Fig. 3.17). This is consistent with the idea that when a beach is trapped between a fixed sea defence and a rising sea level, the beach will experience steepening. Recent evidence has also indicated that the majority of the overall coastline of England and Wales has steepened over the last century (Taylor et al., 2004), and it has been proposed that the removal of sediment from foreshore and the interruption of the landward transgression of the high water line by coastal defence walls are the likely causes of this phenomenon (Crooks, 2004; Taylor et al., 2004). My own field observations also indicate that mean station slope becomes steeper when the beach becomes narrower, reducing the area of local flatter surfaces within the beach. These two environmental variables were significantly related to change in two bivalve species, *M. balthica* and *C. edule* (Table 3.3), which together account for > 77 % of the total biomass in the Humber estuary. Coastal squeeze resulting from sea-level rise is therefore likely to produce steeper and homogenous beach face profiles, which will have a significant impact on total macrobenthic biomass in the Humber.

The response of sediment at Grimsby was somewhat counterintuitive. Sediment was expected to become coarser with a greater degree of coastal squeeze, but median particle size, silt content and organic matter content all showed similar trends, in which the sediment became finer and even muddier as the width of the beach decreased within the study area (Fig 3.3). When coastal squeeze occurs, the removal of soft sediment can be expected in intertidal habitats resulting from steepened beach slope and enhanced wave and tidal energy (Goss-Custard et al, 1990), but this does not appear to be the case in Grimsby and Cleethorpes intertidal area. This could be explained by the estuarine outer region in the Humber being different from open, or exposed, beaches with respect to sediment dynamics. The estuarine site may be more controlled by tidal movement and local siltation rates through such chemical reaction as flocculation rather than the wave action and coastal processes seen on exposed sandy beaches. The study site was also situated along a transitional zone between outer and lower regions of the estuarine system (Barr et al., 1990), where vigorous siltation processes could start to occur, suggesting that sediment can be influenced not only by the local hydrological regime, but also by other estuary-orientated factors.

At the whole estuary scale, the possibility of sedimentary shifts moving up estuary as described in this chapter still remains likely because intertidal habitats from Cleethorpes down to Mablethorpe on the south bank of the Humber, which now are characterised by coarse sandy beach, used to harbour extensive mudflats and saltmarshes before the 13th century (IECS, 1994). The saltmarshes and the fine grained mud were gradually eroded away, confined to upper shore and replaced by sandy beaches, possibly due to local geomorphological change induced by extreme climate events and subsequent exposure to North Sea coastal process (IECS, 1994). Such large-scale change in sediment characteristics from muddy to sandy habitats could have significant implication for estuarine intertidal ecology. Muddy sediment with smaller median particle sizes and higher organic matter content supports a higher macrobenthic abundance and biomass (McIntyre, 1970; Jones, 1988; McLusky, 1989; Meire et al., 1991; Dauer, 1993; Elliott et al. 1998; Beukema, 2002; Ysebaert & Herman, 2002), normally associated with tide-dominated dissipative beaches. However, as beaches become increasingly wave dominated, they tend towards sandy flats and become highly reflective, supporting a poorer infauna (Brown and McLachlan, 1990; Brazeiro, 2001; Rodil and Lastra, 2004). The sedimentary change experienced between Cleethorpes and Mablethorpe may well be ongoing and extended to the inner region of the Humber estuary in the face of rising sea levels and expected increase in extreme climatic events. Although little is known as to how sedimentary components are likely to change in response to sea-level rise at present, the consequences of the change could be catastrophic as indicated by the model simulations and there is an urgent need to improve understanding of such important physical processes.

Although field observations and model simulations have illustrated the negative impacts of slope steepening resulting from coastal squeeze, these results say little about the mechanisms responsible for the relationship between transect slope or median depth and macrobenthic biomass. Goss-Custard et al. (1990) have implied that narrower and steeper beaches become more concave through erosion so that the length of the exposure time at low tide would decrease. Such reduction of exposure time may affect the production of benthic microalgae which plays an important role in estuarine ecosystems (Heip et al, 1995; Herman et al., 1999). Further, beach morphological change may adversely affect macrobenthic larvae settlement onto narrower and steeper beaches, because of the reduction in low-dynamic areas essential for the success of early bivalve recruitment (Bouma et al., 2001). Other biological factors cannot be ruled out. For example, the study area in Grimsby was dominated by *Cerastoderma edule* (Table 3.1, 3.2), known to be affected by predation from crustacean species such as *Crangon crangon* (brown shrimps) and *Carcinus maenus* (shore crabs) (Richards et al., 2002; Huxham and Richards, 2003). *C. maenus* in particular is known to be capable of removing all juvenile *C. edule* from some areas, and their distribution is sediment specific with higher densities in muddy rather than sandy substrata (Richards et al., 1999). Shoals of such predators surge onto intertidal areas with tidal movements and if the squeezed beaches increase the accessibility of such predators by removing the heterogeneous nature of wider beaches, this could result in the removal of *C. edule* considerably from narrower and squeezed beaches.

Clearly, further work is needed to understand the underlying mechanisms of variation in macrobenthic biomass across the study site. However, it seems clear that sea-level rise will cause coastal squeeze leading to narrower and steeper intertidal flats, and if the biological trend found in this study can be generalised within similar geographical regions, the results indicate that sea-level rise could cause a significant reduction of total macrobenthic biomass when beach width is squeezed with adverse affects on the wider estuarine intertidal ecosystem. Further research into the likely impacts of sea-level rise on estuarine ecosystems should attempt to incorporate the relationships between sea-level rise and large-scale sedimentary processes so that the potential negative impacts of macrobenthic loss can be minimised through the coastal and estuarine management.

Chapter 4

Impacts of expected changes in macrobenthic biomass in response to sea-level rise on higher trophic levels in the Humber estuary, UK

4.1 Introduction

Estuaries are characterised by high abundance of microbes, plankton, benthic flora and fauna due to rich supply in nutrients and organic matter from both terrestrial and marine origins, placing them amongst the most biologically productive ecosystems on earth (McLusky, 1989; Kennish, 2002). Estuaries therefore typically harbour immense numbers of invertebrates in fringing intertidal habitats and in subtidal areas providing major food source for higher trophic levels, such as shorebirds, fish and epibenthic crustaceans. For many of the higher trophic levels, estuarine intertidal areas are utilised as their nursery grounds for one or more stages of their life cycle and this is why these habitats are of such high conservation value (Prater, 1981; Baird et al., 1985; Elliott et al., 1990; Elliott & Hemingway; 2002).

Historically, however, estuaries and fringing wetlands have been much modified by urbanization from an ever increasing human settlement, coastal development for harbours, shipping, transportation, wetland reclamation for industries and agriculture, and so on. For many estuaries, these impacts can also be seen in their artificial shapes outlined by the structure of robust sea defence walls protecting almost everywhere over entire longitudinal gradients. The Humber estuary is no exception. It has been estimated that the historical anthropogenic impacts have reclaimed more than 90 % of both the intertidal area and sediment accumulation capacity in the Holocene (Jickells et al., 2000). The remaining intertidal area is now confined and being squeezed between sea defence walls and ever rising sea levels, and there is widespread concern over the ecological impacts of the loss of intertidal habitats on higher trophic levels through a reduction in their food resources.

The aim of this chapter is to investigate the potential impacts of change in macrobenhtic biomass in response to sea-level rise on higher trophic levels, namely, shorebirds, fish and shrimps. First, I describe the current physical and ecological status of the Humber estuary. Then data for the avifauna based on the WeBS Low Tide Count Programme conducted in 2003-2004 (English Nature, 2005) are used to investigate how birds utilise the Humber estuary. 12 key migratory species whose feeding mode was benthivorous were identified in order to assess how such birds utilise estuarine intertidal habitats and how they are associated with their prey resources available at the beginning of their wintering periods, using the macrobenthic survey data in 2003 (Chapter 2). The resulting findings were then used to develop a method for predicting the number of the benthivorous birds in relation to varying amounts of intertidal area and their prey biomass resulting from sea-level rise. Feeding ecology of fish and shrimp was explored to consider the implications of change in the amount of their prey items as a consequence of sea-level rise in the Humber estuary. For this purpose, data on macrobenthic biomass in subtidal habitats obtained from Environment Agency's long-term monitoring data (Environmental Agency, 1980 – 1995) were used.

4.2 The Humber estuary

The Humber estuary is approximately 60 km long, running from the confluence of the rivers Trent and Ouse at Trent Falls to a midpoint between Spurn Head and

Tetney Haven on the east coast of Britain. The Humber is the largest macro-tidal estuary on the British North Sea coast with a mean tidal range of 5.7 m at Spurn Head (English Nature, 2003), and its catchment drains over one fifth of England (24,240 km^2) (Winn et al., 2003). Its width increases from less than 0.5 km at both the river Trent and Ouse to over 8 km at the mouth. The Humber is also the largest estuarine system in England in terms of mean flow (250 m^3 s^{-1}) (Jickells et al., 2000), and it is characterised as a well-mixed estuary with only a small vertical salinity gradient in contrast to a pronounced longitudinal gradient (Barr et al., 1990). Its flushing time varies from 20 days in winter time to 160 days in the summer depending on flowing conditions (Gameson, 1982). The Humber system has an area of approximately 290 km^2 with intertidal and subtidal areas comprising 120 km^2 and 168 km^2, respectively (English Nature, 2003; Winn et al., 2003). Subtidal sediment type is mostly sandy with some patches of gravel and glacial till (Jones, 1988), and these grade into finer particle sediments of silt and clay towards the intertidal areas that surround the main body of the estuary. However, in contrast to the sediment in sheltered muddy extensive intertidal areas in outer north bank, sediments are coarser and sandier towards the outer part on the south bank where the beaches are more exposed to high energy coastal conditions.

The Humber estuary has supported between 140,000 and 180,000 shorebirds and wildfowl in the past decade with a mean annual peak maxima of approximately 154,000 over the period between 1995/6 and 2000/1, providing safe feeding and roosting grounds (English Nature, 2003). These numbers place the Humber among the top 10 European estuaries in order of importance for birds (Winn et al., 2003) and one of the six most important wetland sites in the UK (English Nature, 2003). For example, migratory species such as shelduck (*Tadorna tadorna*), lapwing (*Vanellus vanellus*), knot (*Calidris canutus islandica*), dunlin (*Calidris alpina alpina*), ringed plover (*Charadrius hiaticula*), sanderling (*Calidris alba*), grey plover (*Pluvialis squatarola*) and redshank (*Tringa totanus totanus*) have been recorded in internationally important numbers (Edwards & Winn, 2006), and other key species such as teal (*Anas crecca*), dark-billed brent goose (*Branta bernicula bernicula*), mallard (*Anas platyrhynchos*), wigeon (*Anas* *penelope*), avocet (*Recurvirostra avosetta*), golden plover (*Pluvialis apricaria*), bar-tailed godwit (*Limosa lapponica*), black-tailed godwit (*Limosa limosa icelandica*), oystercatcher (*Haematopus ostralegus*), turnstone (*Arenaria interpres*) and curlew (*Numenius arquata*) can attain nationally or internationally important numbers at times (IECS, 1987; English Nature 2003). The historical records also show a substantial increase of the shorebird population in the Humber estuary during the last two decades, which is primarily attributable to the expansion of the golden plover population, while other species numbers have been essentially unchanged over the same period (English Nature, 2003).

For fish species in the Humber, the estuary has long been recognised as an important nursery area for such commercially important species as sole (*Solea solea*), plaice (*Pleuronectes platessa*) and cod (*Gadus morhua*) (CEFAS, 2002), along with many other demersal species such as flounder (*Platichthys flesus*), dab (*Limanda limanda*) and whiting (*Merlangius merlangus*) (Elliott & Hemingway, 2002). Benthic macrobenthos are known to be the important prey items for many bird and fish species in estuarine environments, and the understanding of how they utilise their prey would be the key to assess the impacts of sea-level rise on the estuarine ecosystems.

4.3 The Humber estuary and birds

Assessing the potential impact of sea-level rise on shorebird populations requires a knowledge of how these shore birds are distributed in relation to intertidal areas and how they utilise the food resources in such areas. The amount of food available in autumn in particularly is an important indicator of the quality of estuaries for overwintering shorebirds (West et al., 2005). In present chapter, I investigate how the spatial pattern of macrobenthic biomass in September (Chapter 2) is associated with the distribution of migratory birds in the Humber estuary in 2003.

4.3.1 The WeBS Bird Count 2003-2004 in the Humber estuary

The data for the avifauna of Humber estuary were derived from the Humber Estuary Low Tide Count Programme 2003-2004 as a part of the WeBS Bird Count Scheme (English Nature, 2005). This programme was organised by the Institute of Estuarine & Coastal Studies (IECS) at the University of Hull on behalf of English Nature (Humber to Pennines Team). In this programme, monthly co-ordinated bird counting on 60 pre-established subdivisions (sectors) of the intertidal habitat was conducted in the period two hours either side of low tide. The detailed map of the subdivisions (sectors) used in the programme in the Humber can be seen in Appendix 5. Missing counts were interpolated for each bird species either spatially or temporally. For spatial interpolation, a missing count was estimated by averaging two density values of adjacent subdivisions multiplied by the area of the site missing. For temporal interpolation, a missing count was estimated by averaging values before and after the missing month. The census results provided in this study deal with the counts of 720 observations with 60 missing observations interpolated, over the period of 12 months from September 2003 to August 2004.

A total of 77 bird species were recorded on the Humber estuary during the census period. 20 key species were extracted based on their numbers in relation to national or international importance criteria, and these were further divided into three groups based on their feeding types (Table 4.1). Shelduck and waders, except for the golden plover, were considered benthivores since they predominantly feed on the intertidal flats. Teal and mallard were categorised as omnivores since they tend to consume plant and other organic matter as well as small macrofauna along the low water edge (Ysebaert et al., 2000). Golden plover and lapwing also fall into this category for their feeding predominantly on terrestrial invertebrates and other organic matter. The remaining species, namely dark-billed brent goose, pink-footed goose and wigeon, were considered herbivores. The mean monthly bird count during the census period was approximately 97,000 for the Humber estuary (Table 4.1).

Table 4.1. Key bird species recorded on the Humber estuary during the census period between September 2003 and August 2004, showing their feeding type, mean monthly count and the number of species within a group.

Key species	Feeding type	mean monthly count	(%)	number of species	(%)
Avocet Bar-tailed godwit Black-tailed godwit Curlew Dunlin Grey plover Knot Oystercatcher Redshank Ringed plover Sanderling Shelduck Turnstone	Benthivorous	57855	59.6	13	16.9
Golden plover Lapwing Mallard Teal	Omnivorous	31186	32.1	4	5.2
Brent goose Pink-footed goose Wigeon	Herbivorous	3341	3.4	3	3.9
Others		4691	4.8	57	74.0
TOTAL		97073	100.0	77	100.0

Figure 4.1 shows the numbers of key birds and other waterbirds counted per month, showing a distinctive seasonal pattern of lowest counts in spring between April and June, and highest in winter between November and January. The annual peak maximum was estimated at approximately 195,000 in November 2003 and the lowest count estimated at 12,000 in June 2004.

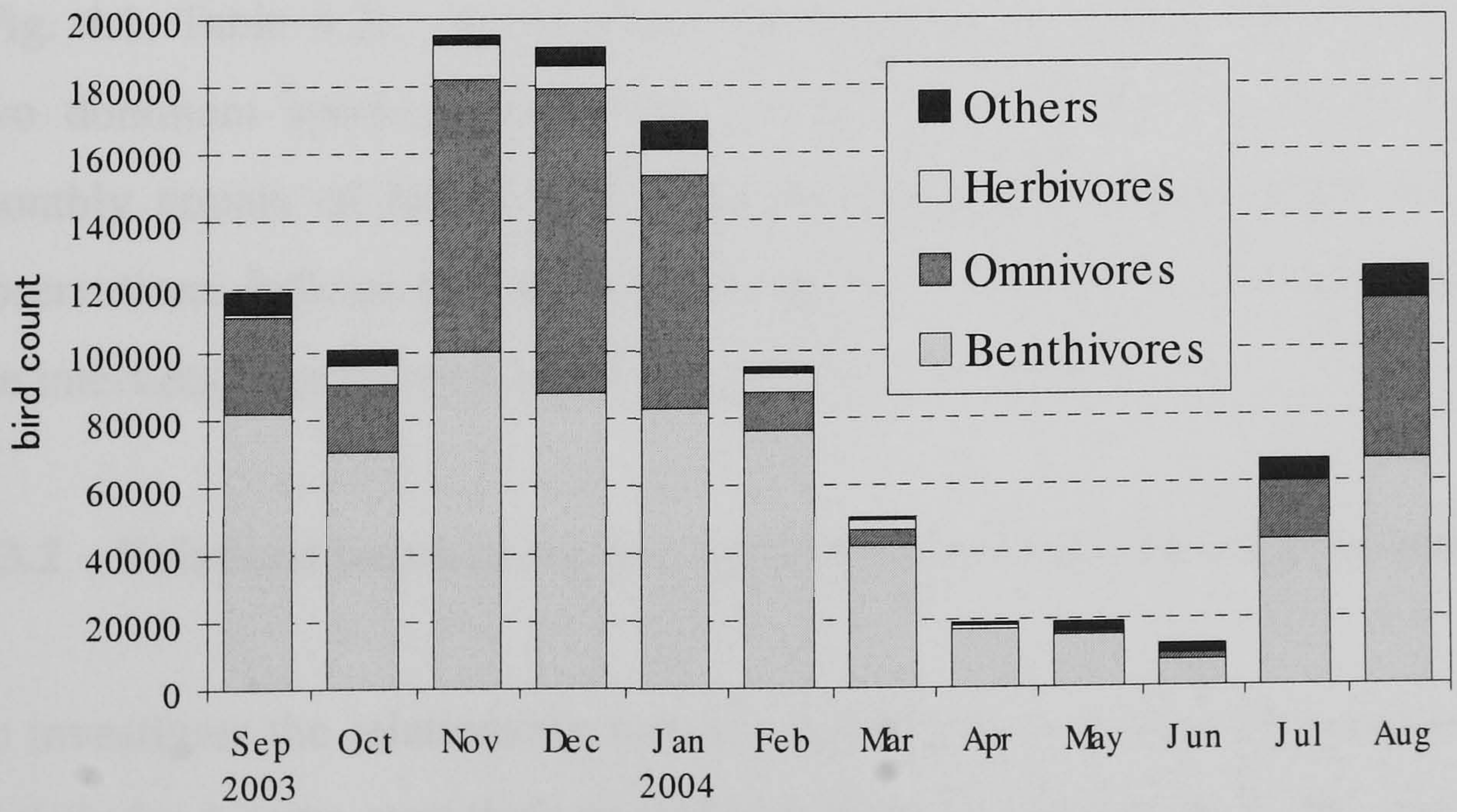


Fig 4.1. Monthly counts of benthivorous, omnivorous, herbivorous and other bird species observed during the census period between September 2003 and August 2004 in the Humber estuary.

Table 4.2. Mean monthly count of the 13 key benthivorous species over the wintering period between September 2003 and February 2004. The species are listed in decreasing rank order of the count.

Key benthivorous species	mean monthly count over wintering period	Percentage (%)	Cumulative percentage (%)
Knot	38468	46.0	46.0
Dunlin	20345	24.3	70.3
Redshank	7260	8.7	79.0
Shelduck	5134	6.1	85.1
Curlew	3131	3.7	88.8
Oystercatcher	2950	3.5	92.4
Bar-tailed godwit	2509	3.0	95.4
Grey plover	1398	1.7	97.0
Black-tailed godwit	900	1.1	98.1
Ringed plover	616	0.7	98.9
Turnstone	520	0.6	99.5
Sanderling	387	0.5	99.9
Avocet	50	0.1	100.0
TOTAL	83667	100.0	

Benthivorous and omnivorous birds dominated the total monthly counts throughout the year (Fig. 4.1), accounting for 59.6 % and 32.1 % of the mean monthly counts, respectively (Table 4.1). Benthivorous, omnivorous and herbivorous birds represented 16.9 %, 5.2 % and 3.9 % of the total number of waterfowl species observed over the census period, respectively (Table 4.1). Although there was a marked increase in omnivore counts from October to November, the number of benthivorous birds remained similar from September to February with a mean monthly count of approximately 84,000, declining in March (Fig. 4.1, Table 4.2). Amongst benthivorous species, knot and dunlin were the two dominant species accounting for 46.0 % and 24.3 % of the total mean monthly counts of benthivores over the wintering period (Table 4.2). These observations indicate that significant number of benthivorous birds depend upon the intertidal over the period from September to February.

4.3.2 Relations between the key benthivorous birds and macrobenthos

To investigate the relationship between benthivorous bird density and availability of their food items over their wintering period, a subset of bird data was extracted from the Humber Estuary Low Tide Count Programme 2003-2004 (HELTCP).

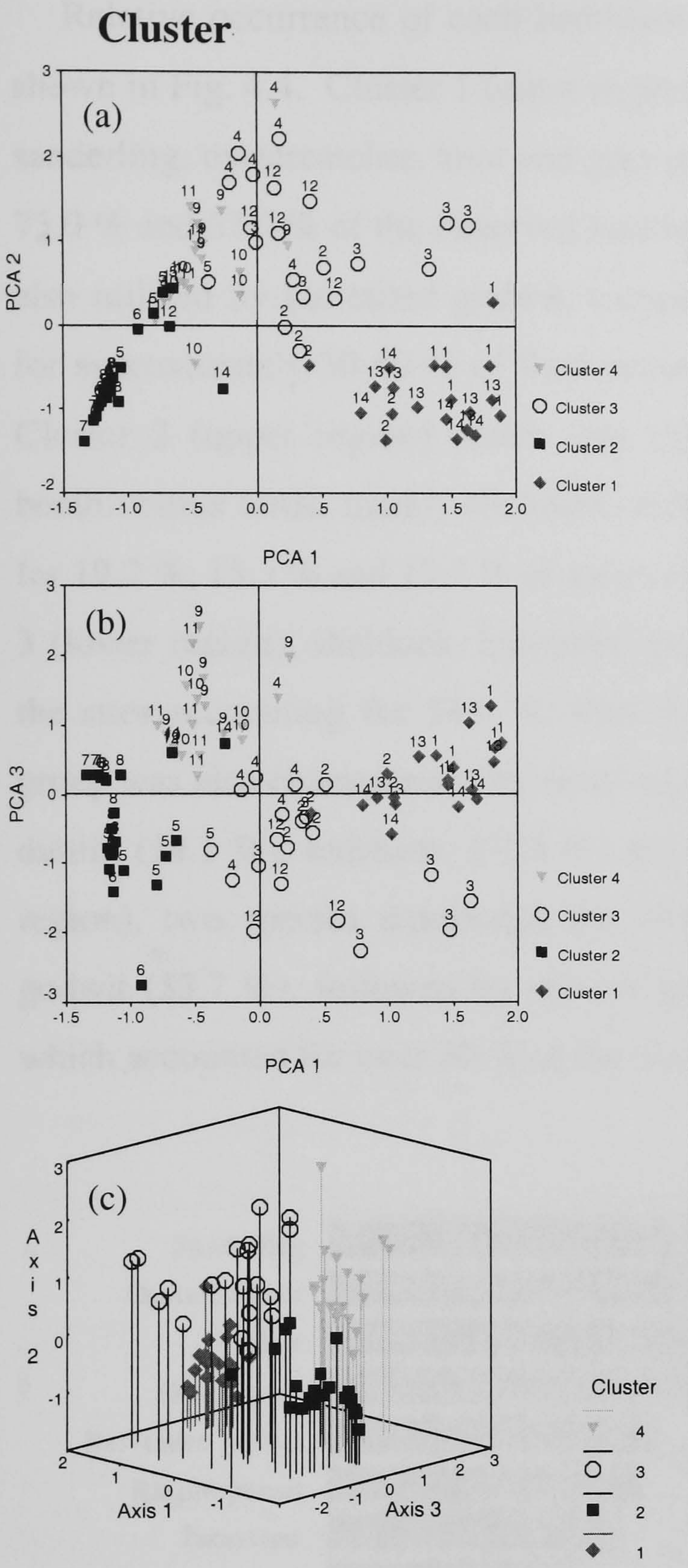
The locations of 14 transects specified in Chapter 2 for macrobenthic sampling were superimposed on the 60-subdivided intertidal map specified in the HELTCP (Appendix 5) so that benthivorous bird densities could be estimated around each macrobenthic transect. When a macrobenthic sampling transect was located within a single HELTCP classified sector, the bird density data of the single sector was used for subsequent analysis. When a macrobenthic transect was located close to a boundary of two or more HELTCP classified sectors, or located across more than two sectors, a mean bird density proportionate to the areas of sectors in contact with the transect was computed. Density values were thus obtained for the key 12 benthivorous birds over the wintering period from September 2003 to February 2004 (6 months) at 14 sites (transects). Sites were then grouped based on bird species composition by classification and ordination. K-means cluster analysis was chosen for the classification because this procedure can identify relatively discrete groups based on simple Euclidian distance, as well as the mean abundance of all species within each cluster. For this analysis, the desired number of clusters, K, must be specified in advance. Cluster centres are defined as the average value on all clustering variables of each cluster's members, and initial cluster centres are chosen in a first pass of the data, then each additional iteration groups observations based on nearest Euclidean distance to the mean of the cluster. Cluster centres therefore change at each pass and the process continues until cluster means do not shift more than a given cut-off value or the iteration limit is reached. Although there needs to be a certain amount of trial and error in selecting the number of clusters, this analysis has the advantage of producing good discrete groups that are usually easy to interpret in ecological context. For the ordination, Principal Component Analysis (PCA) was used. The K-means cluster analysis and the Principal Component Analysis (PCA) were used within the SPSS computer package for Windows. Data on the abundance of birds were double square root transformed prior to analysis to reduce the dominance of the most abundant species such as knot and dunlin (Table 4.2). Avocet was not recorded at the 14 sites and thus could not be included in this analysis.

K-means cluster analysis identified 4 distinctive clusters based on their species composition per month (September – February) within the Humber estuary, and this was confirmed by the ordination when the outputs from the analysis were superimposed on the PCA diagram (Fig. 4.2). In the PCA, the first 3 axes explained 75.1 % of the observed variance with axes 1, 2 and 3 accounting for 42.2 %, 21.2 % and 11.7 %, respectively. In order to explore the relationships between benthivorous bird community structure and environmental factors, the sites were grouped into four regions according to their location along the longitudinal gradient (upper, middle, lower and outer regions of the estuary) defined by Barr et al. (1990). When these new groupings of sites were superimposed on the ordination plot, an almost identical pattern was produced (Fig. 4.3). Clusters 1, 2, 3 and 4 correspond to outer, upper, lower and middle region of the Humber estuary, respectively, indicating that the structure of the wintering benthivorous bird assemblages are strongly influenced by the environmental characteristics which affect macrofaunal biomass (Chapter 2). This was confirmed by Spearman's rank correlation between scores of sites on PCA axis 1 and salinity, beach width and transect slope, in particular, ($p < 0.0001$ for all). PCA axis 2 was significantly positively correlated with sedimentary gradients such as median particle size, organic matter content or silt content ($p < 0.0001$ for all) (Table 4.3). PCA axis 3 was not correlated with any environmental variables significantly, indicating that bird assemblages were not defined by environmental indicators alone, or that 11.7 % of the variance is due to random error.

Table 4.3. Spearman rank correlation coefficients between three PCA axes and physical measurements from three major environmental components specified in Chapter 2.

PCA	LONGITUDINAL		SEDIMENTARY			MORPHOLOGICAL					
	SAL	EXP	MD	ORG	SIL	WID	T-SLO	S-SLO	DEP	M-DEP	DEP-I
Axis 1	(0.89)***	(0.54)***	n.s.	(-0.27)*	n.s.	(0.89)***	(0.89)***	(0.74)***	(-0.45)***	(-0.36)***	n.s.
Axis 2	n.s.	n.s.	(0.67)***	(0.61)***	(0.67)***	n.s.	n.s.	n.s.	n.s.	n.s.	(0.35)**
Axis 3	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.	n.s.

When correlations are significant Spearman rank coefficients are given between brackets. BIO = mean total biomass. ***P<0.001; **P<0.01; *P<0.05; n.s. = not significant. Environmental variables: SAL: salinity; EXP: exposure; MD: median particle size; ORG: organic matter content; SIL: silt content; WID: beach width; T-SLO: transect slope; S-SLO; station slope; DEP; percentage tidal depth; M-DEP: median tidal depth; DEP-I: depth index, for each sampling station.



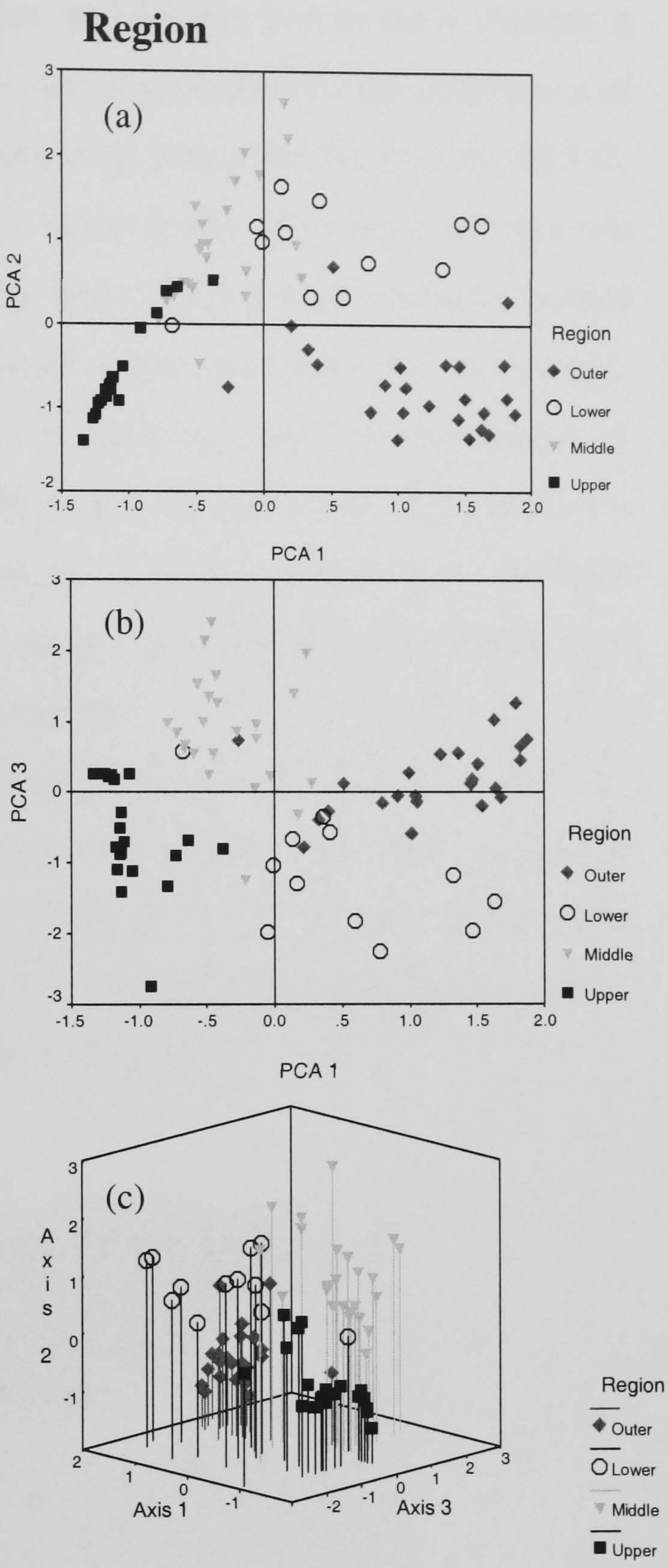


Fig 4.2. PCA diagrams of 14 sites over wintering period between September 2003 and February 2004 (6 months), classified by 4 bird clusters derived from K-means cluster analysis. (a) PCA 1 vs PCA 2, (b) PCA 1 vs PCA 3 and (c) 3-D plot between PCA 1, 2 and 3. See Chapter 2 for the location of sites (transects) 1-14.

Fig 4.3. PCA diagrams of 14 sites over wintering period between September 2003 and February 2004 (6 months), classified by 4 regions (upper, middle, lower and outer) along the longitudinal gradient in the Humber estuary. (a) PCA 1 vs PCA 2, (b) PCA 1 vs PCA 3 and (c) 3-D plot between PCA 1, 2 and 3.

Relative occurrence of each benthivorous bird species across the 4 clusters is shown in Fig. 4.4. Cluster 1 (outer region) was characterised by the dominance of sanderling, oystercatcher, knot and grey plover that accounted for 85.9 %, 80.9 %, 73.0 % and 63.0 % of the observed numbers, respectively. This group of sites was also utilised by bar-tailed godwit, turnstone and ringed plover which accounted for approximately 30-40 % of their occurrence across the study site. In contrast, Cluster 2 (upper region) which was characterised by very low occurrence of benthivorous birds, mainly shelduck, redshank and curlew which accounted only for 19.2 %, 15.3 % and 12.2 % of their observed numbers, respectively. In Cluster 3 (lower region), shelduck, bar-tailed godwit and black-tailed godwit dominated the sites accounting for 54.0 %, 49.6 % and 45.0 % of their occurrence. This group was also characterised by relatively high abundance of grey plover (33.4 %), dunlin (34.1 %), redshank (31.5 %) and curlew (35.0 %). In Cluster 4 (middle region), two species dominated the sites, turnstone (57.2 %) and black-tailed godwit (53.7 %), followed by species such as ringed plover, dunlin and curlew, which accounted for over 30 % of the observed numbers.

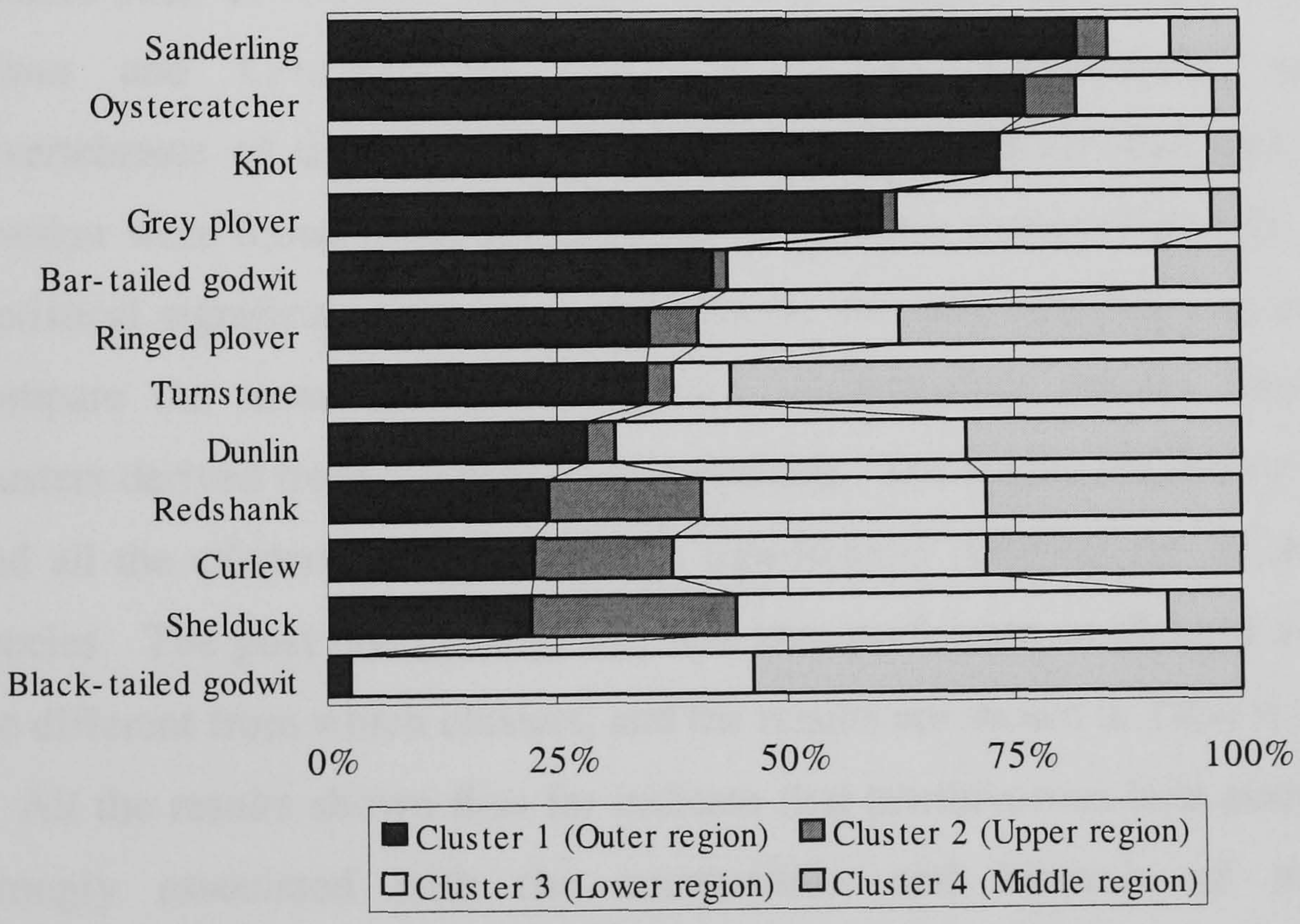


Fig 4.4. Proportion of occurrence for 12 key benthivorous bird species across the 4 clusters derived from the K-means cluster analysis over their wintering period between September 2003 and February 2004.

Figure 4.4 shows that species such as sanderling oystercatcher, knot, grey plover, bar-tailed godwit, shelduck, turnstone and black-tailed godwit had strong associations with clusters, while four other species, ringed plover, dunlin, redshank and curlew were more widely distributed throughout the Humber estuary.

Given the close association between birds and environmental parameters and between macrobenthos and environmental parameters, the relationship between birds and macrobenthos were explored. Figure 4.5 shows the mean biomasses of 15 macrobenthic species which accounted for more than 99.9 % of total benthic biomass in the Humber, across the 4 clusters. These 15 characteristic species illustrated clear differences between the 4 bird groups. For example, Cluster 2 (upper region) is characterised by a high abundance of oligochaetes, *Paranais litoralis*, and Cluster 4 (middle region) is dominated by such species as *Corophium volutator* with relatively high biomass of *Nereis diversicolor*. In Cluster 3, (lower region), *N. diversicolor*, as well as other species such as *Tubificoides benedeni*, *Hydrobia ulvae* and *Macoma balthica* and *Eteone longa* were characteristic. *H. ulvae* was particularly strongly associated with this cluster. While *M. balthica* remained dominant in Cluster 1 (outer region), other important species such as *Nephtys hombergii*, *Nephtys cirrosa*, *Tharyx spp.*, *Scrobicularia plana* and *Cerastoderma edule* were also characteristic macrobenthic invertebrates of this cluster. In addition, both Ampharetidae and *Spiophanes bombyx* were found to be only associated with this cluster (Fig. 4.5). To test the statistical significance for these differences, one-way ANOVA was conducted to compare the mean abundance of each macrobenthic species across the bird clusters derived from k-means cluster analysis. The results are shown in Table 4.4 and all the clusters were statistically significantly different for all macrobenthic species. The post hoc (Tukey) test was also performed to identify which means are different from which clusters, and the results are shown in Table 4.5.

All the results shown thus far indicate that benthivorous bird assemblages are strongly associated with the composition and biomass of macrobenthic invertebrates found in the intertidal flats in the Humber estuary.

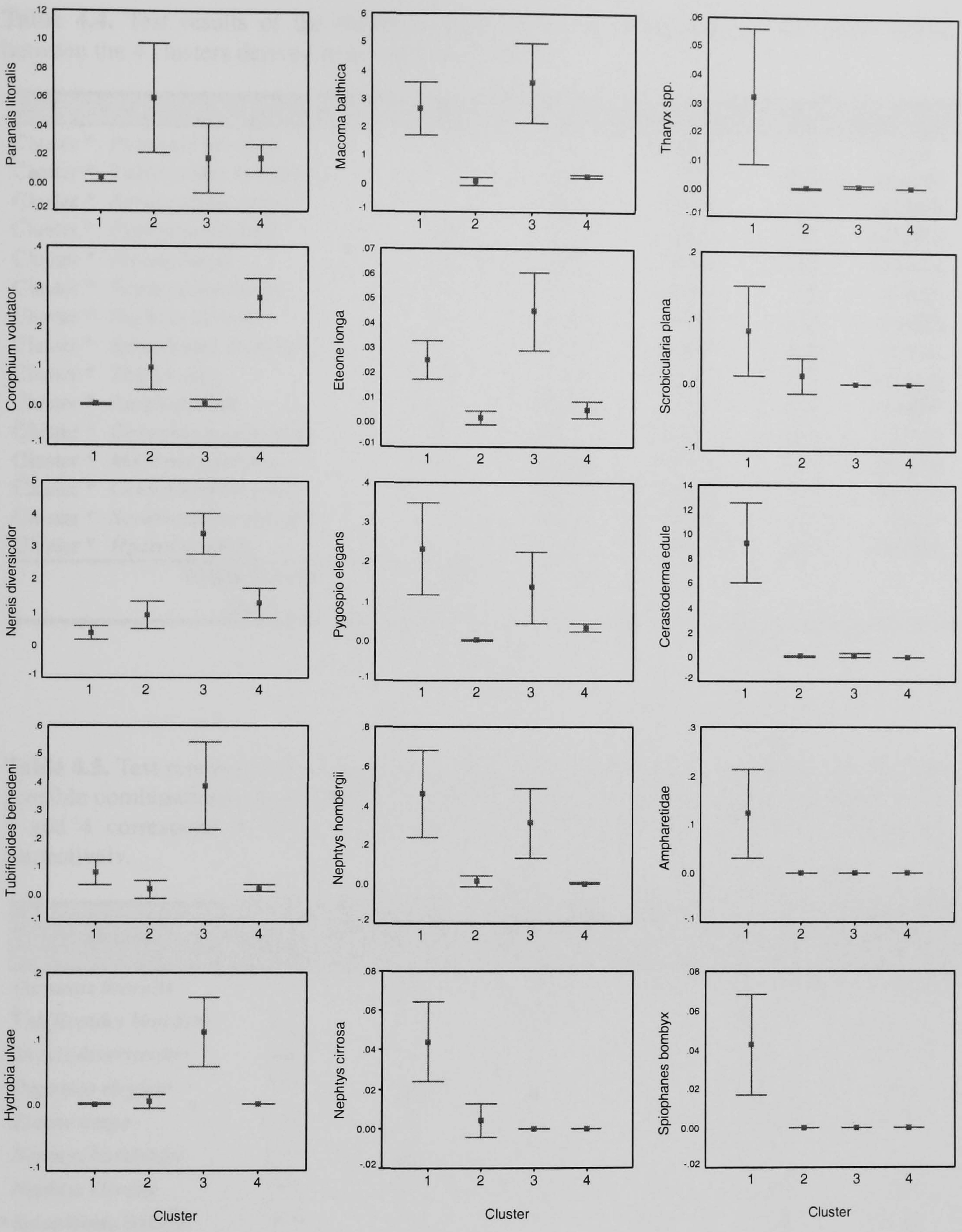


Fig 4.5. Mean values of 15 macrobenthic species biomass (g AFDW m^{-2}) for the 4 bird assemblages (clusters) derived from K-means cluster analysis. Vertical bars show 95 % confident interval. Clusters 1, 2, 3 and 4 correspond to outer, upper, lower and middle region of the Humber estuary, respectively.

Table 4.4. Test results of the statistical significance for differences of the mean values between the 4 clusters derived from one-way ANOVA.

	df	SS	MS	F	P value
Cluster * *Paranais litoralis*	3	0.04	0.014	4.1	<0.01
Cluster * *Tubificoides benedeni*	3	1.91	0.635	20.7	<0.0001
Cluster * *Nereis diversicolor*	3	106.51	35.503	36.1	<0.0001
Cluster * *Pygospio elegans*	3	0.70	0.233	9.9	<0.0001
Cluster * *Eteone longa*	3	0.03	0.008	23.8	<0.0001
Cluster * *Nephtys hombergii*	3	3.16	1.055	11.8	<0.0001
Cluster * *Nephtys cirrosa*	3	0.03	0.009	16.2	<0.0001
Cluster * *Spiophanes bombyx*	3	0.03	0.009	12.6	<0.0001
Cluster * *Tharyx spp.*	3	0.02	0.005	8.6	<0.0001
Cluster * Ampharetidae	3	0.23	0.076	8.7	<0.0001
Cluster * *Corophium volutator*	3	0.91	0.304	39.0	<0.0001
Cluster * *Macoma balthica*	3	188.32	62.774	19.9	<0.0001
Cluster * *Cerastoderma edule*	3	1304.10	434.701	37.7	<0.0001
Cluster * *Scrobicularia plana*	3	0.09	0.030	4.9	<0.01
Cluster * *Hydrobia ulvae*	3	0.18	0.059	18.4	<0.0001
Within Groups	80				
Total	83				

Table 4.5. Test results of the statistical significance for differences of the mean values for all possible combinations of cluster pairs derived from the post hoc (Tukey) test. Clusters 1, 2, 3 and 4 correspond to outer, upper, lower and middle region of the Humber estuary, respectively.

species	combination of clusters					
	1-2	1-3	1-4	2-3	2-4	3-4
Paranais litoralis	**	n.s.	n.s.	n.s.	n.s.	n.s.
Tubificoides benedeni	n.s.	***	n.s.	***	n.s.	***
Nereis diversicolor	n.s.	***	*	***	n.s.	***
Pygospio elegans	***	n.s.	***	*	n.s.	n.s.
Eteone longa	***	**	**	***	n.s.	***
Nephtys hombergii	***	n.s.	***	**	n.s.	*
Nephtys cirrosa	***	***	***	n.s.	n.s.	n.s.
Spiophanes bombyx	***	***	***	n.s.	n.s.	n.s.
Tharyx spp.	***	***	***	n.s.	n.s.	n.s.
Ampharetidae	***	***	***	n.s.	n.s.	n.s.
Corophium volutator	**	n.s.	***	**	***	***
Macoma balthica	***	n.s.	***	***	n.s.	***
Cerastoderma edule	***	***	***	n.s.	n.s.	n.s.
Scrobicularia plana	*	**	**	n.s.	n.s.	n.s.
Hydrobia ulvae	n.s.	***	n.s.	***	n.s.	***

Key: ***P<0.001; **P<0.01; *P<0.05; n.s. = not significant.

4.3.3 Relationship between bird species richness, density and macrobenthic biomass

This section explores relationship between the amount of food present at the beginning of the wintering period and benthivorous bird abundance in the Humber estuary. The mean monthly number of key benthivorous species (species richness) observed between September and February for each site was positively exponentially related to the mean macrobenthic biomass (ANOVA, $F_{1,12} = 28.6$, $p < 0.0001$, $R^2 = 0.70$) (Fig. 4.6). Density of the total key benthivorous birds ha^{-1} showed a similar log-linear relationship, indicating that density of the birds were significantly positively associated with the amount of food available at the beginning of their wintering period (ANOVA, $F_{1,12} = 21.4$, $p < 0.001$, $R^2 = 0.64$) (Fig. 4.7).

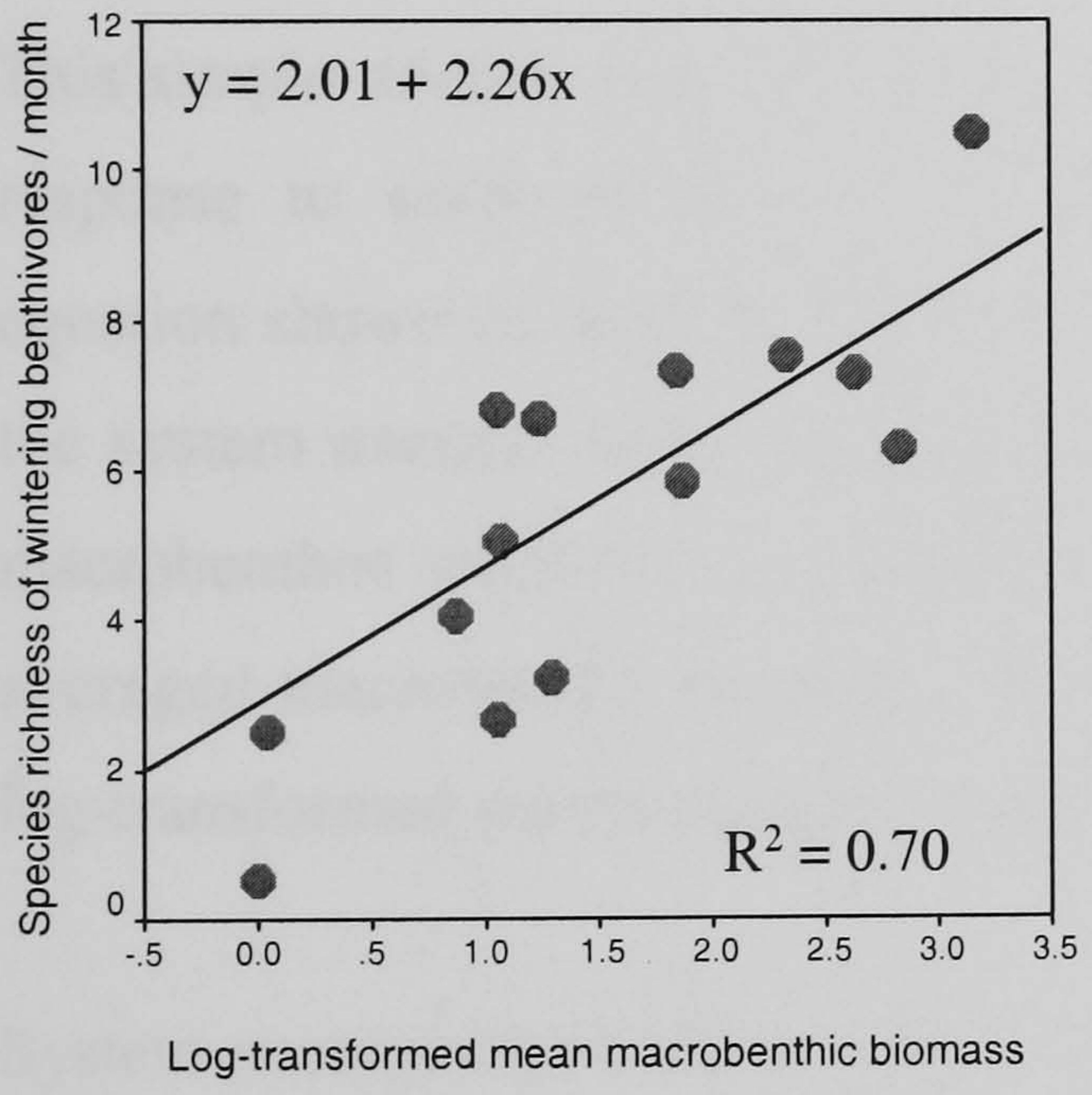


Fig 4.6. Relationship between mean species richness (monthly mean species number of the key benthivorous birds observed between September 2003 and February 2004) and natural log-transformed mean macrobenthic biomass (g AFDW m^{-2}) for 14 sites across the Humber estuary with an equation and R^2 value.

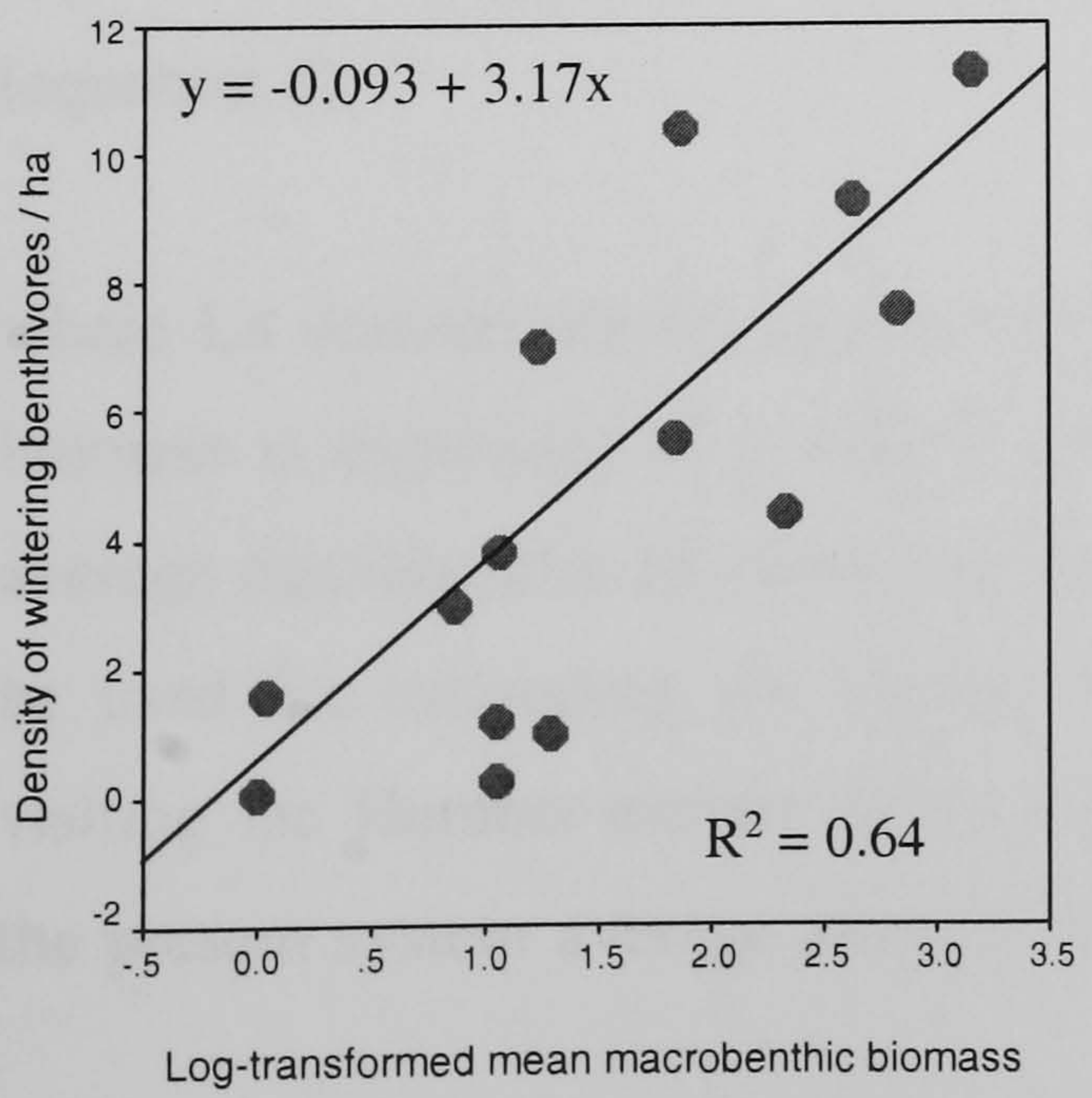


Fig 4.7. Relationship between monthly mean density of the total key benthivorous birds ha^{-1} observed between September 2003 and February 2004, and natural log-transformed mean macrobenthic biomass (g AFDW m^{-2}) for 14 sites across the Humber estuary with an equation and R^2 value.

These relationships suggest that intertidal flats with a higher macrobenthic biomass can support not only more individuals of shorebirds, but also more bird species, depending on the quality (mean macrobenthic biomass) and quantity (area) of the intertidal habitat. This further suggests that macrobenthic biomass could be used as an indicator for the benthivorous bird population dynamics in the context of sea-level rise.

At the estuarine system scale, the total number of benthivorous birds visiting the Humber estuary during the wintering period can be obtained simply by multiplying the area of intertidal flats present within the system by system-averaged density of benthivorous bird as follows:

$$\text{Total number of benthivores} = \text{Intertidal area} \times \text{System-averaged bird density}$$

(equation 1)

This simple relation could be used to predict change in the benthivorous birds in response to sea-level rise in the Humber estuary because the mathematical equation shown in Figure 4.7 provides a means for predicting quantitatively how the system average density of the birds will change in relation to the amount of macrobenthos available at the beginning of autumn expressed as the system-averaged macrobenthic biomass. The linear regression between bird density and log-transformed macrobenthic biomass from Figure 4.7 gives the equation:

$$\text{System-averaged bird density (ha}^{-1}) = -0.093 + 3.17 \times \text{Ln}\,(1 + \text{system-averaged macrobenthic biomass})$$

(equation 2)

where Ln denotes natural log-transformation and system-averaged macrobenthic biomass is expressed in g AFDW m^{-2}. It follows that by inserting the system average macrobenthic biomass into equation 2, the resulting bird density can then be used for estimating the change in the total number of benthivorous birds visiting the Humber estuary in the wintering period (equation 1). For example, the present system average macrobenthic biomass of 8.65 g AFDW m^{-2} (Chapter

2) gives an expected system average density of 7.09 ha^{-1} from equation 2. Because the current area of intertidal flats in the Humber is approximately 1,200 ha, the current total bird number is predicted to be (equation 1):

$7.09 \times 1200 = 85{,}178$

which is close to the observed mean number of benthivorous birds of 83,667 during the wintering period from September 2003 to February 2004 in the Humber (Table 4.2). Starting from this predicted figure, it is now possible to see how this current bird numbers will change in response to changes in intertidal area (equation 1) and macrobenthic biomass (equation 2), both of which respond to changes in sea-level rise. Based on equation 1 and 2, Figure 4.8 shows how bird numbers would be expected to decrease in response to:

— (X) changes in intertidal quality (the percentage loss of the system-averaged macrobenthic biomass) while intertidal area is held constant;

— (Y) changes in intertidal quantity (the area of intertidal flat in the Humber) while intertidal quality is held constant;

— (Z) changes in both quality and quantity.

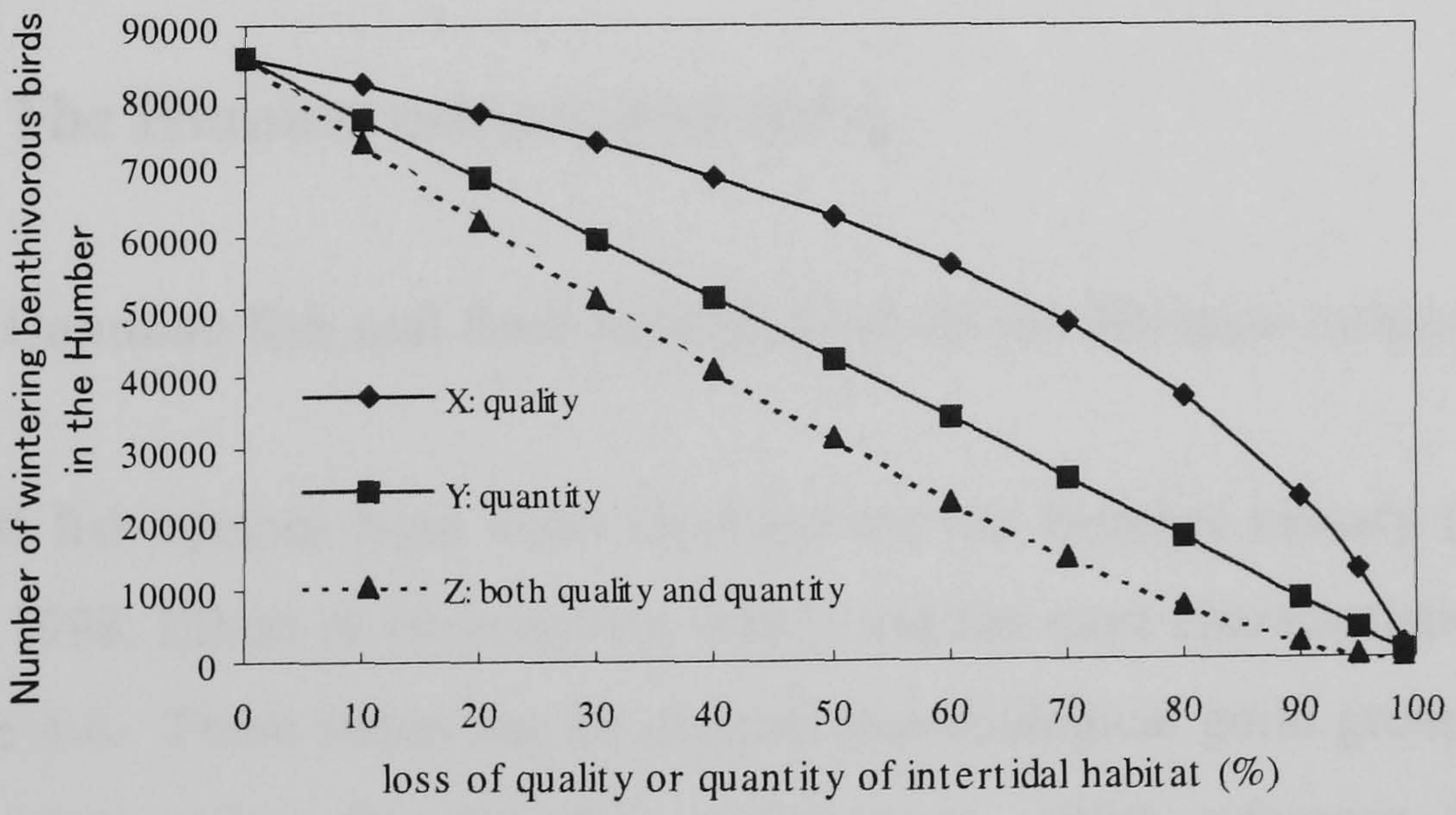


Fig 4.8. Relationships between total number of key benthivorous birds over their wintering period in the Humber and loss of intertidal quality (X), loss of intertidal quantity (Y) and loss of both intertidal quality and quantity (Z), expressed in percentage (%). Quality denotes system mean macrobenthic biomass on intertidal habitat, and quantity denotes the area of the intertidal habitat in the system.

If sea-level rise does not affect the area of intertidal flats, the total number of benthivorous birds is only a function of macrobenthic biomass (quality) as shown in equation 2 (and 1), and this relationship is shown as the curve X in Fig 4.8, indicating that the change would be gradual until much of the macrobenthic biomass has been lost. If sea-level rise affects the area of the intertidal flats (quantity) but the quality of the flats is maintained at the current level, the relationship then becomes linear (equation 1) as shown in the line Y in Fig 4.8, indicating that each area contributes to the number of wintering birds. However, as expected from the simulation results in Chapter 3, the sea-level rise is likely to reduce both the quantity and quality of the intertidal flats reflected in the dotted curve Z in Fig 4.8, showing that bird numbers will be reduced more than expected from simple area loss. Sea level is expected to rise by 0.3 m over the next 50 years (MAFF, 1999), and this is likely to claim 6.7 % of the intertidal area in the Humber estuary (Chapter 3). Furthermore, expected environmental changes such as saline intrusion, slope steepening and sedimentary shift could reduce up to 17.3 % of the quality of intertidal habitats (Chapter 3, Table 3.5). Combining such simulations with the simple equations derived in this chapter, predicts that loss of intertidal area alone would lead to the reduction of the wintering benthivorous bird population by 5707 (7.2 %), and adding the effects of environmental changes could reduce the number of birds by up to 11691, a reduction of up to 15.9 %.

4.4 The Humber estuary and fishes

4.4.1 Common fish and their feeding mode in the Humber estuary

Over 70 fish species have been recorded for the Humber estuary (Marshall & Elliott, 1998, Elliott & Hemingway, 2002) and the most characteristic are shown in Table 4.6. These fishes can be divided into ecological guild groups according to how they utilise the estuarine environment. With reference to Elliott & Hemingway (2002), common goby (*Pomatoschistus micropus*), pogge (*Agonus cataphractus*) and flounder (*Platichthys flesus*), sea snail (*Liparis liparis*) are categorised as typical estuarine resident species, which live in the main body of the estuary throughout the year. Marine species which enter the estuarine habitats for nursery grounds are recognised as marine juvenile migrants, and such flatfish as sole (*Solea solea*), plaice (*Pleuronectes platessa*), dab (*Limanda limanda*), turbot (*Scophthalmus maximus*), brill (*Scophthalmus rhombus*), and such gadidae species as cod (*Gadus morhua*), whiting (*Merlangius merlangus*), pollock (*Pollachius pollachius*), fall into this category. Sprat (*Sprattus sprattus*) uses the estuary as a seasonal migrant, and such fish species as lemon sole (*Microstomus kitt*), haddock (*Melanogrammus aeglefinus*), dragonet (*Callionymus lyra*) and lesser weever (*Echiichthys vipera*) are categorised as marine adventitious species.

Table 4.6. List of characteristic fish species found in the Humber estuary showing their ecological guild type, habitat use for nursery and feeding ground and feeding type (after Elliott & Hemingway, 2002).

common name	species	Ecological guild	nursery habitat	feeding habitat	Feeding type
sole	*Solea solea*	MJ	SS	SS	Db
flounder	*Platichthys flesus*	MJ/ER	TF, SS	TF, SM, IS, SS	Db
plaice	*Pleuronectes platessa*	MJ	SS	SM, IS, SS	Db
pogge	*Agonus cataphractus*	ER	SS	SS	Db
common goby	*Pomatoschistus micropus*	ER	TF, SM, IS, SS	TF, SM, IS, SS	Db
sand goby	*Pomatoschistus minutus*	ER	TF, SM, IS, SS	TF, SM, IS, SS	Db
dab	*Limanda limanda*	MJ	(IS) SS	(IS) SS	Db
sea snail	*Liparis liparis*	ER	SS	SS	Db
dragonet	*Callionymus lyra*	MA	SS	SS	Db
lesser weever	*Echiichthys vipera*	MA	SS	SS	Db
lemon sole	*Microstomus kitt*	MA		SS	Db
whiting	*Merlangius merlangus*	MJ	SS	SM, IS, SS	Dpisc
cod	*Gadus morhua*	MJ	SS	SM, IS, SS	Dpisc
eel	*Anguilla anguilla*	CA		TF, SS	Dpisc
turbot	*Scophthalmus maximus*	MJ	IS	IS (SS)	Dpisc
brill	*Scophthalmus rhombus*	MJ	IS	IS (SS)	Dpisc
pollack	*Pollachius pollachius*	MJ	IS	IS	Dpisc
coalfish	*Pollachius virens*	MA	IS	IS	Dpisc
smelt	*Osmerus eperlanus*	CA/FW	TF, SS	TF, SS	Ppisci
sprat	*Sprattus sprattus*	MS	SS	SM, IS, SS	Pplank
herring	*Clupea harengus*	MJ	SS	SM, IS, SS	Pplank
sandeel	*Ammodytes tobianus*	ER/MA	SS	IS, SS	Pplank
stickleback	*Gasterosteus aculeatus*	CA	TF, SM, SS	TF, SM, SS	Pplank
atlantic salmon	*Salmo salar*	CA			Dpisc
sea trout	*Salmo trutta*	CA			Dpisc
brown shrimp	*Crangon crangon*				
common shore crub	*Carcinus maenas*				
pink shrimp	*Pandalus montagui*				

Key: Ecological guild: MJ, marine juvenile migrant; ER, estuarine resident; MA, marine adventitious; CA, diadromous migrant; FW, freshwater adventitious; MS, marine seasonal migrant. Habitat: SS, subtidal soft substratum; TF, tidal fresh water; SM, saltmarsh; IS, intertidal soft substratum. Feeding type: Db, demersal benthivore; Dpisc, demersal piscivore; Ppisc, pelagic piscivore; Pplank, pelagic planktivore (feeding type adapted from Buchan, 1997)

Finally, fish which use the estuary as a route from the sea to fresh water to breed, or vice versa, are diadromous migrant species, and fish species such as eel (*Anguilla anguilla*), stickleback (*Gasterosteus aculeatus*), Atlantic salmon (*Salmo salar*) and sea trout (*Salmo trutta*) fall into this category in the Humber estuary. *S. solea*, *G. morhua* and *A. anguilla* are the three of the most important fish species taken in good quantity commercially, and some other fish species such as *P. flesus*, *P. platessa*, *M. kitt*, *M. merlangus*, *S. sprattus*, *C. harengus* and *A. tobianus* are also contributors of the commercial catches in the Humber (IECS, 1987; Elliott & Hemingway, 2002; CEFAS, 2002). Additionally, brown shrimp (*Crangon crangon*) and pink shrimp (*Pandalus montagui*) are two macrocrustacean species reported to have a high commercial interest in the Humber estuary (Elliott & Hemingway, 2002). Although diadromous species, *S. salar* and *S. trutta* may have little or no requirement for the estuary on their passage to other habitats (Elliott et al., 1998), most of the characteristic species found in the Humber estuary utilise the estuarine inter- and sub-tidal habitat as important nursery and feeding grounds (Table. 4.6).

Over three-quarters of the characteristic fish species in the Humber are demersal feeding on invertebrates living in or just above the substratum, or on other fish species including their own species (Table 4.6). Because of the high proportion of demersal feeders, dependence of the fish assemblages on the standing crop of benthic macrofauna may be high in the Humber estuary (Table 4.7).

Table 4.7. The proportion of prey groups in the diet of 10 fish species and 2 crustaceans in the Forth estuary, UK. (after Buchan, 1997)

common name	species	intertidal macrobenthos	subtidal macrobenthos	caridean shrimp
flounder	*Platichthys flesus*	0.844	0.003	0.040
plaice	*Pleuronectes platessa*	0.996	0.001	0.000
pogge	*Agonus cataphractus*	0.100	0.274	0.539
common goby	*Pomatoschistus micropus*	0.212	0.582	0.065
dab	*Limanda limanda*	0.000	0.825	0.020
sea snail	*Liparis liparis*	0.000	0.392	0.493
whiting	*Merlangius merlangus*	0.000	0.101	0.565
cod	*Gadus morhua*	0.000	0.170	0.645
sprat	*Sprattus sprattus*	0.000	0.000	0.000
herring	*Clupea harengus*	0.000	0.000	0.000
caridean shrimp	e.g. *Crangon crangon*	0.220	0.604	0.035
common shore crub	*Carcinus maenas*	0.221	0.663	0.109

For example, almost the entire diets of flatfish, *P. flesus* and *P. platessa* comprise intertidal macrobenthos, whereas *P. micropus* and *L. limanda* show a high proportion of subtidal benthic infauna in their diets in the Forth estuary, UK (Buchan, 1997). Other demersal species such as *A. cataphractus*, *L. liparis*, *G. morhua* and *M. merlangus* are less dependent on the benthic macrobenthos, yet they consume a high proportion of the caridean shrimp (e.g. *C. crangon*), which themselves feed on the benthic infauna in the Forth estuary (Table 4.7), indicating the importance of macrobenthos in supporting the estuarine food web. However, fish and shrimp species in estuaries could depend on the consumption of benthic macrofauna both intertidally and subtidally, unlike the benthivorous shorebirds described in the previous section, and sea-level rise would therefore have little implication for estuarine fish assemblages if both inter- and sub-tidal bottom environments are equally rich in the standing crops of macrobenthic biomass. This is because sea-level rise will simply replace the intertidal area with the subtidal area within the same estuarine boundary. However, the quantity of macrobenthic prey items available in subtidal area tends to be smaller than that in intertidal area in estuarine environments (Elliott & Kingston, 1987; McLusky, 1987; Elliott & Taylor, 1989; McLusky et al., 1992; Ysebaert et al., 2003). This quantitative difference in macrobenthic biomass between inter- and sub-tidal environments is a key issue when considering the potential impacts of sea-level rise on populations of fish and shrimps in estuarine ecosystems.

4.4.2 Subtidal macrobenthic biomass in the Humber

Data for the subtidal macrobenthos in the Humber estuary were derived from the Environment Agency's long term monitoring survey data between 1980 and 1995 (Environment Agency, 1980 – 1995). In this survey, 72 sampling stations were arranged in a grid pattern with a spacing of approximately 1-2 km in order to cover entire environmental gradients longitudinally and laterally in the Humber subtidal area (Environment Agency, 1980 – 1995). The data provided species composition and their abundance at each sampling station, and indicated that the subtidal benthic macrofaunal populations in the Humber estuary had not experienced major changes over the survey period. The most recent data available for the species and abundance in 1995 were used to estimate the present subtidal benthic macrofaunal biomass.

From the macrobenthic data set obtained in Chapter 2, the average individual weight of macrobenthic species was calculated by dividing the total weight of each species by its total abundance within the system. Where these intertidal species were present subtidally, their biomasses were obtained by multiplying the average individual biomass values of intertidal species by the subtidal abundance per m^2. For those subtidal species whose average individual weights were unknown, the values of morphologically most similar species found intertidally were used for their biomass estimation. This process enabled the system-averaged subtidal macrobenthic biomass per m^2 to be calculated. 14 subtidal lines were established across the water channel so that each of the line transects corresponded to each of the 14 intertidal transects established in Chapter 2 at the same longitudinal location along the Humber estuary. Mean subtidal macrobenthic biomasses for the 14 subtidal transects were then obtained by averaging the biomass values of subtidal sampling stations on, or closest to, each transect, and they were compared with intertidal counterparts derived in Chapter 2.

Figure 4.9 shows the mean macrobenthic biomass values for the 14 subtidal transects plotted against distance from the mouth. The highest value was 1.2 g AFDW m^{-2} around the lower region and the lowest biomass was recorded as 0.0054 g AFDW m^{-2} around the upper region of the estuary.

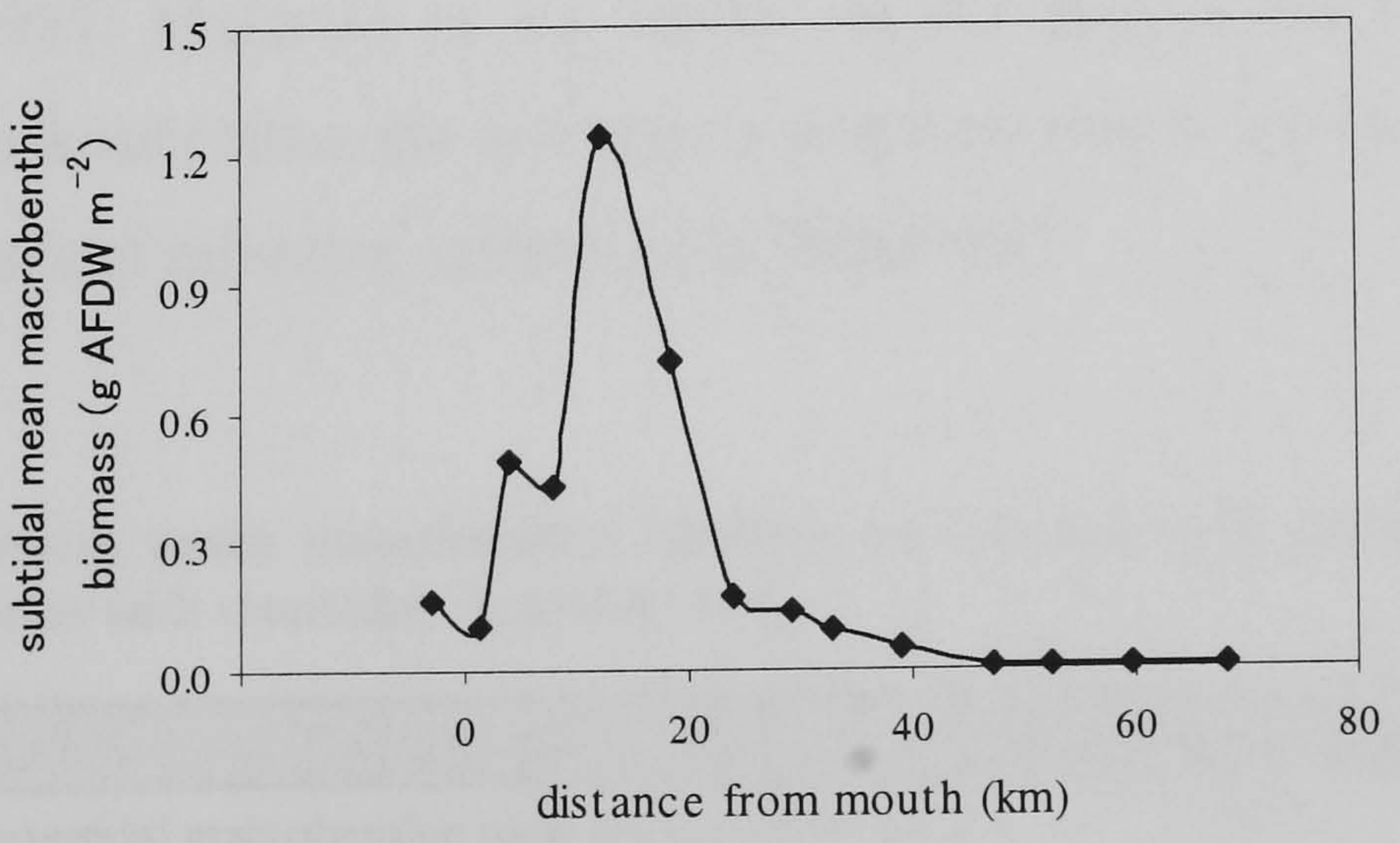


Fig 4.9. Relations between mean macrobenthic biomass (g AFDW m^{-2}) for 14 subtidal transects and distance from mouth (km) in the Humber estuary.

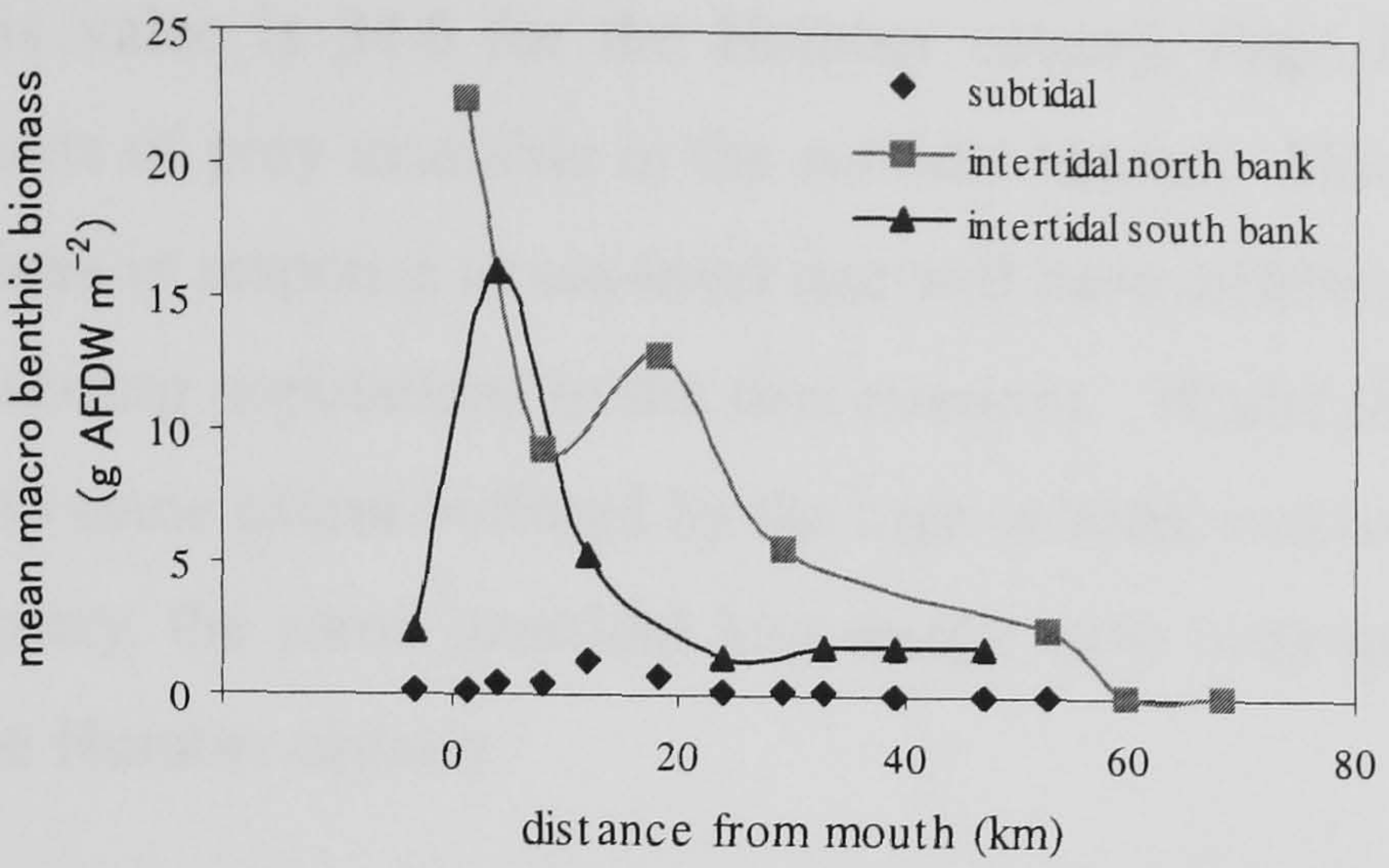


Fig 4.10. Relations between mean macrobenthic biomass (g AFDW m^{-2}) and distance from mouth (km) for 14 subtidal and 14 intertidal transects (7 on the north and 7 on the south bank) in the Humber estuary.

The subtidal mean biomass showed a distinctive spatial pattern with exponential increase from upper towards lower region, but with a sheer drop towards the outer region of the estuary. Although the intertidal transects showed similar trends of macrobenthic biomass distribution, with the values generally increasing towards the mouth with a steep decline at the marine end, the variation of subtidal biomass became too small to be compared with that of intertidal biomass when they were plotted together on the same biomass scale (Fig 4.10), illustrating the magnitude of difference in macrobenthic biomass between the two habitats. Taking account of the proportion of area represented by each transect mean biomass value, system average mean biomass were derived for total intertidal and subtidal area in the Humber, respectively (Table 4.8), and the same calculations were made to obtain the equivalent values for the Forth estuary using data from the literature (Elliott & Kingston, 1987; McLusky et al., 1992). In the case of the Forth estuary, the quantity of macrobenthos per unit area in intertidal area is 3.4 times as high as that in subtidal area at estuarine system scale (Table 4.8).

Table 4.8. System mean macrobenthic biomass on intertidal and subtidal habitats in two different estuaries with intertidal : subtidal ratio.

	Forth estuary	Humber estuary
System mean intertidal macrobenthic biomass (g AFDW m^{-2})	10.49	8.65
System mean sub-tidal macrobenthic biomass (g AFDW m^{-2})	3.05	0.25
intertidal / subtidal ratio	3.4	34.6

However, this value is 34.6 for the Humber estuary, there being substantially smaller amounts of prey available in the subtidal habitat. This suggests that loss of intertidal area in response to sea-level rise will have different consequences for the fish and shrimp populations in the two estuaries. Whilst the loss of intertidal area may be to some extent buffered by the high subtidal macrobenthic biomass in the Forth estuary, the same intertidal loss would have disproportionately greater impacts in the Humber estuary.

4.4.3 Preferred prey items for fish predators in the Humber

With respect to the diet composition of the 8 dominant fish species in the Humber estuary, mysids and amphipods were important, with exceptions of large plaice (*P. platessa*) which have a high consumption of polychaetes and molluscs, and small plaice (*P. platessa*) which take more molluscs, and sole (*S. solea*) which feeds predominantly on polychaetes (Table 4.9, after Marshall & Elliott, 1996). Mysids are tidal migrants, and most of amphipods can be observed both intertidally and subtidally in the Humber. However, standing crops of amphipods are much higher in the intertidal areas than in the subtidal areas. For instance, amphipods have been reported to be the dominant food source for the Humber food web (Marshall & Elliott, 1996), and gammarids (*Gammarus*, *Haustorius*, *Pontocrates* and *Bathyporeia* spp.) and *Corophium volutator* are the commonly found amphipod species in the study area.

Table 4.9. The percentage occurrence of the main prey groups in the diet of 8 common fish species in the Humber estuary. S, M and L denote small, medium and large size, respectively (after Marshall & Elliott, 1996).

	Cod	Flounder			Goby	Plaice		Pogge		Sole	Sprat	Whiting		
		S	M	L		S	L	S	L			S	M	L
Plant	7	10	19	0	0	0	0	0	0	3	0	3	0	4
Polychaeta	29	20	37	25	1	18	84	0	0	84	0	0	4	13
Copepoda	7	0	0	0	0	0	0	0	0	2	45	8	4	0
Mysidae	36	15	7	0	38	18	0	100	18	13	64	74	76	74
Amphipod	50	85	70	75	66	27	42	33	9	39	0	39	55	38
Decapod crustaceans	36	20	0	0	0	0	11	0	64	22	0	0	5	47
Brachyura crustaceans	21	10	0	25	1	0	11	0	18	19	0	0	0	13
Mollusca	7	55	37	0	1	55	84	0	0	14	0	5	10	8
Pisces	14	5	0	25	0	0	0	0	0	6	0	3	5	15
Other	14	40	26	0	2	9	47	17	36	20	0	0	5	13
No. fish	14	20	27	4	140	11	19	6	11	64	11	38	93	53
Size range (cm)	8 - 26	< 15	15 - 25	> 25	2 - 8	< 8	> 8	<7	> 7	7 - 29	4 - 11	< 9	9 - 14	> 14

Table 4.10. System mean abundance of *C. volutator* and *Gammarids* spp on intertidal and subtidal habitats in the Humber estuary and their intertidal : subtidal ratios.

	Corophium volutator	*Gammarids spp.*
System mean intertidal abundance (ind. m^{-2})	470.8	41.6
System mean sub-tidal abundance (ind. m^{-2})	70.0	3.2
intertidal / subtidal ratio	6.7	12.8

The former group has a greater preference for sandy sediments and the latter prefers muddy sediments (Barnes, 1994). They both have a wide tolerance to salinity and they can occur throughout the system intertidally and subtidally, provided their preferred sediment can be found. For this reason, gammarids are confined to the outer sandy region of the south bank in the intertidal areas, but can be found throughout the subtidal area where sandy substrata are present, whereas the longitudinal distribution of *C. volutator* over inter- and sub-tidal areas seems to be similar being common in the upper and middle regions of the Humber estuary. Despite the wider distribution of gammarids in the subtidal areas, however, the system average abundance of gammarids and *C. volutator* in the subtidal area are 13.0 and 6.7 times smaller than in the intertidal areas, respectively (Table 4.10), confirming the importance of intertidal feeding areas for demersal fish in the Humber estuary. McLusky et al. (1992) has reported that 66 % of estuarine fish populations would have been lost in the Forth estuary as a result of around 50 % loss of intertidal areas due to land-claims, assuming that the abundance of fish was controlled purely by benthic infaunal production. Given the magnitude of difference in the standing crop of macrobenthos between inter- and subtidal areas in the Humber estuary, the loss of intertidal area in response to sea-level rise may lead to impacts similar to those which might have been caused in the Forth estuary due to the land-claims.

4.5 General discussion

This chapter has considered the potential impacts of sea-level rise on bird and fish species in response to changes in their food availability in the Humber estuary.

The results for 12 waders and shelduck in the Humber estuary strongly support the view that the amount and the distribution of the prey available at the beginning of wintering period are the major determinants of the density and the distribution of benthivorous shorebirds that utilise the estuarine intertidal areas. Statistically, local mean biomass of the intertidal macrobenthos was significantly and positively correlated with the local number of benthivorous bird species as well as their mean density (Fig. 4.6, 4.7), which suggests that the population of such intertidal dependent birds could be described as a function of the quantity and quality of intertidal habitats at the estuarine system scale. This approach allows an exploration of how prey availability responds to sea-level rise, and could therefore provide a useful framework for maintaining the conservation status of estuarine ecosystems. However, the present analysis could not look at changes in bird populations at the species level due to a limitation relating to dietary differences between benthivorous bird species. Such an approach is beginning to be addressed at the species level (Stillman et al., 2003; West et al., 2004; Stillman et al., 2005). For example, West et al. (2005) suggest that a minimum food : bird ratio above which 'mortality' does not vary and below which 'mortality' increases steadily, be incorporated into management in order to maintain the ability of the site to support shorebirds at their present fitness level, and hence the current carrying capacity of the site. In this approach, 'mortality' is assumed to be influenced by factors such as prey availability, foraging efficiency, interference competition, disturbance and climatic conditions (West et al., 2005). However, the accuracy of such prediction may still depend on the ability for absolute separation and quantification of prey items for each bird species as well as precise prediction of change in their particular prey items in the context of climate change and sea-level rise issues. From this point of view, the simple general bird model suggested in this study may be useful for providing a basis of coastal management at the estuarine system scale when the threat of sea-level rise should be dealt as a matter of urgency.

With respect to shorebird population dynamics at larger geographical scales, it can be argued that a decline in bird numbers within one local estuary does not necessarily indicate a decline of total bird populations at larger scales because of potential changes in intertidal quality or quantity in other neighbouring estuaries (Gill et al., 2001; Norris, et al., 2004; West et al., 2005). Furthermore, at the passage and fly-way scale, changes in climate and sea level within Arctic and sub-Arctic regions where many migratory waterbirds breed, are likely to have as large an impact on populations as changes in the wintering feeding grounds (Watkinson et al., 2004; Boere & Taylor, 2004). From this point of view, it is clear that more surveys and monitoring should be conducted for longer periods of time over varying geographical scales in order to construct ecologically realistic models of population change for migratory shorebirds at an appropriate spatial scale which can be applied to sea-level rise issues. However, it would still be necessary to fully understand the underlying mechanism of population dynamics in relation to their prey availability at the estuarine system scale in order to formulate management plans for safeguarding shorebird populations at larger scales.

With respect to fish assemblages, the Humber is not significantly different from other estuaries in terms of the species composition (Elliott & Hemingway, 2002), and the value of the estuary does not lie particularly in the occurrence of rare species. However, some species are of commercial importance in the estuary or within the adjacent coastal areas (CEFAS, 2002), while other fish species are important in the diet of commercially exploited species, most of which are, in turn, either directly or indirectly dependent upon the high production of benthic infauna in the intertidal areas in the Humber estuary. Estuaries have also long been recognised as important nursery and overwintering grounds for large numbers of fish species (Wolff, 1987; McLusky, 1989; Elliott et al; 1990; Elliott & Hemingway, 2002), and the Humber is no exception, providing safe nursery areas intertidally and subtidally for most of characteristic fish species (Table 4.6). However, the importance of intertidal areas for their feeding is clearly illustrated by the difference in mean macrobenthic biomass between subtidal and intertidal areas in the Humber estuary in present study. The substantially low macrobenthic biomasses recorded in the Humber may be attributable to the high average fresh water flow (250 m^3 s^{-1}) (Jickells et al., 2000), the strong tidal movement (peak velocities between 2.0 – 3.0 m s^{-1}) (IECS, 1987) and extremely high turbidity (suspended particulate matter up to 20 g l^{-1} at the turbidity maximum) (Uncles et al., 1998), in contrast to other estuaries, which may be comparable to respective 38 m^3 s^{-1}, 0.7 – 1.1 m s^{-1} and 10 g l^{-1} in the Forth estuary (Webb & Metcalfe, 1987). Sea-level rise will convert intertidal habitats into subtidal habitats and this would result in loss of 97.4 % macrobenthic biomass per unit area at the system scale. This magnitude of infaunal loss is virtually the same as the impacts of intertidal habitat loss as a consequence of land-claims elsewhere.

Although the present chapter only consider the potential impacts of sea-level rise on the characteristic fish species through their feeding ecology, populations of estuarine fish can be influenced temporally (tidally, seasonally, yearly), physically (salinity, temperature, oxygen concentration, pollution) as well as biologically (food availability, competition, predation, exploitation) (IECS, 1987; Jones, 1988; Marshall & Elliott, 1998; Elliott & Hemingway, 2002). It would therefore be very difficult to predict the impacts of sea-level rise on fish populations quantitatively by simply considering changes in the amount of their prey items unless the direct causal links can be identified between the size of fish populations and any driving factors which affect their survival. The fate of pelagic planktivores such as *S. sprattus* or *C. harengus* in response to sea-level rise is outside the scope of this thesis.

In conclusion, sea-level rise and the resulting loss of intertidal areas is a great threat to estuarine foodwebs. Because of the substantial difference in macrobenthic biomass between intertidal and subtidal habitats, implications for any loss of intertidal area are particularly alarming in the Humber estuary not only for the wintering benthivorous birds which consume macrobenthos exclusively intertidally, but also for fish species which utilise the estuary for important life stages. It is therefore clear that managing the quality and quantity of intertidal habitats is likely to be the key to the success of the maintenance of stable population sizes of both shorebirds and fish species, and therefore the integrity of estuarine ecosystems.

Chapter 5

Management recommendations

5.1 Introduction

Today, half of the global population lives on, or near, the coasts and sea-navigable waterways (Engelman et al., 2002). Average densities within 100 km of a shoreline and within a 100 m of sea level have been estimated as almost 3 times higher than the global average human density (Small & Nicholls, 2003). Historically, the world's coastal and estuarine environments have attracted human settlements through providing suitable sites for urban development, land-reclamation, harbours for trade and transportation, waste disposal, agriculture and biological exploitation. In modern times, the existence of beaches and tidal flats has added not only the recreational amenities but also the natural function of coastal protection (Moller et al., 2001; Winn et al., 2003) to their value. On the other hand, estuaries are important habitats for wildlife and have a high conservation value because estuarine intertidal areas support high benthic invertebrate biomass, especially deposit- and filter-feeding taxa, and this biomass in turn provides essential food sources for higher trophic levels such as epibenthic crustaceans, commercially important fishes and wintering shorebirds, that use the intertidal flats as a nursery area for juvenile stages and/or as adult feeding grounds (Prater, 1981; Wolff, 1987; McLusky, 1989; Elliott et al., 1990; Elliott & Hemingway, 2002). Estuaries therefore are important habitats for both humans and wildlife.

However, many of world's estuaries are changing due to rising sea levels and other aspects of climate change, such as increased storminess and altered wave energy, and on some low-lying coasts, such as southeast England, this is leading to substantial losses of intertidal habitats as a result of coastal squeeze. Intertidal habitats are dynamic features which evolve depending on the interactions between wind direction, wave energy, tidal movements, sediment transportation, rising sea levels and, over time, these habitats change and migrate land-wards as sea level rises (Pethick, 2001). To maintain the current state of estuarine ecosystems, effective conservation involves providing sufficient space to allow habitats to move and evolve in response to the changing environmental conditions (Jones, 1994). On the other hand, rising sea levels are a serious threat to the society that lives on coasts and estuaries due to the increased risk of coastal erosion and flooding. In the context of protecting both society and the conservation value of coastal and estuarine environments, four generic management options have been considered for sustainable coastal defence policies in the UK (Lee, 2001):

1. **do nothing:** carrying out no defence works, except where public safety is involved. This option would lead to continued erosion or flooding of a designated site.

2. **hold the line:** holding the defence line in its present position. This option would result in further coastal squeeze in front of the defences whilst protecting freshwater and brackish habitats inland.

3. **advance the line:** moving the defences seaward. This would result in the loss or degradation of intertidal habitats, amplifying coastal squeeze.

4. **retreat the line:** moving the defence line landward. This would result in the loss or degradation of terrestrial and freshwater habitats behind the current defences, and the re-creation of intertidal habitats.

The most likely option selected by coastal managers have been to either 'do nothing' where the standard of protection would be maintained over the next decades, or to reinforce existing sea defences ("hold the line" approach) in order to reduce the risk of flooding, particularly in front of developed urban areas or high-grade agricultural land, even though this may be unsustainable economically as well as physically, because of increased storminess and altered wave energy expected in response to climate change and sea-level rise. For the purpose of nature conservation, however, the best management option will be achieved by setting back flood defences in the estuary ("managed realignment" approach) because this will allow space for intertidal habitats to evolve and migrate inland, providing ecologically important feeding or nursery grounds for wildlife. Further, this option will create natural salt marshes that also effectively reduce wave energy and therefore the maintenance cost of flood defences (Moller et al., 2001). Where the prospect of sea-level rise makes 'do nothing' or 'holding the line' approaches more costly options, 'managed realignment' (retreat the line approach) has gained acceptance as a sustainable means of defending estuaries (Ledoux et al., 2005; French, 2006). However, identifying suitable sites for a realignment strategy will be a contentious issue since it will involve individuals or communities relinquishing their land and property for the purpose of a wider public good, coastal management. Furthermore, realignment would be a progressive, on-going process – sea levels are expected to continue to rise for at least 500 years (IPCC, 2001). An understanding of the interactions between societal needs, management options and estuarine ecology are therefore important in shaping the direction of sustainable coastal and estuarine environments in the future. The implementation of management policies will be influenced by how society responds to sea-level rise as well as how much ecologists can improve their understanding of the functionality and integrity or vulnerability of estuarine ecosystems.

Clearly, there is a need to find appropriate ways to manage the coastal and estuarine environments that sustain both nature conservation interest and socio-economic needs. In this respect, managed realignment is increasingly seen as a key approach to deliver environmental benefits and at the same time provide economically sustainable coastal flood defences (Ledoux et al., 2005; Edwards & Winn, 2006). However, when such proposals of management realignment are taken forward as schemes, their established criteria for ecological objectives are often unclear, and it seems that the environmental benefits of management realignment have not been maximised to date. The aim of this chapter is to develop a way for providing appropriate ecological objectives in the implementation of future proposed managed realignment schemes in estuarine environments. First, I will outline the context in which management realignment has become preferred option for coastal management in the UK. I then describe those managed realignments that have been proposed and implemented in the Humber estuary to date, and suggest how successful management can be measured through establishing appropriate ecological objectives with particular emphasis on estuarine intertidal ecology. The achievement of such ecological objectives requires the identification of suitable sites, how to determine their necessary area, and how to monitor the schemes after breaching (realignment).

5.2 Two major incentives for managed realignment

Flood and coastal management in England and Wales relies for its planning on Shoreline Management Plans (SMPs) for coasts and estuaries that set out the long-term sustainable strategy for coastal defence (MAFF, 1993, 1995). To date, SMPs have been based on 11 open coast cells defined in terms of sedimentary and coastal processes, rather than administrative boundaries (Winn et al., 2003; Ledoux et al., 2005). Because of this, many flood and defence operating authorities are involved in planning coastal defence works, ranging from individual land owners, local defence committees to larger organizations such as district councils, habitat charities, the Environment Agency and the Department for Environment, Food and Rural Affairs (Defra). The Environment Agency is the executive agency and has discretionary powers for flood and coastal defence under the Land Drainage Act 1991 and the Water Resources Act 1991, and Defra plays a vital role in providing strategic and policy guidance to such operating authorities (Ledoux et al, 2005). In the case of the Humber estuary, a shoreline plan was developed based on the Environment Agency's statutory responsibilities and the opinions of partner organisations (Winn et al., 2003) and its overall policy is that the flood and coastal defence schemes should be technically feasible, economically viable, environmentally appropriate and socially acceptable. In view of such policies, there appear to be two main driving forces for "managed realignment" that explain its major focus in coastal management in recent years. One driver is the pressure for reducing the ever increasing costs of maintaining current sea defences, and the other is the pressure for the UK's obligation to protect the nature conservation value of estuaries.

5.2.1 Economic incentives

Many of the policy makers are now beginning to consider that applying the existing "hold the line" policy for the entire coastal defence is no longer an economically viable option for long-term coastal management. Adopting a "retreat the line" policy is the more economically sensible option because the construction costs are minimal (Hanslip, 2002) and the newly established sea defences will also benefit from having saltmarsh and mudflats in front of them to dissipate wave energy and thus reduce the maintenance costs (Moller et al., 2001; Edwards & Winn, 2006). Generally, the costs of maintaining flood and coastal defences are estimated in terms of maintaining the standard of protection. This in turn is determined by comparing the height of the defences with the water levels and wave heights expected to occur during extreme events (Winn et al., 2003). The standard of defence needed is different in places depending on the value of assets protected, and the Ministry of Agriculture Fisheries and Food (MAFF, 1999) provides guideline for the appropriate standard for different land uses based on a range of return periods (Table 5.1). In the context of future sea-level rise, coastal defences are likely to experience a higher frequency of surges and therefore their return periods will be shorter. The frequency of surges scales logarithmically with surge height (Hulme et al., 2002), so that a relatively modest rise in sea level significantly decreases the standard of coastal protections. Similarly, the energy of waves reaching defences is a function of both local water depth and the height of the waves offshore (Carter, 1988; Crooks, 2004).

Table. 5.1. Indicative standard of protection for property around fluvial and coastal / saline areas. Lower and Higher denote a lower and higher indicative standard of protection (after NADNAC, 2004).

Typical characteristics of land use	Fluvial annual probability of failure (return period in years)		Coastal / saline annual probability of failure (return period in years)	
	Lower	Higher	Lower	Higher
Intensively developed urban areas at risk from flooding and / or erosion	0.02 (50)	0.005 (200)	0.01 (100)	0.003 (300)
Less intensively developed urban areas with some high grade agricultural land	0.04 (25)	0.01 (100)	0.02 (50)	0.005 (200)
Large areas of high grade agricultural land and / or assets of national significance requiring protection with some properties also at risk, including caravans and temporary structures	0.2 (5)	0.02 (50)	0.1 (10)	0.01 (100)
Mixed agricultural land with occasional, often agriculturally related, properties at risk. Agricultural land may be prone to flooding, water-logging or coastal erosion	0.8 (1.25)	0.1 (10)	0.4 (2.5)	0.05 (20)
Low-grade agricultural land, often grass, at risk from flooding, impeded land drainage or coastal erosion, with isolated agricultural or seasonally occupied properties at risk		>0.4 (<2.5)		>0.2 (<5)

Thus, for coastlines where an increase in average wind speed and water depth is predicted, waves will be higher and the wave climate in front of sea defences will be more energetic. This will exacerbate the problem, because less wave energy will be dissipated prior to the wave breaking on the defences, which will undermine the foundation of the structure and thus increase the maintenance costs. MAFF (1999) recommends that for planning purposes an average rate of sea-level rise at 6mm per year should be assumed for the next 50 years, implying that sea levels will rise by 0.3 m relative to land levels over the period. Thus the standard of the coastal defence will be reduced automatically, but maintenance cost will increase inevitably in the face of sea-level rise.

In the case of the Humber estuary, Winn et al. (2003) have reported that approximately 70 % of the land beside the estuary is currently provided with a standard of protection equal to or higher than the indicative standard, but this land area will decrease to less than 40 % if sea levels rise by 0.3 m but defences remain unchanged and not improved. At this rate, the cost of providing acceptable standards of defence is estimated to be of the order of £200–300 million over the next 50 years in the Humber alone (Winn et al., 2003), which amounts to an average maintenance cost of £1400–2100 per linear metre of defence over that period. On a larger scale, a recent risk assessment (NADNAC, 2004) estimated the cost of damage through coastal flooding and erosion for England at £82.7 billion and £2.5 billion, respectively, over the 100 year appraisal period, assuming a "do nothing" approach. However, the expenditure on the defences is estimated to be less than half that required to maintain even the current standard (Ledoux et al., 2005). These figures confirm that the current "do nothing" or "hold the line" policy is no longer perceived economically viable and justifiable.

5.2.2 Environmental incentives

The other major incentive for managed realignment is the UK's obligations to protect the nature conservation interests of the coastal and estuarine habitats. The two main international nature conservation conventions from which the UK's statutory obligations for site selection and protection arise are the Ramsar Convention and the Convention on Biological Diversity. Through ratification of these conventions, contracting nations or governments agree in principle to apply the listed obligations and in some cases to transpose them into national legislation (Boere & Taylor, 2004). For example, under the Ramsar Convention, contracting parties are obliged to designate wetlands for the List of the Wetlands of International Importance and to formulate and implement planning so as to promote conservation and protection of listed sites. The UK Biodiversity Action Plan is an another example of an obligation published in January 1994 in response to Article 6 of the Convention of Biological Diversity, to develop national strategies for the conservation of biological diversity and the sustainable use of biological resources.

In relation to wildlife and nature conservation within Europe, the most influential measures are the Birds and the Habitat Directives which provide for the protection of animal and plant species of European importance and the habitat which support them. These Directives require member states to implement their provisions nationally for the benefit of Europe as a whole, and regulations directly implement EU policy in member states without the need for member states to enact their own legislation. These two directives are the European Union's major response to the Convention of Biological Diversity (1992) which was, in turn, transposed into national law in the UK through the 1994 Habitats and Conservation Regulations. The aims of the Directives are to create a network of designated sites (Natura 2000) which represent areas of the highest value for natural habitats and species of plants and animals which are rare, endangered or vulnerable in the European Community. The Natura 2000 network includes two types of area. Special Areas of Conservation (SACs) are designated under the Habitat Directive where they support rare, endangered or vulnerable natural habitats and species of plants or animals (other than birds), whereas Special Protection Areas (SPAs) are designated under the Birds Directive where areas support significant numbers of wild birds and their habitats. The Habitat Directive calls for measures intended to protect all the Natura 2000 sites from deterioration and damage, and a plan or project likely to have significant effect on these sites must take appropriate steps to determine whether it would reduce the nature conservation interest of the sites. SPAs are often situated over intertidal habitats in front of sea defences in the estuarine environment and it is clear that the regulations have significant implication for the flood and coastal management in terms of the delivery of coastal defence projects and their planning. In the face of tidal advance in response to sea-level rise, the current hold the line policy will inevitably reduce the area of intertidal habitats, and the UK's habitats regulations create a strong presumption to protect such important sites *in situ* unless this cannot be economically or technically justified (Ledoux et al., 2005). In other words, the Habitat Directive is interpreted as a strong requirement for re-creation of intertidal habitats to compensate for habitats lost to any development or coastal squeeze, and this obligation for nature conservation has become another major driver for the implementation of managed realignment.

5.3 Current status of managed realignment

5.3.1 Criteria for site selection

Managed realignment schemes aim to re-design current flood and coastal defences for both economic viability and the maintenance of ecological integrity in the long run. For policy makers, a fundamental requisite for making such a management decision is proper justification of the economics and a sound understanding of ecological functioning. From an ecological point of view, the benthic macrofauna of intertidal habitats plays a central role in supporting the estuarine food web. Because muddy and sandy shores exhibit significant variations in species composition, abundance and biomass of macrobenthic invertebrate assemblages depending on the prevailing physical conditions, opportunistic selection of sites for the creation of intertidal habitats would not necessarily compensate the ecosystem for the loss of ecological integrity induced by coastal squeeze. However, when identifying suitable areas for realignment, little attention has been paid to the ecological benefits for the estuary and it has been carried out primarily on economic or geomorphological grounds. For example, suitable areas for realignment within the UK have been identified by the Royal Society for the Protection of Birds (RSPB) based on the following criteria (Pilcher et al., 2002):

Criterion 1: Location

- The site is adjacent to a tidal estuary or the sea.

Criterion 2: Land use and infrastructure

- No development and minimal infrastructure such as roads or power lines occur in the area.

Criterion 3: Length of realigned defences

- The length of new flood defences required to ensure adjacent areas continue to be protected must be no greater than the length of any existing structures.

Criterion 4: Size

- The size of the realigned area should be at least five hectares, due to cost performance.

These criteria are based purely on economic considerations because undeveloped areas are physically and economically easier to convert into intertidal habitats than developed areas, and shorter defences can reduce the costs of both construction and maintenance. With respect to criterion 4, Defra (2003) also recommends creation of large new habitat rather than that of several isolated smaller areas because not only is this likely to be cost effective, but this also has an ecological advantage. However, Defra (2003) does not provide an explanation why the creation of larger intertidal habitats is ecologically advantageous in estuarine management.

In addition to the list above, Coombes et al. (2004) have identified other criterion for site selection such as elevation and historical context of the area. Elevation is important because the area below the high spring tide level represents the maximum area of intertidal habitat that could be created, and historical context is important because land which was formerly intertidal habitat within the 20th century and has remained undeveloped is more suitable for return to intertidal habitat than that which was reclaimed at an earlier time (Coombes et al., 2004). Based on such criteria, and with the use of geographic information systems (GIS), it should be possible to identify sites that are suitable for management realignment from economic or geomorphological point of view. However, none of these criteria consider the need to maintain the ecological integrity of the ecosystem.

5.3.2 Managed realignment in the Humber estuary

To date, the Environment Agency has identified 9 possible managed realignment sites including the one that opened in October 2003 at Paull Holme Strays (Environment Agency, 2003; Edwards & Winn, 2006). These 9 sites are located throughout the estuary, with 2 sites on river Ouse and Trent, 1 in the inner estuary, 3 in the middle estuary and 3 in the outer estuary (Fig. 5.1).

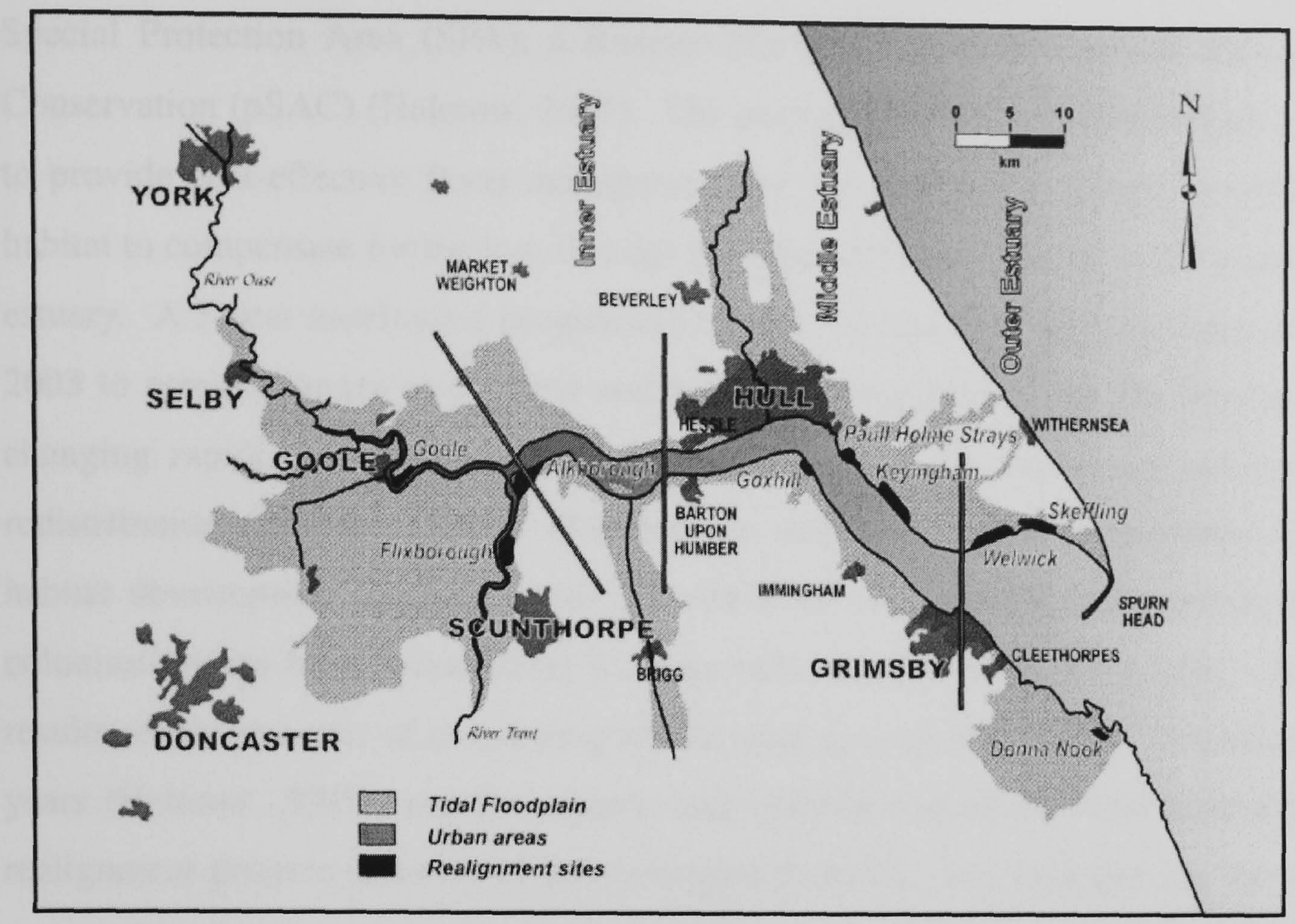


Fig. 5.1. The Humber floodplain with managed realignment sites (from Edwards & Winn, 2006)

The Environmental Agency expects the two potential sites near the rivers Ouse and Trent would provide flood storage and the six seaward of Hull would provide replacement habitat. The site at Alkborough would provide both. These 9 sites could provide a total of 1,992 ha of land for managed realignment, but the Environment Agency are currently reviewing the list to select more preferred sites and the area will be reduced to around 710 ha to compensate for coastal squeeze (Environment Agency, 2003).

5.3.2.1 Paul Holme Strays managed realignment

Of the 9 sites the Environment Agency identified, realignment of the flood defences has now been implemented at Paul Holme Strays, the first major realignment scheme on the Humber. The site is located in the middle estuary on the north bank, approximately 10 km to the south east of Hull (Fig. 5.1). The old tidal embankment was breached in September 2003 and the site now provides approximately 80 ha of newly created intertidal habitat which is adjacent to a

Special Protection Area (SPA), a Ramsar Site and a possible Special Area of Conservation (pSAC) (Halcrow, 2005). The primary objectives of this project are to provide cost-effective flood management for the area and to create intertidal habitat to compensate for the loss through the other defence schemes in the middle estuary. A 5-year monitoring programme for the site has been in place since late 2003 to assess changes in physical and biological properties. The site has been changing rapidly in response to tidal movements and the associated sediment redistribution (Halcrow, 2005). For instance, sediment accretion necessary for habitat development has been observed within the site, and benthic invertebrate colonisation has been followed by increase in bird usage around the area. The results of the first year of monitoring will be used as a baseline for the subsequent years (Halcrow, 2005), and the scheme may become one of the most promising realignment projects in terms of environmental benefits. For example, the site is expected to develop 43 ha and 32 ha of saltmarsh and mudflat, respectively, and expected to support feeding wintering redshank, *Tringa totanus*, dunlin, *Calidris alpina*, shelduck, *Tadorna tadorna*, and curlew, *Numenius arquata*, as well as providing a winter roost for other wintering shorebirds including golden plover, *Pluvialis apricaria*.

However, the location and the size of the site have been determined in a rather opportunistic manner as a result of a coastal defence failure in 1995 (Hanslip, 2003), and the ecological objectives of the scheme do not specify how the creation of this site can compensate for the expected loss of macrobenthic standing crops from the Humber ecosystem. Further, if the ecological target for a project of this kind is to create an identical habitat to the adjacent intertidal habitat in front of the managed realignment site, it may be misleading. This is because the abundance and biomass of benthic invertebrates vary considerably along vertical gradient of the shore profile with highest biomass value often found around mid shore levels (Chapter 2). Almost all realigned sites will be located at the highest level on the shore profile, and such areas tend to have lower benthic invertebrate biomass.

5.3.2.2 Alkborough managed realignment

Alkborough is another site identified for the Shoreline Management Plan in the Humber. The Environment Agency started work here in July 2005 and expect the final breaching to take place in autumn 2006. The site is situated on the south bank of the inner Humber estuary at the confluence of the River Ouse and Trent (Fig. 5.1). One of the characteristics of the scheme is that the project is designed to reduce both the risk of flooding in the inner Humber and around the tidal rivers by allowing previously low-lying arable land to flood. This is based on estuary process studies which indicate that setting back sea defences in the inner estuary or the tidal rivers leading to it may reduce the effects of sea-level rise by lowering water levels locally during high tides or extreme events (Winn et al., 2003). For example, it has been estimated that an area of approximately 400 ha in the river sections could reduce the water level by up to 300 mm (Winn et al., 2003), providing a strong incentive to select sites for alignment around inner rather than in the middle or outer part of estuary. The scheme will create up to 440 ha of intertidal and freshwater wildlife habitats, and is one of the largest intertidal wetland creation projects in the UK, and is therefore expected to bring nature conservation benefits for local wildlife in the inner Humber. This area of the estuary is characterized by extensive *Phragmites* reedbeds, saline lagoons, saltmarsh and wet grazing with fringing mudflats (English Nature, 2003). Reedbeds and freshwater marshes provide nesting sites for a number of bird species, such as marsh harrier, *Circus aeruginosus*, and bearded tit, *Panurus biarmicus* (English Nature, 2003), and bittern, *Botaurus stellaris*, which is the subject of a species recovery programme in the inner Humber. Avocet, *Recurvirostra avosetta*, is associated with shallow lagoons around this site, and mudflats and salt marshes can provide feeding and roosting sites for waterfowl such as mallard, *Anas platyrhynchos*, teal, *Anas crecca*, and shelduck, *Tadorna tadorna* (English Nature, 2003).

However, the large intertidal mudflats developing at Alkborough are likely to have limited importance for overwintering waders dependent on benthic invertebrates for their diet. Intertidal mudflats found around this site are small and have an impoverished infauna mainly due to large salinity variations over the tidal cycle and harsh environmental conditions. The new intertidal flats will be similar to the surrounding habitats in their ability to provide prey items for higher trophic levels, and a creation of large intertidal habitats in the inner estuary will not therefore necessarily lead to a development of vital feeding grounds for all the wintering birds visiting in the Humber. Such habitat creation may be good for a local wildlife, but does not accurately reflect the "no net loss policy" at the estuarine system scale. For example, the intertidal flat near Alkborough supports a mean macrobenthic biomass of 0.04 g AFDW m^{-2} (Chapter 2, transect 6), whereas the large intertidal flat of Spurn Bight at the outer region on the north bank has an average biomass of 17.1 g AFDW m^{-2} (Chapter 2, transect 1 & 2). Thus, in terms of macrobenthic biomass available for higher trophic levels, the creation of 400 ha intertidal habitat around Alkborough could not even compensate for the loss of 1 ha in Spurn Bight.

5.4 Management recommendation from environmental perspective

Given the importance of macrobenthic invertebrates as prey items not only for a variety of wintering birds but also for fish and crustaceans (Chapter 4), this thesis suggests that an urgent need to incorporate intertidal benthic ecology into management planning for maintaining the relevant conservation designations within an estuary. To date, criteria for site and its size selection for managed realignment have been rather opportunistic, and success criteria for ecological objectives have been expressed in simple terms, such as areas of mudflats and salt marshes, or number of birds feeding and/or roosting on site (Leggett et al., 2004; Halcrow, 2005). However, intertidal mudflats and sandflats within an estuary exhibit significant spatial variations in the biomass of macrobenthic prey, and the creation of large intertidal habitats could result in a reduction of macrobenthos overall unless site selection is based on a sound ecological understanding. One of the primary ecological objectives should be to maintain the current standing crops of macrobenthos at the estuary scale. To achieve this goal, two main recommendations are made here with respect to site selection and subsequent monitoring programmes for the Humber estuary.

5.4.1 Site selection for managed realignment

The Humber estuary has an area of approximately 12,000 ha of intertidal habitat, and more than 90% is estimated to be mudflat and sandflat with the remainder being largely salt marshes (Winn et al., 2003). In addition, there are approximately 200 ha of reedbed and 120 ha of several saline lagoons around the estuary (Winn et al., 2003). Currently the Humber has 10 sites of Special Scientific Interest (SSSIs) and its intertidal flats are also designated as a Special Protection Area (SPA) and Ramsar site (English Nature, 2003). Because of its international recognition for the breeding, passage and wintering birds, the entire estuary has also been proposed as a marine Special Area of Conservation (SAC) (Ledoux et al., 2002). On the other hand, there are approximately 235 km of artificial flood defences throughout the estuary (Winn et al., 2003) and there is little undefended area, suggesting that all important intertidal habitats are confined between artificial sea defences and rising sea levels. Defra (2003) has recommended that replacement habitat should ideally be located in or adjacent to the designated site under conservation legislation. Simply retreating current defences to protect conservation designations would of course be impossible due to the potential costs involved in implementing such a scheme. Nevertheless, it is possible to identify a small number of large sites for habitat creation to offset the total loss of intertidal habitats across the estuary.

First, replacement sites must be identified and appropriate size must be calculated on the basis of providing macrobenthic invertebrates for higher trophic levels at the same levels as present for the estuary. The spatial patterns in macrobenthic biomass presented in Chapter 2 & 3 provide a useful starting point for prioritising site selection and calculating the sizes necessary for habitat creation in order to maintain the ecological integrity of the estuary. This approach assumes that newly created intertidal habitats will have the same level of quality (mean macrobenthic biomass per unit area) as the original sites identified for realignment. Figure 5.2 shows beach width, expected loss of beach width and mean macrobenthic biomass in relation to distance from mouth on the north bank and the south bank of the Humber estuary.

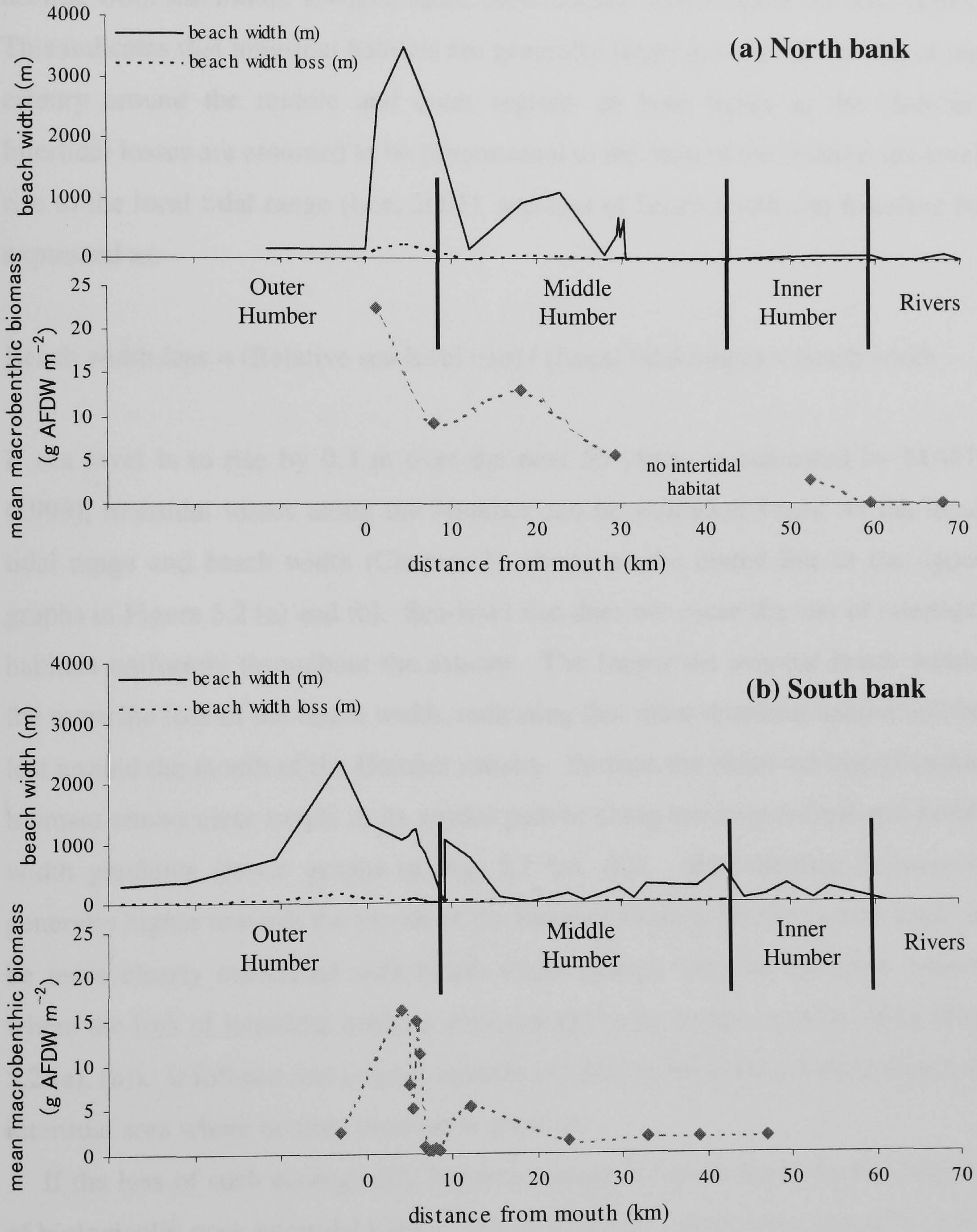


Fig. 5.2. Beach width (m) (solid line), expected loss of beach width caused by 0.3 m sea-level rise (dotted line), and the observed mean macrobenthic biomass at the transect scale (g AFDW m^{-2}) (dotted line with markers), in relation to distance from mouth (km) for (a) north bank and (b) south bank of the Humber estuary. X axis was divided into four regions (Outer, Middle, Inner and Rivers) based on Environment Agency (2003).

In present study, the mouth of the Humber is defined as a mid point between Spurn Head on the north bank and Tetney Haven on the south. In the upper graphs in Figure 5.2 (a) and (b), beach width generally becomes longer towards the mouth from the inner rivers to the outer Humber, but the width also starts to decline from the mouth towards more outer coastal environment on both banks. This indicates that intertidal habitats are generally larger towards the mouth of the estuary around the middle and outer regions on both banks in the Humber. Intertidal losses are assumed to be proportional to the ratio of the relative sea-level rise to the local tidal range (Lee, 2001), and loss of beach width can therefore be expressed as:

$$\text{Beach width loss} = (\text{Relative sea-level rise}) / (\text{Local tidal range}) \times \text{beach width}$$

If sea level is to rise by 0.3 m over the next 50 years, as estimated by MAFF (1999), intertidal losses along the Humber can be estimated based on the local tidal range and beach width (Chapter 2), shown as the dotted line in the upper graphs in Figure 5.2 (a) and (b). Sea-level rise does not cause the loss of intertidal habitats uniformly throughout the estuary. The longer the original beach width, the more the loss of the beach width, indicating that more intertidal habitat will be lost around the mouth of the Humber estuary. Further, the observed macrobenthic biomass shows clear trends in its spatial pattern along the longitudinal and beach width gradients (lower graphs in Fig. 5.2 (a), (b)). Macrobenthic biomass is generally higher towards the mouth of the Humber estuary, but the pattern tends to be more clearly associated with beach width change towards the outer estuary where the loss of intertidal habitats are expected to be higher on both banks (Fig. 5.2 (a), (b)). It follows that coastal squeeze is likely to have the greatest impact on intertidal area where benthic biomass is greatest.

If the loss of such ecologically important areas is compensated by the creation of biologically poor intertidal habitat, this will lead to a significant loss of food for the higher trophic levels. For example, the Environment Agency (2003) estimated that 710 ha of new habitat should be created to compensate for coastal squeeze, showing provisional targets for habitat creation in the four regions across the

Humber estuary (Table 5.2). The potential gain of macrobenthic biomasses (t) for each region and their contribution to the system total biomass through the provisional realignment schemes as well as the two ongoing schemes (at Paull Holme Strays and Alkborough) is shown in Table 5.2. This simple calculation suggests that if the total of 710 ha of intertidal habitats is created as indicated by the Environment Agency (2003), this will generate a further 4.7 % of total macrobenthic biomass available within the system, but current ongoing schemes at Paull Holme Stray and Alkborough could only contribute a further 0.8 % of the total biomass. The former value of 4.7 % could vary depending on how site selection is to be conducted within each region because mean macrobenthic biomass can vary considerably even within each region (Fig 5.2). Thus, if a large area behind high quality intertidal area is selected for managed realignment, this will increase the percentage biomass gain considerably. However, the model simulations in Chapter 3 showed that if sea level rises by 0.3 m, intertidal area will be reduced by 807 ha and the system total macrobenthic biomass will be reduced by between 6.9 % and 22.8 %, depending on how key physical environments change in response to sea-level rise. The two managed realignments at Paull Holme Strays and Alkborough alone could not compensate for such magnitude of losses, and effort needs to be made to identify more suitable sites for managed realignment in order to counteract the future loss of macrobenthic biomass.

Table. 5.2. Provisional estimates of the areas intertidal habitat needed to compensate for losses, the areas of two ongoing realignment schemes (figures in shaded area after Environment Agency, 2003) and expected gain of macrobenthic biomass under such schemes in the Humber estuary.

Loss categories	Outer Humber	Middle Humber	Inner Humber	Rivers	Total
Predicted coastal squeeze (ha)					450
Uncertainty (ha)					150
Total intertidal area loss (ha)	180	360	60	0	600
Estimated compensation for losses (ha)	210	400	95	5	710
Mean macrobenthic biomass (g AFDW m^{-2})	11.6	5.7	2.2	0.04	
Expected gain of macrobenthic biomass (t)	24.4	22.7	2.0	0.0002	49.1
(% gain in relation to system total biomass)					(4.7 %)
		Paull Holme Strays		Alkborough	
Managed realignment taking place (ha)		80		440	480
Mean macrobenthic biomass (g AFDW m^{-2})		10.5		0.04	
Expected gain of macrobenthic biomass (t)		8.4		0.2	8.6
(% gain in relation to system total biomass)					(0.8 %)

The current spatial patterns observed in macrobenthic biomass strongly suggest that potential sites for managed realignment exist around the outer regions on both banks or middle region on the north bank of the Humber estuary (Fig. 5.2), although this approach for site selection is purely based on an ecological perspective and does not take socio-economic imperatives into consideration.

5.4.2 Ecological objectives and subsequent monitoring strategies for individual schemes

Once a potential management site has been identified, the macrobenthic models established in Chapter 3 can be used to estimate how much gain of macrobenthic biomass is obtained for an individual scheme by comparing predicted model outputs under different management options. The models use key environmental parameters known to affect the macrobenthic biomass, namely salinity, sediment characteristics and morphological elements at the transect scale (Chapter 2, 3). Knowledge of how different management options, such as "hold the line" or "managed realignment", affect these key environmental parameters over a given time horizon allows the best possible options to be explored. In addition, this predicted macrobenthic gain can be used as an ecological objective for the individual scheme against which its success or failure can be judged. Figure 5.3 shows a simple habitat change model from present state into the future. The gain of macrobenthic biomass by the creation of new habitat can be calculated as follows:

$$\text{Macrobenthic biomass gain} = \text{Biomass (C)} + \text{Biomass (O)} - \text{Biomass (S)}$$

where C, O and S represent newly created intertidal, intertidal area in front of (or outside) the newly created area, and intertidal habitat squeezed due to sea-level rise, respectively. Although one might expect that biomass gain is simply a calculation of biomass (C), this would be wrong because biomass (O) and biomass (S) will be different depending on the presence or absence of the newly created area (C) due to changes in their physical properties.

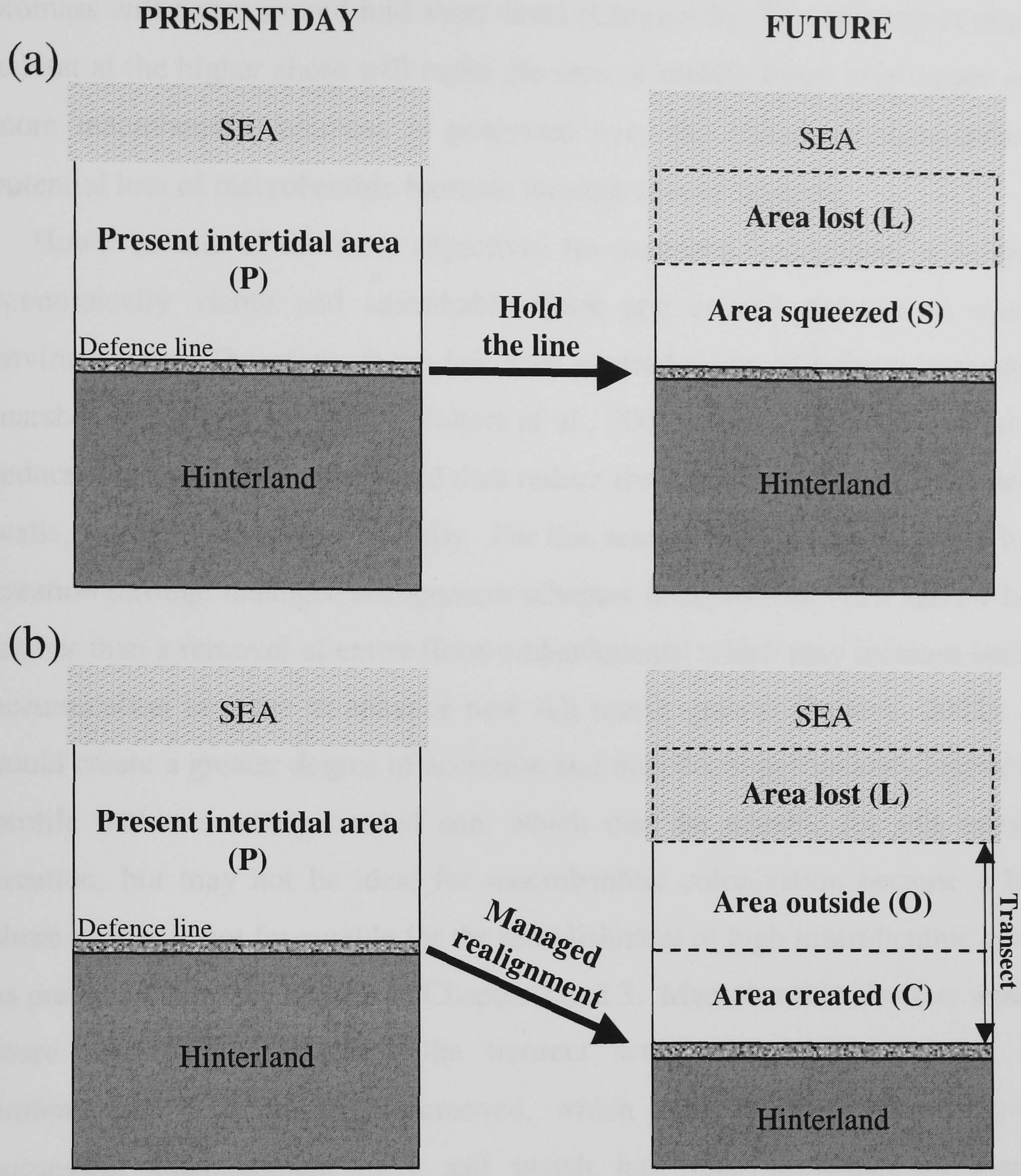


Fig. 5.3. A simple habitat change model for (a) hold the line approach, and (b) managed realignment approach.

For instance, Area (S) is likely to be steeper than Area (O) in terms of their beach face profiles due to the physical processes of habitat creation on Area (O), but of coastal squeeze on Area (S). It will also be important to monitor not only within the newly created area (C) but also in the area outside (O) so that the monitoring programme can cover the entire intertidal profile between high and low tide levels, as shown in Fig. 5.3, because this is the only way to properly measure the ecological effects of managed realignment. If an intertidal transect is established running from the edge of the sea defence to the mean low water level, the highest biomass will occur around mid shore level (Chapter 2). The creation of intertidal habitat at the higher shore will make the area of middle shore level larger so that more macrobenthic biomass is generated over the transect, counteracting the potential loss of macrobenthic biomass through coastal squeeze.

However, one of the main objectives for managed realignment is to provide economically viable and sustainable flood and coastal defence in estuarine environments. Therefore, focus has been centred more on how to re-create salt marshes in realignment sites (Wolters et al., 2005; French, 2006), since this will reduce wave and tidal energy and thus reduce the maintenance cost of the defence walls, and increase coastal stability. For this reason, there are cases where habitat creation through managed realignment schemes is carried out via a narrow breach (rather than a removal of entire flood embankment), which may increase sediment accumulation in order to enhance new salt marsh growth (Pethick, 2002). This could create a greater degree of accretion and thus disproportionately higher shore profile within a newly created site, which may be suitable for salt marsh re-creation, but may not be ideal for macrobenhtic colonization because a higher shore profile is not favourable for the establishment of high macrobenthic biomass as predicted from the models in Chapter 2 and 3. Macrobenthic biomass would be more readily established at the transect scale when the remaining flood embankment was gradually removed, which may be feasible only once a successful establishment of a salt marsh has been confirmed in managed realignment sites. It would therefore be essential to monitor biological, physical and morphological properties over the entire intertidal transect so that the overall effect of a scheme can be assessed.

In summer 2006, two more sites were added to the current ongoing managed realignment schemes in the Humber. The site at Welwick is located at the west end of the Spurn Bight on the north bank (Fig. 5.1), and the site at Chowder Ness is situated near Barton-Upon-Humber around the middle region on the south bank of the Humber estuary (Fig. 5.1). The former has added 47 ha of newly created intertidal habitat and would be promising in terms of producing a future gain of macrobenthic biomass and the latter with 13 ha would be more suitable for providing flood storage as well as a replacement site. These on-going management sites in the Humber will provide a useful test case for sustainable estuarine management which is of relevance to other estuaries in the UK as well as elsewhere trying to cope with sea-level rise.

Chapter 6

Conclusions

6.1 Fulfilment of research objectives and general conclusions

Many estuaries have high conservation value because of their utilisation by fish crustaceans and shorebirds. This thesis emphasises the importance of maintaining the availability of their macrobenthic prey, particularly those species known to be important food sources for higher trophic levels. Extensive areas of intertidal habitats that support such invertebrates could be lost through rising sea levels, which could also cause concomitant environmental changes such as salinity intrusion, beach steepening and reworking of sediment particle size composition in the estuarine intertidal habitat. In the face of such expected environmental changes, this thesis addresses two important research questions. First, how the biomass of benthic macrofauna is likely to change in response to environmental changes resulting from sea-level rise (Chapter 2 & 3) and how this in turn affects the consumers they support (Chapter 4). Second, how to find appropriate coastal and estuarine management approaches that can sustain both nature conservation interests and socio-economic needs (Chapter 5).

In order to address the first question, extensive field surveys were conducted in September 2003 (Chapter 2) and 2004 (Chapter 3). In Chapter 2, the results from the 2003 survey revealed that the dominant species on the Humber were *Cerastoderma edule*, *Macoma balthica* and *Nereis diversicolor*. Multiple regression analysis also indicated that spatial variation in macrobenthic biomass was significantly explained by key environmental variables such as salinity, sediment characteristics and morphological elements, consistent with the general picture claimed for estuaries elsewhere (Chapter 1). In Chapter 3, field observations made in the 2004 survey confirmed that macrobenthic biomass declined and intertidal beaches experienced steepening in response to coastal squeeze. Two data sets from the 2003 and 2004 survey were then combined in order to provide parameters for statistical models aimed at predicting changes in macrobenthic biomass in response to environmental gradients in the Humber estuary. The models indicate that salinity, median particle size, silt content, transect slope, station slope and median depth of the beach were the six key environmental variables which can explain between 80 % and 85 % of the variance in the observed macrobenthic biomasses of *C. edule*, *M. balthica* and the remaining species. Model simulations using these statistical models revealed that if sea level should rise by 0.3 m, between 3.9 % and 22.8 % of macrobenthic biomass could be removed from intertidal habitats, depending on how such key environmental variables actually respond to sea-level change. In Chapter 4, analysis of shorebird distributions on the Humber revealed significant associations between birds and the amount and distribution of the prey at the beginning of wintering period, confirming that the latter are major determinants of the density and distribution of benthivorous shorebirds. This suggests that populations of intertidal dependent birds could be described as a function of the quality (mean macrobenthic biomass) and quantity (area) of intertidal habitats. This simple bird model predicts that if sea level rose by 0.3 m, loss of intertidal area alone would lead to the reduction of the wintering benthivorous bird population by 5707 (7.2 %), whilst adding the effects of environmental changes could reduce the number of birds further by up to 11691, a total reduction of up to 15.9 %. Further, a study of fish diets confirmed that intertidal habitats are important as feeding grounds for many of fish species in the Humber estuary. Any loss of intertidal habitats for fish may be particularly significant in the Humber because of the substantially greater macrobenthic biomass in the intertidal compared to subtidal areas.

For the second question of suitable management approaches, the context in which managed realignment has become the preferred option for shoreline management in the UK is described in Chapter 5. Ideally, managed realignment should aim to re-design current flood and coastal defences for both economic viability and the maintenance of ecological integrity over the long term. However, examination of management realignments that have been proposed and implemented in the Humber estuary to date indicates that efforts need to be made to identify more suitable sites for realignment schemes in order to maximise their ecological benefits. Chapter 5 also argues that the observed spatial patterns in macrobentic biomass provide a useful starting point for prioritising site selection, and the models created in Chapter 3 give an indication of the scale of land area needed for managed realignments in different parts of the estuary and the ecological benefits that could be achieved through such schemes.

Overall, this thesis has confirmed that sea-level rise is a significant threat for estuarine ecosystems and there is a need to find appropriate coastal and estuarine management approaches. In typical European estuaries, 90-95 % of the estuarine intertidal habitats are bare mudflats and sandflats (McLusky and Elliott, 2004) that are important feeding and nursery grounds for higher trophic levels, and it is therefore important to secure such habitats to counteract the future loss of macrobenthic prey that would result from sea-level rise, which should be built into the process of decision making in coastal and estuarine management. Managed realignment has gradually gained acceptance as a preferred option in the context of sustainable coastal and estuarine management in the UK as well as in other parts of the world (Ledoux et al., 2005; Wolters et al., 2005). However, the option remains contentious and has not been adopted on a large scale. Even where the proposals for realignment have been taken forward, such schemes have been often opportunistic, for example, in response to an accidental defence failure (Wolters et al., 2005). Furthermore, economically viable coastal defence is the primary objective for such schemes, and managed realignment is chosen to improve both coastal stability and cost performance by replacing costly artificial "hard" coast protection with less costly natural "soft" coastal landforms. For this reason, the main focus in the implementation of managed realignment has been centred on the re-creation of "salt marshes" in order to provide an expansive energy dissipation system (Pethick, 2002; Walters et al., 2005; French, 2006), and none of the schemes have yet made an attempt to minimise the negative ecological impacts addressed in this thesis.

However, while success criteria for flood defence standards and their cost performance may be easy to define, criteria relating to environmental benefits are more complex and have been difficult to establish (Leggett et al., 2004). Approaches based on cost-benefit analysis often fail to accommodate issues such as environmental sustainability, robustness, or valuation of ecosystem services (Ledoux et al., 2005) and lack of ecological knowledge often means it is difficult to evaluate the success of habitat creation (Leggett et al., 2004) and thus delivery of environmental objectives. Even if environmental objectives for individual schemes are established, they may vary from location to location even within the same estuarine system. For instance, the creation of wet grassland and reedbed in one scheme, and a creation of roosting site for a particular bird species in another. However, the results of the research within the present thesis strongly suggest that there is a need to create overall environmental objectives at the whole estuary system scale in order to maintain ecological integrity. While the creation of salt marsh may be of primary concern for coastal defence objectives, the creation of intertidal sand- and mudflats is none the less valuable for ecological objectives (Garbutt et al., 2006). In view of the large numbers of higher trophic level species that depend on intertidal invertebrates, the maintenance of macrobenthic biomass within an estuary should be one of the main environmental objectives in order to comply with statutory obligations for protecting the nature conservation values of estuaries throughout the UK.

A further consideration is that any delay in implementing appropriate adaptive strategies in the coastal region will reduce their effectiveness in reducing ecological and socio-economic impacts over the medium and long term, when the effects of sea-level rise could be catastrophic. On a regional scale, there is already increasing evidence that intertidal habitats are being lost at an alarming rate as a result of rising sea levels coupled with coastal subsidence. For example, almost 60 % of wetland loss observed along the northern Gulf of Mexico was due to the net effect of sea-level rise and subsidence (White & Tremblay, 1995). Along the Louisiana coast, annual losses of up to 73 km^2 of wetland area were attributable to the increased rate of relative sea-level rise during the 20th century (Eisma, 1998). Thus, the effectiveness of habitat restoration through managed realignment will depend on when, where and how such schemes are implemented through proactive estuarine and coastal management.

Research into the optimum areas, location and type of restoration needed to reduce the risk of flooding and maintain ecological integrity is ongoing in estuarine and coastal environments. It is hoped that the recommendations presented in this thesis will be considered in the implementation of future managed realignment and habitat restorations. Although there are several important issues that could not be incorporated in this thesis, as shown in the next section, it can be concluded that this thesis has successfully fulfilled the research objectives originally set out.

6.2 Future work

The macrobenthic models developed in Chapter 2 and 3 provide a basis for predicting how the biomass of intertidal macrobenthos is likely to change in response to sea-level rise. However, such statistical modelling requires large amounts of data from a number of locations over a wide range of habitats (Thurush et al., 2003). In this thesis, the more intense sampling from Grimsby might over-emphasise this area in the models and this could mean that the predictions for the outer estuary are better than those for the rest of the estuary. In this respect, more data sets from other areas, such as the inner or middle regions of the Humber that has clear beach width gradients, should be incorporated in order to make quantitative predictions with further confidence.

The reliability of the models is also dependent on how accurately the change of the key environmental variables in response to sea-level rise can be predicted. Although sediment distribution was assumed to migrate upstream in the same way as salinity intrusion in the model simulations (Chapter 3), little is actually known as to how the detailed sediment characteristics of the sediment will respond to sea-level rise. However, the consequences of the change in sediment distribution could have the greatest impacts on macrobenthic biomass as indicated in the model simulations and there is an urgent need to improve understanding of such important physical processes.

There are other uncertainties involved in the present models because the modelling approach in this thesis did not incorporate the impacts of temporal factors, such as inter-annual variability in climatic conditions, increase in ambient temperature, increased occurrence of extreme climatic events (e.g. storms and surges) and changes in nutrient load or primary production. In view of the relationships between the above temporal factors which could influence the amplitude of macrobenthic biomass distribution over the entire estuarine scale, and the key environmental variables (spatial factors) which are associated with the local spatial variation within the estuary, the system total macrobenthic biomass in year (*t*) may be described as follows:

$$\text{Biomass}(\text{T}_{\text{n}\,(t)}) = F\{\text{key environmental variables in year } (t)\}$$

where Biomass ($\text{T}_{\text{n}\,(t)}$) denotes mean macrobenthic biomass at transect T_n in year (*t*) expressed as a function of the key environmental variables in year *t* and thus:

$$\text{System Total Biomass in year } (t) = Kcpn\ (t) \times \sum\ (\text{Biomass}(\text{T}_{\text{n}\,(t)}) \times \text{Area}(\text{T}_{\text{n}\,(t)}))$$

where *Kcpn* (*t*) denotes a term determined by factors such as climatic condition (inter-annual variability, ambient temperature or extreme climatic events) *c*, primary production *p* and nutrient load *n* in year *t*, and Area ($\text{T}_{\text{n}\,(t)}$) indicates the area represented by transect T_n in year *t*. Here, *Kcpn* (*t*) could be interpreted as a coefficient for the term $\sum$ (Biomass($\text{T}_{\text{n}\,(t)}$) × Area($\text{T}_{\text{n}\,(t)}$)). These equations therefore indicate that the spatial patterns of observed macrobenthic biomass can be significantly explained by the key environmental variable in any year *t*, yet the system total macrobenthic biomass in year *t* could still fluctuate depending on how the coefficient term *Kcpn* (*t*) varies over time. This thesis fully explored the term $\sum$ (Biomass($\text{T}_{\text{n}\,(t)}$) × Area($\text{T}_{\text{n}\,(t)}$)), assuming that the coefficient *Kcpn* is held constant. However, as reviewed in Chapter 1, factors relating to *Kcpn* are known to strongly influence the ecology of estuarine benthic invertebrate assemblages over time through changes in food availability, fecundity, individual growth and predation (Jensen & Jensen, 1985; Beukema et al., 1993; Olafsson et al., 1994; Heip et al., 1995; Beukema et al., 1997; Herman et al., 1999; Widdows & Brinsley, 2002; Boyes & Elliott, 2006), and long-term studies are essential to improve the understanding of such factors.

In Chapter 4, the future population of intertidal dependent birds was described as a function of the quantity (area of intertidal habitat) and quality (mean macrobenthic biomass per unit area) of intertidal habitats at the estuarine system scale in the Humber. However, this approach could not examine changes in bird populations at the species level because of the dietary differences between benthivorous bird species. West et al. (2005) suggest that a minimum food : bird ratio for individual bird species above which 'mortality' does not vary and below which 'mortality' increases steadily, be incorporated based on factors such as prey availability, food intake rate, interference and so on, to determine the quality and the carrying capacity of estuaries. However, the utility of this approach will still depend on our ability to separately quantify the prey for each of the different bird species, and there is a great need for such investigations.

In Chapter 4, data on the abundance of subtidal macrobenthos in the Humber estuary were derived from the Environment Agency's long term monitoring survey in 1995. Further the biomasses of these species were estimated using average individual biomass values of either the same or morphologically similar species taken from intertidal habitats recorded in the survey in 2003 and 2004. There is an assumption therefore that individual subtidal macrobenthic species in 1995 had similar biomass values to those intertidal species in 2003 and 2004, which may or may not be valid.

In Chapter 5, the site selection process and establishment of ecological objectives provides an indication of the scale of land area needed for managed realignments and the ecological benefits that could be achieved through such restoration schemes. However, because the application of ecological understanding to restoring estuarine intertidal habitats is still in its infancy, good baseline condition survey data and subsequent monitoring are essential if any change in the ecological status of estuarine and coastal habitats in response to sea-level rise is to be recognised. Most monitoring schemes are generally too short term and thus inadequate to identify endpoints of ecosystem or community maturity (Atkinson et al., 2004), and in this respect long-term strategic monitoring should be required to identify processes that determine important ecological outcomes.

Further, there remains uncertainty with the implementation of habitat creation due to a less than perfect understanding of the highly dynamic nature of the physical processes found in estuarine systems, resulting, perhaps, in an emergence of a different habitat and biology from a predicted managed realignment scheme. It is hoped that scientific research on ecological restoration could be continuously updated as knowledge increases to help reduce uncertainty in the design of future estuarine realignment schemes. In this respect, a challenge for ecological restoration in estuarine environments would be to convince scientists from many disciplines such as meteorologists, sedimentologists, biologists, botanists, biochemists, geomorphologists and hydrologists to co-operate more effectively in order to achieve better integration of data necessary to make reliable predictions about estuarine ecology in the future. Incorporation of Geographic Information Systems (GIS) into such multi-disciplinary environmental research will also play a vital role in making a significant contribution to improve the design of future estuarine shoreline schemes.

Finally, this thesis emphasises a need to find appropriate coastal and estuarine management approaches in order to sustain both nature conservation interests and socio-economic needs. However, it was not possible here to fully explore socio-economic aspects of managed realignment. Although managed realignment shows promise for achieving economically viable and ecologically sound management, significant advances have yet to be made in expanding the area of restored intertidal marshes or mudflats. This may be due to lack of technical knowledge, inadequate funding for the scheme, inadequate financial compensation to landowners (Ledoux et al., 2005), and lack of public consensus (Myatt-Bell et al., 2002; Ledoux et al., 2005; French, 2006). Elliott et al. (2006) argue that agreement will be achieved in implementation of successful marine environment management schemes when they are perceived to be environmentally sustainable, economically viable, technologically feasible, socially desirable, administratively achievable, legally permissible and politically expedient. All these issues need to be addressed alongside the establishment of adequate environmental objectives for the sustainable management of estuarine ecosystems.

Appendix 1

Description of each transect with the dates of surveys and OS grid reference for locations of the first and last sampling stations in the field survey conducted along the Humber estuary in September and October in 2003.

Transect No.	Descriptions (date of survey)	Starting point (OS National Grid)	Ending point (OS National Grid)	No. of stations sampled
1	Kilnsea (16/09/03)	541828, 514856	540954, 413833	5
2	Patrington Haven (25/10/03)	533425, 418321	532583, 416563	7
3	Stone Creek (10/09/03)	523105, 419160	522657, 418578	7
4	Salt End (25/09/03)	516571, 426673	516303, 426516	7
5	Brough (09/09/03)	493740, 426037	493692, 425977	7
6	Faxfleet (26/09/03)	485880, 423818	485888, 423752	7
7	Reedness (08/09/03)	479292, 423650	479319, 423737	7
8	South Ferriby (27/09/03)	498546, 421384	498466, 421455	5
9	New Holland (11/09/03)	506015, 423811	505950, 424125	7
10	Goxhill Haven (24/09/03)	511865, 425433	511717, 425613	7
11	North Killingholme Haven (13/09/03)	516416, 420421	516512, 420504	7
12	Pyewipe (15/09/03)	526190, 411281	526455, 411798	7
13	Cleethorpes (14/09/03)	531520, 407903	532385, 408486	7
14	Horse Shoe Point (12/09/03)	538200, 401979	538776, 404460	7

Appendix 2

Detailed results of multiple regression analysis of the four macrobenthic biomass categories against physical variables from three environmental components over two different spatial scales (Chapter 2, Table 2.4). The data on station scale consisted of the average values of replicates (n=94 stations), and whereas the data on transect scale consisted of the average of stations (n=14 transects).

	Model summary							VAR	Model coefficients				
	SS	df	MS	F	p	R^2	S.E.		unstandardised		standardized	t	p
									B	S.E.	Beta		
***M. balthica* (Station scale)**													
Regression	831.6	4	207.9	44.1	2.359E-20	0.65	2.17	(CONS)	-14.262	2.265		-6.30	1.1E-08
Residual	419.7	89	4.7					T-SLO	5.512	0.479	0.847	11.52	2.5E-19
Total	1251.3	93						MD	1.459	0.462	0.331	3.16	2.1E-03
								DEP-I	2.455	0.819	0.187	3.00	3.5E-03
								EXP	-1.475	0.734	-0.229	-2.01	4.8E-02
***C. edule* (Station scale)**													
Regression	305.2	3	101.7	17.0	7.609E-09	0.34	2.44	(CONS)	-0.322	0.918		-0.35	7.3E-01
Residual	537.4	90	6.0					WID	0.001	0.000	0.307	2.69	8.6E-03
Total	842.6	93						S-SLO	0.914	0.374	0.267	2.44	1.7E-02
								SIL	-0.019	0.008	-0.200	-2.24	2.7E-02
Other macrobenthos (Station scale)													
Regression	424.3	3	141.4	95.4	7.398E-28	0.75	1.22	(CONS)	-1.143	0.686		-1.66	9.9E-02
Residual	133.4	90	1.5					SAL	2.174	0.178	0.669	12.23	7.7E-21
Total	557.6	93						ORG	0.907	0.096	0.507	9.44	4.2E-15
								DEP	-0.016	0.004	-0.213	-3.87	2.1E-04
Total biomass (Station scale)													
Regression	557.7	4	139.4	79.1	1.765E-28	0.77	1.33	(CONS)	-4.821	0.689		-6.99	4.8E-10
Residual	156.9	89	1.8					SAL	2.344	0.212	0.637	11.07	2.0E-18
Total	714.6	93						ORG	0.829	0.102	0.410	8.11	2.6E-12
								S-SLO	0.918	0.182	0.291	5.03	2.5E-06
								DEP-I	2.100	0.510	0.212	4.12	8.6E-05
***M. balthica* (Transect scale)**													
Regression	145.5	3	48.5	54.2	1.719E-06	0.92	0.95	(CONS)	-12.699	1.732		-7.33	2.5E-05
Residual	8.9	10	0.9					SAL	3.889	0.451	0.871	8.63	6.0E-06
Total	154.5	13						M-DEP	0.068	0.013	0.543	5.17	4.2E-04
								S-SLO	2.072	0.625	0.408	3.31	7.8E-03
***C. edule* (Transect scale)**													
Regression	192.7	5	38.5	110.5	3.683E-07	0.98	0.59	(CONS)	25.371	2.578		9.84	9.6E-06
Residual	2.8	8	0.3					WID	0.004	0.001	0.876	7.26	8.7E-05
Total	195.5	13						MD	-5.941	0.516	-1.210	-11.51	3.0E-06
								EXP	-6.619	0.729	-0.989	-9.07	1.7E-05
								SAL	3.985	0.569	0.794	7.00	1.1E-04
								T-SLO	-4.355	1.158	-0.661	-3.76	5.5E-03
Other macrobenthos (Transect scale)													
Regression	45.2	2	22.6	40.7	8.237E-06	0.86	0.74	(CONS)	-1.628	0.964		-1.69	1.2E-01
Residual	6.1	11	0.6					SAL	2.056	0.272	0.799	7.56	1.1E-05
Total	51.3	13						ORG	1.084	0.175	0.655	6.20	6.7E-05
Total biomass (Transect scale)													
Regression	58.9	2	29.5	28.7	4.284E-05	0.81	1.01	(CONS)	-1.311	1.310		-1.00	3.4E-01
Residual	11.3	11	1.0					SAL	2.764	0.370	0.918	7.48	1.2E-05
Total	70.2	13						ORG	0.611	0.238	0.316	2.57	2.6E-02

Key: SS: sum of squares; df: degree of freedom; MS: mean square; F: F value; p: p value; S.E.: standard error; t: t value; VAR: environmental variables; CONS: constant; SAL: salinity; EXP: exposure; MD: median particle size; ORG: organic matter content; SIL: silt content; WID: beach width; T-SLO: transect slope; S-SLO; station slope; DEP; percentage tidal depth; M-DEP: median tidal depth; DEP-I: depth index.

Appendix 3

Description of each transect with the dates of surveys and OS grid reference for locations of the first and last sampling stations in the field survey conducted in an intertidal habitat at Grimsby and Cleethorpes in September 2004.

Transect No.	Descriptions (date of survey)	Starting point (OS National Grid)	Ending point (OS National Grid)	No. of stations sampled
1	Transect *a* (14/09/04)	528573, 411146	528708, 411296	9
2	Transect *b* (27/09/04)	528871, 410862	529101, 411128	9
3	Transect *c* (15/09/04)	529196, 410545	529522, 410914	9
4	Transect *d* (28/09/04)	529551, 410202	529927, 410701	9
5	Transect *e* (26/09/04)	529878, 409894	530378, 410532	9
6	Transect *f* (19/09/04)	530309, 409654	530976, 410301	9
7	Transect *g* (18/09/04)	530587, 409277	531353, 410159	9
8	Transect *h* (17/09/04)	530870, 408852	531572, 409644	9
9	Transect *i* (16/09/04)	531130, 408470	531879, 409178	9

Appendix 4

Detailed results of multiple regression analysis of the four macrobenthic biomass categories against physical variables from three environmental components over two different spatial scales (Chapter 3, Table 3.3). The data consisted of the average per station for each transect from the 2003 and 2004 survey (n=23 transects).

	Model summary								Model coefficients				
	SS	df	MS	F	p	R^2	S.E.	VAR	unstandardised		standardized	t	p
									B	S.E.	Beta		
***M. balthica* (Transect scale)**													
Regression	132.3	3	44.095	30.22	1.943E-07	0.80	1.208	(CONS)	-7.414	1.416		-5.24	4.7E-05
Residual	27.7	19	1.4591					TSL	7.382	1.014	1.372	7.28	6.6E-07
Total	160.0	22						SIL	0.044	0.008	0.572	5.22	4.9E-05
								SSL	-2.189	0.851	-0.448	-2.57	1.9E-02
***C. edule* (Transect scale)**													
Regression	322.6	4	80.66	31.626	6.411E-08	0.85	1.597	(CONS)	-45.567	9.113		-5.00	9.3E-05
Residual	45.9	18	2.5504					SIL	-0.412	0.081	-3.501	-5.11	7.3E-05
Total	368.5	22						MD	13.337	2.947	3.302	4.53	2.6E-04
								SAL	3.777	1.000	0.625	3.78	1.4E-03
								SSL	3.383	0.807	0.456	4.19	5.5E-04
Other macrobenthos (Transect scale)													
Regression	47.6	3	15.876	32.322	1.148E-07	0.81	0.701	(CONS)	1.201	0.942		1.28	2.2E-01
Residual	9.3	19	0.4912					SIL	0.036	0.005	0.787	7.78	2.5E-07
Total	57.0	22						SAL	1.584	0.245	0.667	6.48	3.3E-06
								MDP	-0.016	0.006	-0.254	-2.65	1.6E-02
Total biomass (Transect scale)													
Regression	63.2	3	21.083	15.5	2.43E-05	0.66	1.166	(CONS)	-4.447	2.005		-2.22	3.9E-02
Residual	25.8	19	1.3602					SSL	1.515	0.563	0.416	2.69	1.5E-02
Total	89.1	22						SAL	1.887	0.505	0.636	3.74	1.4E-03
								MD	0.872	0.281	0.439	3.10	5.9E-03

Key: SS: sum of squares; df: degree of freedom; MS: mean square; F: F value; p: p value; S.E.: standard error; t: t value; VAR: environmental variables; CONS: constant; SAL: salinity; EXP: exposure; MD: median particle size; ORG: organic matter content; SIL: silt content; WID: beach width; TSL: transect slope; SSL; station slope; DEP; percentage tidal depth; MDP: median tidal depth; DPI: depth index.

Appendix 5 (1)

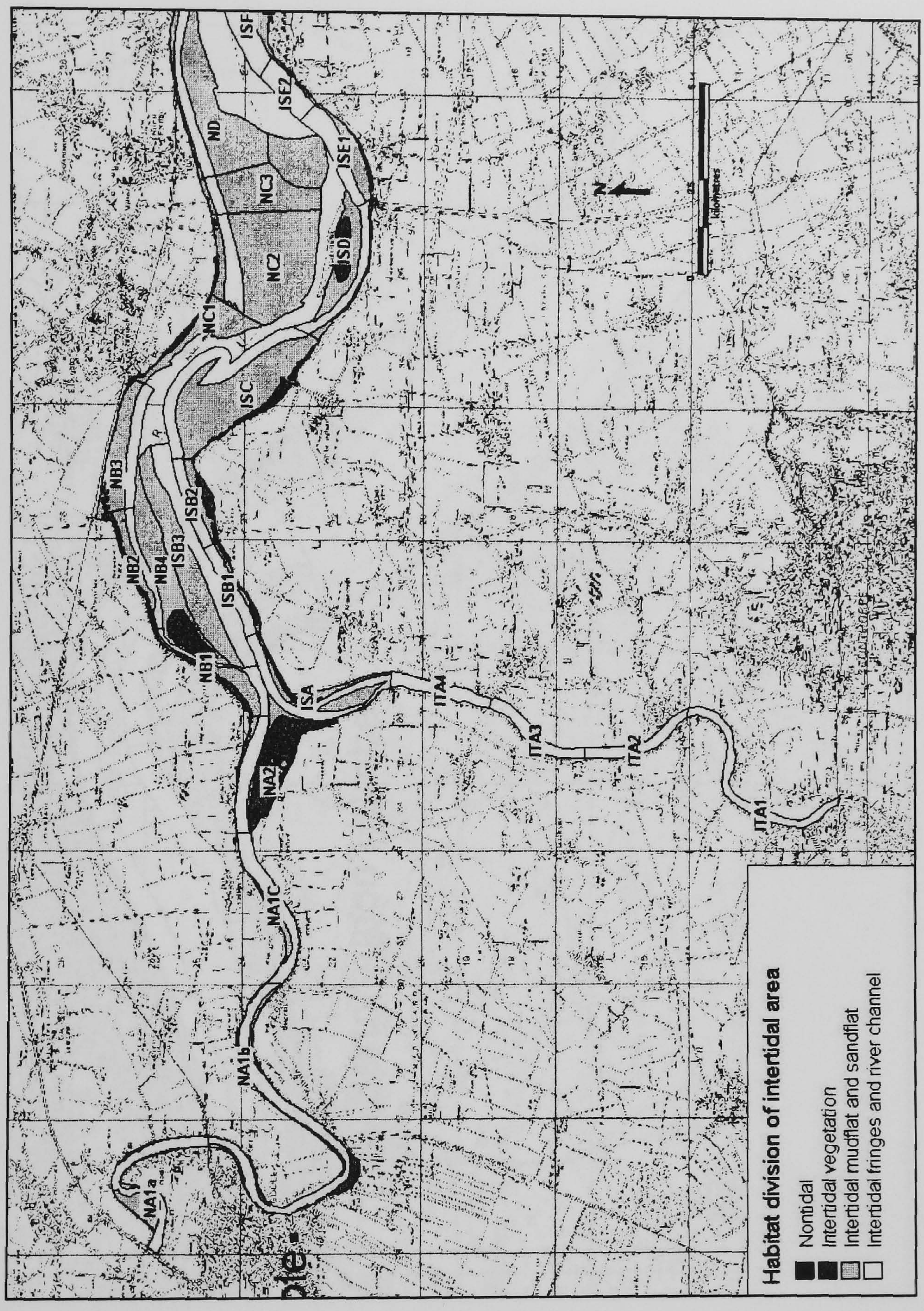


Fig. A-1. Inner Estuary sectors and sub-sectors used in the Humber Estuary Low Tide Count Programme 2003 – 2004 (after English Nature, 2005)

Appendix 5 (2)

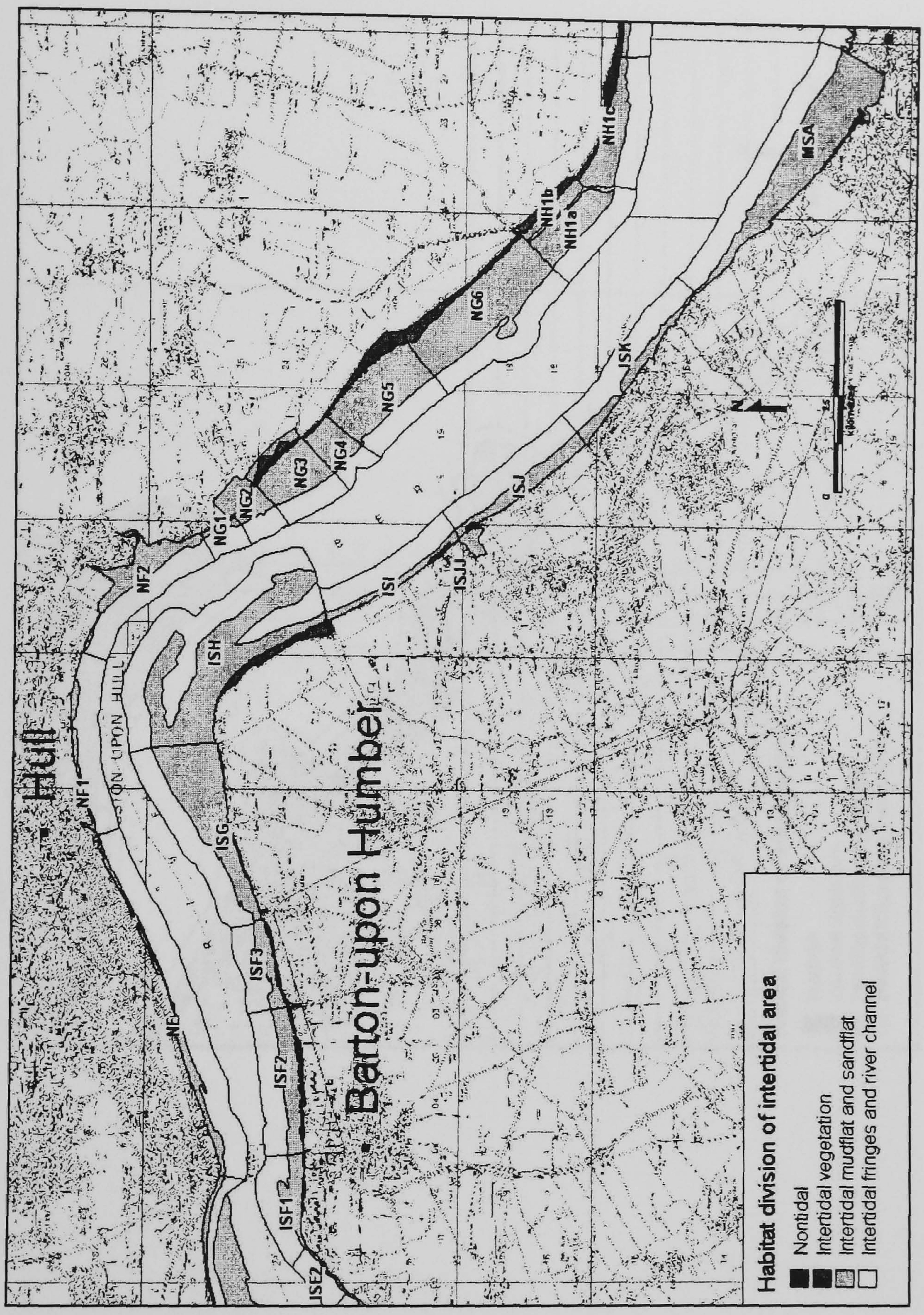


Fig. A-2. Middle Estuary sectors and sub-sectors used in the Humber Estuary Low Tide Count Programme 2003 – 2004 (after English Nature, 2005)

Appendix 5 (3)

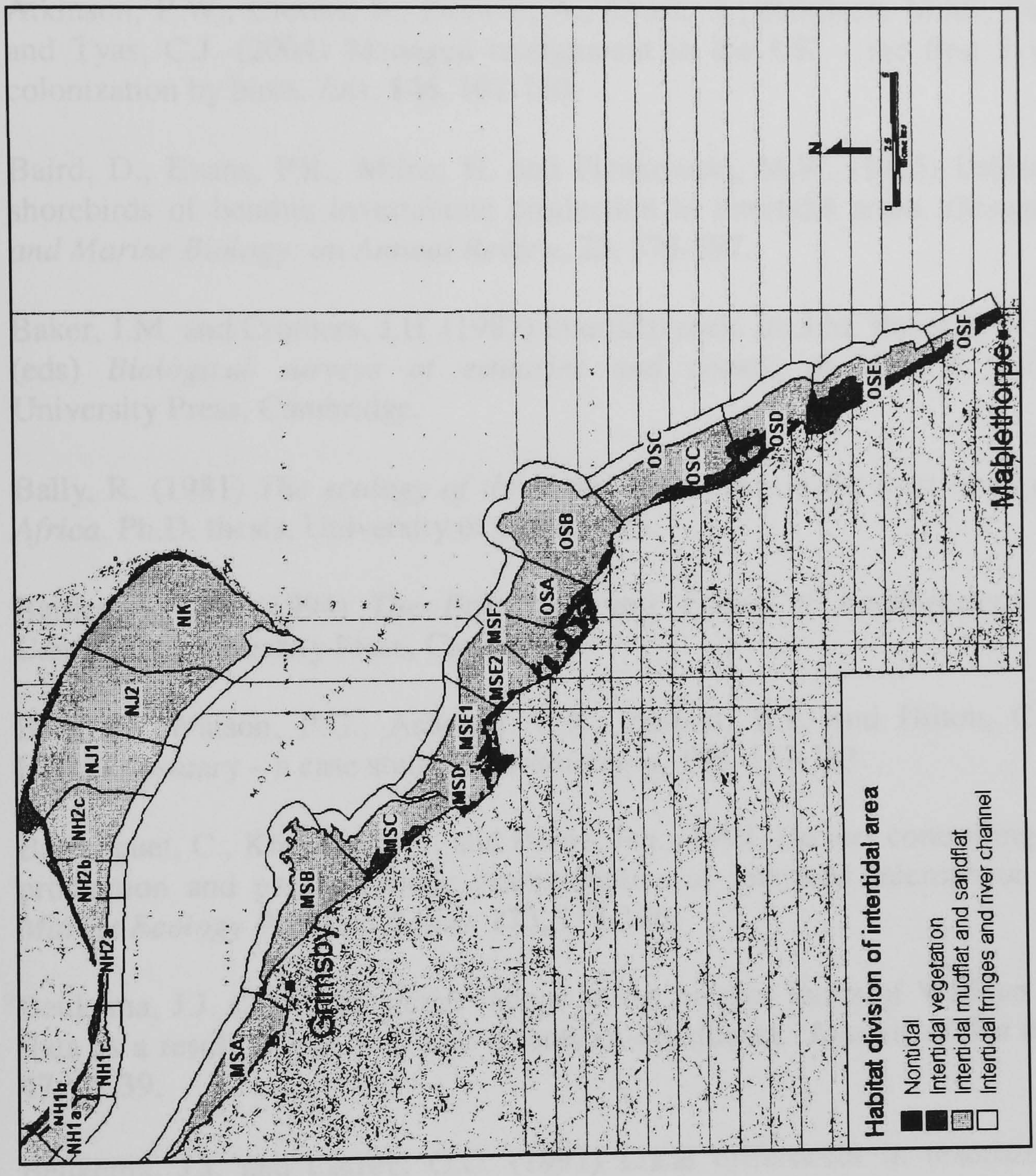


Fig. A-3. Outer Estuary sectors and sub-sectors used in the Humber Estuary Low Tide Count Programme 2003 – 2004 (after English Nature, 2005)

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