Coastal Zone Topics:
Process, Ecology & Management

1. The Changing Coastline

Edited by N.V. Jones

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Preface

The 20th scientific meeting of the Estuarine and Coastal Sciences Association (ECSA) was held at the University of Hull, UK, on 24–28 September 1990. Its theme was "The Changing Coastline". This was a most appropriate subject for a meeting held on the east coast of England where the coastline has been, and is, changing for both natural and anthropogenic reasons.

The programme included sessions on:

- Sea-level rise
- Erosion and deposition
- Biological changes
- Effects of coastal and estuarine barrages
- Conservation and pollution.

It had been planned that the proceedings of the meeting would be published under an arrangement with a publisher in continental Europe. Unfortunately, before negotiations were complete changes in the world economic situation, and in particular changes in exchange rates, made the project not viable. This happened at a time when manuscripts had reached a state almost ready for publication. ECSA Council decided that, because of their responsibility to authors and referees and the widespread interest in the subject, an alternative way of completing the project should be sought.

ECSA Council is, therefore, very pleased that the Joint Nature Conservation Committee (JNCC) has agreed to publish the volume as the first in the new joint (with ECSA) series *Coastal Zone Topics: Process, Ecology & Management.*

It was inevitable that during the gestation period of the volume the manuscript has diverged from simply being the proceedings of the meeting from which it began. Some authors have published papers elsewhere and others were unable to provide revised texts within the time scale. This volume is, therefore, a compilation of papers arising from ECSA 20 that have been modified following comments by referees as well as in some cases being updated to 1993. It is hoped that the volume will provide a useful compilation of information and opinions on this important and multifaceted subject.

Acknowledgements

ECSA 20 was organised by John Pethick, who also undertook some of the early editorial tasks. Both he and Clare Brickle are thanked for their work in the early stages of this project.

The work of all those who refereed papers is very much appreciated, particularly in these days when the arrival of unsolicited manuscripts means that another job has to be pushed down the priority list in order to respond within time limits.

ECSA Council records particular thanks to the authors of the papers which appear in this volume for their forbearance and willingness to respond to requests for revisions. ECSA Council also thanks all those who contributed to a stimulating meeting but whose papers have not, for various reasons, been included in this volume.

N.V. Jones

Sea-level and coastal evolution: Holocene analogues for future changes

I. Shennan

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Variations in the rate of sea-level change, for timescales ranging from the Holocene Epoch to the most recent decades, are difficult to quantify precisely due to the noisy signal and the resolution of the methods of data collection and analysis. Coastal systems show a range of responses to changes in sea-level, with various states of equilibrium separated by periods of rapid change. Non-linear responses to the rate of sea-level rise require further consideration in studies of the impacts of future changes in sea-level.

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Introduction

At present there is much debate and concern regarding future changes in sea-level induced by the enhanced 'greenhouse effect', with the most recent studies showing estimates of rates of sea-level rise by the end of the next century in the order of 7 mm/year, significantly in excess of those for the present century revealed by analyses of tide gauge data, 1 to 2 mm/year (Houghton *et al.* 1990). In turn, the present rates are themselves greater than the longer term averages for the late Holocene, 0 to 1 mm/year. All of the rates quoted are given without inclusion of local rates of crustal deformation, which for the United Kingdom currently varies between extremes of +2 to -2 mm/year (Shennan 1989). The different rates must be considered in any investigation of the sensitivity of coastal environments to changes in sea-level and other factors.

The broad scale linkages between atmospheric carbon dioxide concentrations, climate, ice sheet volume and sea-level are easily hypothesised, although quantification and modelling is very complex. In order to attempt any predictions, ranging from scenarios of sea-level change to physical, environmental and socio-economic impacts, many aspects of complex environmental systems require explanation. Direct measurements of sea-level change and coastline responses are not comprehensive, neither spatially nor through time, and so do not normally provide an adequate base from which to extrapolate future changes. Since most predictions of future changes in sea-level (summarised in Houghton *et al.* 1990) show an increase in the rate of rise during the next century there are two factors which attract attention. Firstly, by concentrating on the absolute amount of sea-level change, rather than the rate, many important components of coastal systems may be poorly modelled. Secondly, disturbances to the system, beyond those which have occurred during the period of direct observation, may induce strongly non-linear changes or threshold responses. There remains a difficulty in integrating data from the Holocene sedimentary record, where change is measured on a coarse timescale of hundreds of years, with the interannual or decadal timescales needed for predicting future changes (Shennan *et al.* 1992). Therefore it is essential to consider the widest range of possible linkages between sea-level change and coastal response. In part this is possible through interpretation of the Holocene record, but the integration of the various data is not easy. During the Holocene coastal lowlands have expanded and contracted over large areas with sea-level change as one, but not the only, important controlling variable. The rates and magnitudes of such changes will give some indication of the range of possible future changes. Relevant data concern all spatial scales from global to site-specific, and temporal scales from thousands of years to seasonal, or even shorter (e.g. Woodworth 1987; Shennan 1989).

Methods

Most studies of sea-level change are essentially inductive, depending on inferences and reasonings, where extrapolations are made to develop generalisations. Too often the method of multiple working hypotheses has been ignored, leading to the possible result of a theory into which all subsequent observations are fitted (see Orford 1987 and Shennan 1987a for discussion of coastal examples). The multiple working hypotheses must be formulated in such a fashion that they can be tested. Untestable hypotheses are of little use in advancing knowledge. Furthermore, progress is made by the rejection of hypotheses which have been falsified. It is erroneous to attempt to proceed by the verification of hypotheses, a situation which is commonly implicit in most studies of local sea-level change where no hypotheses are stated.

It is also valuable to reiterate some general concepts regarding change within geomorphic systems, since observation of change is dependent on the timescale, the area and the data under evaluation. These are illustrated with reference to Figure 1 (after Chorley & Kennedy 1971), with examples applicable to the model of Holocene coastal sedimentary environments in the Fenland (see Shennan 1986b).

Static equilibrium (Figure 1A)

Certain properties of the system remain stable over the time period observed.

Stable equilibrium (Figure 1B)

Systems return towards the previous equilibrium condition following a disturbance to the system, with different modes of recovery (linear, with different lag times; curvilinear; oscillatory) depending on the nature of the negative feedback mechanism. An example is the rate of peat growth in an estuarine environment and the result of a disturbance such as the temporary lowering (or raising) of the watertable. Depending on the resolution of the empirical data available this disturbance may or may not be recorded.

Unstable equilibrium (Figure 1C)

The system crosses a threshold, which is a condition marking the transition from one state of operation to another. If negative feedback mechanisms exist after the threshold (i) a second equilibrium state will exist (e.g. a rise in watertable causes the change from the accumulation of fen carr peat to reedswamp peat). If positive feedback mechanisms exist after the threshold (ii) the system will eventually decay (e.g. organic sediment accumulation comes to an end).

Metastable equilibrium (Figure 1D)

The new stable equilibrium only occurs after the larger disturbance (e.g. a small rise in watertable has no long term effect on fen carr peat accumulation whereas a larger rise causes the change to reed swamp peat).

Steady state equilibrium (Figure 1E)

Properties are invariant with reference to a given timescale but within that period their properties may oscillate (e.g. the rate of sediment accumulation may vary seasonally or annually but appears constant over a longer timescale).

Dynamic equilibrium (Figure 1F)

During period 'X' the fluctuations are so great that the trend of the system is masked and observations over too short a part of the sequence give an impression of steady state equilibrium. With a constantly changing energy environment this is seldom achieved and dynamic equilibrium is probably more realistic. For this reason a short period of direct observations (e.g. annual measurements of sediment accretion; tide gauge measurements) should not be extrapolated to give longer term trends.

Dynamic metastable equilibrium (Figure 1G)

Thresholds allow occasionally great fluctuations to initiate a new regime of dynamic equilibrium. For example the disturbance at '1' is followed by negative feedback and the system recovers (e.g. peat accumulation continues although the overall rate is decreasing), but a second disturbance of the same initial magnitude causes a threshold to be crossed because of the gradual change of the system between '1' and '2'. A second example is the impact on migrating sand barriers of storm surges superimposed on a longer term rise in mean sea-level, whereby some storm surges cause a temporary realignment of the barrier, but eventually, in combination with continued sea-level rise, a storm surge of similar magnitude to the previous ones causes a breakdown of the barrier system.

Past changes in sea-level

Oscillations of sea-level can be identified over a wide range of scales, and at each combination possible thresholds and/or rapid feedback mechanisms can be identified.

At the global scale, the deep ocean sediment oxygen isotope record reveals relatively long interglacial, interstadial and glacial periods of steady state equilibrium, separated by short phases of rapid change. For the transition between the Late Devensian and Holocene steady state equilibria, a number of positive feedback mechanisms have been proposed whereby small changes to the climate system (i.e. a small increase in summer insolation and a small decrease in winter insolation) led to the rapid disintegration of sea-based ice sheets and subsequent rise in sea-level (e.g. Ruddiman and McIntyre 1981).

At time scales more comparable to the present and near future, variations in the rate of sea- level change during the Holocene offer a significant insight into the range of coastal responses. Net accretion of organic and clastic sediments for long periods of the Holocene are

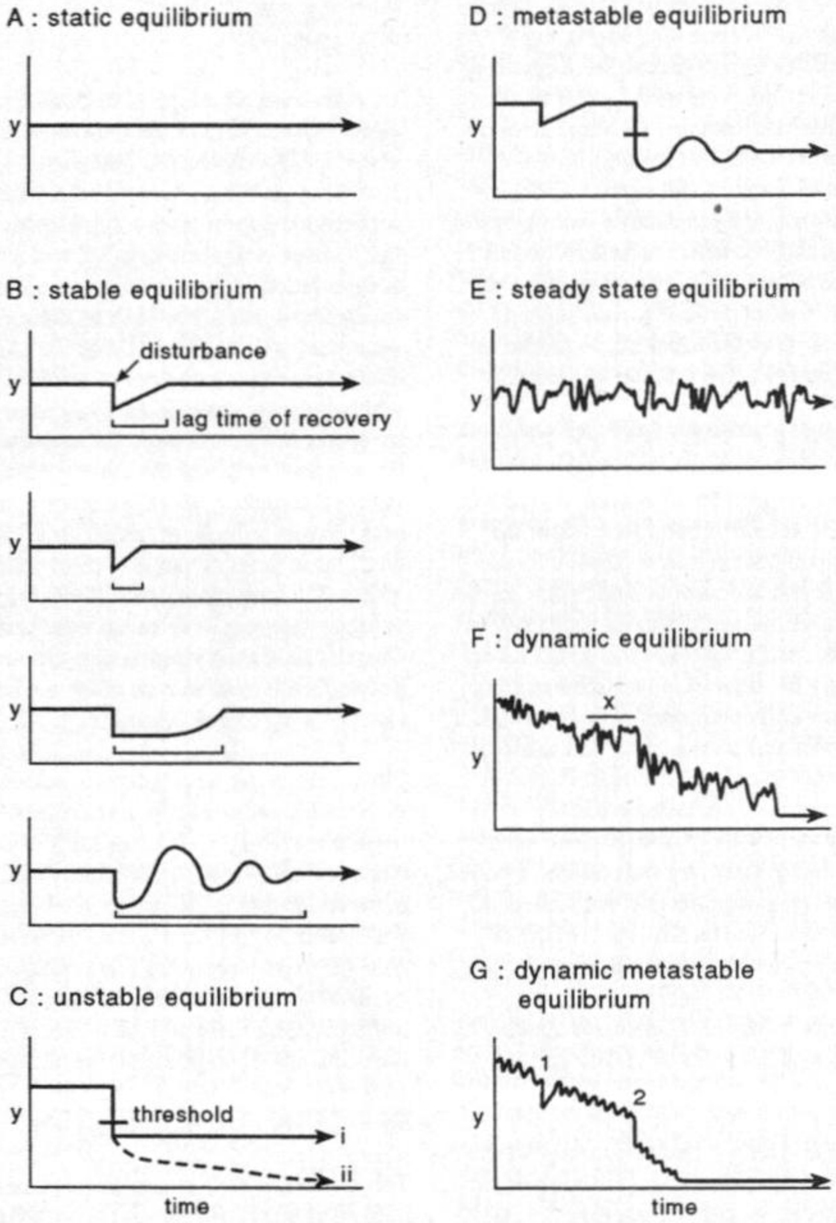

Figure 1 Examples of equilibrium within environmental systems (after Chorley & Kennedy 1971)

characteristic of many coastal lowlands. By applying a standard method of data evaluation, locally and regionally significant sea-level movements may be identified, since the bio-, litho- and chrono-stratigraphic evidence of changes in the rate of sea-level movements follow relatively predictable paths. For example, Haggart (1987) presents such data from sites around the Moray Firth where isostatic uplift has had a great influence on the local relative sea-level history, and rises and falls in relative sea-level can be quantified. The evidence from the Moray Firth can be compared to similar analyses from north-west England (e.g. Tooley 1978), north-east England (Plater & Shennan 1992; Shennan 1992) and eastern England (e.g. Shennan 1986a). Further away from the centres of glacio-isostatic uplift clear evidence of negative sea- level change (relative fall) is harder to identify, but variations in the rate of change can be reflected by the lithostratigraphic and biostratigraphic data. These are directly comparable with rises and falls in sea-level when differential crustal movements are incorporated.

The importance of identifying changes in the rate of sea-level movement is that they may control the advance and retreat of the coastline. Prior to drainage and protection by society the coastal lowlands were extremely susceptible to changes in water level, not only of the sea surface but of ground water and river discharge as well. Altitudinal changes of a few centimetres would result in the landward or seaward shift of the major sedimentary environments. The landward movement is recorded by a transgressive overlap and the seaward movement by regressive overlap. Sea-level change need not be the major controlling variable of this process. Other important and interacting variables included the morphology and angle of the slope over which the sea-level rises or falls; the textural nature and volume of sediment available; the tidal and wave climate; and the terrestrial and fluvial input of sediment and water. Indeed the ages of transgressive and regressive overlaps in the sedimentary basins such as the Fenland vary spatially. Contemporaneously some sites appear in static equilibrium, other in metastable equilibrium and others show change, cf. dynamic metastable equilibrium (Shennan 1986b; Carter *et al.* 1992; Plater & Shennan 1992).

The Holocene chronology is usually established from radiocarbon dates on the contacts of organic layers with clastic beds. Either the scientific requirements or the resource limitations have meant that multiple samples through the organic layers are rarely dated. Thus the rate of organic sediment accumulation is calculated only as an average from dates on the base and top of the layer, even though there may be other evidence for variation. Rates so calculated are underestimates of the original rate due to post-depositional compaction, and this must account for some of the observed differences between beds. Information on the rate of clastic sediment accumulation is even less easily obtained, since such beds are only inferentially dated, by radiocarbon assays on underlying or overlying organic sediments. Recent developments in luminescence techniques (Bailiff & Poolton 1989) will offer new information.

Where there are a series of dated layers there appears to be a difference in the average rates of clastic and biogenic sediment accumulation (e.g. Table 1).

The mean rates given in Table 1 do not account for the counting errors of the radiocarbon data; for example at Goldcliff such allowances would suggest that the clastic sedimentation rate was similar to much of the peat but there was a major change within one of the peats (Smith & Morgan 1990).

By extracting all the pairs of dates from sites within Great Britain, held in the radiocarbon databank set up under IGCP Projects 61, 200 and 274, further relationships can be investigated. Three subsets of data were collected, for rates of basal peat accumulation, intercalated peat accumulation and for clastic sediment accumulation. All rates are given for uncalibrated radiocarbon dates. For each of these classes the data were positively skewed (Table 2). The data sets are also characterised by a number of outliers, in particular where the rate was calculated from two dates separated by only a few centimetres. In such cases allowance for the standard errors of the dates would produce a wide range of estimates of sedimentation rate. Therefore each data set was reduced to lessen such effects by restricting analyses to pairs of samples more than 0.1m apart and 100 to 700 year mean age difference, the upper limit to reduce averaging over an extreme period (Table 3). There is no simple relationship between elapsed time and sediment thickness or mean sediment accumulation rate and age (Figure 2A and 2B).

These results can be related to a number of hypotheses. If differential sediment compaction is not considered important the higher average rates of accumulation for clastic layers (Table 3) may imply a threshold, e.g. between 1.4 and 2.1 mm/year, at which an environment in which both organic and clastic sedimentation can proceed changes to one where organic sedimentation is replaced by clastic sedimentation. If differential compaction is important then there may be no significant difference in the rates, and by implication the change from one type of sediment accumulation to the other is more finely balanced, and variable from site to site.

The mean rates may be misleading and the pattern of outliers more significant. In the whole dataset above 3 mm/year there are twelve cases for clastic sediments,

Table 1 Mean sediment accumulation rates for two Holocene sequences

site	*age*	*thickness metre*	*rate mm/year*	*sediment type*
Adventurers' Land	4500–4180	0.21	0.66	peat
Goldcliff	5950–5530	0.28	0.67	*Phragmites*/wood peat
Goldcliff	5360–5020	0.25	0.74	*Phragmites*/*Cladium* peat
Goldcliff	5020–3440	0.77	0.49	*Sphagnum* peat
Goldcliff	5360–3130	1.10	0.49	average for peat bed
Adventurers' Land	4180–1845	3.38	1.45	clastic
Adventurers' Land	5840–4500	2.68	2.00	clastic
Adventurers' Land	6575–5840	0.93	1.27	clastic
Goldcliff	5530–5360	0.18	1.06	clastic

Data from Shennan 1986a, Smith & Morgan 1990.

Table 2 Summary statistics for rates of sediment accumulation (mm/uncalibrated radiocarbon year), full data set

	basal peat	*intercalated peat*	*clastic layer*
number of samples	35	93	63
mean	1.06	0.77	1.81
median	0.78	0.63	1.09
lower quartile	0.34	0.32	0.58
upper quartile	1.39	1.08	2.45
skewness	1.12	2.02	2.02

Data from radiocarbon database collated by I. Shennan.

Table 3 Summary statistics of the subset of data for rates of sediment accumulation (mm/uncalibrated radiocarbon year), subset of data, see text

	basal peat	*intercalated peat*	*clastic layer*
number of samples	14	40	34
mean	1.36	0.92	2.12
median	1.10	0.74	1.17
lower quartile	0.65	0.42	0.69
upper quartile	1.55	1.14	2.85
skewness	1.08	2.28	1.76

Data from radiocarbon database collated by I. Shennan.

but only two and one from basal and intercalated peats respectively. This could be a better estimate of the threshold at which organic sedimentation is most likely to cease. The thickest clastic beds show average rates greater than 3 mm/year (Figure 2C). Further analyses are required but a suggestion is that in most cases there is a delicate balance between clastic and organic sedimentation when the rate of sedimentation is controlled by sea-level change up to 3 mm/year. Only in isolated cases does more rapid, clastic sedimentation occur. Quantification of compaction will increase the rates given.

Mörner (1980) proposed a model of regional eustatic sea-level change with oscillations of sea-level greater than 1 metre, which is more than most other authors imply. However this model can be used to explore further the effects of changing rates of sea-level, since maximum likely rates, and the possible environmental responses, can be investigated. An alternative to analysing this evidence for changes in the rate of sea-level movement is to apply a model of a smooth Holocene sea-level curve. This second model offers little opportunity for testing any hypotheses of the link between sea-level and changes in the coastal zone. It suggests that sea-level is not a causative factor, raising the question of why there should be so much concern over future sea-level change, rather than other factors resulting in major changes in the coastal zone.

Shennan (1987b) used the regional eustatic curve to model the relationship between relative sea-level change, crustal subsidence and coastal environmental change. From this original analysis it was suggested that for the Fenland/Wash estuary a rate of approximately 5 mm/year sea-level rise may be the general threshold at which coastline retreat would commence (Shennan 1987c). More recent work (Waller 1988; Shennan 1994) has shown that the pattern of coastline advance and retreat is more complex, although there remains relatively good agreement between the regional eustasy model and coastline response, given the limitations of the radiocarbon chronology. Major phases of regressive overlap development occur during periods of predicted negative or low rates of sea-level rise, while above 5 mm/year there is no major regressive overlap development. Slight variations to the model such as reducing the amplitude of the oscillations, more in line with other opinions, produce an even better fit with the empirical data (Shennan 1987b). For the periods of sea-level rise of less than 5 mm/year the interplay of other factors appears to affect the coastal response significantly, but above 5 mm/year there is usually a halt of the seaward expansion of peat-forming communities.

Initially the rates of change were calculated as a 10 year average (in Shennan 1978b). The mean error on radiocarbon dates in the data bank is ± 100 radiocarbon years, although more recently dated peat samples are usually ± 50 radiocarbon years. A 100 year filter maintains the same general pattern, a 200 year filter reduces the amplitude of many of the shorter duration maxima and minima, but a 300 year filter significantly alters the pattern of oscillations (Figure 3), particularly at c. 6200, 4500 and 2300 BP. This 200 to 300 year window is an important consideration for the Holocene evidence. It can be hypothesised that at these periods a poor chronological correlation can be expected. Alternatively, periods for which there is poor agreement may be times of rapid changes in the rate of sea-level movement which will go unidentified with simple reference to the radiocarbon data but could be indicated by detailed stratigraphic analysis. Probable limits to the rates involved can then be estimated for any adequately formulated hypothesis.

Present changes in sea-level

These data have been reviewed and analysed by a number of authors in recent years (see Houghton *et al.* 1990, Tooley & Jelgersma 1992). There remains much debate over the rate at which global sea-level is changing, due to the difficulty of filtering out the effect of regional and local crustal movements and other problems associated with the distribution of tide gauges and the length of the records. Woodworth (1990) concludes that in general there is no evidence for mean sea-level accelerations significantly different from zero over the time period 1870 to the present for European

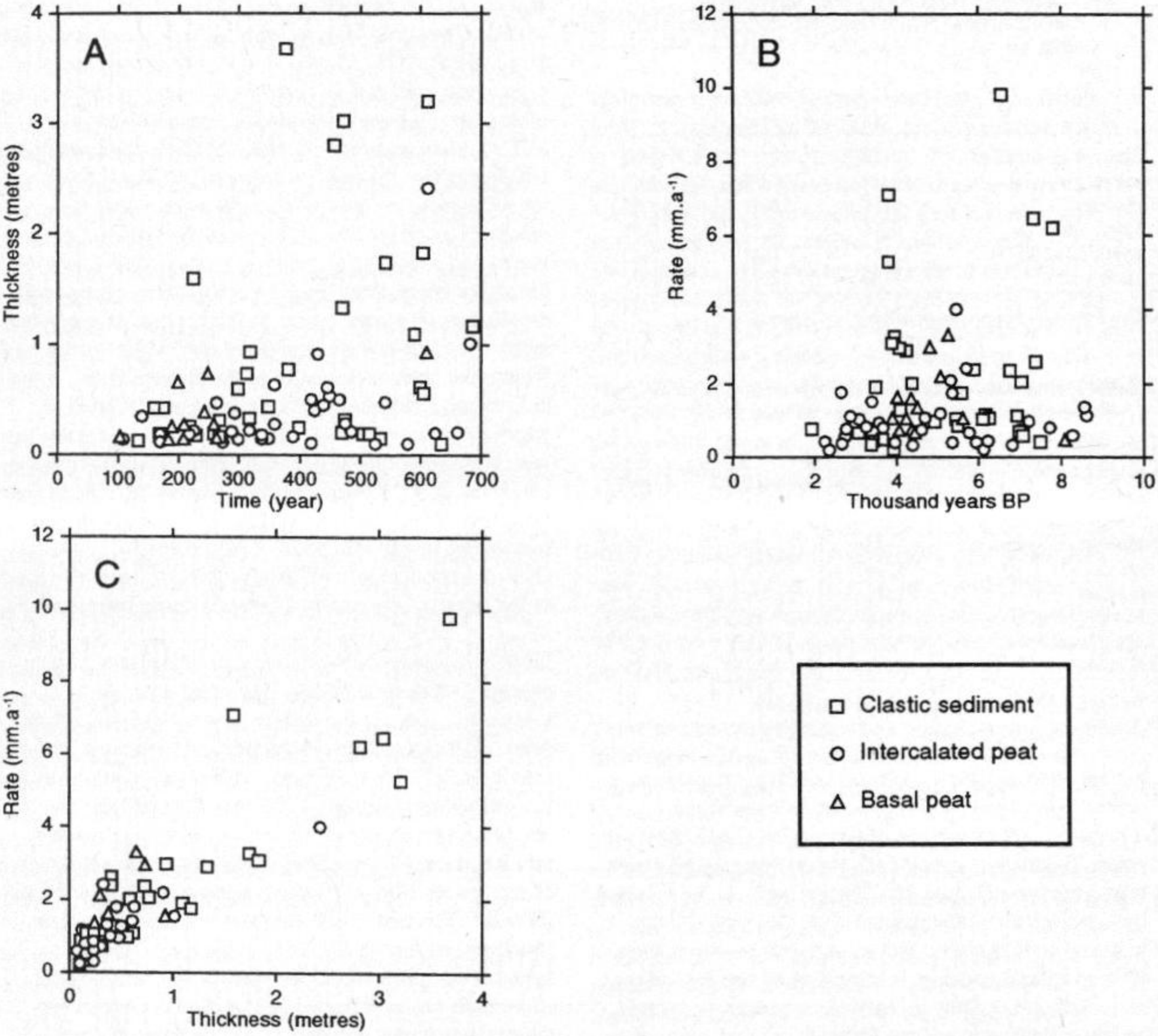

Figure 2 Holocene peat and clastic sediment accumulation, 88 cases from coastal lowlands of the United Kingdom. A, time interval (^{14}C yr) of accumulation and thickness of sediment interval. B, mean sediment accumulation rate and age of sediment (^{14}C yr BP). C, mean sediment accumulation rate and thickness of sediment interval.

tide gauges. During the same period a trend of 1–2 mm/year for global mean sea-level change is evident (e.g. Gornitz & Lebedeff 1987; Shennan & Woodworth 1992), although there have been figures suggested either side of this range (e.g. Pirazzoli 1986; Peltier & Tushingham 1989; Stewart 1989).

There are significant changes in sea-level over decadal and even shorter periods (Pugh & Faull 1982). These result in different long-term trend estimates as different time-envelopes are selected and should also be taken into account when direct observations of estuarine sediment accumulation are analysed. These may well be poor estimates of long term trends. In addition, where the coastline has been significantly altered by society, for example due to reclamation of saltmarshes (Pethick 1981; Allen 1990; and Figure 1), the time lag between disturbance and recovery may be much longer than any period of observation (e.g. Hill 1986). Thus between-site variation and the absence of any clear trend should be expected. A longer time perspective is required to give an understanding of the full range of relationships between sea-level change and coastal response.

Future sea-level rise

The sea-level scenarios published in Barth and Titus (1984) were characteristic of the predictions being made in the early to mid 1980s. More radical values had also been computed but were deemed to be less likely. Recently there has been a convergence of opinion, from research

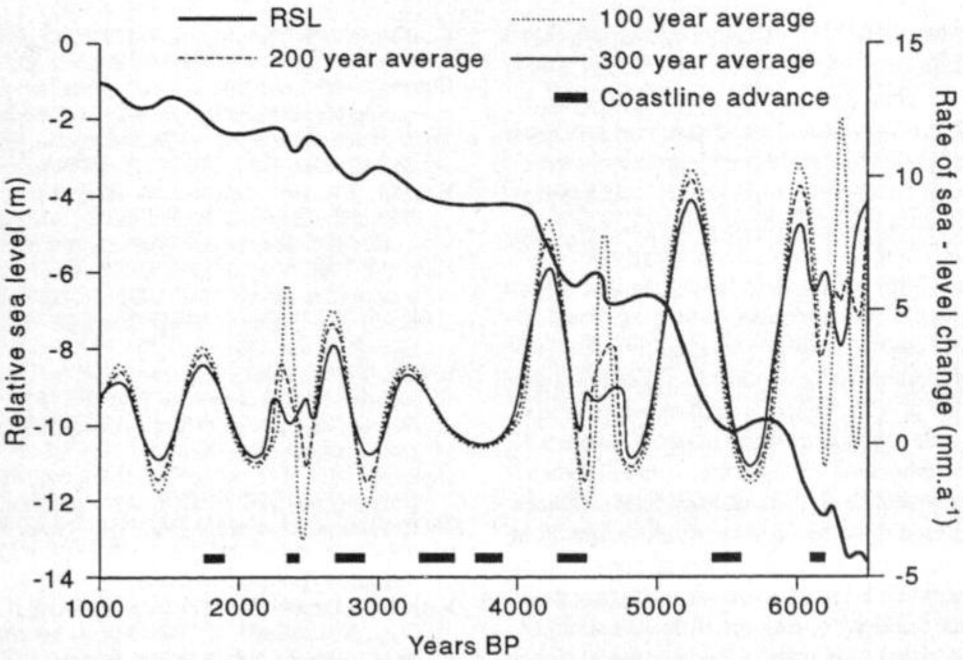

Figure 3 Model of relative sea-level changes, rates of sea-level change (expressed as 100, 200 and 300 ^{14}C year running means) and periods of coastline advance seaward in the Fenland, 6500 to 1500 BP

workers using complex computer models, centred around lower figures. The Intergovernmental Panel on Climate Change (Houghton *et al.* 1990) give a best estimate of 0.60–0.70 metres global average rise by AD 2100 for the 'business as usual' emissions of greenhouse gases, with high and low estimates of c. 1.1 and 0.3 metres. The rates of sea-level rise during the period AD 1990 to 2100 increase with time, and around the middle of the twenty-first century are c. 3.0, 5.5 and 10 mm/year for the low, best and high estimates respectively.

Alternative opinions have been expressed by scientists addressing the topic from different directions to the modellers, including analyses of a range of empirical data sets and re-examination of models of environmental processes (e.g. Idso 1989).

There is decisive evidence for a rise in the level of carbon dioxide in the atmosphere (e.g. Rycroft 1982). Global climate models simulate an associated rise in global air temperatures (increasingly the models are being refined to reveal significant regional variations in temperature response), but there are scientists questioning the empirical evidence for a simple trend or significant cycles in atmospheric temperature, suggesting that prediction will be limited (e.g. Harris 1985). Climate, and also tide gauge (see previous section), data are statistically noisy and if the objective is to relate the response in climate to an increase in atmospheric carbon dioxide levels then local effects must be removed from the data. For each station the local effects included changes in any of the following: the instruments or their location; the observation time; the computational procedures used on the data; and most importantly, changes which have taken place in the surroundings of the station such as building construction, grass replaced by concrete or tarmac and growth of trees (the so-called urban effect). If the urban effect is removed, it has been argued, changes in surface air temperature in the last 100 years are biased (e.g. Wood 1988; Karl *et al.* 1989).

It should also be noted, based on empirical evidence, that there may be the possibility of a very rapid rise in sea-level, decimetres in a few weeks or even metres in a few years. However, the explanations of the causative processes, the accumulation and catastrophic release of subglacial meltwater, have yet to be tested adequately (Shaw 1989; Tooley 1989). Within the Holocene record there is also evidence of extreme water levels attributed to a tsunami induced by massive slumping of sediment from the continental slope (Long *et al.* 1989). Such events may indicate the most extreme, but not the most probable, future scenarios.

Conclusions

Analysis of Holocene sea-level data suggest that there is no reason to suppose that coastal zone response to sea-level change need be linear. Though it is not possible to identify accurately all thresholds some can be suggested for some environments. The balance between organic and clastic sedimentation, transgressive and regressive overlap development, is delicate with sea-level change less than 3–5 mm/year. But above 5 mm/year sea-level change appears to induce widespread coastline retreat within estuaries. Other factors interacting with sea-level change to influence coastal evolution have been frequently stated but are rarely unequivocally proven and quantified. With reference to future changes it must be noted that the period of direct observation coincides with significant modification of the coast by society. The ability of marshes to keep pace with recent sea-level rise may be due to disturbances and imbalances to the system. Any time lag, e.g. the time to attain marsh 'maturity', in the system will obscure potential future problems. Construction of a sea embankment produces an obvious threshold, delaying what would normally be a gradual change. Thus the observation of current rates of intertidal sedimentation up to an order of magnitude greater than the majority during the Holocene should ensure caution rather than complacency in the approach to any future sea-level rise. Non-linear changes should be expected, particularly if rates approach 3–5 mm/year, although changes have occurred at lower rates and the system is most likely not in equilibrium. All possible thresholds and non-linear changes should be identified and their ranges quantified; they must then be presented without being alarmist.

Acknowledgements

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References

Allen, J.R.L. 1990. Constraints on measurement of sea-level movements from salt-marsh accretion rates. *Journal of the Geological Society, London, 147*: 5–7.

Bailiff, I.K., & Poolton, N.R.J. 1989. Charge transfer in quartz and feldspar; factors affecting the dating of young sediments. Extended abstract in *Proceedings of Specialist Workshop on young and old limits of dating sediments*, Occasional Publication No. 6, Research Laboratory for Archaeology and the History of Art, Oxford University, April 1989.

Barth, M.C. & Titus, J.G., *eds.* 1984. *Greenhouse Effect and Sea-level Rise: a challenge for future generations.* New York, Van Nostrand Reinhold.

Carter, R.W.G., Orford, J.D., Jennings, S.C., Shaw, J., & Smith, J.P. 1992. Recent evolution of a paraglacial estuary under conditions of rapid sea-level rise: Chessetcook Inlet, Nova Scotia. *Proceedings of the Geologists' Association, 103*: 167–185.

Chorley, R.J., & Kennedy, B.A. 1971. *Physical Geography: a systems approach.* London, Prentice-Hall.

Gornitz, V., & Lebedeff, S. 1987. Global sea-level changes during the past century. *In*: *Sea-level fluctuation and coastal evolution,* ed. by D. Nummedal, O.H. Pilkey & J.D. Howard, 3–16. SEPM Special Publication 41.

Haggart, B.A. 1987. Relative sea-level changes in the Moray Firth area, Scotland. *In*: *Sea-Level Changes* ed. by M.J. Tooley & I. Shennan, 66–108. Oxford, Blackwell.

Harris, R. 1985. Variations in the Durham rainfall and temperature record, 1847-1981. *In*: *The Climatic Scene,* ed. by M.J. Tooley & G.M. Sheail, 39–59. London, George, Allen & Unwin.

Hill, M. 1986. *Saltmarsh vegetation of the Wash. An assessment of change from 1971 to 1985.* Peterborough, Nature Conservancy Council (Research and survey in nature conservation, No. 13).

Houghton, J.T., Jenkins, G.J., & Ephraums, J.J. (eds.) 1990. *Climate Change: The IPCC Scientific Assessment.* Cambridge, Cambridge University Press.

Idso, S. 1989. *Carbon Dioxide and Global Change; Earth in transition.* Tempe, IBR Press.

Karl, T.R., Tarpley, D., Quayle, R.G., Diaz, H.F., Robinson, D.A., & Bradley, R.S. 1989. The recent climate record: what it can and cannot tell us. *Reviews of Geophysics, 27*: 405–430.

Long, D., Smith, D.E., & Dawson, A.G. 1989. A Holocene tsunami in eastern Scotland. *Journal of Quaternary Science, 4*: 61–66.

Mörner, N. A. 1980. The north-west European 'sea-level laboratory' and regional Holocene eustasy. *Palaeogeography, Palaeclimatology, Palaeoecology, 29*: 281–300.

Orford, J. 1987. Coastal processes: the coastal response to sea-level variation. *In*: *Sea Surface Studies: a global view,* ed. by R.J.N. Devoy, 415–463. London, Croom Helm.

Peltier, W.R., & Tushingham, A.M. 1989. Global sea-level rise and the greenhouse effect: might they be connected? *Science, 244*: 806–810.

Pethick, J.S. 1981. Long-term accretion rates on tidal salt marshes. *Journal of Sedimentary Petrology, 51*: 571–577.

Pirazzoli, P.A. 1986. Secular trends of relative sea-level (RSL) changes indicated by tide gauge records. *Journal of Coastal Research, Special Issue 1*: 1–26.

Plater, A.J., & Shennan, I. 1992. Evidence of Holocene sea-level change from the Northumberland coast, eastern England. *Proceedings of the Geologists' Association, 103*: 201–216.

Pugh, D.T., & Faull, H.E. 1982. Tides, surges and mean sea-level trends. *In*: *Shoreline Protection,* 45–55. Thomas Telford Ltd, London.

Ruddiman, W.F., & McIntyre, A. 1981. The mode and mechanism of the last deglaciation. *Quaternary Research, 16*: 125–134.

Rycroft, M.J. 1982. Analysing atmospheric carbon dioxide levels. *Nature, 295*: 190–191.

Shaw, J. 1989. Drumlins, subglacial meltwater floods, and ocean responses. *Geology, 17*: 853–856.

Shennan, I. 1986a. Flandrian sea-level changes in the Fenland, I. *Journal of Quaternary Science, 1*: 119–154.

Shennan, I. 1986b. Flandrian sea-level changes in the Fenland, II. *Journal or Quaternary Science, 1*: 155-179.

Shennan, I. 1987a. Global analysis and correlation of sea-level data. *In*: *Sea-Surface Studies,* ed. by R.J.N. Devoy, 198–230. London, Croom Helm.

Shennan, I. 1987b. Holocene sea-level changes in the North Sea. *In*: *Sea-Level Changes,* ed. by M.J. Tooley & I. Shennan, 109–151. Oxford, Blackwell.

Shennan, I. 1987c. Impacts on the Wash of sea-level rise. *In*: *The Wash and its environment,* ed. by P. Doody & B. Barnett, 77–90. Peterborough, Nature Conservancy Council (Research & Survey in Nature Conservation, No. 7).

Shennan, I. 1989. Holocene crustal movements and sea-level changes in Great Britain. *Journal of Quaternary Science, 4*: 77–89.

Shennan, I. 1992. Late Quaternary sea-level changes and crustal movements in eastern England and eastern Scotland: an assessment of models of coastal evolution. *Quaternary International, 15/16*: 161–173.

Shennan, I. 1994. Coastal evolution IV. Hypothesis testing. *East Anglian Archaeology,* Report No. 70, 81–84.

Shennan, I., & Woodworth, P.L. 1992. A comparison of late Holocene and twentieth-century sea-level trends from the UK and North Sea region. *Geophysical Journal International, 109*: 96–105.

Shennan, I., Orford, J.D., & Plater, A.J. 1992. Introduction: Quaternary coastal evolution: case studies, models and regional patterns. Final report of the U.K. Working Group, I.G.C.P. Project 274. *Proceedings of the Geologists' Association, 103*: 163–165.

Smith, A.G., & Morgan, L.A. 1990. A succession to ombrotrophic bog in the Gwent Levels, and its demise: a Welsh parallel to the peats of the Somerset Levels. *New Phytologist, 112*: 145–167.

Stewart, R.W. 1989. Sea-level rise or coastal subsidence question. *Atmosphere-Ocean, 27*: 461–477.

Tooley, M.J. 1978. *Sea-level changes in North-West England during the Flandrian Stage.* Oxford, Clarendon Press.

Tooley, M.J. 1989. Floodwaters mark sudden rise. *Nature, 342*: 20–21.

Tooley, M.J., & Jelgersma, S. (eds) 1992. *Impacts of sea-level rise on European coastal lowlands.* Oxford, Blackwell.

Waller, M. 1988. The Fenland Project's environmental program. *Antiquity, 62*: 336–343.

Wood, F.B. 1988. On the need for the validation of the Jones *et al.* temperature trends with respect to urban warming. *Climatic Change, 12*: 297–312.

Woodworth, P.L. 1990. A search for accelerations in records of European mean sea level. *International Journal of Climatology, 10*: 129–143.

Sea-level rise and the future of the British coast

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The way the sea is shaping the present coastline of Britain has been modified by the activities of man in the coastal zone. Natural coastal systems are generally able to respond to changes in sea-level with little overall change. The existence of man-made barriers can impede or prevent this natural adjustment. The impact of sea-level rise on coastal systems is more complex than that of a simple increase in mean sea-level; other aspects of climatic change also interact with the effects of rising sea-level. The problems of predicting the effects of climate change on the coast are increased by these interactions. The likely effects can best be considered in relation to the sensitivity and vulnerability of the different coastal ecosystems. This can then give us a basis on which to formulate plans for the optimum future management of the British coast.

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Introduction

Because of its importance as the interface between land and sea and, more recently, because of its general attractiveness, man has long made use of the coastal zone. This has, however, not been without considerable effects on the coast itself both physically and biologically. Firstly man's activities are generally intolerant of the natural fluctuations in the coastline as erosion or accretion hold sway. Intensive use of the coastal zone has necessitated a fixed boundary, in the form of a sea wall, defining the border between land and sea. Agriculture and forestry, and especially residential or industrial developments, all demand isolation from the dynamic coastal processes.

Natural adjustments of the coast to sea-level change

There is a natural process of adjustment of soft coasts to changes in sea-level, the so-called 'Brun hypothesis' or 'Brun-rule'. As sea-level rises erosion will increase along the seaward edge of the deposits, be they saltmarsh mud, dune sand or shingle. These eroded deposits will then be carried landwards by water or wind and redeposited. By this process the coastal systems will move landwards and upslope until they are re-formed in more or less the same position in relation to the level of the sea with apparently little overall change.

Effect of man on coastal processes

However, wherever barriers to this process of adjustment exist, whether man-made or natural, the process will be inhibited. With erosion continuing and redeposition restricted, a feature may well be reduced in area or lost completely. In general the presence of a man-made fixed structure, be it sea wall, harbour, or dam, will inhibit the natural ability to respond to sea-level rise. The coastal features outside the sea wall, whether beach, mud flat, saltmarsh or dune, will be reduced in area or even destroyed. In the south-east of Britain the land has been sinking relative to the sea at a rate of 3 mm/year for many years. Up to the present, marsh growth has largely been able to accommodate this process. In the future however it will further exacerbate the effect of rises in the level of the sea itself in the very areas that are most sensitive to changes in the relative level of land and sea. Along the east coast of Britain the coast has been made more vulnerable by the forced pace of reclamation and development (Boorman & Ranwell 1977). In past years land was only reclaimed from the sea when it was at a level at which only the highest tides reached it. In such circumstances fresh saltmarsh developed outside the new sea wall. In recent years, however, the pace of reclamation has been such that there has been a severe narrowing of the coastal zone with little development of new habitats seawards. In these circumstances the effects of sea-level rise are greatly accentuated.

Effects of climate change

It is convenient and easy to think of sea-level rise in terms of a simple addition to mean sea-level, but it is far more complicated than this. Data from the Netherlands shows that the present sea-level rise there is made up of a smaller increase in Mean Low Water and a larger increase in Mean High Water (Brouns 1988b). In other words, not only is the sea-level rising but the tidal amplitude is increasing, with consequences for man and for wildlife. Data on this point are not yet available for south-east Britain, but the situation is likely to be similar. The effects of sea-level rise are also felt through the changing patterns of erosion and accretion as the depth of water increases. There will be other more complex effects on the patterns of erosion and accretion. Changes in wind speed and direction and changes in water depth will alter the local wave climate and the patterns of currents, thus further modifying erosion and accretion (de Ronde & Vogel 1988). A major factor affecting coastal processes is the occurrence and severity of extreme events such as storms and the resultant tidal surges. It is peak water levels during severe storms that determine the necessary height of sea defences. Natural ecosystems can also be seriously affected by such events. The rate of development of coastal systems can be greatly changed, in a positive or negative way, by such events. A severe storm can mobilise vast quantities of sediment and give a year or more of normal accretion in a matter of a few hours, or it can destroy a feature built up over many years with similar speed. The pattern and frequency of severe storms and associated tidal surges is likely to change in the years to come. Other aspects of climate change will also have their role, both directly and indirectly. Temperature, rainfall and evaporation will affect the growth of coastal plants in various ways. The flow of rivers could be affected by changing patterns of rainfall inland and this would also have its effect on the coast through changing freshwater inputs into estuaries and changes in the availability of freshwater in the coastal zone. Increased levels of CO_2 will also have a direct effect on plants on the coast that trap and hold the mobile sediments of sand dunes and saltmarshes (Brouns 1988a). However the magnitude of these secondary effects is likely to be small in comparison with effects of changes in sea-level.

Problems of prediction

Over the last few years the magnitude and range of the predicted rises in sea-level have narrowed considerably although the error terms are still large. Typical of the current figures are a 33 cm rise by 2050 and a 66 cm rise by 2100, but these predictions are associated with standard deviations of 32 cm and 57 cm respectively (Oerlemans 1989). The official Dutch report *Sea defence after 1990* uses a figure of 60 cm for the years 1990 to 2090 (Anon. 1989). A recent projection for Sheerness predicts a rise of 65 cm by 2087 (Pugh 1990). However, the future mean sea-level is in some ways less important than the sea-level under extreme conditions. Possible extreme levels, under the range of weather conditions we experience at present, can be predicted. Under certain climatic scenarios, these extremes could increase over and above the predicted increases in sea-level. The important factors here include wind speed and direction – a change of 10% in wind direction and an increase of 10% in wind speed could together effectively double the maximum sea-level to be expected under severe storm conditions (de Ronde & Vogel 1988). There are great problems associated with the prediction of the future course of storm tracks. With the storms of January 1990 and October 1987 very much in British minds, together with the disastrous floods of 1953, we are tempted to think that the number of severe storms is increasing, but the evidence to support this is inconclusive, or at least it was until recently (Zwart 1984). A further factor is the rate of climatic change. In temperate zones a difference of 1 °C is equivalent to a distance of 250 km. Thus a change of 4 °C in 100 years would imply a northwards movement of climatic limits of about 10 km/year. This has considerable implications for the natural response of organisms to climatic change. Different organisms have very different rates of dispersal, and some would not be able to migrate at this speed.

Sensitivity and vulnerability

It is now time to consider the sensitivity and vulnerability of coastal systems to change. Given that there has to be an integration of the human use of the coast with natural processes, what are the key factors responsible for changes along soft coasts? Saltmarshes are composed of fine sediments transported by water and fixed by vegetation. For marsh growth there must be an overall positive accretion balance. This implies a gentle slope and low wave energies but drainage and soil aeration are also needed. The key to healthy saltmarshes lies in a vigorous pioneer zone of plant colonisers. There are fluctuations from year to year in water levels. The lower limit of the establishment of seedlings is determined by the levels of neap tides in the spring. This can vary by 0.5 metres or more from year to year. Mean water levels can also affect the growth and competition of saltmarsh species by affecting the vitality of individual species (Olff *et al.* 1988). Establishment is also affected by the depth to which sediments are reworked – the more sediments are disturbed, the deeper a seedling has to be rooted before it can withstand inundation with its potential for erosion. If marsh growth requires a positive accretion balance, what will reduce this? Effectively this could be a lack of sediment accretion. This can be the result of excess wave action, shortage of suspended sediment in the water or the lack of vegetation cover to fix the sediment. A negative accretion balance will lead to the formation of an erosion boundary in the form of a cliff at the marsh edge. However, even when there is such a negative balance in the pioneer zone, some growth, perhaps even accelerated growth, can occur in the higher marsh zones with good vegetation cover partly as a result of the sediment released by erosion elsewhere. While the existence of a cliff can indicate an eroding situation overall, there can also be cyclical changes of erosion and accretion within the system with little overall change. This was the situation regarding the marshes of the Medway, north Kent, in the 1960s (Kirby 1969). Extensive losses by various patterns of erosion were accompanied by equally spectacular gains elsewhere in the system. This is in contrast to the present situation in Essex where erosion has become the dominant process. What will happen in the future in Britain as a whole will depend on local fine balances between the sediment supply, the magnitude of the sea-level rise and the potential for saltmarsh growth.

Sand dune development will also be affected by rising sea-levels. The main effect will be increased erosion at the seaward edge with the potential, not necessarily realised, for growth in the landward areas. Dune stability and thus dune morphogenesis could also be affected by climatic effects on the vegetation. Increased evapotranspiration and decreased rainfall in the summer could reduce plant cover with a resulting increase in dune mobility. This would not necessarily reduce the value of the dune for sea defence but would increase the width of the dune zone needed for a particular degree of safety. It would also have implications for nature conservation. An increase in the dynamic activity of the dunes, as a result of climate change and sea-level rise, could well increase the diversity and thus the biological value of the system (Beijersbergen & Beekman 1989). Sea-level rise could also, locally, raise the level of the freshwater table in dune soils with the corresponding rejuvenation of dune slack communities.

Shingle banks are probably less sensitive to sea-level rise than other communities as they are less dynamic, although by the same token any reworking of these shingle deposits and their displacement landwards could lead to the loss of some of their biological diversity for considerable periods. Some of the communities associated with extensive shingle deposits have developed over hundreds of years and, if destroyed, they could be lost at least for similar periods of time if not for ever. Earth cliffs and cliffs of glacial sands, clays and gravels are particularly vulnerable to rising sea-levels.

The direct attack by the sea is the factor that determines the rate of erosion. Vegetation, where present, is mainly a sign of stability rather than a cause of it. Changing patterns in the flow of ground water could have some effect on the rate of erosion but, like vegetation, the effects will be very small in comparison with the dominance of marine erosion. It has already been noted, however, that in some areas the erosion of earth cliffs provides a vital supply of sediment for the growth of marshes and sand dunes elsewhere and thus changes in the rates of cliff erosion, whether positive as a result of sea-level rise or negative by sea defence measures, could have major effects elsewhere. In addition to the direct impacts on the coast itself, there could also be considerable effects in the para-maritime zone landwards. Higher sea-levels could greatly increase salt penetration inland. In certain areas salt water penetrates considerable distances inland and a rise in sea-level could increase this. In addition, buried salt or brackish water deposits could rise nearer the surface, effectively increasing soil salinity. Direct penetration of salt water up rivers will increase, particularly if high sea-levels coincided with low river water levels. Conversely, in the winter increased rainfall and higher sea-levels could increase the extent and frequency of flooding, with implications for sea defence, river management, agriculture and nature conservation.

Planning for the future

The Dutch have identified four alternatives (Anon 1989):

1. Retreat and concentrate on the protection of the polders.
2. Maintain protection of areas of special importance and value.
3. Maintain the present line.
4. Advance seawards and thus protect the whole coast.

The key approach on this list is the fourth one, the use of extended shore defences combined with offshore protection. The overall effect is to reduce marine erosion and to increase accretion and thus to shift the balance from an eroding coastline to an accreting one. By this means, the required degree of direct shore protection in the future will remain unchanged, or it may even be possible to reduce it. The big disadvantage of this approach will of course be the very high capital cost, and thus future Dutch sea defence policy will be based on maintaining the present line, albeit in a more dynamic state. In the case of the British coast, planning for the future is generally not so advanced (Boorman 1989). There is an important difference in that at least some of the low lying coastal areas are local and discrete. For these areas abandonment to the sea may be an economic necessity. For the larger low-lying areas, such as the Fens south of the Wash and most of the Essex coast of the Thames estuary, the value of the reclaimed land far exceeds the cost of increased protection. For some estuaries in Britain it is appropriate to consider protection by dams or storm surge barriers. Generally, however, the capital costs of such barriers are high compared with the costs of improving even extensive lengths of conventional sea wall. Thus it is unlikely that estuarine barriers will be generally used to shorten the length of the coastline that needs protection. The Thames barrage is an exception to this and will need extensive modification to cope adequately with predicted rises in sea-level. Other barriers are only likely to be built if there are other benefits e.g. the generation of tidal power (Severn Barrage) or possibly in conjunction with road building. It is relatively easy to assess the benefits of different sea defence options in terms of the increased safety of people, their houses and their factories. It is also possible to assess the benefits in terms of the protection of agricultural land. What is more difficult is to assess the impact of different options on nature conservation and on the natural environment. We should try to compare like with like and compare the costs of different sea defence options with the impact on the natural environment in cost benefit terms. This is not easy but I think that we do have to attempt to do it.

How do we put a price on our natural heritage? One formula that has been suggested is to combine the value of the present use (what people pay to visit the area) with the potential value of use (what might be earned from the area) and with a figure representing the intrinsic value of the feature (e.g. what would be considered as fair compensation for loss or the possible cost of recreating a similar feature elsewhere) (Renkema & Wossink 1989). Problems naturally arise when there are differences over what people are willing to pay or willing to accept. It is worth thinking positively about the possible benefits of sea defence changes in relation to nature conservation. It is easy to think that with rising sea-levels nature will be squeezed out against the sea wall. This is the worst situation but at some cost and with careful planning this can be avoided (Louisse & Kuik 1989).

A rising sea-level implies a more dynamic coast (Beijersbergen & Beekman 1989) and thus, given a wide enough coastal zone for natural processes to operate, more rather than less opportunities for nature and wildlife. There are also both problems of habitat loss and of opportunities for habitat creation in the para-maritime zone inland of the dunes and saltmarshes. Some areas may be lost to agriculture by increase salt seepage, but equally new brackish water habitats could be created. It is clear that we can expect great changes in all our coastal habitats. By careful planning we can avoid the worst effects and exploit the beneficial or potentially beneficial ones. One problem we have to face is that coastal habitats, like their inland counterparts, may become more fragmented. To ensure both the health of these remaining habitats and the retention of the present range of habitats or their future equivalents as a result of global warming, we also have to have the appropriate ecological infrastructure. By this I mean the optimum spatial and functional interrelationships of the individual component areas.

A question that is often asked is 'do we need a disaster before we do anything about predicted rises in sea-level?' I think that for the Netherlands the answer must be 'no'. For the United Kingdom, I hope that we, like the Dutch, learnt from 1953, but I am not completely sure. Unlike 1953 however, we do have the warnings about what we are likely to face and thus we have the chance to act now.

References

Anon 1989. *Kustverdediging na 1990.* Den Haag: Rijkswaterstaat.

Beijersbergen, J., & Beekman, F. 1989. Slufters en gaten in de duinen: naar een natuurlijker kustlandschap. *De Levende Natuur, 1989*: 118 –123.

Boorman, L.A. 1989. *Climatic change, rising sea levels and the British coast.* London, HMSO.

Boorman, L.A., & Ranwell, D.S. 1977. *Ecology of Maplin Sands and the coastal zones of Suffolk, Essex and North Kent.* Norwich, Institute of Terrestrial Ecology.

Brouns, J.J.W.M. 1988a. The impact of elevated carbon dioxide levels on marine and coastal ecosystems. *Texel, Rijksinstituut voor Natuurbeheer, R.I.N. Rapport*, 88–55.

Brouns, J.J.W.M. 1988b. The impact of sea level rise on the Dutch coastal ecosystems. *Texel, Rijksinstituut voor Natuurbeheer, R.I.N. Rapport*, 88–60.

Kirby, R. 1969. *Sedimentary environments, sedimentary processes and river history in the lower Medway, Kent.* Ph.D. Thesis, Queen Mary College, London.

Louisse, C.J., & Kuik, A.J. 1989. Kustverdediging: een 'spel' tussen natuur en mens. *De Levende Natuur, 1989*: 96–101.

Oerlemans, J. 1989. A projection of future sea level rise. *Climatic Change, 15*: 151–174.

Olff, H., Bakker, J.P., & Fresco, L.F.M. 1988. The effect of fluctuations in tidal inundation frequency on salt marsh vegetation. *Vegetatio, 78*: 13–19.

Pugh, D.T. 1990. Is there a sea level problem? *Proceedings of the Institution of Civil Engineers, Part 1.88*: 347–366.

Renkema, J.A., & Wossink, G.A.A. 1989. De zorg voor het milieu in economisch perspectief. *Landbouwkundig Tijdschrift, 101*: 10–12.

Ronde, J.G. de, & Vogel, J.A. 1988. Kustverdediging na 1990 *– Technisch Rapport 6 – Zeespiegelrijzing.* Nota GWAO 88.015. Den Haag: Rijkswaterstaat.

Zwart, B. 1984. Neemt het aantal zware stormen toe? *Nautisch Technisch Tijdschrift, 9*: 72–75.

Changes in sedimentation and contamination in the Ribble estuary

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Beresford Hartwell, P.R., Horsington, R.W., Mamas, C.J.V., Randle, K., Taylor, E.W., West, J.R., & Sokhi, R. S. 1995. Changes in sedimentation and contamination in the Ribble estuary. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 14–19.

Chemical elements have been used as tracers in this preliminary study to investigate the nature of sedimentation in the Ribble estuary. Signatures of Zinc and Cs-137 are of particular interest, as they exhibit variations consistent with temporal changes in industrial inputs. Higher concentrations are observed in the mid-estuary rather than at the marine or freshwater ends. The rate of accretion at the Nazemount site is estimated to be 40 mm/year which is significantly higher than that found in other Irish Sea estuaries. Depth and longitudinal profiles of the contaminants are discussed and possible reasons for this high accretion rate are suggested.

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Introduction

Estuarine environments are considered to be dynamic as they are subject to time dependent tidal forces. This introduces stresses such as salinity and temperature changes as well as exposing the system to continual movements of sediments. Superimposed on the regular temporal tidal variations are the episodic events that result in rapid, large scale deposition or removal of fine grained materials. In addition to these natural processes estuaries are also exposed to anthropogenic stresses, such as, discharges of effluents, navigational and recreational activities and industrial development.

The Ribble estuary has been subject to various engineering practices over the last 200 years associated with navigational and dock development activities (Barron 1938, Drakes 1986). Overall, the Ribble estuary appears to be affected by accretion more than erosion, with average accretion figures of 14 mm/year reported for the years 1909–1947 and 9 mm/year for the period 1947–1967 (O'Connor 1987). The rate of accretion is high because the estuary is still adjusting to the various engineering works (dredging of the main channel from the mid 19th century until the mid-1960s, construction/decay of a training wall over the same period, and land reclamation), and to the changing natural environment. The Ribble estuary is not expected to reach a new equilibrium state until well into the next century (O'Connor 1987).

The present study forms an integral part of a major programme to investigate the transport of heavy metal and radionuclide contaminants in UK estuaries. The aims of this study are to identify the nature of the sediment deposits within the Ribble estuary and to establish depth and longitudinal distributions of the contaminants at selected site locations. In addition, an estimate of the current rate of accretion for the mudbanks of the estuary is made, in order that future accretion/erosion patterns may be predicted. The data reported in this paper were collected over a period of ten months.

Methods

This study makes use of chemical tracers present in the estuary from a variety of sources to determine the nature and rate of sedimentation within the estuary. In order to do so, core and surface sediment samples were taken from four sites, selected to show the depth and longitudinal contamination patterns in the marine brackish and freshwater regions of the estuary: Lytham, Nazemount, Preston (dock site) on the north bank and Preston (A6 bridge site) on the south bank (Figure 1). In order to prevent contamination of the samples the cores were collected using plastic tubes driven into the mud. After the cores were removed, they were sealed with plastic coverings. Surface samples were collected with plastic implements and stored in sealed plastic bags.

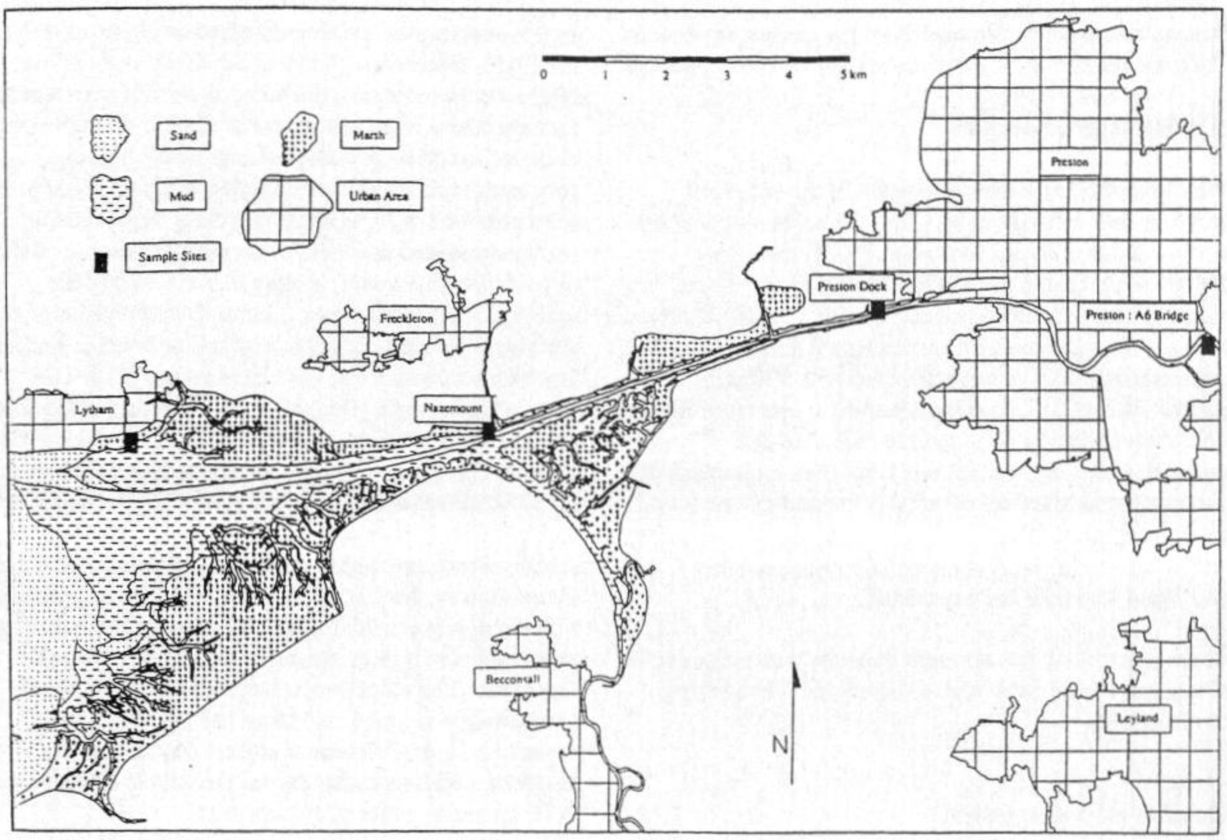

Figure 1 The Ribble estuary and the sampling sites (Lytham, Nazemount, Preston Docks and Preston A6 bridge)

The samples were analysed for Cs-137 and various other non-radioactive elements. Some of the elements occur naturally while others arise from anthropogenic inputs and thus reflect discharge patterns of local industry. Cs-137 is an artificial radionuclide and acts as a convenient tracer as it is mainly released from a well defined discharge source (British Nuclear Fuels Ltd (BNFL) Sellafield reprocessing plant). By combining the signatures of radioactive and non-radioactive elements it is possible to identify peaks of discharges and hence assign a date to the sediments and thus calculate the rate of sedimentation. The shear strength of the sediment was also estimated along the transverse of the intertidal zone at points between the cliff at the seaward edge of the saltmarsh and the low water mark. Similar studies have been conducted in other Irish Sea estuaries, for example, Carr & Blackley (1986), although with different techniques.

One commonly identified problem associated with this kind of survey is that the natural biota in the estuary disturbs the sedimentation pattern and mixes the sediment layers (by burrowing, for example). The Ribble sediment studied, however, tend to be fine, black and anaerobic, containing few macroinvertebrates with no evidence of burrowing molluscs. Consequently, mixing of the sediments by bioturbation is not expected to be significant. In isolated zones of more stable mud a small polychaete worm (*Capitella capitata*) was found (Popham 1966), but these are surface deposit feeders which are unlikely to create much bioturbation to disrupt the sediments.

The sediment samples were exposed to the following analyses.

(i) Particle size

Particle sizing was performed using a Malvern Instruments laser particle sizer. It is important to determine the particle size fractions as pollutants tend to become adsorbed preferentially onto smaller, fine particles because of their large surface area to volume ratio.

(ii) Radionuclide measurements

A hyperpure germanium computer-based gamma-ray spectrometry system was employed for the identification and quantification of Cs-137 in the sediment samples. This radionuclide results predominantly from Sellafield discharges. The sediment samples were dried and prepared in to fine powders before counting them in a Marinelli beaker, ensuring a fixed geometry. The concentrations were deduced from the gamma-ray spectra with the aid of a Canberra Spectran AT software package.

(iii) Heavy metal analysis

Analysis of other elements present in the sediment samples was carried out by using the accelerator-based Proton Induced X-ray Emission (PIXE) technique. Pellets of dried and powdered sediment were placed in a vacuum chamber and bombarded with a 2.5 MeV proton beam from a Dynamitron accelerator. The resulting characteristic X-rays were detected with a lithium drifted silicon, Si(Li), x-ray detector. A spectrum fitting computer code was employed to calculate the concentrations of the elements from their characteristic x-ray spectra. More details of this method can be found in Sokhi *et al.* (1990).

(iv) Shear strength measurement

Shear strength of the sediment deposits was measured *in situ* using a hand held Pilcon shear vane when taking core samples.

Results and discussion

Longitudinal profiles were produced from the results of the analyses for a number of elements present. For some elements, the pattern of the profile is much as expected. Chlorine concentration, for example, shows a definite decline with distance upstream from Lytham (Figure 2). Ratios of concentrations relative to Lytham fall to 0.14 at the A6 bridge site in Preston. This is to be expected since chlorine is associated with sea water. Similarly, some land-derived elements (such as Fe, Zn, Mn and Cu, see Figure 2) occur with a stronger signature landwards from Lytham.

In general there is little depth variation of most of the non-radioactive elements at any of the sites. At Nazemount, however, the sediments show a variation in the concentration of Zinc both in the longitudinal profile and with depth (Figures 2 and 3). Values for Zn concentration with depth are listed in Table 1. The peak in the longitudinal profile is possibly caused by the increased adsorption of Zinc onto the fine suspended particles as a result of higher salinity in this region of the estuary (Mance & Yates 1984). An alternative or an additional effect may be the enhancement of organic coating of the Clifton Marsh sewage works and thus leading to increased adsorption. The vertical variation may be due to the closure several years ago of a local industrial plant discharging Zinc.

The profiles for Cs-137 also show Nazemount to be an area of preferential concentration (Figure 4 and Table 2). In this case the source of contamination is assumed to be the BNFL reprocessing plant at Sellafield in Cumbria. Highest concentrations, therefore, would be expected at Lytham where more adsorption of Cs-137 on to the mud particles would be possible because of the higher concentrations. Large concentration of Cs-137, however, were observed at Nazemount and these may reflect a higher suspended sediment concentration associated with the turbidity maximum, leading to a more efficient scavenging from the water column. Depth profiles at each site also show some possible anomalies. In other studies in which sediment cores have been taken, of the Dee estuary for example (Bonnett *et al.* 1988), a peak level of concentration has been observed at 20–30 cm below the surface. This peak is thought to be associated with peak Cs-137 discharges from Sellafield in 1975 (Howarth & Kirby 1988). This expected peak was not found in the Ribble estuary, even at depths of 600 mm. Since the Zn signal shows a distinct pattern of concentration variation with depth it is considered unlikely that the sediment has been subject to mixing through multiple erosion and deposition. The absence of a detectable peak of Cs-137 concentration suggests that there has been an accumulation of sediment of at least 600 mm on the intertidal mud flats at this part of the Ribble estuary since 1975: an average rate of 40 mm/year.

The discrepancy between the value of 40 mm/year quoted and the 9 mm/year value previously reported (O'Connor 1987) may not be as large as it appears. The reduction in estuary capacity upon which the 9 mm/year value is based applies only to the main channel; this study is concerned primarily with the intertidal muds seawards of the saltmarsh cliff, a relatively dynamic environment in which much larger volumes of material are deposited. In these circumstances, a much higher value for accretion on the mud banks may be expected.

The recorded shear strength values also indicate that the Nazemount site is of a different nature from the others in the estuary. Shear strength variations at Lytham and Preston show some evidence of consolidation at depth (Figures 5 and 6). The sharp increase in the shear strength at a depth of about 800 mm at Preston may be caused by the influence of bank protection engineering structures below the surface. Values observed in May 1990 at Nazemount (Figure 5) show no such consolidation; some slight strengthening of the material is evident just below the surface but no further dewatering or compaction seems to have occurred below about 500 mm. Although this may indicate a highly mobile beach environment in which sediment is periodically stripped from the beach and is eventually (or fairly rapidly) replaced by newer or resuspended sediment, it seems unlikely, as there would then be little depth variation of the tracers. Organic coating of the particles which would inhibit both compaction and

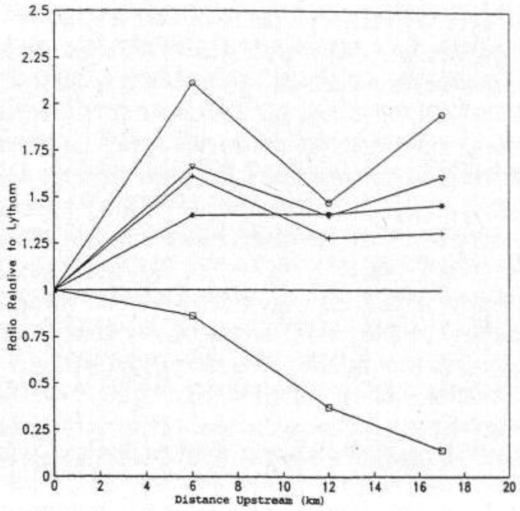

Figure 2 Ratios of elements measured in sediment samples along the estuary (▽ Mn, ○ Zn, + Cu, ✱ Fe and □ Cl)

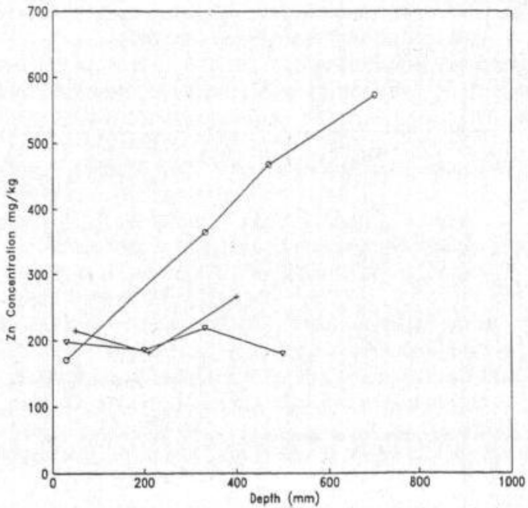

Figure 3 Zn concentrations with depth at Lytham (▽), Nazemount (○) and Preston Dock (+)

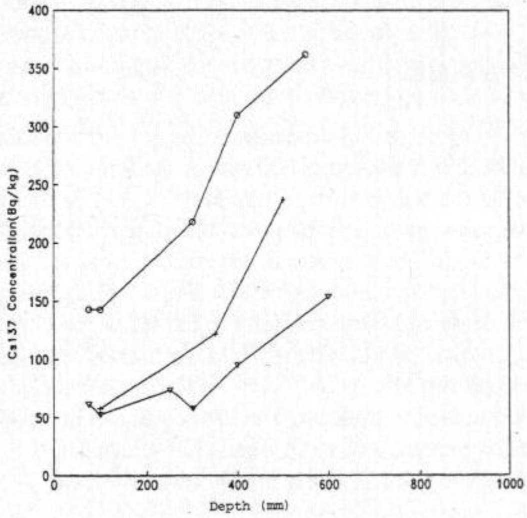

Figure 4 Cs-137 concentrations with depth at Lytham (▽), Nazemount (○) and Preston Dock (+)

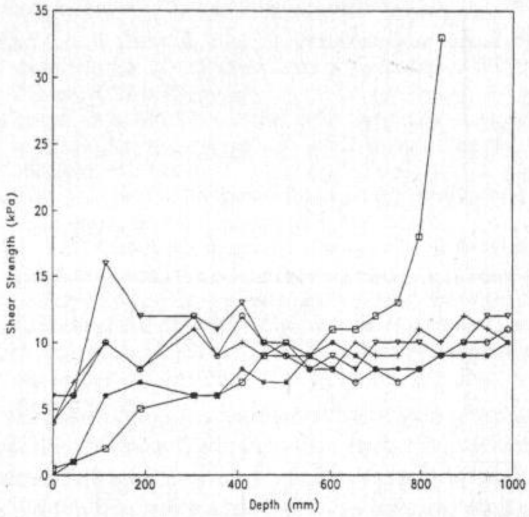

Figure 5 Variation of shear strength with depth at Nazemount and Preston Dock for various distances from the top of the beach (▽ 0 metres, ○ 4.5 metres, + 9 metres, ✱ 13.5 metres for Nazemount and □ 5 metres for Preston Dock)

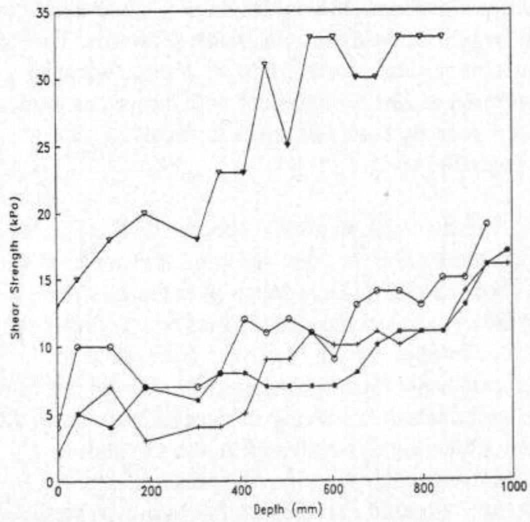

Figure 6 Variation of shear strength with depth at Lytham for various distances from the saltmarsh cliff (▽ 0.5 metres, ○ 3 metres, + 6 metres and ✱ 9 metres)

Table 1 Zn concentrations (mg/kg) in the sediment samples (uncertaintities* are typically ± 10%)

depth (mm)	*Lytham*	*Nazemount*	*Preston Dock*
30	198	171	-
50	-	-	215
200	186	-	-
210	-	-	182
330	219	365	-
400	-	-	266
470	-	468	-
500	180	-	-
700	-	573	-

* These uncertainties reflect the precision of the PIXE analysis technique and do not include errors due to sampling.

Table 2 Cs-137 concentrations (Bq/kg) in the sediment samples (uncertaintities* are typically ± 1%)

depth (mm)	*Lytham*	*Nazemount*	*Preston Dock*
75	614	1430	-
100	527	1430	579
250	738	-	-
300	578	2180	-
350	-	-	1220
400	951	3100	-
500	-	-	2370
550	-	3620	-
600	1540	-	-

* These uncertainties reflect the counting statistics and do not include errors due to sampling.

dewatering may provide an explanation. Overall, the shear strength values observed are consistent with those found in the Severn estuary, where surface sediments have little resistance but subsurface material exhibit much more shear strength (West 1990).

As particle sizing revealed all of the sites consisted of mainly fine-grained (< 0.063 mm) sediments further analysis was not undertaken since it is widely considered that particulate pollutants will tend to accumulate on the fine-grained fractions. Although some reports have stressed the importance of the < 0.002 mm fraction, in the Ribble estuary there seems to be a consistency of particle size d_{50} around 0.010 mm. The distribution of Rb, often used as a proxy for particle size effects (Allen & Rae 1987), also indicates that there is little longitudinal variation in the particle size along the estuary. Ratios relative to Lytham remain close to 1.0 as far as the A6 bridge site.

In addition to a distinct longitudinal pattern of shear strengths, a strong transverse pattern was also found at each site. In each case, points at the lower end of the beach have generally lower shear strengths than points further up. This is to be expected since the lower zones will be covered by the tide for longer periods, with a consequently shorter period of dewatering and consolidation. At Lytham, Figure 6, the shear vane measurements are consistent with this reasoning at distances of 0.5 metres and 3 metres from the saltmarsh cliff. At lower beach positions (between 3 metres and 9 metres), on the other hand, the shear strength profiles are less consistent and measurements at a given depth do not appear to show any transverse correlation. This discrepancy, however, may not be significant because of the inherent uncertainty associated with the shear vane technique, making it difficult to resolve these shear vane signals. The strength found at the highest station (0.5 metres from the saltmarsh cliff) probably reflects the shorter periods of submersion of this part of the beach. This part of the beach will be submerged beneath active waters only on the higher (spring) tides, and even then the period of submergence will be of short duration.

Conclusions

(i) The shear strength measurements at each site show similarities in the upper 500 mm of the mud. At all sites the shear strength varies in the range 0–15 kPa. However, each site shows a complex structure, transversely and with depth. The structure is particularly clear at Lytham where there is a pronounced increase in shear strength with depth and with proximity to the saltmarsh cliff. The Nazemount site shows a remarkable consistency of shear strength values with only very little strengthening landwards or with depth. The Preston dock site shows a similar pattern to that at Lytham, but with a sharp increase to over 33 kPa at about 800 mm depth.

(ii) The ratios for non-radioactive elements, relative to the Lytham values, show that Cl (a mainly marine element) increases seawards: Zn, Fe, Cu and Mn (which are largely land-derived) decrease seawards. The almost constant vertical profile of the elements (with the exception of Zn) is consistent with either constant anthropogenic inputs or, more probably, a natural background level.

(iii) The Zn concentrations observed at Nazemount were much higher than at other sites and also indicate that the levels of discharge have fallen over the past decade. Surface values obtained at all sites may indicate the natural level of zinc in the river. The enhanced concentrations found at Nazemount may be due to the increased salinity resulting in more adsorption of Zn ions. Alternatively, or in addition to the above mechanism, adsorption may be facilitated by the enhanced organic covering on the sediment particles in this part of the estuary due to the increased nutrients provided by the Clifton Marsh sewage works 2 km upstream.

(iv) The longitudinal Cs-137 data show an enhancement at Nazemount, although the variation with depth is also clearly observed at the other sites. Concentrations of Cs-137 have been declining in the Ribble estuary over a considerable period of time (at least 15 years). This is in accord with most Irish Sea estuaries and coincides with the peak discharges from Sellafield in 1975. It is possible to speculate as to why the Nazemount concentrations are high and the Lytham concentrations relatively low, when the reverse might be expected. This may be related to the availability of particles on which the Cs-137 can be adsorbed, large quantities of suspended sediment becoming available in the turbidity maximum region. Levels at Preston would not be expected to be high; by the time tidal waters reach this point most of the marine-derived free elements, including radioactive materials, will have been diluted or adsorbed.

(v) Accretion at the Nazemount site has been at least 600 mm over the past 15 years: an average rate of 40 mm/year. Mixing of the sediment by alternating periods of erosion/deposition is thought unlikely because of the distinct vertical profiles obtained for Zn and Cs-137.

Acknowledgements

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References

Allen, J.R.L., & Rae, J.E. 1987. Late Flandrian shoreline oscillations in the Severn estuary: a geomorphological and stratigraphical reconnaissance. *Transactions of the Royal Society of London, B315*: 185–230.

Barron, J. 1938. *A History of the Ribble navigation – From Preston to the Sea.* Guardian Press.

Bonnett, P.J.P., Appleby, P.G., & Oldfield, F. 1988. Radionuclides in coastal and estuarine sediments from the Wirral and Lancashire. *The Science of the Total Environment, 70*: 215–236.

Carr, A.P., & Blackley, M.W.L. 1986. Implications of sedimentological and hydrological processes on the distribution of radionuclides: the example of a saltmarsh near Ravenglass, Cumbria. *Estuarine, Coastal and Shelf Science, 22*: 529–543.

Dakres, J.M. 1986. *The Last Tide – A History of the Port of Preston.* Carnegei Press.

Howarth, J.M., & Kirby, C.R. 1988. *Studies of environmental radioactivity in Cumbria, Part II: modelling the dispersion of radionuclides in the Irish Sea.* Report AERE-R 11734, DoE/RW/88.058.

Mance, G., & Yates, J. 1984. *Proposed environmental quality standards for List II substances in water: zinc.* WRc Technical Report TR209.

O'Connor, B.A. 1987. Short and long term changes in estuary capacity. *Journal of the Geological Society, London, 144*: 187–195.

Popham, E.J. 1966. The littoral fauna of the Ribble estuary, Lancashire, England. *Oikos, 17*: 19–32.

Sokhi, R.S., Davis, A., Randle, K., West, J.R., & West, M.S. 1990. Elemental analysis of sediments from the Severn estuary. *Nuclear Instruments and Methods in Physics Research, B49*: 328–331.

West, M.S. 1990. *Aspects of intertidal sediment dynamics in the Severn estuary.* Ph.D. Thesis, University of Birmingham.

Biogenic stabilisation and altered tidal range: preliminary observations

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Colloidal carbohydrate concentrations, as a measure of diatom exopolymer production, were found to correlate with sediment stability on the high intertidal of an estuarine mudflat. This relationship did not hold for similar stations treated with formaldehyde or for untreated mid- and low-shore stations. Four different groupings of diatom species were defined by principal component analysis, of which a *Nitzschia epithemioides* assemblage was associated with the most stable substratum.

Measurements of sediment bed height revealed periods of accretion interrupted by events of rapid erosion due to high winds blowing up the estuary. Formaldehyde treatment had the effect of reducing the variation in bed height when compared with untreated stations. Treated sediments became compacted and easily dewatered, leading to the formation of desiccated breccia which were never found on untreated sites. Formaldehyde treatment had no effect on the mean size of sediment particles although untreated stations were better sorted than treated stations. The differences between treated and untreated stations were considered to be direct or indirect consequences of the inhibition of bioturbation and/or algal development. The construction of a barrage on the Severn estuary is predicted to reduce the tidal range and alter the cyclic emersion and immersion of intertidal communities. The likely effect of this reduced range on algal assemblages and sediment stability is discussed.

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Introduction

Recent work has shown that epipelic diatom communities on estuarine mudflats can help to stabilise the surface layer of the sediment through the production of mucopolysaccharide exopolymers (Holland *et al.* 1974; Coles 1979; Frostick & McCave 1979; Vos *et al.* 1988). These exopolymers, produced as a by-product of diatom locomotion (Edgar & Pickett-Heaps 1984), can form a matrix that binds sediment particles at the sediment surface (Paterson 1989). The mechanism of this binding relies upon the production and decay characteristics of the exopolymer. In the intertidal region the efficiency of binding is likely to be related to the population density, species composition and physiological state of the diatom assemblages as well as physical changes due to assemblage position in relation to the tidal range. This paper examines the relative importance of this algal stabilisation at different tidal levels and compares the results with stations treated with formaldehyde to remove biological activity. The results are discussed in the context of a reduced tidal range, such as would be present behind the proposed Severn tidal barrage.

Methods

Site

A moderately sloping mudflat, bordered by *Spartina* marsh, at Aust on the Severn Estuary, UK (OS ref. ST 562 887), was examined. Two permanent transects were established in February 1990, and three stations (upper shore – MHWN, mid shore – MW, and low shore – MLWN) on each were marked by metal poles. The relative height of the sediment was monitored from a bar across the top of these reference poles. One transect was treated by spraying with formaldehyde at weekly intervals (200 ml conc. CH_2O m^{-2}).

Measurements

Measurements were taken during Spring tides at approximately monthly intervals with the exception of bed height measurements which were made at weekly intervals to increase temporal resolution of sediment accretion and erosion. Salinity, temperature and shear strength were measured *in situ*, the latter with a modified shear vane measuring only the surface 0.5 cm of sediment. Erodibility was measured using a cohesive strength meter (see Paterson 1989), adapted to fire a jet of water parallel to the sediment surface. This allows direct calibration with shear stress (Paterson & McArthur, unpublished data). Samples of surface sediment were analysed in the laboratory for particle size distribution (Elzone particle sizer), water content, total organic matter, protein content (Buchanan & Longbottom 1970), chlorophyll *a* (Jensen 1978), and carbohydrate (Kochert 1978). Epipelic diatom communities were sampled using the lens tissue method (Eaton & Moss 1966), and identified to species using acid cleaned subsamples. Data on wind direction and strength was obtained from the Bristol Weather Centre's station at Avonmouth (ST 505 787).

Results

Physical changes

Both surface salinity and temperature showed positive increases with time between March and August ($P < 0.05$ and $P < 0.001$ respectively, Figure 1a–b). Sediment water content of the untreated transect was initially between 40% and 50% (Figure 1c), and showed a significant increase with time ($P < 0.05$). Water content was slightly more significantly correlated with station position down the shore on the untreated than the treated sites ($P < 0.05$ for both transects, Figures 1c–d).

Sediment shear strength was lowest on the low-shore stations for both transects ($P < 0.05$ and 0.01 for untreated and treated respectively, Figures 1e–f). The stations on the treated transect also showed a significant decrease in shear vane readings over the course of the experiment ($P < 0.001$) which was due to the frequent presence of a very fluid layer of mud overlying more compacted sediment (this also accounted for high water content readings). The bulk of the sediments (by weight) on both transects were smaller than 62.5 μm and were generally dominated by particles in the 7.8 μm to 31.3 μm size range (5–7 phi units). There was no difference in mean particle size between transects although sediments were better sorted in the untreated stations (Table 1).

There was a net accretion of sediment on all stations between March and May (Figure 2). Accretion and erosion events were significantly ($P < 0.001$) affected by strength, direction, and duration of the wind, winds from the south-west causing the greatest erosion. Stormy weather between weeks 15 and 17 removed most of the redeposited sediment from the sites, although the treated upper shore station lost less sediment than the untreated. Accretion of soft sediments continued after this event. Variations in bed height were most pronounced in the untreated stations. Periods of accretion and erosion took place at the same time on the two transects but the extent of level change during either erosion or accretion was less on the treated than the untreated stations (Figure 2). During periods of sediment accretion the beds contained proportionally more water and organic matter as demonstrated by weeks 10 and 19 where accretion of 0.017 and 0.031 metres took place on the lower shore and 0.01 and 0.014 metres on the upper sites, treated and untreated stations, respectively (Figure 2). Sediments during accretion also had a greater mean particle size. The sediment on the formaldehyde-treated upper and mid-shore stations became compacted and laminated during the experiment, the upper and mid-shore stations showing remarkably clear horizontal laminations. During periods of fair weather the sediment of the treated upper and mid-tidal stations became dry and cracked forming desiccation breccia. The surface of the treated areas was consistently smoother than adjacent untreated areas.

Biological changes

Although total organic carbon, protein and bulk carbohydrate concentrations were measured, no significant pattern with respect to time or station position was found with the exception of a general increase in total organic matter with time ($P < 0.05$).

Concentrations of colloidal carbohydrate and chlorophyll *a* correlated significantly with each other on both the untreated ($\text{Chl } a = 0.063 \text{ (coll. carb)} + 29$, $r^2 = 90.7\%$) and treated ($\text{Chl } a = 0.076 \text{ (coll. carb)} - 1.71$, $r^2 = 97.5\%$) transects (Figures 3a–d). Both showed greater variability on the untreated transect. No significant relationship with either time or position on the shore was found (Figures 3a & c). On the treated transect, both chlorophyll *a* and colloidal carbohydrate showed a significant increase over the course of the experiment on all stations ($P < 0.01$) (Figures 3b & d).

The values of shear stress for the surface layer of sediment ranged from 0.2 to 1×10^4 N m^{-2} (for clay substrates). A significant linear regression ($P < 0.01$, $r^2 = 89.2\%$) between ln (critical stress + 1) and colloidal carbohydrate concentration was present on the upper shore site of the untreated transect (Figure 3e). There were no significant relationships between these two variables on the mid- and lower-shore stations on the untreated transect or at any station on the treated transect (Figure 3f).

Principal components analysis of the epipelic diatom assemblages on the transects over the experiment showed that four types of diatom community occurred (Figure 4). Community Type 1 was dominated by a small (15 μm) naviculoid diatom (*Navicula pargemina*

Table 1 Mean particle size, sorting, and skewness of the surface sediment on the two transects at five times during a nineteen week period

			treated			*untreated*	
	weeks	*MHWN*	*MW*	*MLWN*	*MHWN*	*MW*	*MLWN*
Mean size	0	5.80	5.50	5.75	4.90	6.30	5.70
	4	6.30	6.20	6.10	6.20	6.30	6.10
(phi)	10	4.85	5.35	5.30	4.95	5.60	4.60
	17	6.30	6.50	7.00	7.00	6.30	6.00
	19	5.55	5.60	5.45	5.55	5.60	5.45
Sorting	0	3.4	2.73	2.83	1.78	2.22	1.40
	4	1.05	1.15	1.15	1.10	1.20	1.12
	10	1.48	1.38	1.03	1.12	1.52	1.15
	17	1.08	1.42	1.67	1.43	1.25	1.30
	19	1.30	1.50	1.30	1.30	1.50	1.30
Skewness	0	-0.50	0.13	-0.33	-0.37	0.73	-0.03
	4	0.20	0.01	0.10	0.05	0.05	0.07
	10	1.18	-0.02	0.03	0.01	0.02	0.00
	17	0.07	0.02	0.03	0.03	0.05	0.20
	19	0.10	0.15	0.05	0.10	0.15	0.05

MHWN = mean high water neaps, MW = mean water, MLWN = mean low water neaps.

Underwood & Yallop). Community Type 2 was dominated by the larger (45 µm) *Nitzschia epithemioides*, while community Type 3 was characterised by *Rhaphoneis minutissima* and *Navicula flanatica*. Community Type 4 occurred in association with community Type 2, and was dominated by both *Nitzschia epithemioides* and *Navicula phyllepia*.

The diatom communities on the low shore stations of both transects were characterised as Type 3 communities throughout the experiment. During March, April, and May, the upper and mid-shore diatom communities on the untreated transect were Type 1 (Figure 4). However, the corresponding communities on the treated transect were of Type 3. From May onwards, the upper and mid-shore treated communities developed into Types 2 and 4, a pattern which also occurred on the untreated transect.

Discussion

Measurements began after a period of stormy weather which removed the soft sediment revealing an underlying layer of compacted clay. The water content and shear vane data for March reflect this fact. Net accretion of fine sediments on top of this layer continued throughout the experiment, depending on wind conditions. Erosion events were greatest when the wind direction was between 130° and 250°, this direction corresponding to the maximum fetch, and therefore maximum wave energy. No significant changes in sediment particle size distribution were found with season, which agrees with data from the Wash (Coles 1979). The threshold of erosion for the sediment of both transects was in broad agreement with the results of Amos & Mosher (1985) who found intertidal sediments to be up to 80 times more resistant to erosion than subtidal deposits, and this difference in general strength is also likely to occur between natural and resuspended deposits measured in the laboratory.

Treated stations

The compaction and smoothness of the treated upper and mid-shore stations appears to have been due to the removal of bioturbating animals, which can significantly increase sediment mixing (Branch & Pringle 1987). The water content of sediments was more related to sample time than to station position. A result of the compaction on the treated upper shore site was that less erosion occurred during the mild storm conditions in July. The mid-shore station was subject to greater erosion where sediment laminations appeared to allow the bed to erode in large sheets. During periods of erosion, treated sites revealed more compacted layers with low organic and water contents. The critical bed shear stress of cohesive sediments is related to the consolidation or bulk density of the bed (Terwindt & Breusers 1972) and as more compacted layers are revealed, a greater shear stress is required to erode them. An increase in bulk density and, therefore, yield stress with depth is a common result in laboratory experiments using reconsolidated cohesive sediments (A. Delo, Hydraulics research, pers. comm.). This may explain the reduced variation in erosion from the treated stations. Erosion from the upper stations was occasionally compounded by stripping of desiccation breccia formed during exposed calm periods (weeks 15–17, Figure 2).

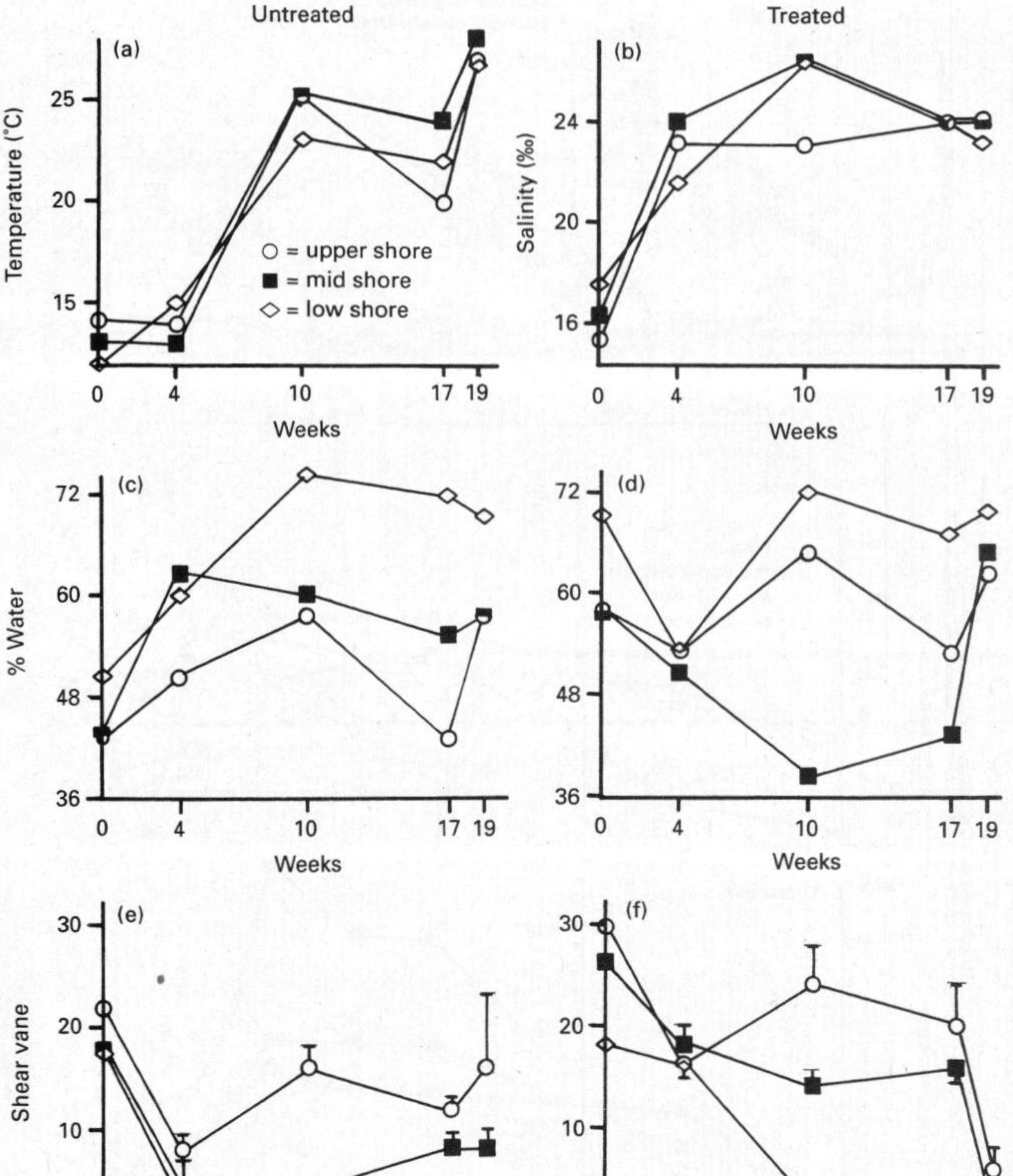

Figure 1 Temperature (a), salinity (b), water content (c and d) and shear vane (e and f) measurements for the untreated and treated transects over a five month period. Each point mean of 5 replicates, + S.E., except (a), where n = 2. Where S.E. are not shown, they are smaller in height than the symbol.

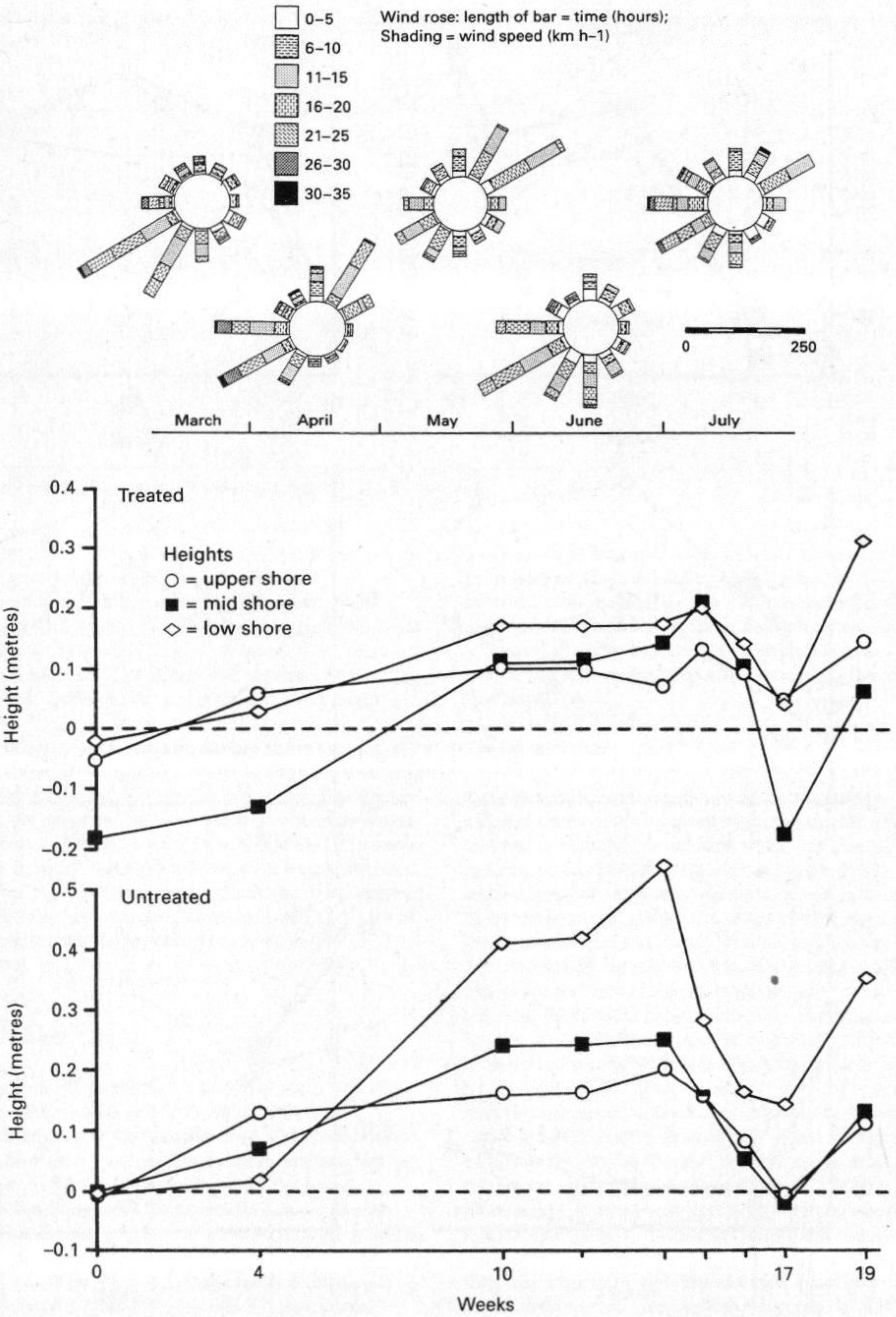

Figure 2 Wind rose diagrams and relative changes in sediment bed height on the two transects over the period of the experiment

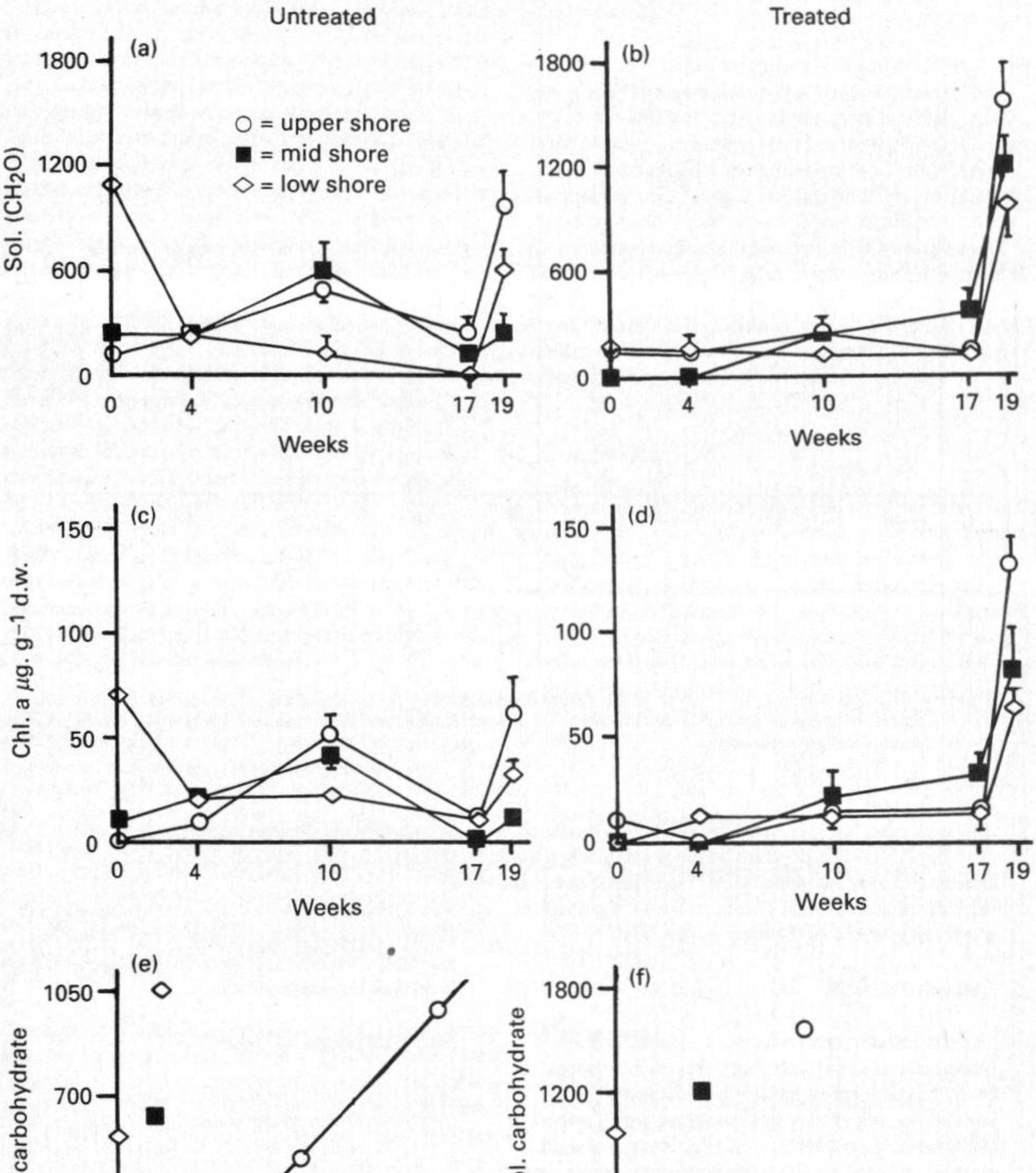

Figure 3 Colloidal carbohydrate concentration as μg glucose equiv. g dry weight sediment^{-1} (a and b), chlorophyll *a* concentrations (c and d), and the relationship between col. CH_2O and shear stress, on the untreated and treated transects (e and f). (Mean + S.E., n = 5 for Figures 3a–3d.)

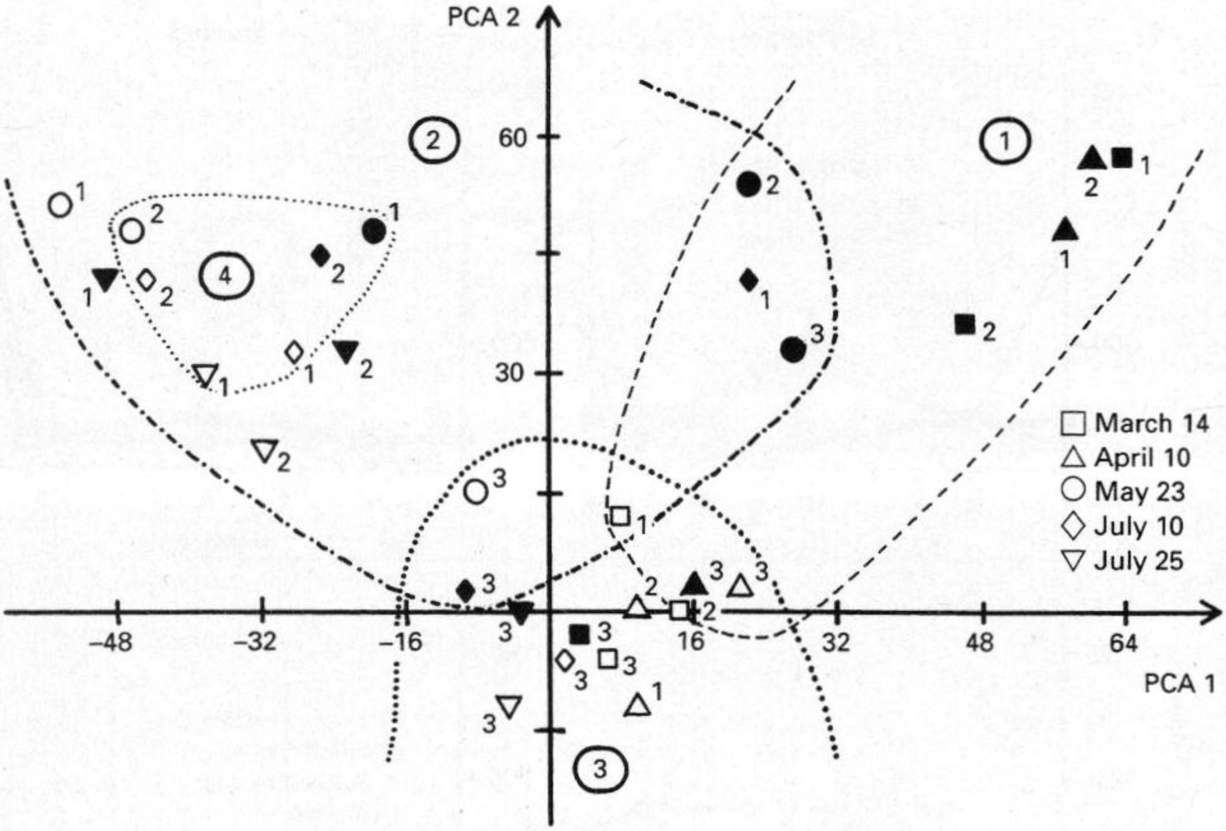

Figure 4 Principal Components Analysis of the diatom communities on the untreated and treated transects. Different community types delineated by hatched lines. 1,2,3: upper, middle and lower shore. Solid symbols untreated, open symbols treated communities.

Poisoning with formaldehyde killed the diatoms present on the surface, but these areas were quickly recolonised (4–5 days). Grazing pressure on the recolonising diatom communities on the treated sites was reduced due to the removal of macrofaunal grazers, and this appeared to allow large blooms to develop in late July.

Untreated stations

The untreated transect was more variable than the treated stations in almost every aspect. The greater extent of sediment turnover was indicated by less variable sorting of particles compared with treated sediments. Water content of the sediment was more closely related to station position than to sample time, unlike treated areas. There was almost no variation in the physical appearance of the untreated site with consistent evidence of bioturbation by *Nereis diversicolor* and *Hydrobia ulvae* (own observations).

Variation in bed height of untreated stations fell into recognisable periods of behaviour, accretion during calm weather followed by major erosive events corresponding with wind-generated wave action. Field observations during weeks 14–16 indicated sediments were high in water content with surface algal blooms. The presence of the algae and the mild conditions may account for the greatest accretion (mid-shore, 21 cm) found during the experimental period.

The biomass of diatoms on the untreated transect showed no pattern of seasonal or spatial variation, in agreement with the results from the Dutch Wadden Sea (Cadée & Hegeman 1974).

Biological effects on stability

In general, erosion was less pronounced on the formaldehyde-treated stations which contrasts with the results of de Boer (1981), who found that poisoned sandy sediments eroded rapidly compared with sediments colonised by microalgae. This highlights the care that must be taken by biologists to define the sedimentology of systems which can produce such widely varied results in terms of biogenic stabilisation. Meadows & Tait (1989) and Meadows *et al.* (1990) showed in laboratory experiments that burrowing animals enhanced sediment stability. The reasons for these differences in results may relate to the timescale and conditions under which the experiments were performed. More *in situ* work is required to investigate the influence of infauna.

A significant relationship between colloidal carbohydrate concentration and critical surface stress was demonstrated for the upper shore, untreated site. The algal origin of the polysaccharides measured is suggested by the strong relationship between colloidal carbohydrate and Chl. *a*. An increase in sediment stability associated with dense algal communities has been reported by various authors (see Introduction). Paterson *et al.* (1990) showed that the stability of upper and mid-shore sediments increased more over the period of emersion for sites with dense diatom populations. They also found no enhanced stabilisation at the lowest tidal stations which is in agreement with our results. The data presented were collected at low tide ± 1 hr, and it is likely that sediment stability would have increased further before the return of the tide. The epipelic diatoms on the more stable sediments (on the upper and mid-shore sites of both transects) were predominantly *Nitzschia epithemioides* assemblages, with less stable sediments colonised by *Navicula pargemina*, and community Type 3 species. This suggests either that *Nitzschia epithemioides* favours firmer sediments, or that this species is a significant stabiliser through exopolymer production, a hypothesis that remains to be tested *in situ*, although Holland *et al.* (1974) have shown that different species of diatom can have different stabilising potentials.

The absence of any significant relationship between sediment stability and algal biomass on the mid- and low-shore untreated sites may have been due to a combination of factors. The lower shore sites were often liquid mud, and therefore it was impossible to take cohesive strength meter measurements. The species of diatom found on the wetter sediments (Type 3) may not produce large enough quantities of exopolymer to stabilise the liquid sediment, or the high water content helps dissolve the mucopolysaccharides therefore preventing binding.

The lack of any relationship between algal biomass and stability on the poisoned sites may have been due to the increased water content and/or the predominance of low-shore diatom species on these sites. Also high stability values but low algal biomass were often recorded when the lower compacted sediments were exposed by erosive events.

Formaldehyde treatment acts as a model for abiotic sediment behaviour but is not entirely successful since even when spraying at weekly intervals algae recolonised extremely quickly. Preliminary bacterial counts also indicated that bacteria were even less affected (Underwood, unpublished data). The role of bacteria in sediment dynamics has received little attention with the exception of Grant & Gust (1987) who found a considerable influence on sediment erosion by submerged bacterial communities.

Tidal barrage

The effect of raised water levels due to the construction of a tidal barrage would be to reduce the period of exposure to the air of the upper shore mudflats, and to completely immerse the lower shore regions. This may have the effect of reducing sediment stability through increasing the sediment water content preventing mucopolysaccharide accumulation and restricting the algal communities to those typical of lower (wetter) shores. As the salinity of the water behind a barrage is likely to decline, then Type 2 diatom communities which favour higher salinity and are potentially better at sediment stabilisation, may decline resulting in more mobile sediments.

Conclusions

Poisoning with formaldehyde had a greater effect on the upper shore sites than on the lower shore sites. Poisoned sites showed signs of compaction, lamination, dewatering, and reduced sediment movement and bioturbation. Sediment surface stability was correlated with algal biomass on the untreated upper shore site, the more stable sediments being colonised mainly by *Nitzschia epithemioides*. Wind generated wave action was the most important factor in the control of erosive events.

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References

Amos, C.L., & Mosher, D.A. 1985. Erosion and deposition of fine-grained sediments from the Bay of Fundy. *Sedimentology 32*: 815–832.

Branch, G.M., & Pringle, A. 1987. The impact of the sand prawn *Callianassa krausii* Stebbing on sediment turnover and on bacteria, meiofauna, and benthic microflora. *J. Exp. Mar. Biol. Ecol. 107*: 219–235.

Buchanan, J.B., & Longbottom, M.R. 1970. The determination of organic matter in marine muds: the effect of the presence of coal and the routine determination of protein. *J. Exp. Mar. Biol. Ecol. 5*: 158–169.

Cadée, G.C., & Hegeman, J. 1974. Primary production of the benthic microflora living on tidal flats in the Dutch Wadden Sea. *Neth. J. Sea Res. 8*: 260–291.

Coles, S.M. 1979. Benthic microalgal populations on intertidal sediments and their role as precursors for salt marsh development. *In*: *Ecological Processes in Coastal Environments,* ed. by R.L. Jefferies & A.J. Davy, 25–42. Oxford, Blackwell.

De Boer, P.L. 1981. Mechanical effects of micro-organisms on intertidal bedform migration. *Sedimentology 28*: 129–132.

Eaton, J.W., & Moss, B. 1966. The estimation of numbers and pigment content in epipelic algal populations. *Limnol. Oceanogr. 11*: 584–595.

Edgar, L.A., & Pickett-Heaps, J.D. 1984. Diatom locomotion. *In*: *Progress in Phycological Research, 3*, ed. by F.E. Round & D.J. Chapman, 47–88. Bristol, Biopress.

Frostick, L.E., & McCave, I.N. 1979. Seasonal shifts of sediment within an estuary mediated by algal growth. *Est. Coast Mar. Sci. 9*: 569–576.

Grant, J., & Gust, G. 1987. Prediction of coastal sediment stability from photopigment content of mats of purple sulphur bacteria. *Nature, 330*: 244–246.

Holland, A.F., Zingmark, R.G., & Dean, J.M. 1974. Quantitative evidence concerning the stabilisation of sediments by benthic marine diatoms. *Mar. Biol. 27*: 191–196.

Jensen, A. 1978. Chlorophylls and carotenoids. *In*: *Handbook of Phycological Methods, Physiological and Biochemical Methods*, ed. by J.A. Hellebust & J.S. Craigie, 59–74. Cambridge, Cambridge University Press.

Kochert, G. 1978. Carbohydrate determination by the phenol-sulfuric acid method. *In*: *Handbook of Phycological Methods, Physiological and Biochemical Methods*, ed. by J.A. Hellebust & J.S. Craigie, 95–98. Cambridge, Cambridge University Press.

Meadows, P.S., & Tait, J. 1989. Modification of sediment permeability and shear strength by two burrowing invertebrates. *Mar. Biol. 101*: 75–82.

Meadows, P.S., Tait, J., & Hussain, S.A. 1990. Effects of estuarine infauna on sediment stability and particle sedimentation. *Hydrobiol. 190*: 263–266.

Paterson, D.M. 1989. Short-term changes in the erodibility of intertidal cohesive sediments related to the migratory behaviour of epipelic diatoms. *Limnol. Oceanogr. 34*: 223–234.

Paterson, D.M., Crawford, R.M., & Little, C. 1990. Sub-aerial exposure and changes in stability of intertidal estuarine sediments. *Est. Coast. Shelf Sci. 30*: 541–556.

Terwindt, J.H.J., & Breusers, H.N.C. 1972. Experiments on the origin of flaser, lenticular and sand-clay alternate bedding. *Sedimentology, 19*: 85–98.

Vos, P.C., de Boer, P.L., & Misdorp, R. 1988. Sediment stabilisation by benthic diatoms in intertidal sandy shoals; Qualitative and quantitative observations. *In*: *Tidal-influenced Sedimentary Environments and Facies* ed. by P.L. de Boer *et al.*, 511–526. D. Reidel Pub.

The niche of *Spartina anglica* on a changing coastline

A.J. Gray, E.A. Warman, R.T. Clarke & P.J. Johnson

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The greatest single agent of change in many temperate-zone saltmarshes this century has probably been the perennial grass *Spartina anglica*. Here we describe a simple model of the species' niche based on multiple regression of 27 physical and mainly tide-related variables. More than 90% of the variation in the upper and lower vertical limits of the species can be accounted for by variation in tidal range and modifying variables such as fetch, estuary area and latitude. The main change which is likely to occur as the species evolves is a narrowing of this niche by dieback at the lower limits and invasion by competitors at the upper limits. Both will bring major coastal changes. However, the effects of global warming and of direct human intervention by management or tidal barrage construction must be considered and can partly be predicted from a knowledge of *Spartina*'s niche and biology.

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Introduction

Arguably the single event which has wrought the greatest change in temperate-zone saltmarshes this century has been the spread, and partial retreat, of the perennial grass *Spartina anglica* C.E. Hubbard (hereafter simply *Spartina*). The origin and evolution of this remarkable allopolyploid species have been very well documented (Lambert 1964; Marchant 1967, 1968; Gray *et al.* 1990a; Raybould *et al.* 1991a, 1991b; Gray *et al.* 1991) as have its introduction and spread around the world (Goodman *et al.* 1959; Hubbard & Stebbings 1967; Ranwell 1967; Doody 1984; Gray & Benham 1990). Around 10,000 ha of intertidal marsh in Britain are currently dominated by *Spartina* (Charman 1990). Although dying back in many parts of the south and east coasts of England (and south Wales), it continues to spread in several Welsh estuaries and in the north-west of England (Doody 1984; Charman 1990; Gray *et al.* 1991).

The reasons for the species' success are manifold and undoubtedly include a suite of biological properties related to its hybrid origin, heterozygosity and its utilisation of the C4 photosynthetic pathway. However, a major factor enabling it to invade large areas during the last 100 years appears to be its occupancy of a formerly vacant niche on intertidal mudflats to seaward of the previous lower limit of perennial vegetation (Gray 1986). In this contribution we describe an empirical model of the plant's niche in part of its British range and examine how niche breadth may alter, and be altered by, future natural and man-made coastal change.

The niche of *Spartina anglica*

In a study described in detail elsewhere (Gray *et al.* 1989) data were collected from 107 line transects across saltmarshes in 19 estuaries in south and west Britain from Poole Harbour to Morecambe Bay. Surface heights were recorded at intervals by levelling from nearby Ordnance Survey benchmarks, and from previously established temporary benchmarks in the Severn and Dee estuaries. The levels of the extreme upper and lower limits of *Spartina*, including pioneer clumps, and of the main *Spartina* sward, if present, were recorded for all transects. Transects were taken only through apparently healthy *Spartina* swards, often with outlier clumps to seaward, avoiding those which were fronted by erosion cliffs or where the population was small. Transect data from earlier studies on Poole Harbour (Gray 1985a and unpublished reports), Milford Haven (Dalby 1970), the Dyfi estuary (Chater 1986), the Conwy (Gray, unpublished), the Ribble (Mullins 1985) and Morecambe Bay (Gray & Bunce 1972) were also incorporated into the model.

A total of 27 variables for each of the 107 transects was obtained from the field survey, maps, aerial photographs and tide tables. These included measurements of tide range and levels, marsh size and slope, exposure, estuarine area and aspect, latitude and sediment type. The relationship between these physical parameters and *Spartina* elevational limits was assessed using multiple regression.

In fact the elevational range of the plant is closely related to tidal range alone, as can be seen from Figure 1, which included ten range lines not included in the regression model, based on data extracted from the literature (Gray *et al.* 1989; that for Poole Harbour from Ranwell *et al.* 1964). Broadly speaking, *Spartina* is distributed between Mean High Water Neap tides (MHWN) and Mean High Water Spring tides (MHWS), with a tendency to extend downshore below the MHWN level in areas of spring tidal range less than 7 metres,

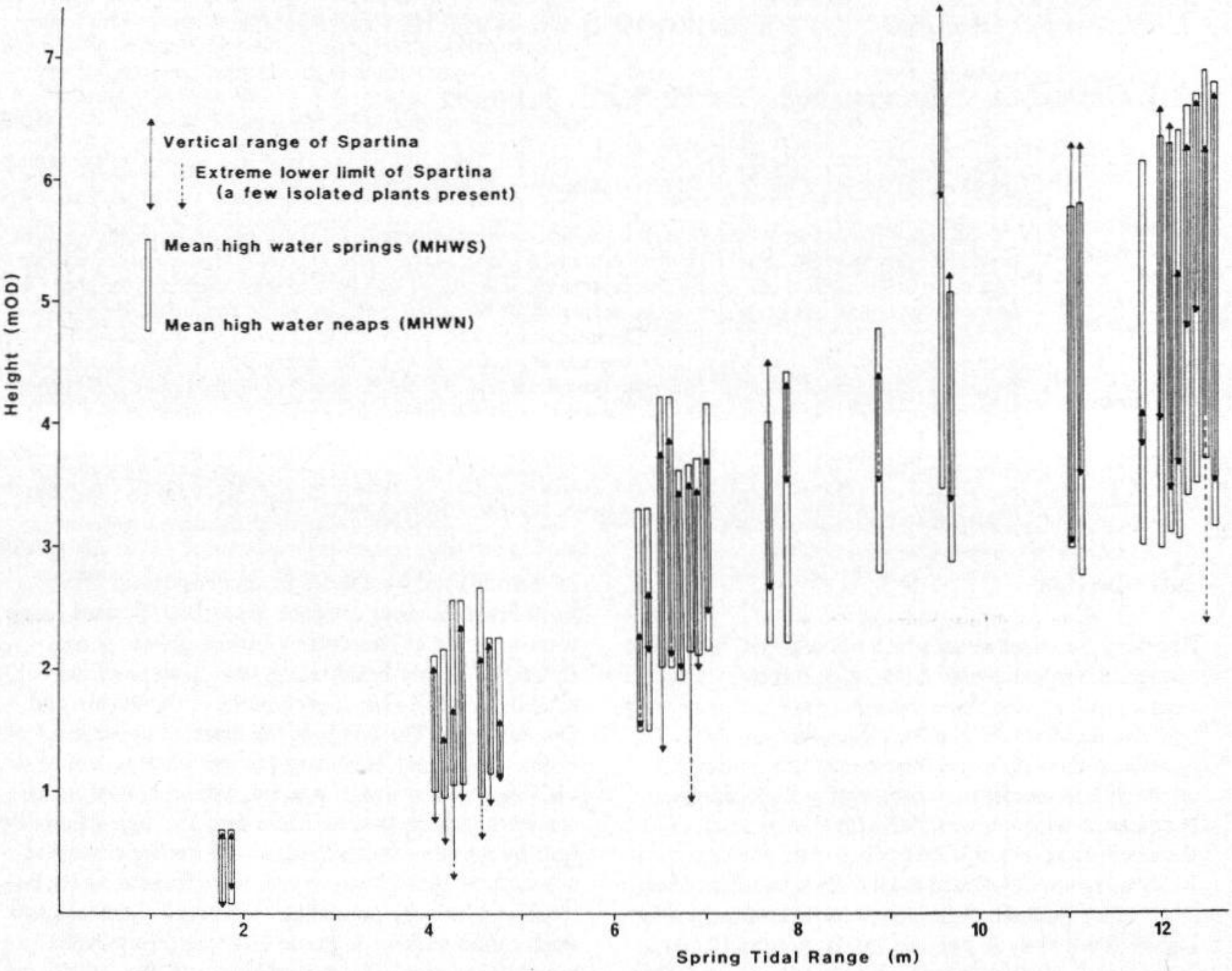

Figure 1 Vertical range of *Spartina* marsh and spring tidal range for 32 sites from 20 estuaries. From left to right: Poole Harbour (2 lines, the first from Ranwell *et al.* 1964), Foryd Bay, Teign, Dyfi, Mawddach, Traeth Bach, Tamar, Tavy, Milford Haven (2), Tywi, Taf, Red Wharf Bay, Conwy, Lavan Sands, Loughor, Dee, Ribble, Mersey, Shepperdine, Morecambe Bay, Severn (10 sites).

and to extend above MHWS level in areas of tidal range greater than 7 metres. More precisely the upper and lower limits of *Spartina* (in metres OD Newlyn) were described from the following simple linear regression against spring tidal range:

$$\begin{array}{lll} \text{Upper limit} = & -0.571 & +\ 0.528.\ \text{Range ST} \\ & (0.132) & (0.018) \end{array}$$

($R^2 = 88.7, S = 0.54$)

$$\begin{array}{lll} \text{Lower limit} = & -0.826 & +\ 0.451.\ \text{Range ST} \\ & (0.127) & (0.018) \end{array}$$

($R^2 = 86.1, S = 0.52$)

(Here, and in the equations below, the standard errors of the regression coefficients are given in brackets below each coefficient – the significance of the particular variable can be tested using Student's t computed as $t = b/SE(b)$ which is distributed as t with $N - 1 - p$ degrees of freedom, where N = the number of transects and p = the number of regressor variables in the equation. S = residual standard deviation in metres).

Although tidal range accounted for 89% and 86% respectively of the variation in upper and lower limits, other variables did significantly improve the prediction of these limits. The lower limit of *Spartina* (in metre OD Newlyn) was best described by the equation:

$$\begin{array}{lllll} LL = -0.805 & +\ 0.366\ (SR) & +\ 0.053\ (F) & +\ 0.135\ (\mathrm{Log}_e A) \\ \quad (0.102) & (0.019) & (0.016) & (0.025) \end{array}$$

($R^2 = 93.7, S = 0.35$)

where SR = spring tidal range (metres), F = fetch in the direction of the transect (km) and $\mathrm{Log}_e A = \log_e$ estuary area (km^2).

Thus, *Spartina* extends further down the shore than would be predicted from tidal range effects alone on transects with a shorter seaward fetch and in smaller estuaries. The upper limit of *Spartina* was described by:

$$\begin{array}{lllll} UL = 4.74 & + 0.483\,(SR) & + 0.068\,(F) & - 0.099\,(L) \\ \quad (2.29) & (0.028) & (0.020) & (0.045) \end{array}$$

$(R^2 = 90.2, S = 0.50)$

where SR = range and F = fetch as in the equation above and L = latitude (in degrees north expressed as a decimal).

The upper limit is similarly affected by fetch, but also varies significantly with latitude; the further north the marsh the lower down the shore, relatively speaking, is the upper limit of the *Spartina* sward.

An important feature of these equations is the very high values of R^2, more than 90% of the variation in both upper and lower limits being accounted for. This is a remarkably high proportion for a biological model. The possible reasons for such apparent precision are discussed in detail elsewhere (Gray *et al.* 1989, 1991) but are broadly threefold. First, tidal range is a good general estimator of a large number of correlated factors such as turbidity, current speed and submergence rates which may limit the species distribution. Second, we believe that *Spartina*'s occupancy of a niche downshore of the former marsh leads to its limits, and particularly the lower limit, being controlled mainly be physical tide-related factors (for which there is good evidence: Hubbard 1969; Morley 1973; Groenendijk 1986). The control of species limits by largely biological factors such as competition is less amenable to quantification even in relatively species-poor systems such as saltmarshes (Russell *et al.* 1985). Third, the relatively recent origin of *Spartina anglica* and its narrow genetic base (the species is genetically uniform at the level of electrophoretically-detectable variation – Raybould *et al.* 1991a) have precluded the extensive population differentiation, and hence niche extension, found in species such as *Puccinellia maritima* and *Aster tripolium* (Gray & Scott 1980; Gray 1985b, 1987).

The current niche of *Spartina* is, therefore, likely to change relatively rapidly both in response to physical coastal changes and as a result of ecological and evolutionary changes within the species itself as its interaction with its environment, biotic and abiotic, unfolds.

The changing niche

What are likely to be the major changes in *Spartina*'s distribution and ecological niche during its second century? This question is discussed below in terms of *Spartina*'s response to natural changes on the one hand and to man-made change on the other (although the distinction is somewhat artificial and the effects inseparable).

'Natural' change

A major change in *Spartina* marshes, which began on the south coast of England as early as the mid-1920s, has been the 'dieback' of swards, particularly at the lower levels of the shore. This is exemplified by the two range lines for Poole Harbour, Dorset, in the bottom left-hand corner of Figure 1, one based on the data of Ranwell *et al.* (1964) and the other on surveys in the mid-1980s. Whilst they have an identical upper limit, the lower limit in 1964 extended more than 15 cm further down the intertidal zone. This parallels the extensive reduction in *Spartina* marsh area, in the Harbour, which declined from a maximum of 775 ha in about 1925 to around 415 ha in 1980, and continues today (Gray & Pearson 1984; Gray 1985a). Whatever the proximal cause of dieback, which is now widespread in southern Britain and Europe – and most explanations refer to the highly anaerobic soil conditions which prevail in dieback areas – the phenomenon can be viewed in the longer term as part of the process of niche adjustment. By causing the rapid accretion of large volumes of sediment, and irreversibly changing the physiographic and hydraulic characteristics of the lower vegetated zones, *Spartina* has paved the way for its own destruction.

The removal in this way of the lower part of *Spartina*'s vertical niche has enormous implications for coastal and estuarine change. The release of sediments, and the nutrients they contain, is likely to have profound effects. In Poole Harbour for example, where up to 1.8 metres of newly-accreted sediment has been recorded below *Spartina* swards (Hubbard & Stebbings 1968), the capture of more than 7 million cubic metres of sediment and subsequent release of at least 3.5 million cubic metres is reflected in the changes in bed levels of the major navigation channels (Figure 2). The possible effects of releasing nutrients bound in the sediment, and of the decay of *Spartina* roots and rhizomes, on estuarine eutrophication and the spread of green algae in many southern estuaries have not been quantified, but are likely to be significant. It is likely that such changes will continue, sediment continuing to be released especially in those areas of fine silts and muds in the south of its geographical range where the species has long been established. The rate of change may be increased in these areas by rising sea-levels – a higher-than-average relative sea-level rise of 5 mm/year has been recorded over a 19-year period at Portsmouth (Woodworth 1987).

Evidence from our studies in Poole Harbour indicates that at least some of the sediment released by *Spartina* is being deposited at higher levels on the marshes. This process may accelerate the further narrowing of *Spartina*'s niche by the downshore invasion of other species. The successional replacement of *Spartina* at its upper limits by species such as *Puccinellia maritima* has been noted from several marshes (see Gray *et al.* 1991 for review). Interestingly, this replacement occurs more quickly after *Spartina*'s establishment, and at a point relatively further down the tidal range, as the current northern limits of the species' geographical range are approached (as indicated in the regression by the significant negative effect of latitude on the upper limit). Unfortunately, in both Britain and the Netherlands the effects of increasing latitude on competition between *Spartina* and *Puccinellia maritima* are difficult to separate from those of increasingly sandy substrata as one goes north. However, the importance of early seasonal growth in the C_3 grass *Puccinellia maritima* in gaining a competitive advantage over *Spartina* has been demonstrated experimentally by Scholten & Rozema (1990). Similarly Long *et al.* (1990) have shown that the productivity of *Spartina*, in accord with its utilisation of C_4 synthesis, has a strong dependence on temperatures and that the species is probably prevented from spreading northwards by low temperature during the period April–October.

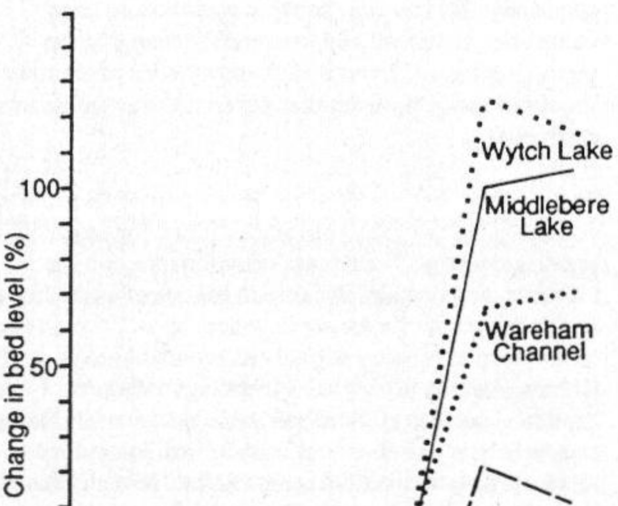

Figure 2 The percentage change in bed levels of four major channels in Poole Harbour, Dorset, between the Admiralty surveys of 1849 and those of 1934, 1954 and 1980. Note the deepening during the expansion of *Spartina* marsh and shallowing during its decline from 1934 onwards (from Gray *et al.* 1990).

Man-made change

In the absence of other major changes we might predict a gradual narrowing of *Spartina*'s realised niche (*sensu* Hutchinson 1957) as physical changes, largely driven by the plant itself, cut off the lower end of its vertical range, and invasion by well-established competitors occurs at the upper end. This process may be halted or reversed by genetic changes in the species, although currently these are difficult to envisage – indeed the genetic uniformity of *Spartina* and its consequent vulnerability to pathogens may hasten its demise (Gray *et al.* 1990b, 1991). However it is more likely that human intervention, both direct and indirect, will play a role in *Spartina's* future.

First, the projected climatic and sea-level changes accompanying global warming will have profound effects which are difficult to predict in detail. Predictions of changes in net primary production indicate that higher temperatures will increase biomass in *Spartina* by decreasing photoinhibition and accelerating the development of the leaf canopy (Long 1990). This suggests increased growth and competitive ability and possibly an accelerated northward spread of the species. However the productivity of *Puccinellia maritima* is also expected to increase (for different reasons related to increased conversion efficiency in a higher CO_2 environment – Long 1990), producing a new level of competitive interaction between the two species. Furthermore major uncertainties about the effects of sea-level rise on accretionary balance and vegetation change in the lower zones of saltmarshes (e.g. Dijkema *et al.* 1990) argue for caution and more research on the interplay between sea-level change and increased production in *Spartina*.

Second, more direct human intervention is likely to continue to affect *Spartina*, as it has in the past, by the deliberate spread or, increasingly likely, the control and eradication of the species in those estuaries where it is perceived to be a threat to nature conservation interest. Doody (1990) considers the arguments for and against controlling the species.

Third, and looking forward to later contributions, the construction of tidal barrages and barriers, principally for energy generation, will produce major changes to which *Spartina* may be expected to respond. That response can be predicted to some extent from the regression model discussed earlier. For example, in those parts of post-barrage estuaries where tide curves and cycles remain sinusoidal and fluctuate approximately about mean sea-level (e.g. immediately down-estuary from many barriers) the equations predicting *Spartina* limits in natural estuaries remain directly relevant. Thus in general, should tidal range in such areas be reduced by a barrage, other things being equal, *Spartina* swards may be expected to move downshore. However tidal barrage schemes reduce tidal range above a barrage mainly by cutting off the lower part of the natural range (as water is held back for discharge through the turbines on the ebb tide) and direct predictions from the natural estuary situation are impossible. More relevant, but again to be applied with caution, may be predictive equations which relate *Spartina* vertical limits to their deviations from standard tidal levels such as MHWNs or MHWSs. For example, the equation:

$$\begin{array}{l} LL - MHWN = \\ \qquad \begin{array}{ccccc} -\,0.192 & +\,0.044\,(F) & +\,0.200\,(\mathrm{Log}_e A) & -\,0.400\,(E) \\ (0.33) & (0.015) & (0.025) & (0.165) \end{array} \\ (R^2 = 72.4,\ S = 0.36) \end{array}$$

where F and $\mathrm{Log}_e A$ are as above and E = 'estuarineness' (the distance from the upstream tidal limit as a proportion of the total distance from there to the estuary mouth) may be helpful in predicting lower limits in post-barrage estuaries where mean sea-levels are altered above a barrage. Ideally, however, the post-barrage response of *Spartina* should be predicted from a detailed quantitative knowledge of the tidal relations which obtain at the upper and lower limits, and which are thought to be the major determinants of *Spartina* niche. Models based on the limiting effects of tidal submergence would enable more accurate predictions of those situations where submergence patterns do not match those of natural tides. A preliminary attempt to provide such a model (Gray *et al.* 1989) suggests that the niche limits of *Spartina* are not defined by a narrow range of values for submergence-related variables, although the model is qualitatively useful in highlighting those variables which ought to be measured in more detail in particular estuaries where barrages are planned.

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References

Charman, K. 1990. The current status of *Spartina anglica* in Great Britain. *In*: Spartina anglica *– a research review,* ed. by A.J. Gray & P.E.M. Benham, 11–14. London, HMSO.

Chater, E.H. 1986. *Spartina* in the Dyfi estuary. *In*: *Ynyslas Nature Reserve Handbook 4, The Estuary Edn. 5,* 169–180. Aberystwyth, University College of Wales and Nature Conservancy Council.

Dalby, D.H. 1970. The salt marshes of Milford Haven, Pembrokeshire. *Fld. Scd. 3*: 297–330.

Dijkema, J.S., Bossinade, J.H., Bouwsema, P., & De Glopper, R.J. 1990. Salt marshes in the Netherlands Wadden Sea: rising high-tide levels and accretion enhancement. *In*: *Expected effects of climatic change on marine coastal ecosystems,* ed. by J.J. Beukema, W.J. Wolff, & J.J.W.M. Brouns, 173–188. Dordrecht, Kluwer Academic Publishers.

Doody, J.P., *ed.* 1984. Spartina anglica *in Great Britain.* Peterborough, Nature Conservancy Council. (Focus on Nature Conservation No. 5.)

Doody, J.P. 1990. *Spartina* – friend or foe? A conservation viewpoint. *In*: Spartina anglica *– a research review,* ed. by A.J. Gray & P.E.M. Benham, 77–79. London, HMSO.

Goodman, P.J., Braybrooks, E.M., & Lambert, J.M. 1959. Investigations into 'dieback' in *Spartina townsendii* agg. I. The present status of *Spartina townsendii* in Great Britain. *J. Ecol. 47*: 651–677.

Gray, A.J. 1985a. *Poole Harbour. Ecological sensitivity analysis of the shoreline.* Huntingdon, Institute of Terrestrial Ecology.

Gray, A.J. 1985b. Adaptation in perennial coastal plants – with particular reference to heritable variation in *Puccinellia maritima* and *Ammophila arenaria. Vegetatio 61*: 179–188.

Gray, A.J. 1986. Do invading species have definable genetic characteristics? *Phil. Trans. R. Soc. Lond. B 314*: 655–674.

Gray, A.J. 1987. Genetic change during succession in plants. *In*: *Colonisation, Succession and Stability*, ed. by A.J. Gray, M.J. Crawley & P.J. Edwards, 273–293. Oxford, Blackwell Scientific Publications.

Gray, A.J., & Bunce, R.G.H. 1972. The ecology of Morecambe Bay. VI. Soils and vegetation of the salt marshes: a multivariate approach. *J. Appl. Ecol. 9*: 221–234.

Gray, A.J., & Scott, R. 1980. A genecological study of *Puccinellia maritima* (Huds.) Parl. I. Variation estimated from single-plant samples from British populations. *New Phytol. 85*: 89–107.

Gray, A.J., & Pearson, J.M. 1984. *Spartina* marshes in Poole Harbour, Dorset, with particular reference to Holes Bay. *In*: Spartina anglica *in Great Britain,* ed. by J.P. Doody, 11–14. Peterborough, Nature Conservancy Council. (Focus on Nature Conservation No. 5.)

Gray, A.J., & Benham, P.E.M. *eds.* 1990. Spartina anglica *– a research review.* London, HMSO.

Gray, A.J., Clarke, R.T., Warman, E.A., & Johnson, P.J. 1989. *Prediction of marginal vegetation in a post-barrage environment.* Wareham, Institute of Terrestrial Ecology. (Report No. ETSU-TID-4070.)

Gray, A.J., Benham, P.E.M., & Raybould, A.F. 1990a. *Spartina anglica* – the evolutionary and ecological background. *In*: Spartina anglica *– a research review,* ed. by A.J. Gray and P.E.M. Benham, 5–10. London, HMSO.

Gray, A.J., Drury, M.G., & Raybould, A.F. 1990b. *Spartina* and the ergot fungus *Claviceps purpurea* – a singular contest? *In*: *Pests, Pathogens and Plant Communities,* ed. by J.J. Burdon & S.R. Leather, 63–79. Oxford, Blackwell Scientific Publications.

Gray, A.J., Marshall, D.M., & Raybould, A.F. 1991. A century of evolution in *Spartina anglica. Adv. Ecol. Res. 21*: 1–62.

Groenendijk, A.M. 1986. Establishment of a *Spartina anglica* population on a tidal mudflat: a field experiment. *J. Envir. Management 22*: 1–12.

Hubbard, J.C.E. 1969. Light in relation to tidal immersion and the growth of *Spartina townsendii (s.l.)*. *J. Ecol. 57*: 795–804.

Hubbard, J.C.E., & Stebbings, R.E. 1967. Distribution, dates of origin, and acreage of *Spartina townsendii (s.l.)* marshes in Great Britain. *Proc. Bot. Soc. Bri. Isl. 7*: 107.

Hubbard, J.C.E., & Stebbings, R.E. 1968. *Spartina* marshes in southern England. VII. Stratigraphy of the Keysworth Marsh, Poole Harbour. *J. Ecol. 56*: 707–722.

Hutchinson, G.E. 1957. Concluding remarks – *Cold Spring Harb. Symp. on Quantitative Biol. 22*: 415–427.

Lambert, J.M. 1964. The *Spartina* story. *Nature 204*: 1136–1138.

Long, S.P. 1990. The primary productivity of *Puccinellia maritima* and *Spartina anglica*: a simple predictive model of response to climatic change. *In*: *Expected effects of climatic change on marine coastal ecosystems,* ed. by J.J. Beukema, W.J. Wolff & J.J. Brouns, 33–39. Dordrecht, Kluwer Academic Publishers.

Long, S.P., Dunn, R., Jackson, D., Othman, S.B., & Yaakub, M.H. 1990. The primary productivity of *Spartina anglica* on an East Anglian estuary. *In*: Spartina anglica *– a research review,* ed. by A.J. Gray & P.E.M. Benham, 34–38. London, HMSO.

Marchant, C.J. 1967. Evolution in *Spartina* (Gramineae). I. History and morphology of the genus in Britain. *Bot. J. Linn. Soc. 60*: 1–24.

Marchant, C.J. 1968. Evolution in *Spartina* (Gramineae). II. Chromosomes, basic relationships and the problem of the *S. × townsendii* agg. *Bot. J. Linn. Soc. 60*: 381–409.

Morley, J.V. 1973. Tidal immersion of *Spartina* marsh at Bridgewater Bay, Somerset. *J. Ecol. 61*: 383–386.

Mullins, P.H. 1985. *The seed biology of* Spartina anglica *(C.E. Hubbard) in the southern Ribble estuary.* Ph.D. Thesis, Liverpool Polytechnic.

Ranwell, D.S. 1967. World resources of *Spartina townsendii (s.l.)* and economic use of *Spartina* marshland. *J. Appl. Ecol. 4*: 239–25.

Ranwell, D.S., Bird, E.C.F., Hubbard, J.C.E., & Stebbings, R.E. 1964. *Spartina* marshes in southern England V. Tidal submergence and chlorinity on Poole Harbour. *J. Ecol. 52*: 627–641.

Raybould, A.F., Gray, A.J., Lawrence, M.J., & Marshall, D.F. 1991a. The evolution of *Spartina anglica* C.E. Hubbard (Gramineae): origin and genetic variation. *Biol. J. Linn. Soc. 43*: 111–126.

Raybould, A.F., Gray, A.J., Lawrence, M.J., & Marshall, D.F. 1991b. The evolution of *Spartina anglica* C.E. Hubbard (Gramineae): variation and status of the parental species in Britain. *Biol. J. Linn. Soc. 44:* 369–380.

Russell, P.J., Flowers, T.J., & Hutchings, M.J. 1985. Comparison of niche breadths and overlaps of halophytes on salt marshes of differing diversity. *Vegetatio 61*: 171–178.

Scholten, M., & Rozema, J. 1990. The competitive ability of *Spartina anglica* on Dutch salt marshes. *In*: Spartina anglica *– a research review* ed. by A.J. Gray & P.E.M. Benham, 39–47 London, HMSO.

Woodworth, P.L. 1987. Trends in U.K. Mean Sea Level. *Mar. Geodesy 11*: 57–87.

Change of saltmarsh vegetation following the construction of the Oosterschelde storm surge barrier

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De Leeuw, J., De Munck, W., Apon, L.P., Herman, P.M.J., & Beeftink, W.G. 1995. Change of saltmarsh vegetation following the construction of the Oosterschelde storm surge barrier. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 35–40.

In 1986 the Oosterschelde storm surge barrier was completed. This caused a strong reduction in inundation frequency of the saltmarshes along the Oosterschelde estuary. In this study the changes in the vegetation were analysed. The vegetation of 14 permanent plots recorded before and after construction of the storm surge barrier was compared. Following completion of the barrier (1986) the vegetation of the middle marsh became invaded by species from the high marsh, while typical middle marsh disappeared. Higher marsh communities were not as strongly affected.

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Introduction

In 1986 the Oosterschelde storm surge barrier was completed. This tidal barrier was designed to combine optimal coastal defence with preservation of the estuarine Oosterschelde ecosystems. The barrier allows tidal movement in the estuary under normal conditions, while during periods of storms it can be closed. In the early 1980s a change of the cross-sectional area of the mouth of the estuary from 80,000 to 14,000 square metres was foreseen. Hence a reduction in tidal range by 23% and reduced high water levels were predicted (Smies & Huiskes 1981).

The reduction of the high water levels would lead to less frequent inundation of the saltmarshes. A strong response of the vegetation to this tidal reduction was predicted (Smies & Huiskes, 1981). This corresponds with the view that the species composition of saltmarsh vegetation is related to marsh level and determined by the ability of species to withstand seawater inundation (Long & Mason 1983, Van Diggelen 1988). However, Groenendijk *et al.* (1987) considered it unlikely that the zonation of saltmarsh vegetation would show a large scale change as a result of tidal reduction.

In this article a preliminary analysis is presented of the changes of saltmarsh vegetation following the reduction of the tides caused by the construction of the Oosterschelde storm surge barrier.

Material and Methods

The storm surge barrier

In 1976 it was decided to construct a storm surge barrier in the mouth of Oosterschelde estuary (Figure 1). Two additional structures, the Philipsdam and the Oesterdam, were projected in the rear of the estuary. The barrier consists of 65 prefabricated piers which support sliding steel gates that can be closed when necessary (Watson & Finkl 1991). In 1984 the piers were placed, while the sill beams and upper beams were installed between 1984 and 1986. The continuing placement of these structures in the mouth of the estuary gradually reduced its cross section from 80,000 square metres in the pre-barrier situation to 17,550 square metres in April 1986. The tidal exchange started to be influenced by mid 1985 when the cross section was reduced below 35,000 square metres. For various reasons a variable number of gates of the storm surge barrier have been closed during most of 1986 and the first months of 1987. This partial closure of the barrier lasted until the closure of the Philipsdam in April 1987. From then on the barrier has been operated as intended (Smaal & Boeije 1991).

Field sampling

The study was conducted at the Stroodorpepolder saltmarsh situated on the southern edge of the Oosterschelde estuary (Figure 1). Species composition of the vegetation was recorded in 1984 and 1990 in fourteen

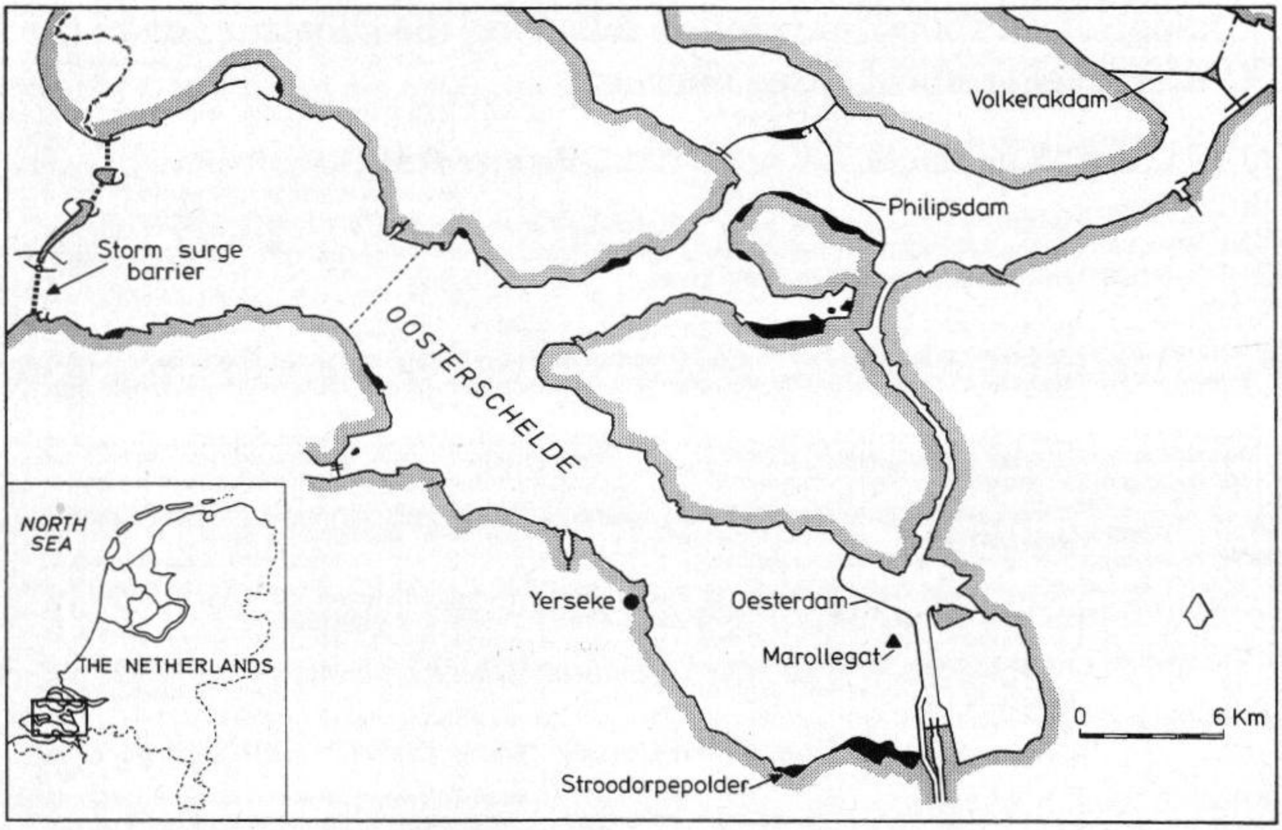

Figure 1 Map of the Oosterschelde estuary indicating the storm surge barrier, the Marollegat tide-gauge station and the Stroodorpepolder salt marsh study area

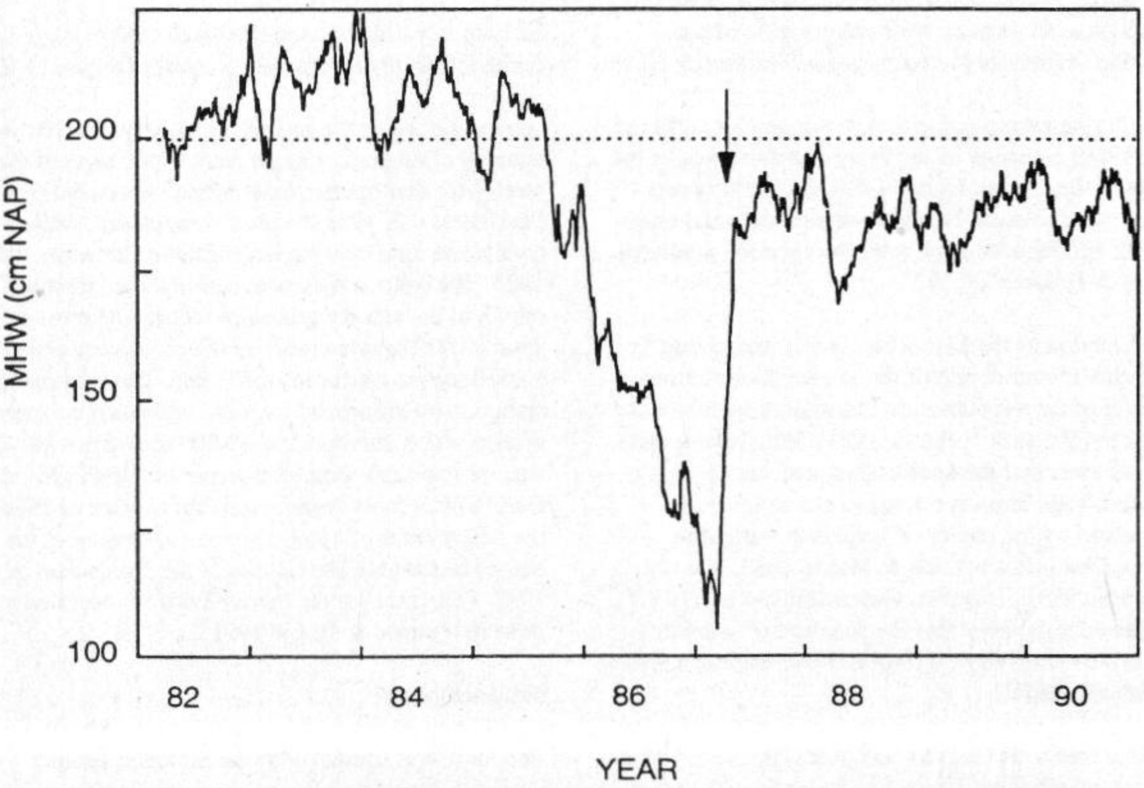

Figure 2a Running average (two lunar cycles) of high water (cm NAP) as recorded at the Marollegat tide-gauge station between 1982 and 1990

Figure 2b A 1987 SPOT satellite image of the storm surge barrier showing the restriction of tidal flow into the estuary on the flood tide; flow is from left to right in this image (© CNES/NLR 1987)

12 square metre permanent quadrats established in the 1960s by Beeftink and co-workers. The vegetation in the quadrats was representative for the vegetation at the Stroodorpepolder marsh as a whole. The cover and the number of individuals for each individual species was estimated by eye according to the combined scale of Doing-Kraft (1954) and transformed to numerical values according to Van Der Maarel (1979). Twinspan (Hill, 1979) was used to analyse the vegetation data. Elevation of permanent plots with respect to NAP (Dutch Ordnance Level) was measured in 1984 and 1990 using a theodolite. High water data recorded at the Marollegat station were obtained from Rijkswaterstaat (Dutch Coastal Engineering Service).

Results

Tidal movement, marsh level and inundation frequency

In the early 1980s mean high water fluctuated between 190 and 220 cm NAP (Figure 2). The high water levels started to decrease in the second half of 1985. The partial closure of the barrier in 1986 and the first months of 1987 further reduced the high water levels. As a consequence the marshes were not flooded during the growing season (April through September) of 1986 (Figure 3). From April 1987 onward the storm surge barrier functioned as intended: the barrier remained open during more than 99% of the tides, closing during extreme storm surges (> 3 metres + NAP) only. However, the barrier restricts the exchange of tidal waters between the estuary and the North Sea (Figure 2) and therefore causes a reduction of the high water levels. Since the elevation of the permanent quadrats changed very little (Table 1 and Table 2) one elevation range for the permanent plots is indicated in Figure 3. In 1990 the inundation frequency of the permanent plots had been reduced by 50% or more with respect to the situation in 1984 (Figure 3). Note that at present the lower plots have approximately the same inundation frequency as the highest plots in 1984.

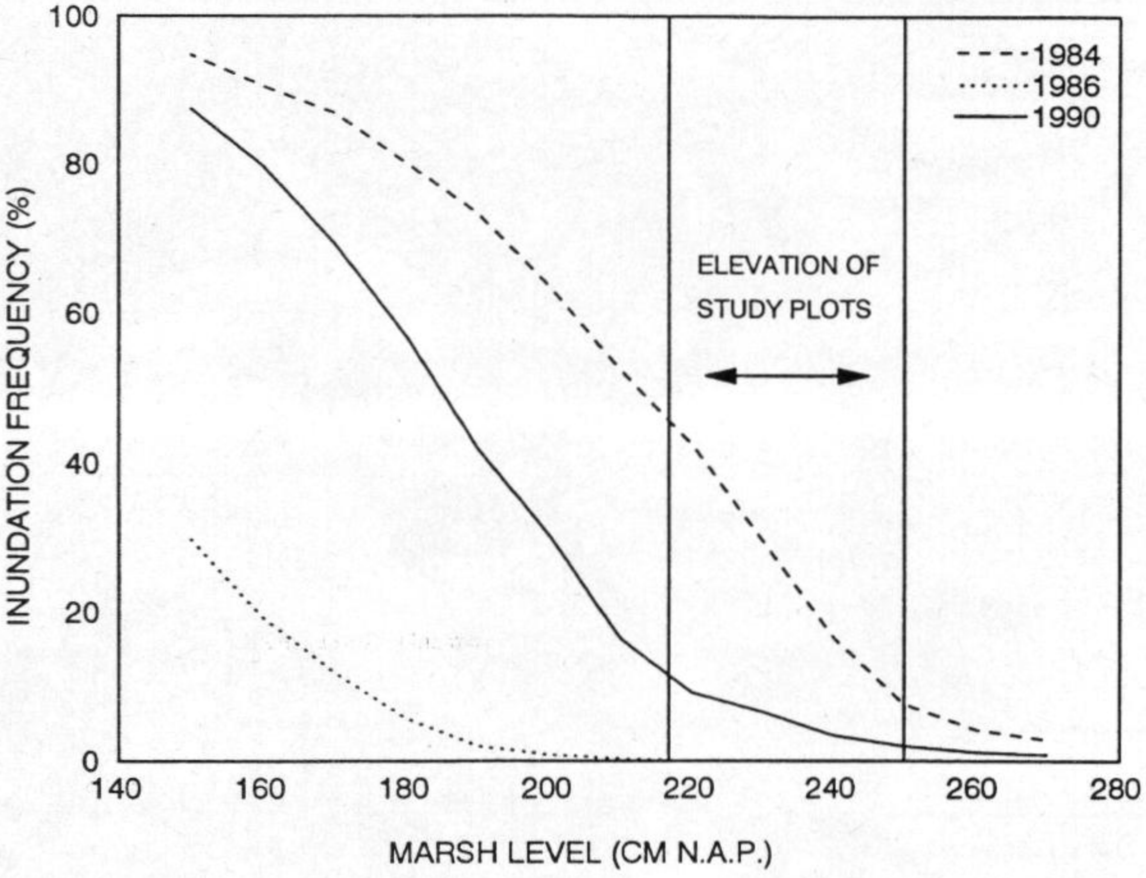

Figure 3 Inundation frequency during the growing season (April through September) in 1984, 1986 and 1990 for the Stroodorpepolder marsh as related to the elevation of the marsh surface (based on high water recorded at the Marollegat tide-gauge)

Change in the vegetation

Table 1 shows a tabular arrangement generated by Twinspan of the vegetation of the fourteen permanent quadrats in 1984. The main dichotomy for the species is indicated by the horizontal line, the main dichotomy for the plots by the vertical line. The top eleven species are more abundant in the plots to the left of the primary division. *Limonium vulgare* (Plant nomenclature follows Tutin *et al.* 1964–1980) and *Triglochin maritima* are differential species for this species group. Several of the other species in this group, such as *Spartina anglica* and *Puccinellia maritima,* are indicative for lower and wetter marsh sites (Westhoff & Den Held 1975). This corresponds with the fact that the plots are situated at lower elevations; all but one were situated below 236 cm NAP. The last four species are more abundant in the plots to the right of the primary division. *Elymus pycnanthus* is the best differential species for this group. Together with *Halimione portulacoides* and *Atriplex hastata, Elymus* indicates well drained conditions which prevail in the high marsh and on creek banks. All but one plot were situated above 235 cm NAP. These results suggest that the main dichotomy in the Table is related to site elevation. In the following we will therefore refer to the upper left hand corner of Table 1 as middle marsh species and plots, and to the lower right hand corner as high marsh species and plots.

In Table 2 the species composition of the vegetation in the same fourteen permanent plots in 1990 is presented. To allow comparison the order of species and plots used in Table 1 has been maintained. Table 2 shows that the vegetation of the middle marsh plots changed drastically. Many species present in 1984, like *Plantago maritima*, *Triglochin*, *Limonium* and *Spartina* had disappeared altogether in 1990. A group of middle marsh species (*Puccinellia*, *Salicornia*, *Suaeda maritima* and *Spergularia media*) remained present in the lower middle marsh plots (3, 4, 17, 51 and 61), while all middle marsh species but *Festuca rubra* disappeared from the three highest plots (5, 6 and 58). These middle marsh species have been replaced by upper marsh species: *Atriplex* and *Elymus* invaded the middle marsh plots. The vegetation of the high marsh plots changed less drastically. However, all middle marsh species and one upper marsh species disappeared (*Puccinellia*, *Salicornia*, *Suaeda*, *Spergularia* and *Aster tripolium*)

Table 1 The vegetation of the fourteen permanent plots at the Stroodorpepolder marsh in 1984, ordered and classified in a two way table by Twinspan. Horizontal and vertical line indicate the main dichotomy in the table for species and plots respectively.

Plot number	*17*	*61*	*3*	*4*	*51*	*58*	*6*	*5*	*10*	*18*	*8*	*15*	*7*	*71*
Elevation (cm + NAP)	*233*	*217*	*229*	*227*	*226*	*241*	*235*	*236*	*250*	*248*	*239*	*235*	*239*	*231*
Festuca rubra						4		1						
Glaux maritima						3	3	4						
Juncus gerardi							4	1						
Limonium vulgare	20	10	25	4	10	2	60	45	1					
Plantago maritima	1				2		15	3						
Triglochin maritima	75	50	30	30	5	5	5	35						
Spartina anglica				4	40	5	4	20						
Puccinellia maritima	10	65	45	20	2	75	10				1	3	2	10
Salicornia europaea	2		1	4	4		4	4	1		4			
Suaeda maritima	2		2	3	2	2	4	2	1		5	2		
Spergularia media	1	1	1	2	4		2	3	1		1			
Aster tripolium	2	10	3	5	4	10	5	10	3	3	10	4	3	4
Halimione portulacoides			2	10	75	40	5	1	40	1	30	20	65	95
Elymus pycnanthus						5	2		60	100	55	80	40	4
Atriplex hastata										3		1		
Atriplex littoralis														

Table 2 The vegetation of the fourteen permanent plots at the Stroodorpepolder marsh in 1990. Species and plots have been arranged in the same sequence as in Table 1. Horizontal and vertical line indicate the main dichotomy for species and plots in Table 1.

Plot number	*17*	*61*	*3*	*4*	*51*	*58*	*6*	*5*	*10*	*18*	*8*	*15*	*7*	*71*
Festuca rubra	1				1	15	15	35						
Glaux maritima														
Juncus gerardi														
Limonium vulgare														
Plantago maritima														
Triglochin maritima														
Spartina anglica														
Puccinellia maritima	2	10	10	25	1									
Salicornia europaea	1		2		1									
Suaeda maritima	50	10	90	50	2									
Spergularia media	2			2	2			1						
Aster tripolium	20	10	3	10	3									
Halimione portulacoides	2	20	1	10	90	1	5	1					5	45
Elymus pycnanthus	10	50		4	4	90	90	50	90	100	90	95	90	50
Atriplex hastata	2	2	15	4			2	20	5		1		1	
Atriplex littoralis											1		1	1

and *Halimione* decreased in the high marsh. *Elymus* now dominates the high marsh plots more strongly than before.

Discussion

The results indicate that tidal reduction was followed by a change of the vegetation towards a species composition characteristic for higher marsh levels. This corresponds to a downward shift of the vegetation zonation. Such a downward shift has been reported for marshes along the Baltic coast where land uplift causes a gradual reduction of the flooding frequency (Ericson 1981, Cramer & Hytteborn 1987). The opposite, an upward movement of the zonation pattern and regression of the vegetation has been reported for increased flooding (Beeftink 1979, 1987). Olff *et al.* (1989) were able to relate fluctuations in abundance of a number of saltmarsh species to year-to-year variation in tide level. In all of these studies a change in height of the flooding waters was followed by a change of the vegetation. This forms strong evidence for the hypothesis that the zonation of saltmarsh vegetation is determined by the height of the flooding waters.

Groenendijk *et al.* (1987) considered it unlikely that the zonation of saltmarsh vegetation would show large scale change in response to tidal reduction. There are several possible explanations for the discrepancy between the expectations of Groenendijk *et al.* (1987) and the results reported in this paper. First of all the predictions were based on short term experiments including middle marsh species only. The species which increased most strongly, *Elymus pycnanthus*, was not included in the experiments. Secondly the predictions concerned the impact of a sudden permanent reduction in high water level. The actual change of the high water levels has been more complex; after the marshes had not been flooded for one year (May 1986 till April 1987) they were flooded since April 1987 at a lower frequency than before 1986. The marsh sediments were excessively drained during the period of non-flooding, as demonstrated by the shrinkage of the sediments (de Leeuw *et al.* 1994). The absence of flooding may have affected the vegetation since the soils became desalinised and oxidized (Oenema 1988). This may have facilitated the expansion of upper marsh species into the middle marsh. Besides, the severe frost in the winters of 1984/85 and 1986/87 may have influenced the response of the vegetation. De Leeuw *et al.* (1994) reported that *Halimione* and *Spartina* were damaged by frost in 1985. This indicates that one has to be cautious before attributing the response of the vegetation to the permanent reduction of the tides since April 1987.

References

Beeftink, W.G. 1979. The structure of salt marsh communities in relation to environmental disturbances. *In*: *Ecological processes in coastal environments*, ed. by R.L. Jefferies & A.J. Davies, 77–93. Oxford, Blackwell.

Beeftink, W.G. 1987. Vegetation responses to changes in tidal inundation of salt marshes. *In*: *Disturbance in grasslands*, ed. by J. van Andel *et al.*, 97–117. Dordrecht, Junk.

Cramer, W., & Hytteborn, H. 1987. The separation of fluctuation and long-term change in vegetation dynamics of a rising seashore. *Vegetatio 69*: 157–167.

De Leeuw, J., Apon, L.P., Herman, P.M.J., de Munck, W. & Beeftink, W.G. 1994. The response of saltmarsh vegetation to tidal reduction caused by the Oosterschelde storm surge barrier. *Hydrobiologia 282/283*: 335–353.

Doing-Kraft, H. 1954. l'Analyse des carrés permanents. *Acta Botanica Neerlandica 3*: 421–425.

Ericson, L. 1981. Aspects of the shore vegetation of the Gulf of Bothnia. *Wahlenbergia 7*: 45–60.

Groenendijk, A.M., Spieksma, J.G.J., & Vink-Lievaart, M.A. 1987. Growth and interactions of salt marsh species under different flooding regimes. *In*: *Vegetation between land and sea*, ed. by A.H.L. Huiskes *et al.*, 236–258. Dordrecht, Junk.

Hill, M.O. 1979. *TWINSPAN – a FORTRAN program for analysing multivariate data in an ordered two-way table by classification of the individuals and attributes.* New York, Cornell University.

Long, S.P., & Mason, C.S. 1983. *Salt marsh ecology*. Glasgow, Blackie.

Oenema, O. 1988. Pyrite accumulation in salt marshes in the Eastern Scheldt, southwest Netherlands. *Biogeochemistry 9*: 75–98.

Olff, H., Bakker, J.P., & Fresco, L.F.M. 1988. The effects of fluctuation in tidal inundation frequency on a salt marsh vegetation. *Vegetatio,* 78: 13–19.

Smaal, A.C., & Boeije, R.C. 1991. Veilig getij, de effecten van de waterbouwkundige werken op het getijmilieu van de Oosterschelde. *Nota GWWS 91.088*, DGW/Directie Zeeland, Middelburg. (In Dutch).

Smies, M. & Huiskes, A.H.L. 1981. Holland's Eastern Scheldt Estuary barrier scheme: some ecological considerations. *Ambio 10(4)*: 158–165.

Tutin, T.G., Heywood, V.H., Burges, N.A., Moore, D.M., Valentine, D.H., Walters, S.M., & Webb, D.A. 1964–1980. *Flora Europaea.* Cambridge, Cambridge University Press.

Van Der Maarel, E. 1979. Transformation of cover-abundance values in phytosociology and its effects on community similarity. *Vegetatio* 39: 97–114.

Van Diggelen, J. 1988. *A comparative study on the ecophysiology of salt marsh halophytes.* Ph,D. Thesis, Free University Amsterdam.

Watson, I., & C.W. Finkl Jnr. 1991. State of the art storm–surge protection: the Netherlands Delta Project. *J. Coast. Res. 6*: 739–764.

Westhoff, V., & Den Held, A.J. 1975. *Plantengemeenschappen in Nederland.* Thieme, Zutphen.

The consequences of a one-year tidal reduction of 35% for the saltmarshes in the Oosterschelde (south-west Netherlands)

D.J. de Jong & Z. de Jong

De Jong, D.J., & de Jong, Z. 1995. The consequences of a one-year tidal reduction of 35% for the saltmarshes in the Oosterschelde (south-west Netherlands). *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 41–50.

During the final phase of construction of the storm surge barrier in the Oosterschelde (south-west Netherlands), the tidal range was reduced by around 35% for the duration of about one year. As a result, there was no, or very little, flooding of the lower parts of the saltmarshes for about 1 year and of the higher parts for about 1.5 years. This has had significant consequences for the soil, morphology and vegetation of the saltmarshes. This paper deals with these changes, which are summarised below and in Table 2. The possible consequences for the future resulting from these changes are also discussed.

Main consequences of the temporary absence of the flooding are:
- physical ripening, including cracking and settling of the soil, enhanced by severe frost in 1986 and 1987
- chemical ripening, involving oxidation of Pyrite in the formerly anaerobic soils (neutralised by $CaCO_3$) and desalination
- collapsing of saltmarsh creek banks, producing 'shoulders'
- sedimentation of mud in the creeks, which was only partly removed in the new situation
- enhanced saltmarsh cliff erosion, due to decreased cliff strength and to lowered tidal flats in front of the cliffs
- an invasion of nitrophylous species and glycophytes which are tolerant to salt
- a decline of *Spartina anglica*, in some places even completely.

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Introduction

The aim of the Delta Project, which started over thirty years ago, was to protect the south-western Netherlands against flooding. The project involved the construction of a system of dams along the North Sea coast, which has considerably shortened the coastline. The final stage of the project involved the construction of a storm surge barrier in the mouth of the Oosterschelde, in conjunction with two partitioning dams in the eastern part (Figure 1) (Knoester *et al.* 1984; Nienhuis & Smaal 1994). One consequence of the presence of this barrier is that the original average tidal range of 3.6 metres at Yerseke is reduced by about 12% to approximately 3.2 metres.

During the final stage of the construction of the storm surge barrier in 1986–87, however, the tidal range was temporarily reduced further by, at maximum, 35% to approximately 2.5 metres. This period of extra tidal reduction gradually began in the autumn of 1985, reached its peak in the autumn and winter of 1986/87 and ended abruptly in April 1987 when the last partitioning dam was closed. Since then the new tidal regime mentioned above has been maintained in the Oosterschelde (Figure 2) (Vroon 1994).

During the period with 35% tidal reduction, the vegetated parts of the saltmarshes were flooded either sporadically or not at all (Figure 2). From April 1987 onward the lower parts were flooded again at regular intervals. However, as the higher water levels occur mainly in the stormy winter period, the period without flooding of the higher parts actually continued until the autumn/winter of 1987/88. Thus the lower parts were not flooded for about 1 year, and the higher parts for about 1.5 years.

Moreover, the winter of 1986/87 was characterised by a long period of severe frost (Table 1). The absence of tidewater and its moderating effect enabled the frost to exercise an especially strong influence on the saltmarsh system.

Table 1 Averaged monthly air temperature (°C) and number of ice days (days with maximum temperature <0°C) at Vlissingen (data of the Royal Dutch Meteorological Institute)

	long term averaged		*1985*		*1986*		*1987*	
	°C	*ice days*	*°C*	*ice days*	*°C*	*ice days*	*°C*	*ice days*
January	3.1	3	-1.5	13	3.4	1	-1.3	13
February	3.1	2	-0.3	10	-3.6	12	2.5	0

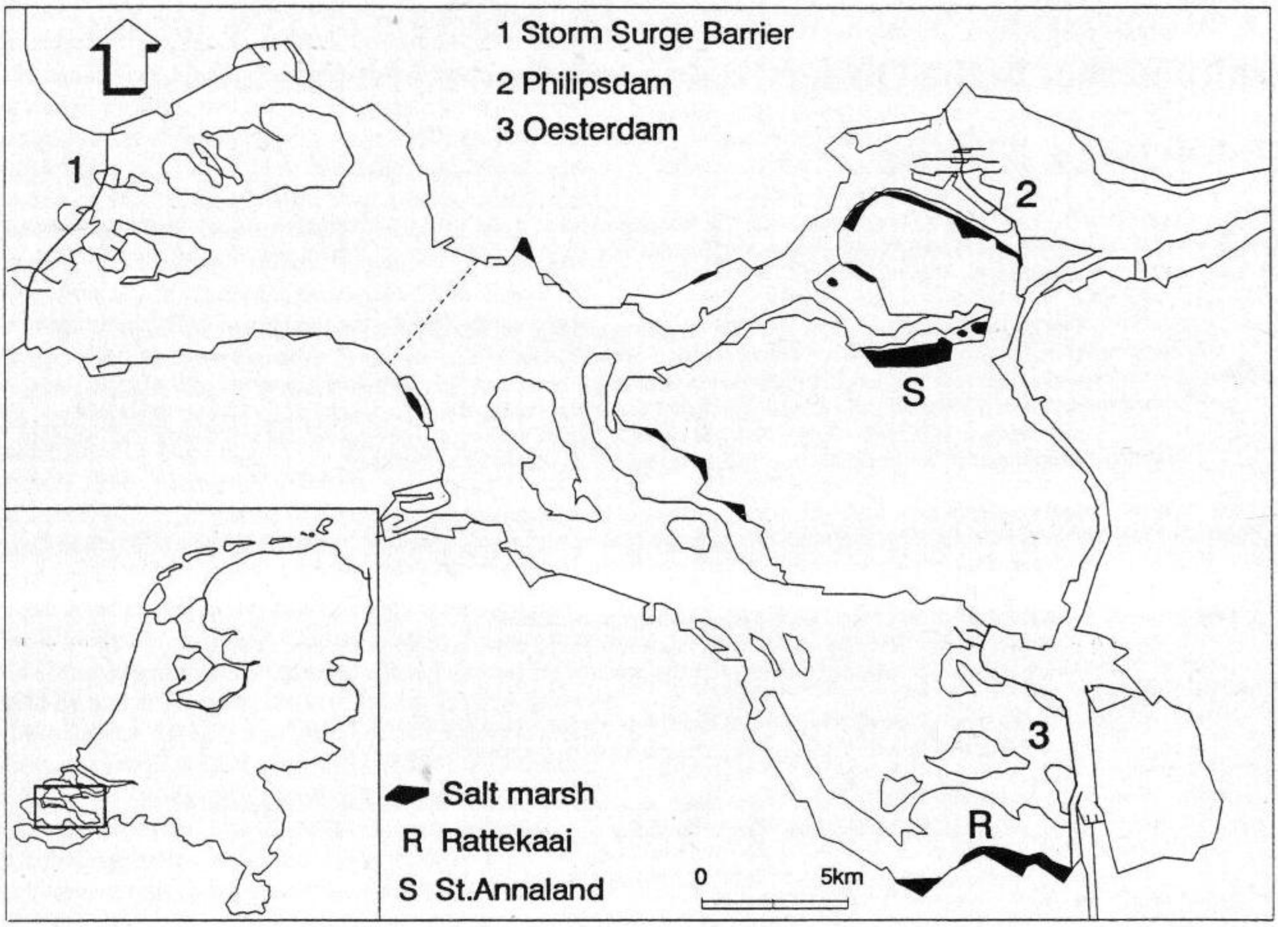

Figure 1 The Oosterschelde (south-west Netherlands) with the main dams and the main saltmarshes studied

In this paper the main consequences of this 1–1.5 year absence of regular flooding for the saltmarsh system will briefly be described. The next section briefly describes the methods used. Following this, a rough indication is given of the base situation in 1984/85 of the two saltmarshes studied. Next, the consequences of the period without flooding are briefly described, and soil, morphology and vegetation are considered in turn. These consequences are described until the summer of 1988, since for large parts of the saltmarshes the floodings did not begin again until the winter (i.e. the stormy season) of 1987/88. The final section gives a summary of the main differences between the new situation in 1988 and the original situation in 1984/85 and discusses possible long term consequences of these. The study focuses on two saltmarshes, namely those at St Annaland and Rattekaai (Figure 1), but observations were also carried out on other saltmarshes in the area.

Methods

The observations and measurements described in this paper are mainly carried out on two saltmarshes, but general observations on other saltmarshes have been made as well. The measurements include the following items:

1. Vegetation measurements have been carried out in 28 transects of about 10–20 metres, being a cross-section from the basin towards the creek. The vegetation is described yearly by means of percentage cover in adjacent plots of 0.5 × 0.5 metres from 1984 onwards.
2. Cliff erosion has been measured in fourteen profiles perpendicular to the saltmarsh edge. The measurements have been carried out by means of a theodolite, three to four times a year from 1982 onwards.
3. Settling was measured in the vegetation transects mentioned earlier by means of a theodolite; the measurements were carried out about two or three times a year from 1985 to 1990.
4. Soil structure was investigated in the vegetation transects. In 1990 calcium carbonate content and the pH value to a maximum depth of about 0.6 metres were surveyed. This was done by taking soil samples in subsequent layers of 5 cm, which were further analysed in the laboratory for calcium carbonate (by adding HCl and measuring the liberated CO_2) and pH (KCl).

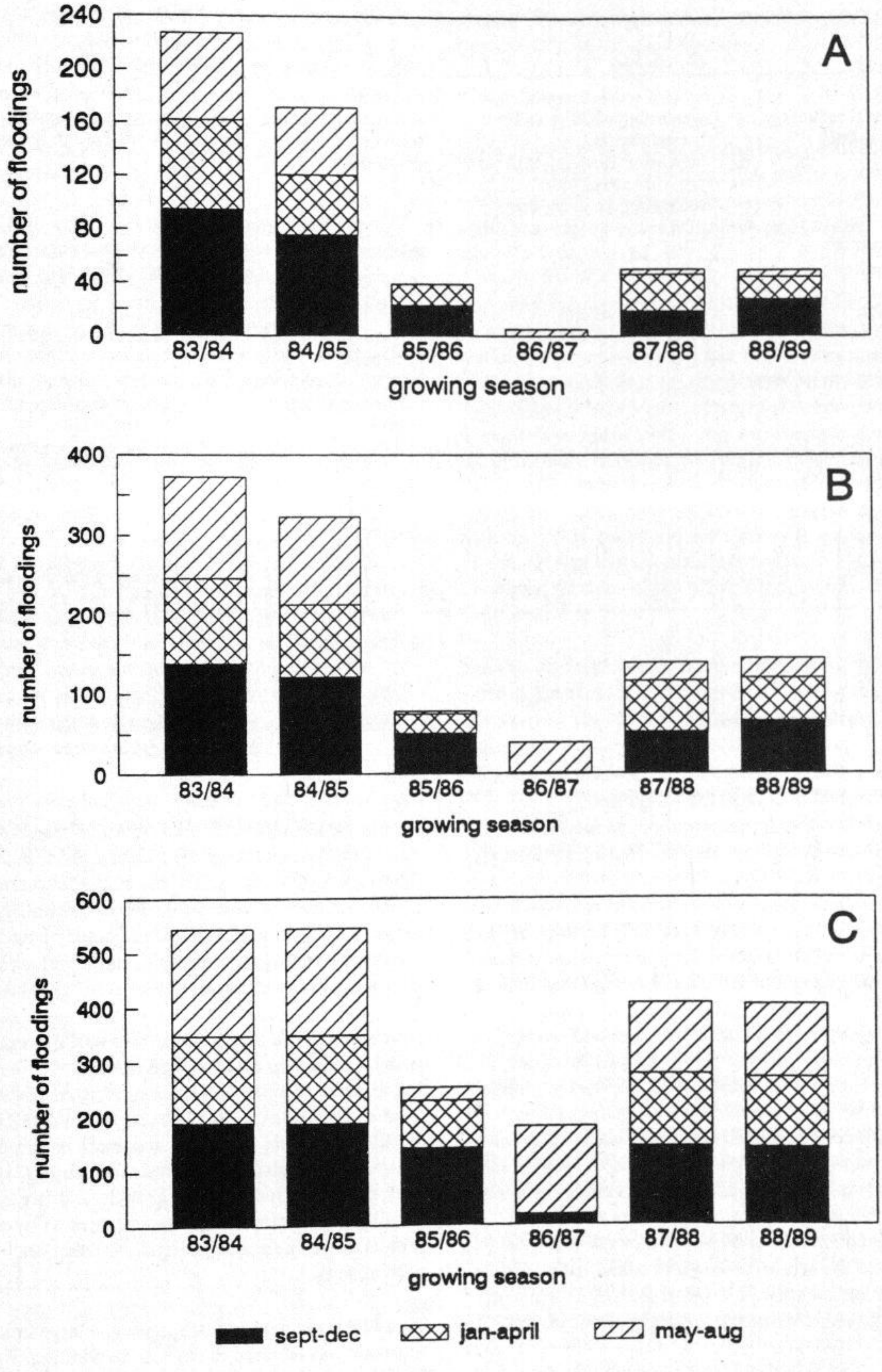

Figure 2 Flooding frequency in six subsequent growing seasons for three elevation-zones of the saltmarsh: A, high marsh (= ± 20 cm above MHW in the pre-barrier period); B, middle marsh (= ± around MHW); C, low marsh (= ± 30 cm below MHW). (Growing season: e.g. summer 1984 = September 1983–August 1984, etc.) The figure shows clearly the strong decrease in flooding frequency during the interim period (growing season 1985/86 and 1986/87).

Table 2 Tabulated summary of the main phenomena observed in the saltmarshes from 1986–1988

chronology	*morphology*	*soil*	*vegetation*
spring 1986 65% tidal reduction (no flooding)	no cliff erosion weakening of cliffs and creek banks increasing elevation difference basin–creek banks sedimentation in creeks	desiccation soil ripening settling desalination	decrease original vegetation increase nitrophytes introduction salt tolerant species
winter 1986/87 (severe frost)		increased ripening	strengthened impact on plants (root freezing)
April 1987 88% tidal reduction (flooding lowest parts)	increased cliff and creek bank erosion	low saltmarsh: ripening processes ± stops middle high saltmarsh: ripening processes continue	low saltmarsh: recovery vegetation middle high saltmarsh: further decrease → → barren areas increase of glycophytes
winter 1987 (flooding all parts)		middle high saltmarsh ripening processes diminish	
summer 1988			maintaining barren areas disappearance of glycophytes

Base situation in 1984/85

The saltmarshes are mainly low and middle saltmarshes, i.e. up to a level of about Mean High Spring Tide (minimum flooding frequency about twenty times a year). The low basins are characterised by an (almost) unripened soil and a vegetation dominated by *Spartina anglica* (nomenclature follows Tutin *et al.* 1964–1980). The middle basins have a soil which is half ripened to a depth of several decimetres. They are covered with a more mixed vegetation which includes species such as *Triglochin maritima*, *Puccinellia maritima* and *Limonium vulgare* in addition to *Spartina anglica*. The vegetation on the low/middle creek banks consists mainly of *Halimione portulacoides* and on the middle/high creek banks of species such as *Festuca rubra, Artemisia maritima* and *Elymus pycnanthus.* The soil is generally sandy loam to clayey loam in the basins and more sandy on the creek banks. The average depth of the underlying sandy layer ranges from about 0.7 metres at St Annaland to 1.5 metres at Rattekaai. The edge of the saltmarsh largely consists of a cliff subject to annual erosion in the order of 0.1–1 metres (Figure 3). Accretion of the saltmarshes occurs only locally.

The total area of saltmarshes in the Oosterschelde is 1,725 ha of which 1,080 ha was closed off from the tide by the partitioning dams, leaving 645 ha in the remaining tidal part of the Oosterschelde.

Consequences of the absence of the tide

Soil

Due to the absence of flooding the soil became extremely dry, and large deep cracks (up to 0.5 metres deep) appeared in the soil of the basins as early as the summer of 1986. These cracks led to considerable aeration of the soil, which had previously been completely or largely anaerobic, and therefore to a rapid physical and chemical ripening.

As a result of the physical ripening settling occurred, which reached up to 0.1 metres in the clayey basins, but remained limited to a few centimetres on the sandier creek banks. Also because of the cracking the stratified structure of the soil was more or less disturbed. Due to the combination of physical ripening, severe desiccation and reduced root growth (and in places possibly significant burrowing activities by mice and rabbits as well), the soil strength also declined significantly.

As a result of the chemical ripening, a large amount of nutrients was released in the soil. In addition, Pyrite (FeS_2) was oxidised, in some areas on a large scale, e.g. the Rattekaai saltmarsh:

$$4\ FeS_2 + 15\ O_2 + 10\ H_2O \rightarrow 4\ FeOOH + 8\ H_2SO_4$$

(Houtekamer 1991, see also Oenema 1988, Vranken *et al.* 1990.)

In many places, e.g. Rattekaai, this process of pyrite oxidation was visible on the saltmarsh as a significant oozing out of iron and elementary sulphur at the creek-heads. In addition, part of the calcium carbonate present in the soil neutralised the sulphate released:

$$CaCO_3 + H_2SO_4 \rightarrow CaSO_4 + H_2O + CO_2.$$

(Van Breemen 1973.)

The ripening processes occurred in every basin up to depths of about 0.4–0.5 metres and led to the saltmarsh soils attaining a significant degree of ripening over large areas as early as the summer of 1986. Considering the rapid introduction of salt-tolerant glycophytes, desalination of the soil also occurred in the summer of 1986. This process occurred most quickly and to the greatest degree on the creek banks.

In addition, during the winter of 1986/87 the physical ripening was further strengthened by the period of severe frost in February 1987.

After the lowest parts had again been flooded in the summer of 1987, the ripening and desalination processes more or less stopped there. On the higher parts of the saltmarshes, however, there was still no flooding during that summer and as a result the ripening processes continued. In view of the further increased number of salt tolerant glycophytes, desalination continued as well on the higher parts during this period.

It was only after the higher parts had been flooded again significantly in the autumn of 1987 that the processes of ripening were strongly reduced and desalination stopped. A situation had arisen in which the soil of the saltmarshes had generally ripened more than was in keeping with the new tidal range of 88%; a nearly ripened soil was flooded again.

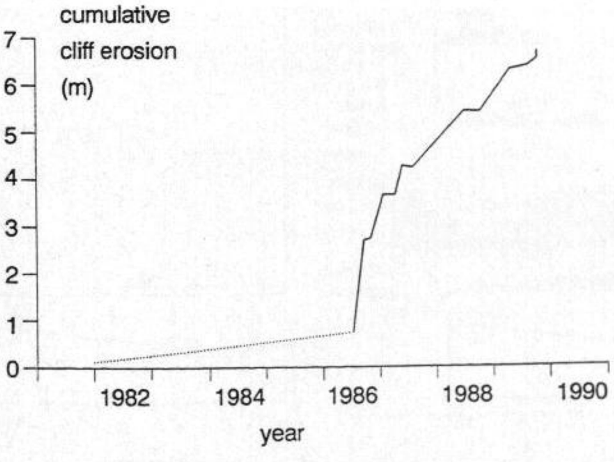

Figure 3 Cumulative cliff erosion in the period 1982–1990 of a cliff at the St Annaland saltmarsh; the erosion in the period 1982–1986 is based on only two measurements, in each year respectively

Morphology

In the period with a 35% reduced tide, the tide water tended to reach the saltmarsh cliffs rarely, if at all, leading to little or no wave impact on the cliffs. As a result the erosion of the saltmarsh cliffs almost stopped completely in this period.

The process of reduction in soil strength of the cliffs resulting from desiccation, mentioned above, was further strengthened by the fact that the vegetation also declined as a result of the dry period; plant roots, depending on the species, strongly influence the soil strength (van Eerdt 1985). The result was that when the cliffs were flooded again with tidewater from April 1987 onwards and were saturated with water to a significant degree, they became too heavy for the residual soil strength and more liable to wave erosion. The result was that by gravity, whether or not helped by waves, sections broke off or slipped down on a large scale.

Besides, in many places the elevation of the tidal flat in front of the cliff had been lowered, causing the boundary of the clayish saltmarsh soil with the underlying tidal sands to become an outcrop. This led to an undermining and with that a weakening of the cliffs and, due to that, to increased erosion of the cliffs.

From the summer of 1987 onwards, all this led to more substantial cliff erosion in many places than before 1986, having an order of magnitude of 1–2.8 metres per year, a process which has continued in 1990, see Figure 3.

The same decline in soil strength also occurred on the creek banks, creating a high risk of parts of them sliding down or breaking off; however, here it was mainly because of the current in the creeks. As a result of the erosion of the creek banks in many places 'shoulders' arose in the creeks, i.e. lowered edges near the creek bank (see Figure 4) on which plants characteristic of low saltmarshes (such as *Suaeda maritima* and *Spartina anglica*) established.

Another process which occurred in the period of a greatly reduced tide was a significant degree of local sedimentation of fine material in a number of saltmarsh creeks. This was caused by the fact that the tidewater no longer reached the basins, but remained in the creeks. As a result, during slack high water the water stagnated in the creeks and the entrained suspended sediment settled to the bottom. On account of the small quantity of water in relation to the size of the creek system, the ebb current velocities were too low to resuspend the

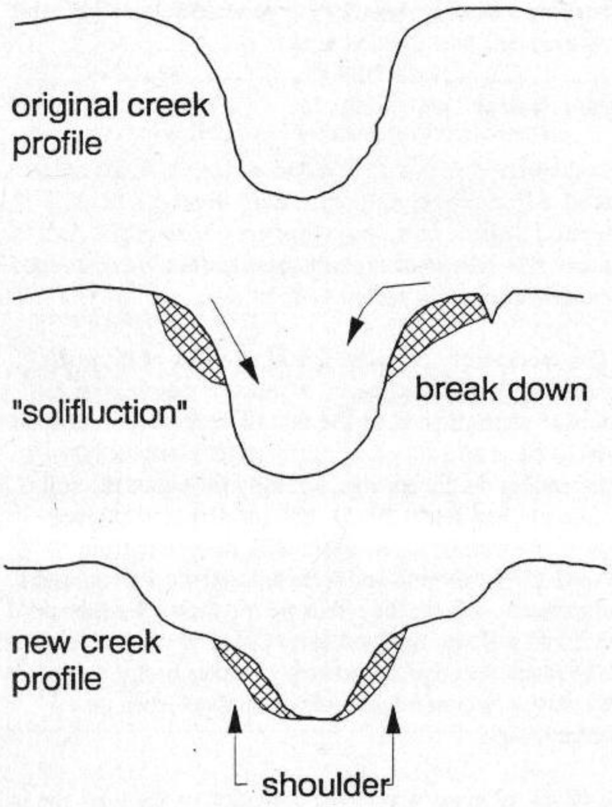

Figure 4 Scheme of the change in creek morphology

deposited sediment. Since the sharp decline in current velocities in the Oosterschelde meant that there was virtually no more sand in the water, this suspended sediment consisted almost entirely of fine silt, mainly from the mudflats lying in front. This process of the silting up of the saltmarsh creeks therefore occurred mainly in the saltmarshes where there was a large mudflat area lying in front. When the tidal range increased again, after April 1987, and the current velocities in the creeks therefore also increased again, part of the deposited sediment was gradually eroded.

Vegetation

The consequences of the significant tidal reduction for the vegetation varied widely but largely corresponded with the developments in a saltmarsh at which the influence of the tides had just been totally abandoned.

In the summer of 1986, the original vegetation cover of the basins declined quite considerably, especially *Spartina anglica*. It was replaced by annual nitrophytes – *Suaeda maritima* in the lower parts of the basins and *Atriplex littoralis* and *A. prostrata* in the higher parts. *Aster tripolium* also spread widely, see also Figures 5 and 7. Since many nutrients were released, these species sometimes grew very large: *Suaeda maritima* grew to

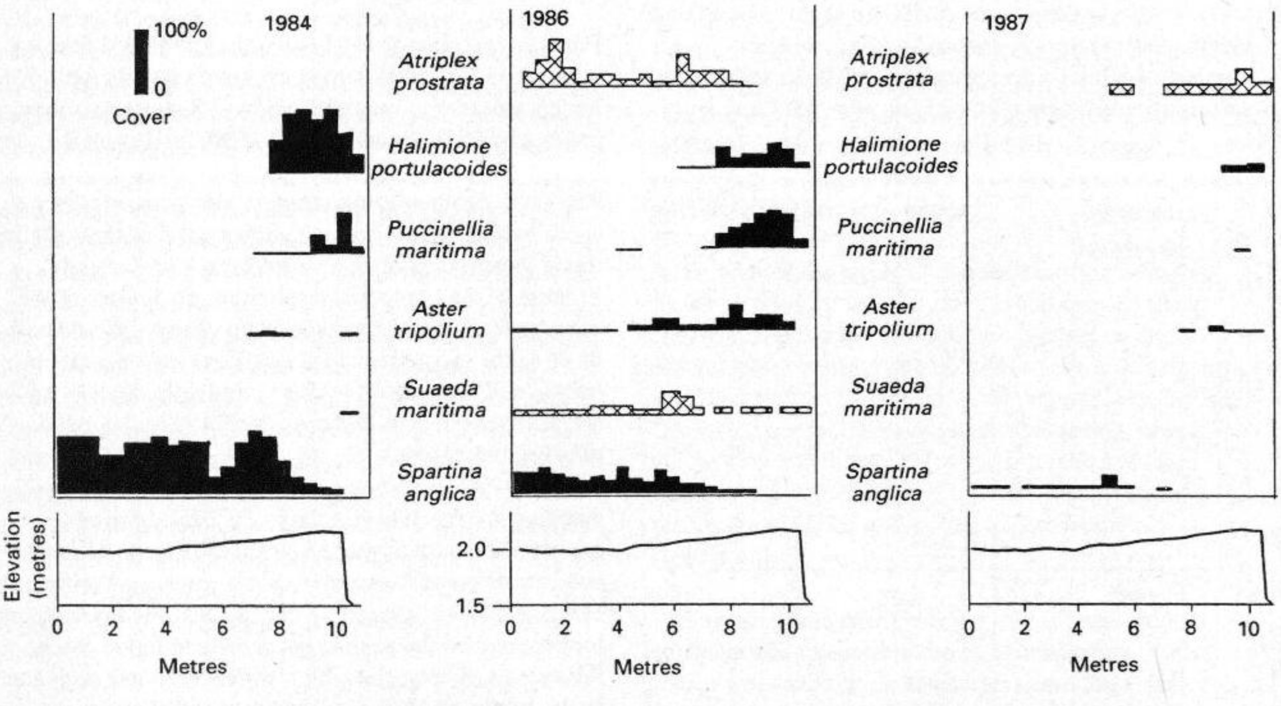

Figure 5 The vegetation development in the middle high part of the Rattekaai saltmarsh, showing the invasion of nitrophites in 1986 and the dieback of the vegetation in 1987 (below the elevation profile of the transects is indicated relative to Mean Sea-level; the presence of the plants is indicated as percentage cover of the soil) (MHW–1984 = 2.1 metres; MHW–1987 = 1.8 metres)

about 0.7–1 metre and *Aster tripolium* reached a height of 2 metres in places, some two or three times higher than usual. On the low/middle creek banks *Halimione portulacoides* maintained its position fairly well and on the middle/high creek banks *Elymus pycnanthus* grew very well. In places the vegetation was interspersed with glycophytic species which are to some extent salt-tolerant, and that are also characteristic of a newly reclaimed saltmarsh, such as *Matricaria maritima* ssp. *inodora* and *Sonchus arvensis.*

In the summer of 1987, the lowest parts were reflooded from time to time. As a result, the *Spartina* vegetation of the lowest saltmarsh, which had been able to maintain its position in 1986, also survived in 1987, although the plants tended to grow less dense (Figure 6) and less tall than they had done before.

On the higher parts, however, during the summer of 1987 the processes of desalination and soil ripening continued. In that summer on the middle-high part of the Rattekaai saltmarsh the *Spartina* and *Halimione* vegetation died off almost completely over large areas, most probably as a result of these continuing soil processes, in combination with either the severe frost in February 1987 (which had a much greater effect on the saltmarsh vegetation, due to the absence of the moderating effect of the tidewater) or the presence of a thick layer of debris which prevented seeds from germinating. As no other species had taken their place, these areas remained almost bare throughout the summer (Figure 5). During the following summer too (1988), there was still hardly any vegetation to be found.

On the other parts of the saltmarshes vegetation did, however, remain, though in many cases the original species had significantly declined in cover, while the space which became available was largely occupied by the above mentioned species *Aster*, *Suaeda* and *Atriplex*,

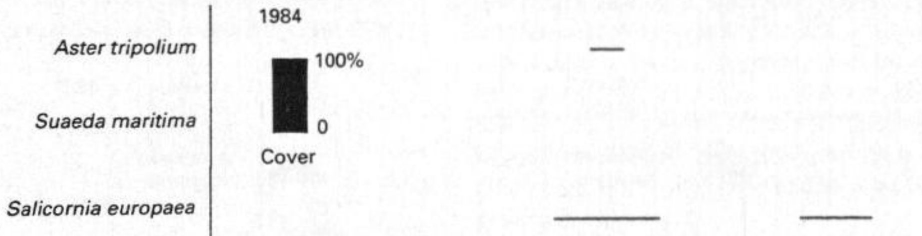

Figure 6 The vegetation development on the lowest saltmarsh (see also Figure 5) (MHW–1984 = 2.1 metres; MHW–1987 = 1.8 metres)

see Figure 7. A noteworthy fact, however, is that the last two species, in contrast to the extreme individual growth in 1986, in 1987 frequently occurred in high densities, but remained very small, at 2–5 cm (max 10 cm). Possible causes include exhaustion of soil nutrients, the plants having adverse effects on one another, a reduced pH and major fluctuations in salinity, but these factors have not been studied further. The 1986 elevation limit described between *Suaeda* and *Atriplex* was also also found in 1987, and more or less corresponded to the level of Mean High Spring Tide in that summer.

Many more glycophytes were found in 1987 than in 1986; species such as *Solanum nigra* and *Sonchus arvensis* were found regularly, but *Plantago major* and *Cirsium arvense* and even *Sambucus nigra* were also present. These glycophytes were found also in 1987 mainly on and near the creek banks.

The main causes for the large changes in the vegetation cover and plant composition are ascribed to the direct influence of the absence of flooding as well as the concomitant changes in soil conditions, e.g. the changes in aeration, ripening and the oxidation of Pyrite, which affects the growth of the saltmarsh plants (see Van Diggelen 1988).

It is striking that the main changes in the vegetation occurred on the low and middle saltmarshes. This is probably due to the fact that the relative changes in flooding frequency were greatest in this zone.

The main changes observed during the study period are summarised in Table 2.

The new baseline situation in 1987/88

As a result of the biotic and abiotic developments described above, a new baseline situation had arisen in 1987/88, which differed in many aspects from the original baseline situation of 1984/85. Several aspects may have significant repercussions on future developments in the saltmarshes. The five main aspects are discussed below.

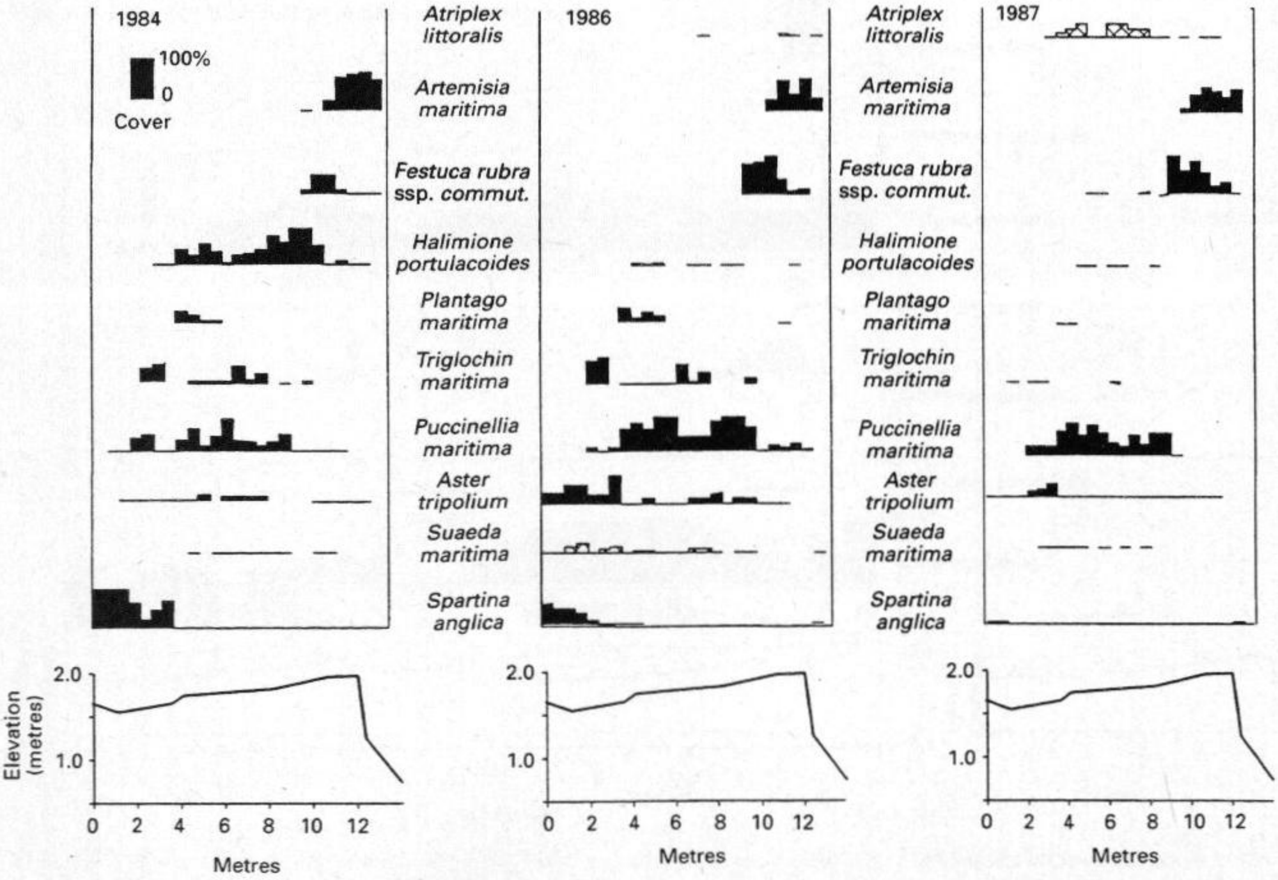

Figure 7 The vegetation development in the middle high part of the St Annaland saltmarsh (10 main species; see also Figure 5) (MHW–1984 = 1.9 metres; MHW–1987 = 1.6 metres).

1. The physical and chemical ripening of the soil has been much more marked and has penetrated much deeper than would have been the case without the period of 35% tidal reduction. An important consequence is the disturbance of the structure of the soil, which may have implications, *inter alia*, for the geohydrological condition of the saltmarsh. For instance, it may lead to a more rapid infiltration and a more rapid and deeper percolation of floodwater and rainwater. This may result in shortages of moisture in the future during dry periods, especially in the basins. Furthermore, faster and more extreme fluctuations in soil salinity are also conceivable.

2. Pyrite has been oxidised, in some areas on a large scale. This had led to a demand on the carbonate present, which acts as the first buffer for the neutralisation of the sulphuric acid released. If this supply of carbonate is used up in basins before all the sulphuric acid has been neutralised, the basins may rapidly acidify down to pH values of 3 (Van Breemen 1973).

During studies on five saltmarshes by Houtekamer (1991) in 1990, it was found that in about 65% of 50 basins investigated, the calcium carbonate content in the rootzone declined to less than 2% and quite often even to almost zero, see Figure 8. This means that these basins bear a significant risk of reduced pH values. In addition, in 1990, during measurements in three of seven basins further studied, pH values of 6–6.5 were found (normally 7.5–8), which shows the possible threat of acidification of the soil (Figure 8). In 1986 a pH value of about 3 was found in one basin (Vranken *et al.* 1990), but this low value was not measured again in 1987 (Vranken *et al.* 1990) and 1990 (Houtekamer 1991). If acidification actually occurs, it will have serious consequences for the vegetation, leading to its disappearance, locally or over fairly large areas, and temporarily or permanently.

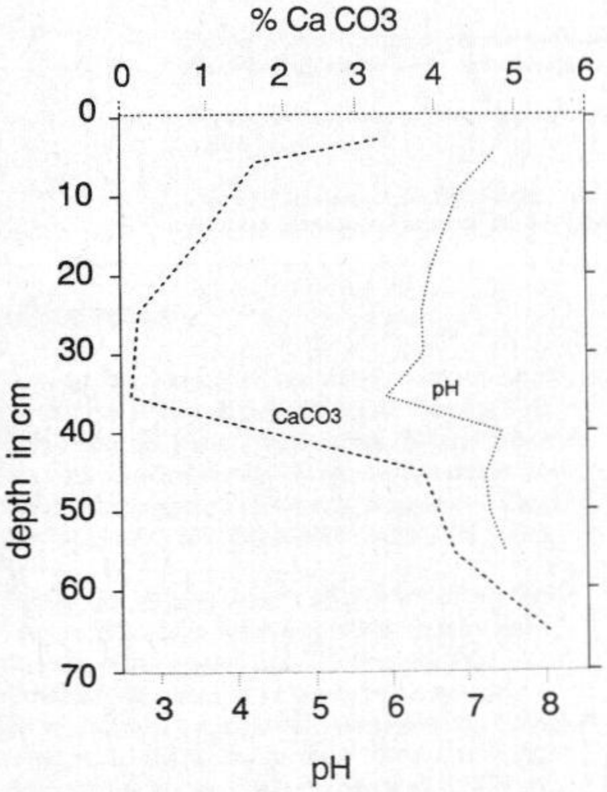

Figure 8 Calcium-carbonate content and pH in the soil at different depths

3. The soil strength of cliffs and creek banks has declined significantly. Since it is unlikely that the soil strength will recover and return to the original values, this may mean that the current increased level of cliff erosion will also continue in the future. This may also apply to the current level of erosion of the creek banks.

4. The *Spartina* vegetation which used to occupy the low basins on a large scale has largely or completely disappeared. Although at first this often gives saltmarshes a desolate appearance, it may ultimately have a positive effect. The significant and rapid decline of *Spartina* has created space for other species to become established in the newly arisen habitats. Otherwise they would have had to 'invade' this persistent vegetation.

5. The difference in elevation between basin and creek bank have increased. This has led to a steepening of the transition zone, causing a decrease of the width of this elevation and vegetation gradient. This is particularly evident in the case of the low and middle creek banks.

De Jong *et al.* (1994) and De Jong & Van de Pluym (1994) are referred to for further information on the developments in the new situation 1988–1991.

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References

De Jong, D.J., & Van de Pluym, A.M. 1994. Consequences of a tidal reduction to the salt marsh vegetation in the Oosterschelde (The Netherlands). *Hydrobiologia 282/283*: 317–333.

De Jong, D.J., de Jong, Z., & Mulder, J.P.M. 1994. Consequences of a reduction of the tidal range for some abiotic aspects on saltmarshes in the Oosterschelde (The Netherlands). *Hydrobiologia 282/283*: 303–316.

Houtekamer, N.L. 1991. Schorren in de Oosterschelde: onderzoek naar het verzuringsrisico van schorbodems door de oxidatie van pyriet (Salt marshes in the Oosterschelde: research into acidification risk of the soils by oxidation by Pyrite). State University Utrecht, *Geographical Institute, Report GEOPRO 1991.05* (IRO) (in Dutch).

Knoester, M., Visser, J., Bannink, B.A., Colijn, C.J., & Broeders, W.P.A. 1984. The Eastern Scheld Project. *Wat. Sci. Tech. Vol. 16:* 51–78.

Nienhuis, P.H., & Smaal, A.C. 1994. The Oosterschelde estuary, a case study of a changing ecosystem: an introduction. *Hydrobiologia 282/283*: 1–14.

Oenema, O. 1988. *Early diagenesis in recent fine-grained sediments in the Eastern Scheldt.* Ph.D. Thesis, University of Utrecht.

Tutin, T.G., Heywood, V.H., Burges, N.A., Moore, D.M., Valentine, D.H., Walters, S.M., & Webb, D.A. 1964–1980. *Flora Europaea*. Cambridge, Cambridge University Press.

Van Breemen, N. 1973. Soilforming processes in acid sulphate soil. *In*: *Proc. of the Int. Symp. on Acid Sulphate Soils,* ed. by H. Dost. Wageningen, ILRI-publ. 18.

Van Diggelen, J. 1988. *A comparative study on the ecophysiology of salt marsh halophytes*. Ph.D. Thesis, Free University of Amsterdam.

Van Eerdt, M.M. 1985. The influence of vegetation on erosion and accretion in salt marshes of the Oosterschelde, The Netherlands. *Vegetatio 62*: 367–373.

Vranken, M., Oenema, O., & Mulder, J.P. 1990. Effect of tide range alterations on salt marsh sediments in the Eastern Scheldt, S W Netherlands. *Hydrobiologia 195*: 13–20.

Vroon, J. 1994. Hydrodynamic characteristics of the Oosterschelde in recent decades. *Hydrobiologia 282/283*: 17–27.

Long-term changes in the intertidal macrobenthic fauna at eight permanent stations in the Oosterschelde – effects of the construction of the storm surge barrier: preliminary results

J. Seys, P. Meire, J. Coosen & J. Craeymeersch

Seys, J., Meire, P., Coosen, J., & Craeymeersch, J. 1995. Long-term changes in the intertidal macrobenthic fauna at eight permanent stations in the Oosterschelde – effects of the construction of the storm surge barrier: preliminary results. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 51–60.

In the period 1983–1989, the macrozoobenthos of eight permanent stations in the Oosterschelde estuary (south-west Netherlands) was studied to evaluate the effects of changed environmental conditions during and after the construction of a storm surge barrier in the mouth of this estuary and two secondary dams in the eastern part and the northern branch, respectively.

In this preliminary analysis of the temporal changes in the benthic communities, no long-term changes in total biomass, biomass of the trophic groups and species richness over the study period could be demonstrated for most of the stations. However, at species level, a substantial decrease of *Hydrobia ulvae* was observed at five of the eight stations. This might be related to increased wave-action around mean sea water level and removal of silt from the sampling stations, caused by a reduction in tidal range, tidal volume and current velocities after the completion of the storm surge barrier. Another possible explanation of the gradual collapse of the *Hydrobia* population in the Oosterschelde is its short life span and the absence of good recruitment since 1985.

Strong seasonal fluctuations were caused by several factors such as three successive severe winters, changes in predator/prey numbers (*Nephtys hombergii*/*Scoloplos armiger*), and reduced inundation times. Variability in time proved to be larger for filter feeders than for omnivores/predators and deposit feeders.

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Introduction

Over the last thirty years, benthic invertebrates have been studied intensively all over the world. They are a vital food source for higher trophic levels (fishes, birds) and react upon environmental changes, which makes them suitable for long-term monitoring programmes (Agger 1989; Kohnke 1989; Reise 1989; Wolfe *et al.* 1987).

In these monitoring studies, large seasonal, year-to-year and long-term variations in both density and biomass of species are usually observed. This variability is caused by differing environmental factors, such as e.g. climate, hydrodynamics, biological interference and food availability (Beukema 1989; Buchanan & Moore 1986; Pearson *et al.* 1986). Additional variability can be induced by man (fisheries, bait digging, pollution, dredging and engineering works).

After a disastrous stormflood in south-west Netherlands in 1953, extensive coastal engineering works have been carried out to protect the area from new floodings. In the Oosterschelde, the construction of a storm surge barrier was completed in 1986, and two compartment dams were completed respectively in 1986 (Oesterdam) and 1987 (Philipsdam) (Figure 1). Although these constructions aimed at maintaining the ecological characteristics and values of the estuary, some changes occurred (Rijkswaterstaat, 1991). Being a unique opportunity to study the effects of coastal engineering works on the ecosystem, a comprehensive research programme was started (Smaal *et al.* 1991). In this paper, some preliminary results on temporal variation of the intertidal macrobenthic fauna of the Oosterschelde and the possible impact of the hydraulic engineering works are discussed. Although mussels (*Mytilus edulis)* and cockles (*Cerastoderma edule*) form a major part of total benthic biomass in the Oosterschelde, they will not be dealt with in this paper: mussels are absent at the

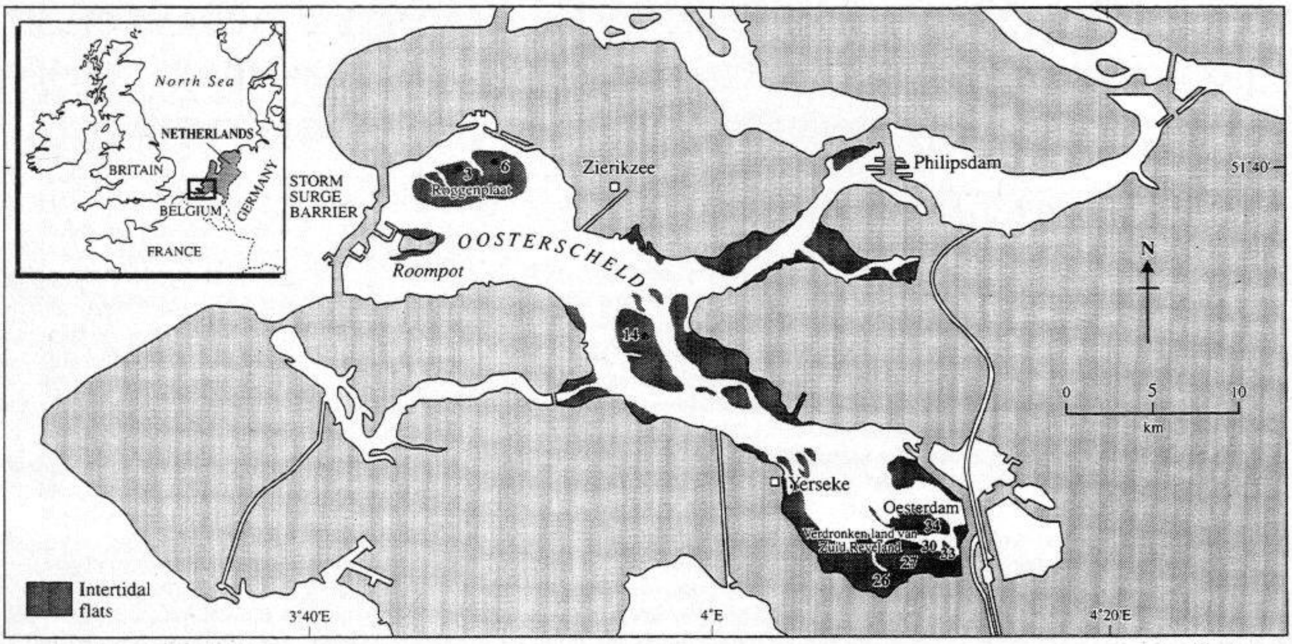

Figure 1 Location of the eight monitoring stations in the Oosterschelde

sampling stations and the results of cockles will be presented elsewhere (Coosen *et al.* 1994).

Methods

Study area and study period

The Oosterschelde can be characterised as an estuary with a low degree of pollution. The freshwater input is small compared to the tidal volume, hence salinity is high and stable (Wetsteyn *et al.* 1990). The construction of the storm surge barrier in the mouth and compartment dams in the eastern part and northern branch of the estuary has resulted in a further decrease of the freshwater load (from 70 $m^3 s^{-1}$ to 25 $m^3 s^{-1}$), a smaller tidal range (from 3.7 metres to 3.25 metres) and lower maximal flow velocities (from 1.5 $m s^{-1}$ to 1.0 $m s^{-1}$). Seston and nutrient loads decreased and transparency of the water increased (Wetsteyn & Bakker 1991). As a result, the total annual primary production remained unchanged in most parts of the estuary. The chlorinity has slightly increased: in 1988–1989 it was about 17–18 g Cl l^{-1} in the mouth and 16–17 g Cl l^{-1} in the eastern and northern part of the estuary. In the pre-barrier system chlorinity amounted to 16.5 g Cl l^{-1} and 14.5–15 g Cl l^{-1} respectively (Wetsteyn *et al.* 1990). The reduction of the tidal amplitude has increased the effects of wave action around mean sea-water level (MTL) with an average up to 80%. This has resulted in the removal of silt to the deeper gullies (Rijkswaterstaat 1991). As a result, median grain size is lower now and silt percentages in the upper sediment layer have decreased on six of the eight sampling stations (Table 1) (Seys *et al.* 1992).

Table 1 Characterisation of the eight monitoring stations: sampling station, tidal flat, tidal elevation (MTL + cm), median grain size (phi-units) and silt percentage (< 53 μ) before (1983) and after (1989) the completion of the major engineering works

station	*elevation*	*median grain size*		*silt percentage*	
		1983–84	*1989*	*1983–84*	*1989*
3	+ 36	2.65	2.31	4.8	0.9
6	+ 50	2.74	2.48	0.5	0.1
14	- 50	3.00	2.31	1.2	0.3
26	- 50	3.09	2.85	2.2	2.1
27	+ 110	3.39	3.22	3.9	2.7
30	- 70	3.12	2.82	0.5	1.1
33	+ 60	3.47	3.26	12.0	7.8
34	- 35	3.04	2.73	0.7	0.4

In the study period, winter conditions differed widely: three severe winters (1984/85, 1985/86 and 1986/87) were followed by two extremely mild ones (1987/88 and 1988/89). The winter 1983/84 showed average temperatures.

Sampling programme

Eight intertidal stations, situated in the mouth (Roggenplaat), the central part (Galgenplaat) and the eastern part (Verdronken Land van Zuid-Beveland) of the Oosterschelde (Figure 1) were sampled twice a year

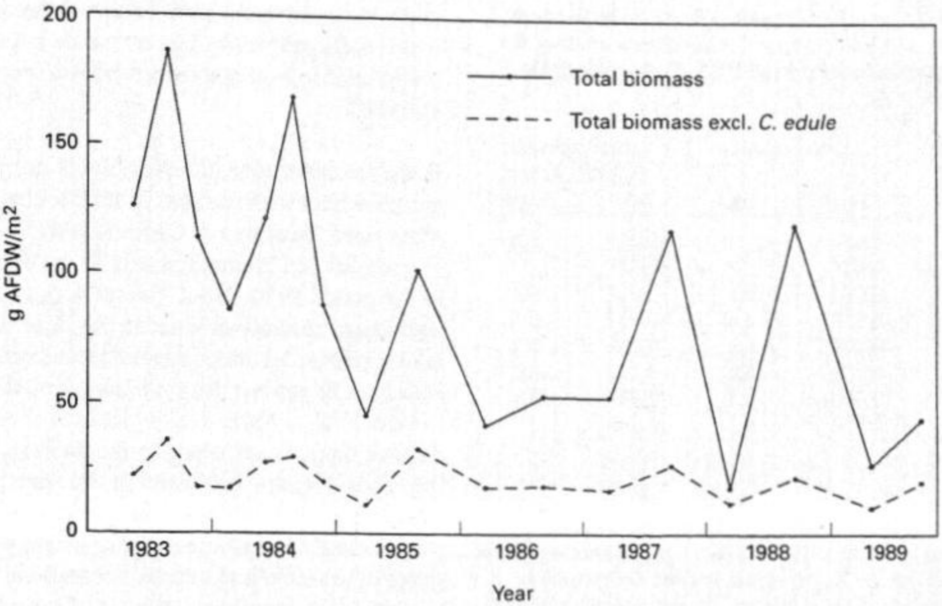

Figure 2 Total macrobenthic biomass (g ADW m^{-2}) with and excluding *Cerastoderma edule* in the period 1983–1989 at station 3

(March–April and August–September) between 1983 and 1989. In 1983 and 1984 additional samples were taken in May–June and November–December. Three sets of five core samples, each 10.3 cm in diameter (83 cm^2), were taken to a depth of 30 cm, sieved on a 1 mm mesh-sieve in the field and fixed with 7% neutral formalin. Additionally, two square metres were dug out to a depth of 10 cm and sieved on a 3 mm mesh-sieve in the field. The deeper sediment layer (down to 40 cm) was handsorted for large *Arenicola marina* L., *Neireis diversicolor* O.F. Müller, *Nephtys hombergii* Savigny, *Mya arenaria* L. and *Scrobicularia plana* Da Costa. This material was also fixed with 7% neutral formalin.

Laboratory methods

The small (83 cm^2) samples were stained with Rose Bengal and for the period 1987–1989 all organisms were sorted and identified up to species level (except for Oligochaeta, Nemertini and Anthozoa). From the samples of the period 1983–1987 only the eleven most important species in biomass (*A. marina, Ceratoderma edule* L., *Hydrobia ulvae* Pennant, *M. balthica, M. arenaria, Heteromastus filiformis* Claparède, *Lanice conchilega* Pallas, *N. hombergii, N. diversicolor, Scoloplos armiger* O.F. Müller and *S. plana*) were counted. Therefore, in the following analysis, only the eleven species mentioned above will be considered. Shell-length of bivalves was measured to the nearest millimetre. Ash-free dry weight (AFDW) biomass was obtained by weighing all individuals per species per station after drying for either a minimum of 72 hours at 80 °C, and ashing for 2 hours at 570 °C (1983–1987), or 12 hours at 110 °C and weighing again after incinerating for 2 hours at 550 °C (1987–1989) and subtracting the two values. All bivalves were weighed including the shell. Data from the square metre samples were only used for larger and often deeper living species such as *A. marina, N. diversicolor, N. hombergii, M. arenaria, S. plana.* If there were less than five larger (> 3 cm) individuals in the fifteen core samples, a combination was made of the larger specimens (> 3 cm) from the square metre samples with the small ones (< 3 cm) from the core samples.

In the data-analysis, variability in time is expressed as the coefficient of variation of biomass, i.e. 100 times the standard deviation (s) divided by the mean (m). Coefficients were calculated for the more common species (on average more than 2% of the total biomass) and for the different trophic groups, for all values as well as for spring values only.

Results and discussion

Long-term trends

Total biomass

On the eight stations total biomass shows clear seasonal fluctuations, with low values in spring and high biomasses in late summer. With the exception of stations 3 (Figure 2) and 6, which showed a gradual decrease, no significant

Table 2 Kendall Tau rank correlation coefficients for total biomass with and without *Cerastoderma edule* at the eight stations in the period 1983–89 (* = $p < 0.05$; ** = $p < 0.01$).

station	*N*	*total biomass*		*total biomass without cockles*	
		Tau	*sign*	*Tau*	*sign*
3	17	-0.456	**	-0.206	n.s.
6	17	-0.471	**	-0.353	*
14	10	-0.067	n.s.	-0.067	n.s.
26	16	0.117	n.s.	0.167	n.s.
27	17	-0.176	n.s.	-0.162	n.s.
30	12	-0.121	n.s.	-0.152	n.s.
33	17	0.059	n.s.	0.044	n.s.
34	17	0.250	n.s.	0.309	*

Table 3 Total biomass (g ADW m^{-2}) with and without *Cerastoderma edule* in the survey of late summer 1985 and 1989 on the Roggenplaat and the Galgenplaat (mean ± 95%CI; N=120 and 110 respectively)

tidal flat	*total biomass with* C. edule		*total biomass without* C. edule	
	1985	*1989*	*1985*	*1989*
Roggenplaat	58.92 ± 20.73	103.48 ± 33.51	21.45 ± 10.68	20.48 ± 7.21
Galgenplaat	49.17 ± 22.80	113.31 ± 37.38	25.81 ± 17.52	22.31 ±11.18

Table 4 Kendall Tau rank correlation coefficients for biomass of deposit feeders, filter feeders and omnivores/predators at the eight stations in the period 1983–89 (* = $p < 0.05$; ** = $p < 0.01$).

station	*N*	*biomass deposit feeders*		*biomass filter feeders*		*biomass omnivores/ predators*	
		Tau	*sign*	*Tau*	*sign*	*Tau*	*sign*
3	17	0.000	n.s.	-0.485	**	-0.485	**
6	17	-0.221	n.s.	-0.515	**	-0.221	n.s.
14	10	-0.022	n.s.	0.022	n.s.	0.378	n.s.
26	16	0.250	n.s.	0.133	n.s.	0.117	n.s.
27	17	-0.162	n.s.	-0.176	n.s.	0.191	n.s.
30	12	-0.152	n.s.	-0.182	n.s.	0.394	*
33	17	0.279	n.s.	0.265	n.s.	0.029	n.s.
34	17	0.338	*	0.265	n.s.	0.176	n.s.

trend over the seven year study period is found (Table 2). Stations 3 and 6 are dominated by cockles *C. edule* (respectively 73% and 39% of total biomass). Without the cockle, the negative trend in total biomass disappears in station 3 (Figure 2) and is weaker in station 6 (Table 2). This is confirmed by the results of a survey in late summer 1985 and 1989 (Meire *et al.* 1991 & 1992; Van der Meer *et al.* 1989), in which no difference in total biomass excluding cockles was found on the Roggenplaat and the Galgenplaat (Table 3). However, these surveys revealed a higher cockle biomass in 1989, compared to 1985. This might be explained by local effects of the cockle fishery, which had a profound effect on cockle populations in the Oosterschelde in periods of intensive fishery (Lambeck *et al.* 1988).

In the Oosterschelde, the decrease in nutrient input has not coincided with changes in total biomass. On the other hand, Beukema & Cadée (1986) found an increase in numbers and biomass in half of the common species in the period 1970–1984. Though a causal relationship between simultaneously occurring increases of nutrient levels, primary production and secondary production could not be proven, they mentioned such a relationship to be natural. Similar positive effects of eutrophication on total biomass are found in the western Baltic Sea and the Sound (Jensen 1986) and in the Kattegat (Pearson & Rosenberg 1978). On the other hand, Dörjes *et al.* (1986) concluded after a study of seven years that the range of seasonal and annual fluctuations was too large to distinguish long-term changes of the whole benthic community from eutrophication-induced effects.

Trophic groups

As for total biomass, the biomass of the different trophic groups shows neither a decrease nor an increase at most stations (Table 4). The decrease of filter-feeders at station 3 and 6 is caused by the decline in cockle biomass. At station 3, *N. diversicolor* and *N. hombergii* are most abundant in 1983–84. Low biomass values of the omnivores/predators in 1985–87 at this station can probably be explained by the occurrence of three severe winters.

Species

Since only eleven species were sorted before 1987, we can not compare species richness on these stations before and after the construction of the storm surge barrier. In the benthic survey of late summer 1985 (Meire *et al.* 1991; Van de Meer *et al.* 1989) the same total of species (53) was found as in late summer 1989 (54) (Meire *et al.* 1992). Taking into account the assumed number of different species not identified within some genera, respectively 62 and 63 species were present. Species not found in either one of the sampling years (*Abra alba* Wood, *Crepidula fornicata* L., *Angulus fabulus* Meuschen, *Ophiura* sp., *Pectinaria* sp., *Pholoe* sp., *Haustorius arenarius* Slabber, *Orchestia* sp., *Idotea* sp.) are rare species, difficult to sample or restricted to specific habitats. The American species *Ensis directus* Conrad which expanded into the Dutch coastal waters in recent years (Essink 1985) was found on the Roggenplaat in 1989.

Although total biomass of trophic groups do not show any clear trend, at least the abundance of one species seems to be affected. Between 1983 and 1989, both density and biomass of *Hydrobia ulvae* decreased with

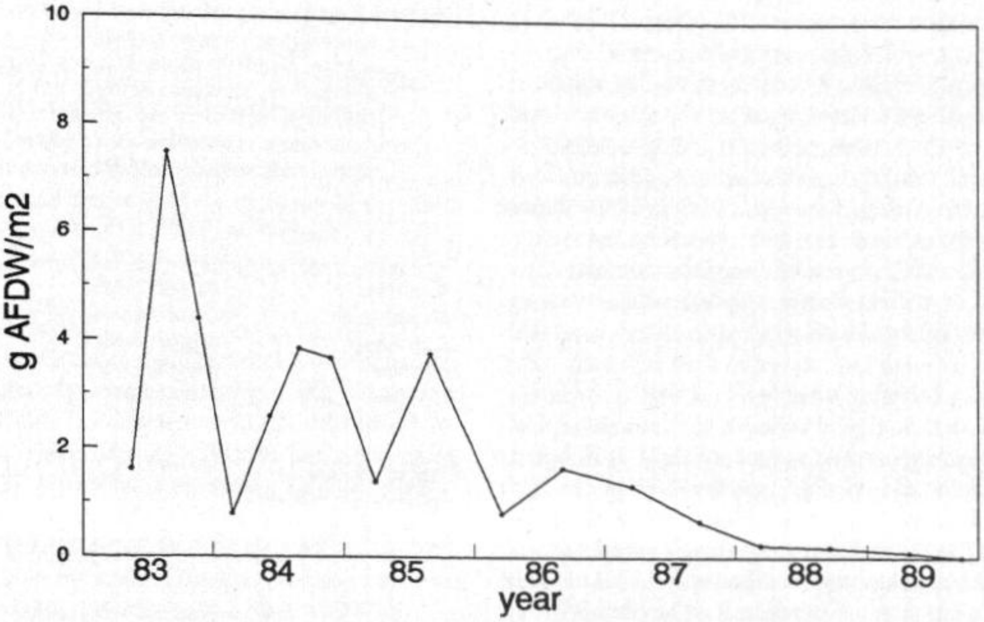

Figure 3 Biomass of *Hydrobia ulvae* at station 6 in the Oosterschelde in the period 1983–1989

more than 90% at five of the eight stations. The other three stations show a normal pattern with (station 27) or without (station 33) low abundancy in spring 1987, or with a peak value in spring 1986. Particularly at stations 3 and 6 (Figure 3) on the Roggenplaat, a gradual decline was observed. A decrease in biomass was also found in the benthic surveys of late summer 1985 (Meire *et al.* 1991; Van der Meer *et al.* 1989) compared to 1989 (Meire *et al.* 1992): on the Roggenplaat, a reduction from 1.41 ± 0.48 g ADW m^{-2} to 0.08 g ADW m^{-2} was observed; on the Galgenplaat values of respectively 4.21 ± 1.08 g ADW m^{-2} to 0.50 ± 0.22 g ADW m^{-2} were recorded.

One reason for the decrease in this species might be the removal of silt from parts of the intertidal area. The five stations mentioned above are those with the smallest silt fraction in the upper sediment layer in 1983–84, and the silt content was further reduced from 1983–84 to 1989 (maximum 1% < 53 μ). As a surface deposit feeder, *H. ulvae* will leave places where the silt fraction becomes too small.

Secondly, *H. ulvae* prefers sheltered areas (Walters & Wharfe 1980) in estuaries, whereas the construction of the storm surge barrier in the Oosterschelde has increased wave-action in the intertidal area by on average up to 80%.

Thirdly, *H. ulvae* has an irregular (temporal as well as spatial) and unpredictable recruitment, described by several authors (e.g. Beukema & Essink 1986; Dörjes *et al.* 1986). Besides irregular recruitment, they also denote the variability in *Hydrobia* populations due to predation effects, good mobility and a short life span (normally not exceeding 2.5 years: Clay 1960; Dekker 1979; Fish & Fish 1974). A short life span and a bad recruitment since 1985 might also explain the gradual collapse of the *Hydrobia* population in the Oosterschelde.

Patterns of variability

Seasonal patterns

Although a comparison of biomass values of individual species and trophic groups in 1983 and 1989 reveals no clear trends, these values do vary within the study period. This variability is caused by many different factors and can mask underlying trends. The most obvious pattern of variability is seasonal variability. The biomass of most species reaches maximum values at the end of the summer–beginning of autumn, and decreases during wintertime (due to predation, mortality and weight loss) to reach minimum values in spring (Beukema 1974) (Figure 2). During severe winters, such as the three successive ones in 1984/85, 1985/86 and 1986/87, mortality is much higher. This is illustrated by the low biomasses of *C. edule* in each of the three subsequent springs. The effect of winter frost can also be demonstrated for intertidally living *L. conchilega*. In station 3, this species was very rare between spring 1985 and spring 1988 (Figure 4). Different from the observed effects after the severe winter 1963 (Crisp 1964) is the slow recovery here. Other species (*A. marina, S. armiger, M. balthica, M. arenaria, H. filiformis*) known to be more tolerant to severe winter conditions (Beukema 1979 & 1985; Crisp 1964; Dörjes 1980; Ziegelmeier 1964), do not show a decrease in the Oosterschelde during this period. Dörjes (1980) showed that in subtidal areas mortality is caused by low temperatures; in the intertidal zone ice-cover and ice-movement, as well as shorter inundation times, can decimate entire populations.

Variability and trophic groups

Variability in time differs between species and trophic groups. The coefficient of variation (Table 5) is high for *C. edule* and *N. diversicolor*, intermediate for *M. balthica, S. armiger* and *H. ulvae* and low for *M. marina* (Table 5). Filter feeders, as a group, are clearly more variable in time than deposit feeders. This is in agreement with the hypothesis of Levinton (1972), which states that the size of populations of suspension feeding marine benthic species should fluctuate widely in numbers, both temporally and spatially, whereas numbers of deposit feeders living in the same area should be relatively constant. This difference between the two groups would be related to the degree of constancy of their respective food supplies, being unpredictable and heavily fluctuating for suspension feeders and relatively constant for deposit feeders (Levinton 1972). Our results are similar to those of Beukema *et al.* (1983), who tested the hypothesis of Levinton on the benthic populations of the Western Wadden Sea. However, there were some differences in the methods of data analysis. We used biomasses instead of densities to calculate coefficients of variation, and *N. diversicolor* and *N. hombergii* were considered as omnivores/predators instead of deposit feeder and predator/deposit feeder respectively. Whereas both species showed a rather low variability in the Wadden Sea (ranking immediately behind the most stable species *A. marina* and *M. balthica*), we found *N. diversicolor* to be the most variable of the six biomass-dominating species when considering the entire dataset, and the second most variable when spring values are compared.

Table 5 Coefficient of variation of the six dominant (> 2% of total biomass) species *Cerastoderma edule, Arenicola marina, Hydrobia ulvae, Macoma balthica, Nereis diversicolor, Scoloplos armiger* and of the deposit feeders, filter feeders and omnivores/predators. Coefficients of variation are calculated on the entire dataset and on spring values only.

species/trophic group	*feeding type*	*variability in time*	
		total	*spring*
A. marina	deposit feeder	62	61
M. balthica	deposit feeder	103	109
S. armiger	deposit feeder	113	94
H. ulvae	deposit feeder	125	108
C. edule	filter feeder	155	170
N. diversicolor	omnivore	171	153
deposit feeders		46	39
omnivores/predators		125	122
filter feeders		147	155

Our results suggest that winter conditions, more than food availability are responsible for the observed

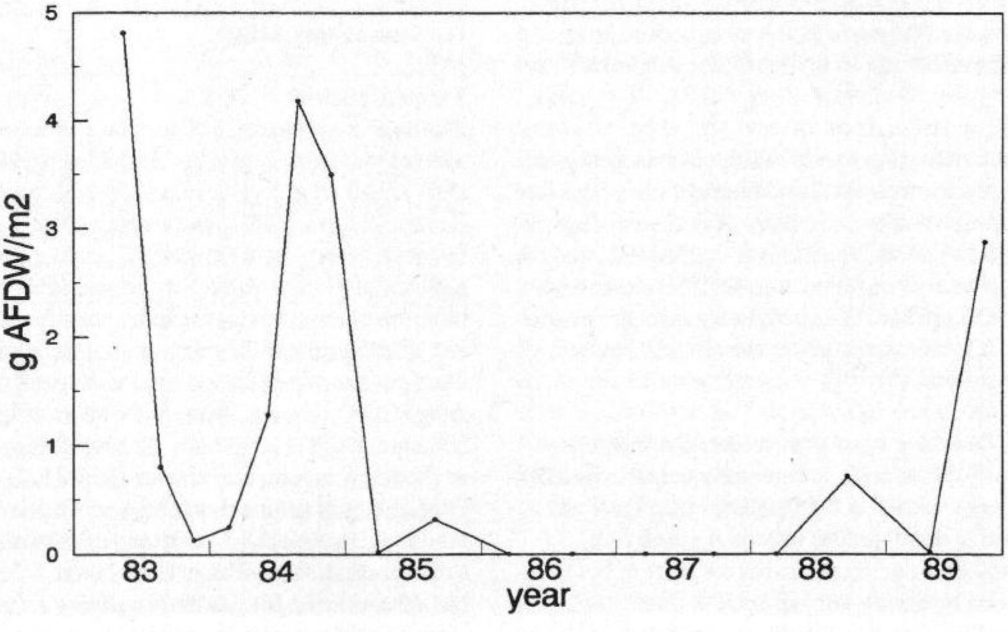

Figure 4 Biomass of *Lanice conchilega* at station 3 (Roggenplaat) in the period 1983–1989

variability. Ranking of the six dominating species as to temporal variability, demonstrates that the most winter-sensitive species (*C. edule, N. diversicolor*) are most variable, whereas more winter-resistant organisms (*A. marina, M. balthica, S. armiger, H. ulvae*) have lower coefficients of variation. Also Beukema *et al.* (1983) mentioned that at least the extremes of temporal variability appeared to be caused by mechanisms that have no direct connection to the type of food source.

The role of predation

Changes in the abundance of one species can have clear effects on other organisms (Commito 1982; Schubert & Reise 1986). This is illustrated at station 6 on the Roggenplaat, where a strong reduction in abundance of the predator *N. hombergii* from spring 1985 onwards (due to the severe winter) coincides with a sharp increase in density and biomass of the prey *S. armiger* (Figure 5). Here, the critical *Nephtys* biomass which can control a *Scoloplos* population is 0.60 g AFDW m^{-2}. This is near to the 0.30 g AFDW m^{-2} found by Beukema (1987) in the Dutch Wadden Sea. For the other stations, initial biomasses of *N. hombergii* were mostly much lower and were further reduced during the severe winter period in all stations above MTL.

Changes in inundation time

Species such as *M. balthica, M. marina, H. ulvae* and *S. armiger* are tolerant to severe winter conditions. Nevertheless, at the high station 27 (MTL + 110 cm) they show a marked biomass decrease in spring 1987, causing a reduction in total biomass (Figure 6). This is probably caused by the closure of the secondary dams in October 1986 and March/April 1987, with a prolonged reduction in tidal amplitude, resulting in higher exposure times on the intertidal flats above MTL (Figure 6). Up to 40% of all high waters did not reach MTL + 110 cm between October 1986 and April 1987. This means that in March/April 1987 up to ten successive high waters did not reach this level, whereas in normal situations this was restricted to maximally one high tide. From experimental work by Hummel *et al.* (1986), we know that nearly all common species show high mortality after more than four days exposure, and more sensitive species such as *Tharyx marioni* Saint-Joseph, *N. hombergii* and *S. armiger* after 2–4 days. At station 27 (MTL + 110 cm), we can therefore expect a rather high mortality for all species in spring 1987, exactly what was found. Species most sensitive to exposure, such as *S. armiger* (Figure 6) needed more time to recover than for example *A. marina, H. ulvae* or *M. balthica.*

Conclusion

Temporal variability in benthic systems can be studied at different time scales and taxonomical/functional levels. Variability in time was larger for filter feeders than for deposit feeders at eight Oosterschelde stations and the importance of several factors causing fluctuations (severe winter weather, changes in predator/prey impact, shorter inundation time) has been demonstrated.

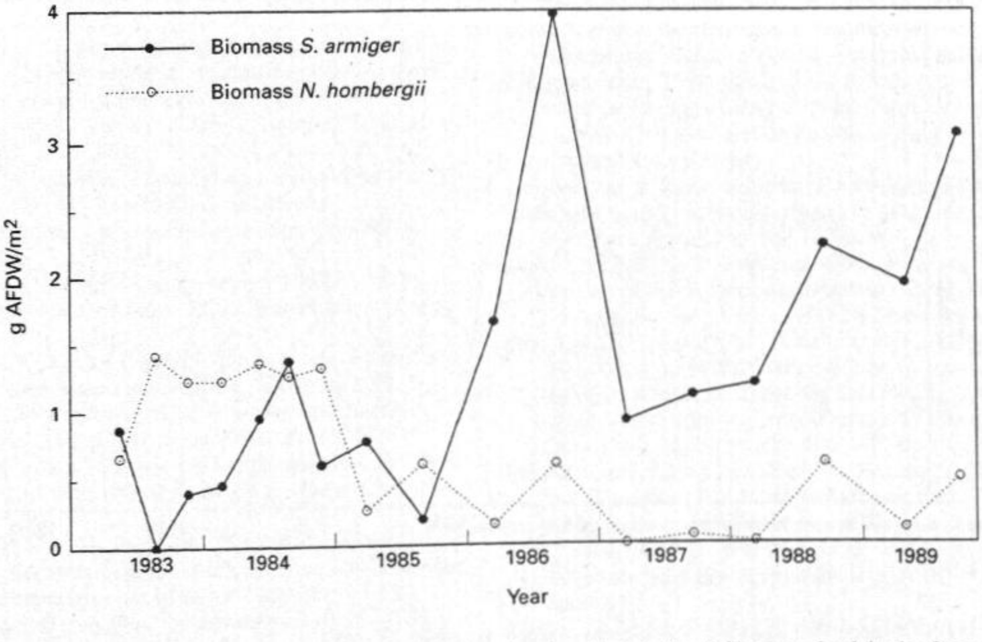

Figure 5 Biomass of *Scoloplos armiger* and its predator *Nephtys hombergii* at station 6 (Roggenplaat) in the Oosterschelde in the period 1983–1989

On a longer term, no change in total biomass, species composition or biomass of trophic groups was found. Since major environmental factors (salinity, primary production) in the system have not changed between 1983 and 1989, this result is not surprising. However, from these preliminary results at only eight different stations, we can not conclude now that nothing has changed in the benthic communities of the whole Oosterschelde after the completion of the storm surge barrier. The gradual decrease of *H. ulvae* in large parts of the estuary indicates that at least at species level some changes might have occurred. Further analysis of this and other datasets (1. late summer surveys on three tidal flats in 1985 and 1989; 2. results of sampling at six permanent stations on the Slikken van Vianen, i.e. in the northern branch of the estuary) will be done in the near future.

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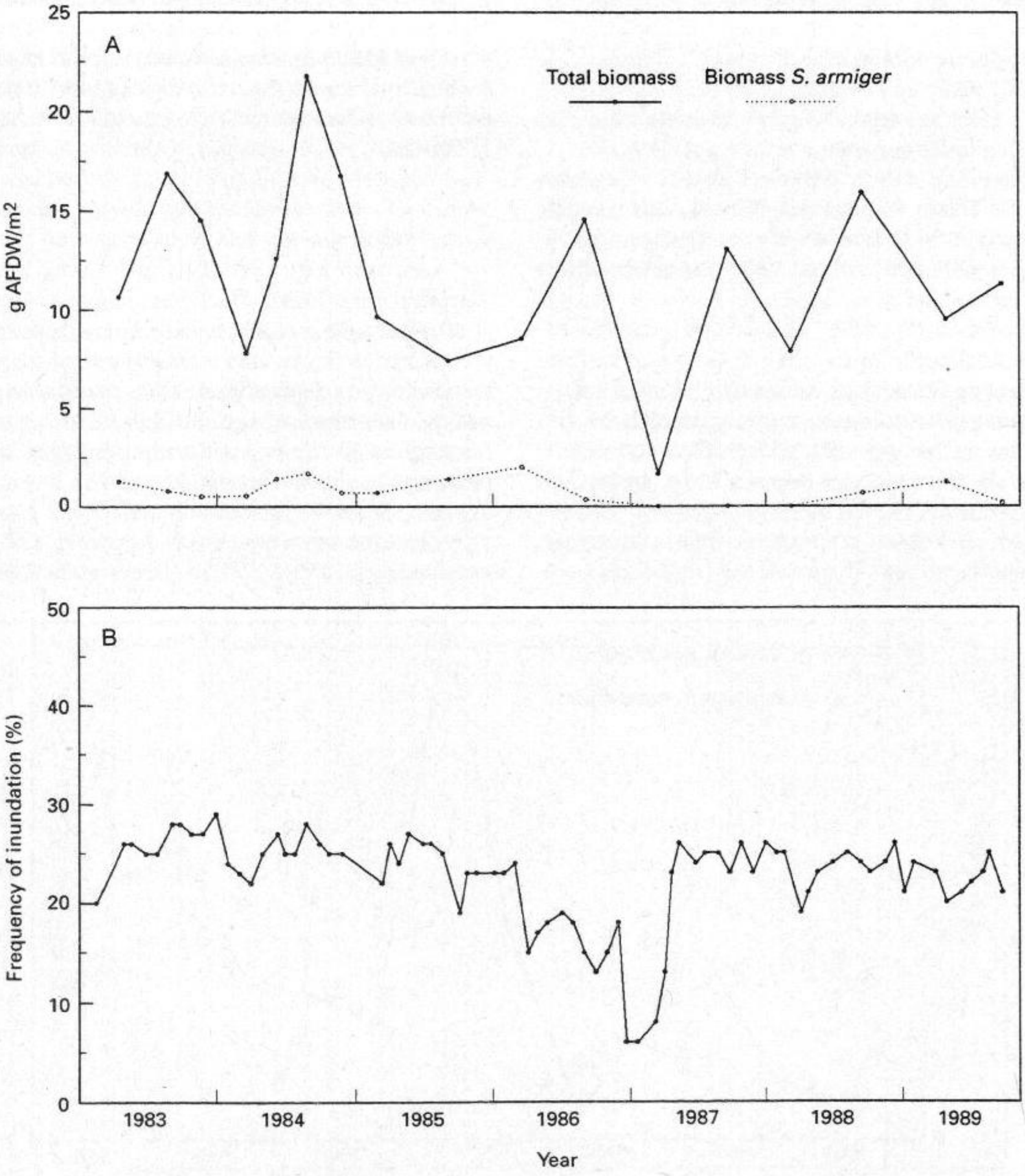

Figure 6 A: Total biomass and biomass of *Scoloplos armiger* at station 27 (MTL + 110 cm) in the period 1983–1989.
B: Percentage of high waters per month reaching the level MTL + 110 cm, near Yerseke (Oosterschelde) in the period 1983–1990.

References

Agger, P. 1989. Principles for the Danish biomonitoring programme. *Helgoländer Meeresunters 43*: 309–317.

Beukema, J.J. 1974. Seasonal changes in the biomass of the macro-benthos of a tidal flat area in the Dutch Wadden Sea. *Neth. J. of Sea Res. 8 (1)*: 94–107.

Beukema, J.J. 1979. Biomass and species richness of the macrobentic animals living on a tidal flat area in the Dutch Wadden Sea: effects of a severe winter. *Neth. J. of Sea Res. 13 (2)*: 203–223.

Beukema, J.J. 1985. Zoobenthos survival during severe winters on high and low tidal flats in the Dutch Wadden Sea. *In*: *Marine Biology of Polar Regions and Effects of Stress on Marine Organisms,* ed. by Gray & Christianse, 351–361. John Wiley & Sons Ltd.

Beukema, J.J. 1987. Influence of the predatory polychaete *Nephtys hombergii* on the abundance of other polychaetes. *Mar. Ecol. Progr. Ser. 40*: 95–101.

Beukema, J.J. 1989. Long-term changes in macrozoobenthic abundance on the tidal flats of the western part of the Dutch Wadden Sea. *Helgoländer Meeresunters 43*: 405–415.

Beukema, J.J., Cadée, G.C., & Hummel, H. 1983. Differential variability in time and space of numbers in suspension and deposit feeding benthic species in a tidal flat area. *Oceanol. Acta. Actes 17º Symposium de Biologie Marine, Brest, 27 Septembre–1 Octobre 1982*: 21–26.

Beukema, J.J., & Cadée, G.C. 1986. Zoobenthos responses to eutrophication of the Dutch Wadden Sea. *Ophelia 26*: 65–76.

Beukema, J.J., & Essink, K. 1986. Common patterns in the fluctuations of macrozoobenthic species living at different places on tidal flats in the Wadden Sea. *Hydrobiologia 142*: 199–207.

Buchanan, J.B., & Moore, J.J. 1986. A broad review of variability and persistence in the Northumberland benthic fauna 1971–85. *J. mar. biol. Ass. U.K. 66*: 641–657.

Clay, E. 1960. Hydrobia ulvae *Pennant,* H. ventrosa *Montagu and* Potamopyrgus jenkinsi *Smith, literature survey of the common fauna of estuaries, 8.* I.C.I. Paints Division, Brixham Res. (Memorandum PVM45/A/381).

Commito, J.A. 1982. Importance of predation by infaunal polychaetes in controlling the structure of a soft-bottom community in Maine, USA. *Mar. Biol. 68*: 77–81.

Coosen, J., Twisk, F., Van der Tol, M., Lambeck, R.H.D., Van Stralen, M., & Meire, P. 1994. Variability in stock assessment of cockles (*Cerastoderma edule* L.) in the Oosterschelde (in 1980–1990), in relation to environmental parameters. *Hydrobiologia 282/283*: 381–395.

Crisp, D.J. 1964. The effects of the winter 1962/63 on the British marine fauna. *Helgol. Wiss. Meeresunt 10 (1–4)*: 313–327.

Dekker, R. 1979. *Numbers, growth, biomass and production of organic and calcareous matter of* Hydrobia ulvae *(Gastropoda : Prosobranchia) in the western Dutch Wadden Sea.* Interne Verslagen, NIOZ Texel 1979-15.

Dörjes, J. 1980. Auswirkungen des karlten winters 1978/79 auf das marine makrobenthos. *Natur u. Museum 110*; 109–115.

Dörjes, J., Michaelis, H., & Rhode, B. 1986. Long-term studies of macrozoobenthos in intertidal and shallow subtidal habitats near the island of Norderney (East Frisian coast, Germany). *Hydrobiologia 142*: 217–232.

Essink, K. 1985. On the occurrence of the American jack-knife clam *Ensis directus* (Conrad, 1843) (Bivalvia, Cultellida) in the Wadden Sea. *Basteria 49*: 73–80.

Fish, J.D., & Fish, S. 1974. The breeding cycle and growth of Hydrobia ulvae in the Dovey estuary. *J. mar. biol. Ass. UK 54*: 685–697.

Hummel, H., Meijboom, A., & de Wolf, L. 1986. The effects of extended periods of drainage and submersion on condition and mortality of bénthic animals. *J. Exp. Mar. Biol. Ecol. 103*: 251–266.

Jensen, K. 1986. Changes of the macrozoobenthos at 3 monitoring stations in the western Baltic Sea and the Sound. *Hydrobiologia 142*: 129–135.

Kohnke, D.P. 1989. Monitoring activities in the Federal Republic of Germany pursuant to international conventions and national legislation. *Helgoländer Meeresunters, 43*: 319–324.

Lambeck, R.H.D., Hannewijk, A., & Brummelhuis, E.B.M. 1988. *A survey of the cockle (*Cerastoderma edule*) on two tidal flats in the Oosterschelde in November 1987: Effects of fishery?* Delta Institute for Hydrobiological Research, Yerseke. Report 1988–6 (in Dutch).

Levinton, J. 1972. Stability and trophic structure in deposit-feeding and suspension-feeding communities. *The American Naturalist 106 (950)*: 472–486.

Meire, P.M., Buijs, J., Van der Meer, J., van den Dool, A., Engelberts, A., Twisk, F., Stickvoort, E., de Jong, D., Lambeck, R.H.D., & Polderman, J. 1991. *Macrobenthic fauna in the Oosterschelde in autumn 1985: a presentation of the data*. Report DIHO, RWS-DGW, RUG (in Dutch).

Meire, P.M., Buijs, J., Seys, J., Twisk, F., & Stickvoort, E. 1992. *Macrobenthic fauna in the Oosterschelde in autumn 1989: a presentation of the data.* Report DIHO, RWS-DGW, RUG, JN (in Dutch).

Pearson, T.H., & Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. mar. Biol. Ecol. 92:* 157–206.

Pearson, T.H., Duncan, G., & Nuttal, J. 1986. Long term changes in the benthic communities of Loch Linnhe and Loch Eil (Scotland). *Hydrobiologia 142*: 113–119.

Pielou, E.C. 1974. *Population and community ecology. Principles and methods.* New York, Gordon & Breach.

Reise, K. 1989. Monitoring the Wadden Sea – an introduction. *Helgoländer Meeresunters 43,* 259–262.

Rijkswaterstaat. 1991. *Veilig getij. Effects of the engineering works on the tidal habitats of the Oosterschelde.* Dienst Getijdewateren – Directie Zeeland. (Nota GWWS 91.088/ AX 91.091; in Dutch).

Schubert, A., & Reise, K. 1986. Predatory effects of *Nephtys hombergii* on other polychaetes in tidal flat sediments. *Mar. Ecol. Progr. Ser. 34*: 117–124.

Seys, J., Meire, P., & Coosen, J. 1992. *Macrobenthos on eight stations in the Oosterschelde estuary 1983–1989: presentation data.* Report W.W.E., Gent (in Dutch).

Smaal, A.C., Knoester, M., Nienhuis, P.H., & Meire, P.M. 1991. Changes in the Oosterschelde ecosystem induced by the Delta works. *In*: *Estuaries and Coasts: spatial and temporal intercomparisons (Proc. 19th Symp. E.C.S.A. Caen 1989)* ed. by M. Elliott & J-P. Ducrotoy, 375–384.

Van der Meer, J., van den Dool, A., Engelberts, A., de Jong, D.J., Lambeck, R.H.D., & Polderman, J.M. 1989. *The macrozoobenthos of some intertidal flats in the Oosterschelde. Results of a survey in late summer 1985.* Rijkswaterstaat (Delta Institute Report GWAO 89.307; in Dutch).

Walters, G.J., & Wharfe, J.R. 1980. Distribution and abundance of *Hydrobia ulvae* (Pennant) in the Lower Medway estuary, Kent. *J. Moll. Stud. 46*: 171–180.

Wetsteyn, L.P.M.J., Peeters, J.C.H., Duin, R.N.M., Vegter, F., & de Visscher, P.R.M. 1990. Phytoplankton primary production and nutrients in the Oosterschelde (The Netherlands) during the pre-barrier period 1980–1984. *Hydrobiologia 195*: 163–177.

Wetsteyn, I.P.M.J., & Bakker, C. 1991. Abiotic characteristics and phytoplankton primary production in relation to a large-scale coastal engineering project in the Oosterschelde (The Netherlands): a preliminary evaluation. *In*: *Estuaries and Coasts: spatial and temporal intercomparisons (Proc. 19th Symp. E.C.S.A. Caen 1989)* ed. by M. Elliott & J.-P. Ducrotoy, 365–373.

Wolfe, D.A., Champ, M.A., Flemer, D.A., & Mearns, A.J. 1987. Long-Term Biological Data Sets: Their Role in Research, Monitoring and Management of Estuarine and Coastal Marine Systems. *Estuaries 10 (3)*: 181–193.

Ziegelmeier, E. 1964. Einwirkungen des kalten winters 1962/63 auf das Makrobenthos im OsEtteil der Deutschen Bucht. *Heolgol. wiss. Meeresunters 10*: 276–282.

Effect of habitat loss and habitat change on estuarine shorebird populations

J.D. Goss-Custard

Goss-Custard, J.D. 1995. Effect of habitat loss and habitat change on estuarine shorebird populations. *In: Coastal Zone Topics: Process, Ecology & Management 1*: 61–67.

The fate of overwintering migratory shorebirds is one of the major concerns expressed when the effects on estuaries of environmental changes, such as land-claim, barrage construction, global warming and sea-level rise, are being considered. The environmental conditions in an estuary may affect bird numbers at two geographic scales. First, they could affect the ability of a particular estuary to support birds in winter, i.e. its carrying capacity. Second, they could affect the balance between the annual rates of survival and reproduction, i.e. the equilibrium population size. The paper proposes a framework for thinking about how environmental changes in winter might affect both the carrying capacity of a particular estuary and the year-round equilibrium size of the population as a whole. It stresses the difficulty of doing the wide-ranging empirical studies needed to make predictions in particular systems. Accordingly it also briefly describes a simple technique, and its limitations, by which preliminary predictions may be made while our understanding of these systems is gradually being improved.

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Introduction

The fate of overwintering migratory shorebirds is one of the concerns expressed when the effects on estuaries of environmental changes, such as land-claim, barrage construction, global warming and sea-level rise, are being considered. These birds occur on estuaries mostly outside the breeding season when overwintering or stopping briefly in spring and autumn to build up fat reserves to fuel their onward migration to or from their more northerly, often arctic, breeding areas. Locally, the concern is that an environmental change might affect an estuary's ability to support birds. At a larger geographic scale, the population size of the species or subspecies might be affected (Goss-Custard & Moser 1988).

Because their main activity on estuaries is feeding, attention has naturally focused on how environmental change might affect the feeding conditions. To satisfy its food requirements, a bird must collect food, or supply itself, at a rate that achieves its demands in the time available. Table 1 shows how environmental changes might affect each of these three components of the demand-supply equation, the net effect defining the new feeding conditions. Since as bird density increases, and competition for food intensifies, the intake rate will decrease (see below), the Table shows the reciprocal of bird density so that a downward-pointing arrow, for example, signifies a decrease in the rate of intake as it does in the other three columns under supply-rate. The rate at which a shorebird can feed also depends on the absolute density of the prey and on their availability, i.e. on how easily a bird can detect and catch them, this usually depending on the prey's behaviour in, or on, the substrate (Goss-Custard 1984). Since an environmental change may have a different effect on prey density and availability, they are shown separately in the Table.

Table 1 shows how some environmental changes, if sufficient in magnitude, could affect both the demand and the supply sides of the equation. In terms of its effect on the feeding conditions experienced by birds, the effect could differ between the two sides of the equation; for example, a rise in temperature could decrease energy demands, thus making them easier to satisfy, yet decrease the rate at which food can be collected by reducing prey abundance and availability. It is therefore necessary to predict the net effect of an environmental change on the feeding conditions. Sometimes, the net effect within one side of the equation, or even one component within it, may need to be predicted. Consider, for example, the effect of a tidal-power barrage on the rate of food acquisition: while any good feeding areas at the bottom of the shore might become permanently inundated and so unavailable to birds, the prey biomass in the remaining upshore areas might increase by being covered for longer at high tide, especially if the intertidal environment as a whole became generally more benign (Warwick *et al.* 1991). This example also illustrates how the effects of an environmental change on the ecosystem as a whole that supports the birds must also be understood and quantified. As the example of global warming and associated sea-level rise illustrates, the changes in the ecosystem may be extremely difficult to forecast at our present level of knowledge. For all these reasons, the net effect of environmental changes on the balance between the rates of demand and supply for individual birds may be complex and difficult to predict.

Table 1 How environmental changes might affect both the demand of shorebirds for food and their ability to obtain it. Key: ↑ signifies increase; ↓ signifies decrease; – signifies no change; numbers refer to the explanations, where needed, and references listed below; asterisks indicate uncertainty.

	demand rate		*supply rate*		
			rate of intake:		
environment change	*food requirements*	*time available to feed*	*bird density*	*prey density*	*prey availability*
Global warming:					
Sea-level rise	–	↓1*	↓2*	↓3*	↓4*
Rise in temperature	↓	–	–	↓–↑*5	↑–↓*6
Increase in winds	↓	↑–↓7	↑–↓8	↓9	↓10
Increase in rainfall	↑11*	↓12	↓13	↓–↑14*	↓↑15
Developments:					
Tidal-power barrage (ebb generation)	–	↓16	↓17	↓↑18*	↑19*
Reservoirs; land-claim	–	–↓20	↓	↑–↓21*	–
Activities:					
Disturbance on flats at low water	↑22	↓23	↓24	–	–
Disturbance at high water roosts	↑22	–	–	–	–
Bait-digging, collecting shellfish, etc	↑22	↓25	↓26	↓↑27*	–
Pollution:					
Organic and nutrient enrichment	–	–	–	↓↑28	–

Explanations and references: 1–2. Increased sea-levels and lowered levels of the intertidal flats through increased rates of erosion could reduce both the exposure time of the flats and their area (Goss-Custard, McGrorty & Kirby 1990). 3. Increased rates of erosion and reduced productivity of the inshore waters could reduce invertebrate abundance (Goss-Custard, McGrorty & Kirby 1990). 4. Increasingly coarse and hard sediments could make many prey organisms harder for birds to detect and catch (Goss-Custard, McGrorty & Kirby 1990). 5. The abundance of prey could increase or decrease depending on their temperature preferences and those of their intertidal predators (Beukema 1990). 6. By becoming more active at warmer temperatures, some prey may be easier for birds to detect and catch (Goss-Custard 1984), unless the sediments dry out more, thus reducing prey availability. 7–8. Depending on wind direction in relation to direction of tidal flow, exposure time and area exposed could be increased, decreased or remain the same. 9. Increased winds could reduce inshore productivity and thus prey abundance (Emerson 1989). 10. Prey may be harder to detect because the birds are buffeted and the sediment-surface swept (Evans 1976). 11. By reducing feather insulation, heat-loss is likely to increase. 12–13. Higher water levels could reduce exposure time and area of flats available. 14. Depending on the responses of prey to reduced salinities, they could increase or decrease in abundance, or remain unchanged. 15. Rain can make some intertidal prey harder to catch at low water, but increases the rate at which earthworms can be caught at high water by some bird species (Goss-Custard 1984). 16–17. By holding back the ebbing tide, exposure time and intertidal area are reduced. 18. Such a large change in the ecosystem could affect the abundance of various prey species in either direction (Warwick *et al.* 1991). 19. In some hypertidal estuaries, sediment changes could increase prey accessibility to birds (Goss-Custard *et al.* 1991). 20. If sited at the top of the shore, this can reduce exposure time unless new upshore flats develop. 21. Prey densities could change if the development significantly changed the hydrodynamics of the estuary. 22. Increased time spent flying increases energy requirements. 23–26. Inability to settle on the flats reduces feeding time and the area available in which to feed. 27. Exploitation of invertebrates may directly reduce invertebrate densities (Horwood & Goss-Custard 1977) but removing predatory crabs may increase their abundance. 28. Depending on the concentrations, the abundance of invertebrate prey may be increased or decreased (Pearson & Rosenberg 1978).

Yet it is important to be able to do so if we are to be able to anticipate how environmental change will affect shorebird numbers. A change in the feeding conditions may affect a bird's decision whether to remain on an estuary or to move elsewhere. It may affect the bird's body condition and thus its chances of surviving the winter, migrating and breeding successfully. It may also indirectly affect the bird's vulnerability to enemies if hungry shorebirds are more vulnerable to predators (Whitfield, Evans & Whitfield 1988) and, perhaps, to parasites and diseases. Forecasting the effects of environmental change therefore involves not only predicting the net effects on the feeding conditions themselves, but also the effect of a change in these on complex and interacting processes that determine shorebird numbers.

Though the goal is far from being achieved, there is a broad understanding of how environmental changes might affect shorebird numbers. Using mainly published reviews rather than source references, this paper sketches a framework for considering how bird numbers might be affected by predicted changes in the feeding conditions, both locally and globally.

Individual variation in the response to changes in the feeding conditions

Individuals vary greatly in how they exploit the feeding opportunities provided by an estuary; individuals eat different prey species, use different feeding techniques, feed in different places and at different times and respond in different ways to conspecifics. It is most unlikely that all the individuals' foraging 'policies' are equally successful. Even amongst oystercatchers *Haematopus ostralegus* opening mussels *Mytilus edulis* with the same technique, the rate of food intake can vary twofold between individuals (Goss-Custard & Durell 1988). The foraging performance of individuals also changes with age as they gain experience (Goss-Custard & Durell 1987), adding further to the individual variation in supply-rates. Indeed the food requirements of individuals may vary according to their body-size, general level of activity and metabolic efficiency; if so, the supply-rate needed to match the demand-rate may also vary between birds. Though the balance between supply and demand may be averaged over some time by birds storing energy as fat to help them survive periods of harsh winter weather (Evans 1976), it is most unlikely that the risk of failing to achieve the threshold rate at which supply equals demand is distributed equally amongst the individuals of a population. Rather, some birds will be more at risk than others.

On failing to achieve the threshold intake rate, birds can either respond behaviourally, by emigrating, or remain where they are and risk death. Either way, bird numbers change, if only locally. The proportion affected by an environmental change depends simply on how many birds manage to achieve the threshold intake rate in the new feeding conditions. Figure 1a illustrates this general idea. It considers a situation in which a constant number of birds faces a change in the feeding conditions from poor to good, or vice versa. Each bird must balance its demand-supply equation by crossing the critical threshold at which demand and supply are equalised in the time available. Any change in the demand for food, or in the time available to feed or in the rate of food intake could cause the proportion of birds failing to reach the critical threshold to change. Birds that fail to do so will either emigrate or die and the proportion failing to do so will change as the feeding conditions change. The exact change in the proportion failing, or the 'response curve', will depend on how individual foraging performances are distributed throughout the population and phenotypic variation between birds in combination with the heterogeneity in their environment.

The role of competition between birds

An environmental change may not simply act independently on each individual in the shorebird population. Rather, the presence of other birds, of the same (Goss-Custard 1980) and other (Zwarts 1978; Evans *et al.* 1979) species, may have an influence on the feeding conditions experienced by individuals. These interactions are usually competitive. Most environmental changes cause bird density to increase because they reduce the area of intertidal flats available for the birds (Table 1). For this reason, understanding the effect of increased competition on the feeding conditions is likely to be critical for predicting how environmental changes will affect bird numbers at both the local and global levels of scale.

Competition can be due to interference between foraging birds and to depletion of prey stocks. Interference may be caused by direct social interactions between birds or by more subtle means. Oystercatchers exemplify the former: they attack each other to steal mussels and the average intake rate decreases when this occurs frequently (Goss-Custard & Durell 1987). A subtle form of interaction, with no observable social interactions, occurs in redshank *Tringa totanus*: prey take predator-avoidance action when redshank are near, so making them less accessible to the birds which then feed more slowly (Selman & Goss-Custard 1988). Depletion competition occurs because shorebirds can eat a substantial proportion of their prey, thus reducing prey density and the birds' intake rate during one winter (Goss-Custard 1980) and, perhaps, over the longer term (Horwood & Goss-Custard 1977).

The role of competition in environmental changes is most easily seen in the many changes that reduce intertidal area. But any kind of change will have a similar knock-on effect if competition already has a significant effect on birds' foraging performance. For example, lower ambient temperatures in winter increases the demand for food by oystercatchers at a time when reducing food stocks make it more difficult for it to be met. Consequently, birds attack each other more frequently, thus intensifying the severity of interference at a particular bird density (Goss-Custard & Durell 1987).

The precise effect of competition depends on the contribution it makes to the distribution of foraging performances, and therefore to the response of emigration and death rates to changing bird density. In part, it depends on whether some individuals can maintain their own intake rates at the expense of others which, in turn, depends on exactly how competition operates. Consider the effect of an increase in population size in oystercatchers (Fig. 1b). Dominant birds are able to maintain their high intake rates as bird density and competition increase (Ens & Goss-Custard 1984), so the upper limit of the frequency distribution remains fixed. As more birds fall below the threshold,

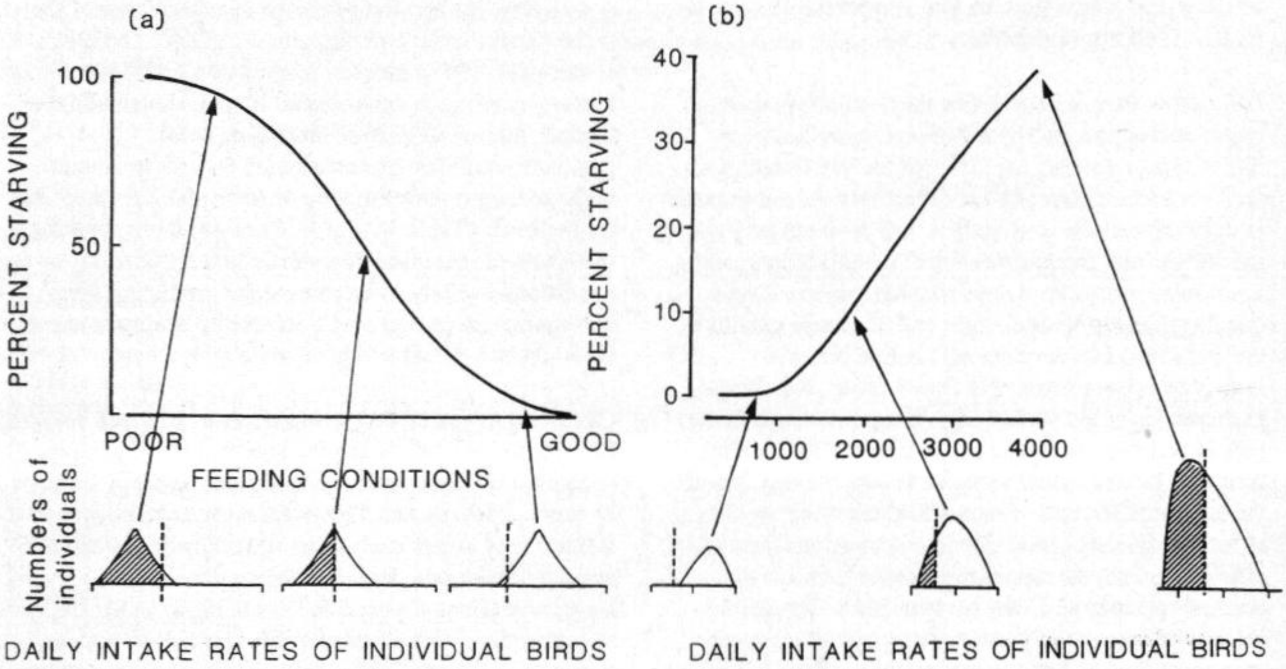

Figure 1 The proportion of birds failing to achieve the threshold rate of food intake required to meet demand for food in the time available in relation to changing feeding conditions. In (a), a constant number of birds face a change in the environment, such as improving weather. In (b), the environment stays the same but the number of birds increases, thus increasing bird density and so intensifying competition; this is equivalent to habitat loss, which will also increase bird density, though bird numbers stay the same.

In both cases, each bird must achieve the critical threshold rate of daily food intake to survive, this being indicated by the vertical dashed line. Individual birds vary greatly in their ability to do this, and this variation is shown in the frequency histograms of individual daily feeding performance. The shapes of these frequency distributions are hypothetical but are important because they determine how the death rate from starvation changes as the feeding conditions change. In (a), the shape of the distribution remains the same throughout because the bird's ability to forage is assumed to remain the same irrespective of the weather conditions. But in (b), the histograms skew gradually to the left as population size and density increase because some dominant individuals are assumed to be able to maintain their own intake rate as competition intensifies.

the distribution becomes increasingly skewed to the left, causing the response curve to increase sharply. But in other cases, all individuals may be affected by rising competition: interference in redshank may be an example. Here the upper point, is not fixed and the whole distribution simply moves to the left and the response curve rises more gently as bird density rises. These examples represent extremes, and normally may occur in combination. For example, if food abundance generally declines, all oystercatchers would find it more difficult to forage, though the dominants might be least affected by remaining in the best areas and stealing from subdominants. Both the point at which competition causes emigration and mortality rates to increase, or the 'intercept', and the subsequent rate at which they increase with bird density, or 'slope', can, in theory, vary greatly. But at present, no study has measured either slope or intercept in a shorebird population.

Carrying capacity of the wintering grounds

The three components of feeding conditions vary from place to place and at several geographic scales. Large scale regional differences in food requirements occur due to different weather conditions and distances from breeding areas (Pienkowski & Evans 1985). Because of spatial variations in prey density at several scales (Goss-Custard 1983), intake rates may vary down the length of a shore and between different shores, estuaries and, perhaps, regions of the wintering range. The wintering range may be viewed as a series of gradients in the feeding conditions occurring at several interlocking scales.

Field studies on several species suggest that certain parts of these gradients may be occupied first as birds arrive on the wintering grounds, so becoming, by definition, preferred areas. Bird numbers here may eventually reach a plateau, presumably because of intensifying competition: in effect, there comes a point when, for every bird that arrives, another inferior competitor leaves. When this happens, the carrying capacity of the area may have been reached (Goss-Custard 1985). Such limits to local density occur at several scales; for particular zones of a shore, for different parts of an estuary and for whole estuaries (Goss-Custard 1985; Moser 1988). Further increases in bird numbers mainly occur, by definition, in other less preferred areas where carrying capacity has not yet been reached.

The concept of carrying capacity has been used in different ways, leading to confusion and doubts as to its usefulness (Dhondt 1988). Nonetheless, the concept has merit because it encapsulates the common-sense notion that a limit to the numbers of birds that can winter in an area must exist. Problems arise when the term is used without definition or in such a way as not to capture this simple idea; for instance, as noted by Dhondt (1988), when it is used as a synonym for equilibrium population size. But when the numbers of shorebirds in part of their feeding gradient apparently reach a plateau, despite a ready supply of additional birds, it would serve little purpose to introduce a new term that encapsulates the same basic idea.

In shorebirds, the difficulty in its use is empirical rather than conceptual. It comes in deciding whether the maximum possible density, as distinct from the maximum density yet observed, has been reached. That numbers reach an apparent ceiling does not necessarily imply that they could not, in new circumstances, reach even higher levels. At these small scales, ceiling densities will mainly be achieved by individuals deciding whether or not to remain in a particular part of the feeding gradient. The decisions may, indeed, often be taken by young birds prospecting for places to overwinter for the rest of their lives (Evans & Townshend 1988; Myers, Schick & Castro 1988). But as overall numbers rise, or the gradient is gradually reduced by habitat loss, intensifying competition may cause the basis of decisions to change, with a consequential effect on bird density in preferred areas. For example, by defending an exclusive feeding territory, an individual may protect itself from interference and depletion, and this could set a limit to local density. But as competitor pressure increases, it may no longer be economical to defend a feeding area (Myers, Connors & Pitelka 1984). Territories would then break down, allowing bird densities to rise to a new limit based on different decisions, or even determined by prey depletion. Ultimately, studies of animal decision-making and fitness will provide the only sound basis for predicting the 'top-line' above which local bird density cannot rise, this being the main idea behind the concept of carrying capacity. In no case where plateau densities have been detected is it yet known whether still higher bird densities might not be reached if competitive pressure were to intensify still further through the redistribution of birds that inevitably follows habitat loss (Goss-Custard 1977). The problem of how to distinguish the maximum density possible from the maximum that has yet been observed remains unresolved.

Equilibrium population size

If conditions deteriorate locally through environmental change, the most likely response of the birds is to try to find other areas not yet at capacity. This means that, in net terms, birds forced to leave the preferred areas may have to move to less favoured areas where their chances of survival are probably less. In the longer term, therefore, a series of local habitat losses or degradations would lead to a greater proportion of the birds wintering in less suitable regions of their feeding conditions gradient, so the mortality rate of the population as a whole would be expected to increase. If, conversely, the local changes favoured the birds, their survival rates would be expected to increase. Though local environmental changes are likely to affect local bird numbers mainly by bird movements, the aggregate effect of many small environmental changes would eventually be expected to produce a change in the mortality rate of the population as a whole. The task then is to predict what effect this would have on the size of the population as a whole.

At the level of the world population, numbers are determined solely by the balance between the mortality rate and the birth rate. Either, or both, of these can be assumed to act in a regulatory way because shorebird populations do not normally increase to infinity or go extinct. However, the equilibrium population size will be affected by a change in either the regulatory (density-dependent) or non-regulatory (density-independent) rates. So long as there is not an exactly compensating factor present, anything that affects the birth and death rates will affect the balance between them, and thus the equilibrium population size (Sinclair 1989; Goss-Custard 1993).

The death rate during the non-breeding period will depend on how many of the birds can find places in the various food gradients comprising their winter food supply that provide intake rates that enable them to survive. If an increasing number of these are already at capacity, an increasing proportion of birds might be unable to find adequate feeding places, and so die: indeed, many prospecting juveniles may die before they find a place in which to settle at all. Simulations with a shorebird population dynamics model show how the total, year-round population size is sensitive to increased winter mortality rates, whether or not they are dependent on density, and even if they only affect young birds, and even if strong regulation also occurs on the breeding grounds (Goss-Custard 1981; Goss-Custard & Durell 1990). This means that changes in winter feeding conditions could have a considerable effect on the equilibrium population size, even though considerable constraints on population increase also occur on the breeding grounds in the summer. Since the equilibrium population size, in turn, affects the supply of birds to particular estuaries, the overall state of the winter feeding conditions can then affect the numbers of birds living in particular estuaries that are not yet at capacity.

By the same argument, the equilibrium population size also depends on the mortality and reproductive rates during the breeding season and on migration. Unless feeding conditions are generally improving in Britain through organic enrichment, increased survival and birth rates in summer may explain why the winter numbers of some species continue to increase in Britain (Salmon, Prys-Jones & Kirby 1989), despite the apparently large number of threats to their winter food resources. As the supply of birds available to spend the winter in Britain increases and the most preferred areas reach apparent capacity, birds seem to spread further down the gradient of feeding conditions, whether within one estuary or between estuaries. This illustrates why equating carrying capacity in these migratory populations with the equilibrium population size of the whole population would be so inappropriate. The equilibrium population size in winter reflects as much processes occurring on distant breeding grounds as in the winter itself. Applying the notion of carrying capacity in these circumstances loses the basic idea of the concept which is that a particular place can only support a certain number of birds. It is therefore more useful just to refer to the carrying capacity of particular wintering areas.

The practical problem of making predictions in a particular case

This paper has sketched out a framework for thinking about the consequences for shorebird numbers of various environmental changes. It has stressed the need to forecast the new trade-off between changes that may make it both easier and more difficult for individuals to balance their demand/supply equation. It has also stressed the need to think about the consequences at both the local and global scales; in the latter case, there is a need to recognise that population size is at least as much affected by processes occurring on distant breeding grounds as in the estuaries themselves. This may not only require a change in the concepts used to think about shorebird populations (Goss-Custard 1993) but also represents a considerable practical scientific challenge. Yet predictions are required long before anything like a full understanding is available. There is an urgent need now to develop methodologies that, however imprecise and provisional, allow predictions to be made that are better than guesswork.

One approach is to predict future bird densities on estuaries from correlations between their existing densities on estuaries and environmental variables that can themselves be predicted. For example, shorebird density often correlates with the density of their prey or some aspect of the substrate, or both (Goss-Custard 1983). Future bird densities might therefore be predicted from predictions of future prey densities, for example. The problem here, though, is that the future prey densities can also be difficult to predict (Goss-Custard *et al.* 1991; Warwick *et al.* 1991). Another approach is suggested by the work of Emerson (1989) who showed that invertebrate production can be predicted reasonably well from certain physical and hydrodynamical variables, such as wind-stress, tidal range and ambient temperature. These 'master variables', that appear to drive the biological productivity of the system, are often easier to predict following environmental change than are biological variables. If so, it might be possible to predict future bird densities directly from these master variables without working through all the intermediate ecological processes that affect the abundance of their prey, these being only imperfectly understood anyway.

Even at this early stage in the development of such techniques, it is important to recognise the limitations. First, unless bird densities on estuaries are everywhere at carrying capacity, which seems unlikely, such empirical relationships may only give a worst-case prediction. If forced to do so, bird densities at a particular level of invertebrate productivity might be able to increase above the levels observed, without increases in bird mortality. Second, the outcome of environmental change on estuaries will depend on the interaction between population processes occurring there and on the breeding grounds, and this is only understood in very general terms. This means that, while a simple methodology for predicting future bird densities does need to be developed, there is an important need at the same time to increase our understanding of the processes and systems involved. This is a two-pronged approach, involving the development of simple predictive methodologies on the one hand while improving our basic understanding on the other. It may provide a means by which ever more realistic forecasts can be achieved while, at the same time, providing the best available answers to immediate and pressing practical questions.

References

Beukema, J.J. 1990. Expected effects of changes in winter temperatures on benthic animals living in soft sediments in coastal North Sea areas. *In*: *Expected Effects of Climatic Change on Marine Coastal Ecosystems,* ed. by J.J. Beukema, W.J. Wolff & J.J.W.M. Brouns, 83–92. Dordrecht, Kluwer Academic Publishers.

Dhondt, A.A. 1988. Carrying capacity: a confusing concept. *Acta Oecologia 9*: 337–346.

Emerson, C.W. 1989. Wind stress limitation of benthic secondary production in shallow, soft-sediment communities. *Mar. Ecol. Prog. Ser. 53*: 65–77.

Ens, B.J., & Goss-Custard, J.D. 1984. Interference among oystercatchers, *Haematopus ostralegus*, feeding on mussels, *Mytilus edulis*, on the Exe estuary. *J. Anim. Ecol. 53*: 217–231.

Evans, P.R. 1976. Energy balance and optimal foraging strategies in shorebirds: some implications for their distributions and movements in the non-breeding season. *Ardea 64*: 117–139.

Evans, P.R., Henderson, D.M., Knights, P.J., & Pienkowski, M.W. 1979. Short-term effects of reclamation of part of Seal Sands, Teesmouth, on wintering waders and shelduck. *Oecologia 41*: 183–206.

Evans, P.R., & Townshend, D.J. 1988. Site faithfulness of waders away from the breeding grounds: how individual migration patterns are established. *Acta XIX Cong. Int. Ornithol.*: 594–603.

Goss-Custard, J.D. 1977. The ecology of the Wash. iii. Density-related behaviour and the possible effects of a loss of feeding grounds on wading birds (Charadrii). *J. Appl. Ecol. 14*: 721–739.

Goss-Custard, J.D. 1980. Competition for food and interference among waders. *Ardes 68*: 31–52.

Goss-Custard, J.D. 1981. Role of winter food supplies in the population ecology of common British wading birds. *Verhand. Orn. Gesell. Bay. 23*: 125–146.

Goss-Custard, J.D. 1983. Spatial and seasonal variations in the food supply of waders (Charadrii) wintering in the British Isles. *Proc. 3rd Nordic Cong. Orn.* 85–96.

Goss-Custard, J.D. 1984. Intake rates and food supply in migrating and wintering shorebirds. *In*: *Shorebirds: migration and foraging behaviour* ed. by J. Burger & B.L. Olla, 233–270. New York, Plenum Press.

Goss-Custard, J.D. 1985. Foraging behaviour of wading birds and the carrying capacity of estuaries. *In*: *Behavioural ecology: ecological consequences of adaptive behaviour,* ed. by R.M. Sibly & R.H. Smith, 169–188. Oxford, Blackwells.

Goss-Custard, J.D. 1993. The effect of migration and scale on the study of bird populations: 1991 Witherby lecture. *Bird Study 40*: 81–96

Goss-Custard, J.D., & Durell, S.E.A. le V. dit, 1987. Age-related effects in oystercatcher *Haematopus ostralegus*, feeding on mussels, *Mytilus edulis*. 3. The effect of interference on overall intake rate. *J. Anim. Ecol., 56*: 549–558.

Goss-Custard, J.D., & Durell, S.E.A. le V. dit, 1988. The effect of dominance and feeding method on the intake rates of oystercatchers *Haematopus ostralegus*, feeding on mussels, *Mytilus edulis*. *J. Anim. Ecol., 57*: 827–844.

Goss-Custard, J.D., & Durell, S.E.A. le V. dit, 1990. Bird behaviour and environmental planning: approaches in the study of wader populations. *Ibis 132*: 273–289.

Goss-Custard, J.D., McGrorty, S., & Kirby, R. 1990. Inshore birds of the soft coasts and sea-level rise. *In*: *Expected Effects of Climatic Change on Marine Coastal Ecosystems,* ed. by J.J. Beukema, W.J. Wolff & J.J.W.M. Brouns, 189–193. Dordrecht, Kluwer Academic Publishers.

Goss-Custard, J.D., & Moser, M.E. 1988. Rates of change in the numbers of dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. *J. Appl. Ecol. 25*: 95–109.

Goss-Custard, J.D., Warwick, R.M., Kirby, R., McGrorty, S., Clarke, R.T., Pearson, B., Rispin, W.E., Durell, S.E.A., & Rose, R.J. 1991. Towards predicting wading bird densities from predicted prey densities in a post-barrage Severn estuary. *J. Appl. Ecol. 28*: 1004–1026.

Horwood, J., & Goss-Custard, J.D. 1977. Predation by the oystercatcher, *Haematopus ostralegus* (L.), in relation to the cockle, *Cerastoderma edule* (L.), fishing in the Burry Inlet, South Wales. *J. Appl. Ecol. 14*: 139–158.

Moser, M.E. 1988. Limits to the numbers of grey plovers, *Pluvialis squatarola*, wintering on British estuaries: an analysis of long-term population trends. *J. Appl. Ecol. 25*: 473–486.

Myers, J.P., Connors, P.G., & Pitelka, F.A. 1984. Optimal territory size and the sanderling: compromises in a variable environment. *In*: *Foraging Behaviour,* ed. by A.C. Kamil & T.D. Sargent, 135–158. New York, Garland Press.

Myers, J.P., Schick, C.T., & Castro, G. 1988. Structure in sanderling (*Calidris alba*) populations: the magnitude of intra- and interyear dispersal during the nonbreeding season. *Int. Orn. Congr. 19*: 604–615.

Pearson, T.H., & Rosenberg, R. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanogr. Mar. Biol. Ann. Rev. 16*: 229–311.

Pienkowski, M.W. & Evans, P.R. 1985. The role of migration in the population dynamics of birds. *In*: *Behavioural ecology: ecological consequences of adaptive behaviour,* ed. by R.M. Sibly & R.H. Smith, 331–352. Oxford, Blackwells.

Salmon, D.G., Prys-Jones, R.P., & Kirby, J.S. 1989. *Wildfowl and wader counts 1988–89*. Slimbridge, The Wildfowl & Wetlands Trust.

Selman, J., & Goss-Custard, J.D. 1988. Interference between foraging redshank, *Tringa totanus*. *Anim. Behav. 36*: 1542–1545.

Sinclair, A.R.E. 1989. Population regulation in animals. *In*: *Ecological Concepts,* ed. by J.M. Cherrett, 197–242. Oxford, Blackwells.

Warwick, R.M., Goss-Custard, J.D., Kirby, R., George, C.L., Pope, N.D., & Rowden, A.A. 1991. Static and dynamic environmental factors determining the community structure of estuarine macrobenthos in SW Britain: why is the Severn estuary different? *J. Appl. Ecol. 28*: 329–345.

Whitfield, D.P., Evans, A.D., & Whitfield, P.A. 1988. The impact of raptor predation on wintering waders. *Int. Orn. Congr. 19*: 674–687.

Zwarts, L. 1978. Intra- and interspecific competition for space in estuarine bird species in a one-prey situation. *Int. Orn. Congr. 17*: 1045–1050.

Land-claim on British estuaries: changing patterns and conservation implications

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Estuaries are one of Britain's most important wildlife resources. They have also been extensively utilised by man for many centuries. Man has claimed land from the productive intertidal parts of estuaries since at least Roman times, first mainly by enclosing saltmarshes for agriculture, but more recently also for urban, industrial, port and recreational facilities. Land-claim has been progressive and piecemeal, generally at a rate of 0.2%–0.7%/year, and has taken over 25% of intertidal from many estuaries. These rates are typical of the rates of wetlands loss world-wide. Overall there is known to have been land-claim of at least 89,000 ha on just eighteen of the 155 British estuaries – a habitat loss of 37% of their former area, and a 23% loss from the overall British resources. Widespread land-claim, much of it rubbish and spoil disposal, is continuing. The 123 land-claims current in 1989 affected 45 estuaries, and 135 proposals affected 55 estuaries (35% of the total number of British estuaries). In 1989 proposals were mostly for housing schemes, marinas and barrages, which if built would claim at least a further 8% of the remaining intertidal resource. Many proposals affect estuaries of known international importance for their wildlife. Continuing estuarine land-claim has serious implications for the future maintenance of healthy estuaries and the international obligations to safeguard their wildlife.

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Introduction

Wetlands are amongst the most important and threatened wildlife habitats in the world. Estuarine wetlands are widely regarded as some of the most productive ecosystems in existence, largely as a consequence of the continual input and cycling of nutrients. Estuaries are generally characterised by a low diversity of macrobenthic organisms, but these occur in large numbers and with a very high biomass. This provides a large food resource which is exploited by abundant predators such as other invertebrate species, fish, mammals and birds.

Man has also long exploited these resources of estuarine plants, shellfish, fish and wildfowl for food, and the saltmarsh and grassland habitats for stock grazing. Furthermore the shelter afforded by estuaries has for many centuries also provided safe anchorage for the shipping so vital in carrying the imports and exports for an island people such as the British.

Nevertheless in many places in Britain and elsewhere man has been progressively and extensively modifying estuarine habitats for a variety of purposes. Such land-claim, often referred to as 'reclamation' or 'winning land from the sea', has generally been undertaken with the underlying view that estuarine intertidal (and occasionally subtidal) land is freely available and of no intrinsic value. Its conversion and modification has been viewed as being of benefit since it brings more land under direct human control. The idea of intertidal land of estuaries being wasteland suitable only for covering up with landfill seems to have become more prevalent with the increasing urbanisation of many estuaries since the Industrial Revolution. Formerly the major modifications to estuaries were for agricultural purposes; most other forms of human exploitation centuries ago depended on the maintenance and health of the intertidal flats and marshes from which, for example, to gather fish, shellfish and wildfowl.

This paper briefly reviews the patterns of overall estuarine habitat destruction set into the context of Britain's remaining estuarine resource. It brings together the information on land-claim for each of over 250 different human activities for which information has been collated by the former Nature Conservancy Council's Estuaries Review (now forming part of the Joint Nature Conservation Committee's Coastal Review) so as to provide an overview of the variety of patterns of land-claim that have taken place on different estuaries. This is followed by a synthesis of the current distribution and causes of habitat loss, and the implications of the further proposals active in 1989, throughout the estuarine resource in Britain. The wider context of these habitat losses and the implications for wildlife conservation complete the paper. Although this paper restricts its coverage to Britain such land-claims have been and are widespread world-wide (e.g. Gosselink & Baumann 1980; Frayer *et al.* 1983; Tiner 1984; Howe 1987; Dugan 1993; Hötker & Kölsch 1993).

The information presented here is derived and updated from the more detailed analyses reported in the main Estuaries Review report *Nature conservation and estuaries in Great Britain (*Davidson *et al.* 1991), to which reference should be made for details of methodology. That report provides an overview of wildlife, its conservation, and human activities on estuaries throughout Britain. It provides a basis for the development of future estuarine conservation strategies.

We deal here only with the direct loss of wildlife habitat, and focus on the intertidal and subtidal parts of estuaries. Additional information is, however, presented where relevant for the variety of marine-influenced terrestrial habitats around the estuaries. Further analysis of coastal habitat losses, particularly on the variety of associated coastal terrestrial habitats, is under way (Doody in prep.) and current information about rates of habitat loss, and projections of future losses, have recently been summarised by Pye & French (1992).

It is important to recognise that these direct losses of habitat, although very important, are only one of the types of pressure exerted by human activities on estuarine wildlife. The estuarine areas that remain after the land-claims described here are subject to a variety of pressures such as degradation through high levels of pollution and waste discharge, and damage to habitats and disturbance to wildlife by high levels of visitor and recreational pressure (Davidson & Rothwell 1993). Since, as described below, the area of estuarine habitat continues to diminish whilst recreational pressures tend to increase, the remaining areas of high wildlife conservation importance face continuing and increasing pressures.

The episodes of land-claim described in this review are chiefly 'terrestrial-based', i.e. they have originated from human extension of activities out from the existing shoreline. Hence these encroachments have, in general, had a disproportionately large impact on the upper tidal flats and saltmarshes of estuaries. This has particular implications for the ways in which the remaining parts of estuaries can be used by wildlife (see for example Davidson & Evans 1986a). Only more recently have such 'land-based' encroachments begun to claim areas covering the whole range of intertidal shore levels down to low-water mark and below, as technologies of land-claim have permitted the rapid covering of extensive areas of estuaries for docks and industrial developments and other similar purposes. These major land-claims can also have substantial and long-term indirect effects on the remaining habitats of an estuary, through their impact by reduction of the tidal prism and hence modification to tidal currents, sediment transport and the location of tidal channels.

For the purposes of this paper we treat land-claim as any deliberate action by people to convert naturally functioning intertidal and subtidal areas into land, and to convert terrestrial wildlife habitat into agricultural, urban or industrial use of no direct wildlife benefit. Land-claims vary greatly in size and many, although individually small, contribute considerably to the overall piecemeal pattern of land-claim (see Figure 6): hence in our analyses we have included all known land-claim cases whatever their size.

Britain's estuarine resource

The Estuaries Review developed a definition of estuaries derived from NERC (1975) and Cowardin *et al.* (1979): a partially enclosed area at least partly composed of soft tidal shores, open to saline water from the sea, and receiving fresh water from rivers, land run-off or seepage. For collection of data we defined a core intertidal and subtidal area for each estuary and collected data also for an associated terrestrial area of semi-natural habitats surrounding each estuary.

Using this definition 155 parts of the British coastline were identified by the Estuaries Review as estuarine (Figure 1). In total these estuaries have over 530,000 ha of intertidal and shallow inshore marine habitats. Of this area over 310,000 ha is intertidal flats and saltmarshes, about 83% of the total intertidal area of the British coastline. The estuaries in Britain vary greatly in size, from the Wash (66,600 ha) and the Severn Estuary (55,700 ha) to the 76 estuaries each with an area of less than 500 ha (Davidson *et al.* 1991). This large and varied estuarine resource is of major national and international importance for its wildlife, and the variety in number and size, and also in geomorphological type and tidal range, make Britain's estuaries of great interest and importance in both European and world terms.

Estuarine habitats are amongst the most naturally-functioning in Britain (e.g. Ratcliffe 1977; Doody in prep.). Estuaries are of great significance in wildlife conservation terms for this naturalness and for the wide range of wildlife features including their coastal habitats, aquatic communities, rare plants and invertebrates, fish, sea and aquatic mammals, and breeding, migrant and wintering bird populations. This great range of wildlife features is described in detail in Davidson *et al.* (1991) and is summarised in, for example, Davidson (1990) and Doody *et al.* (1993).

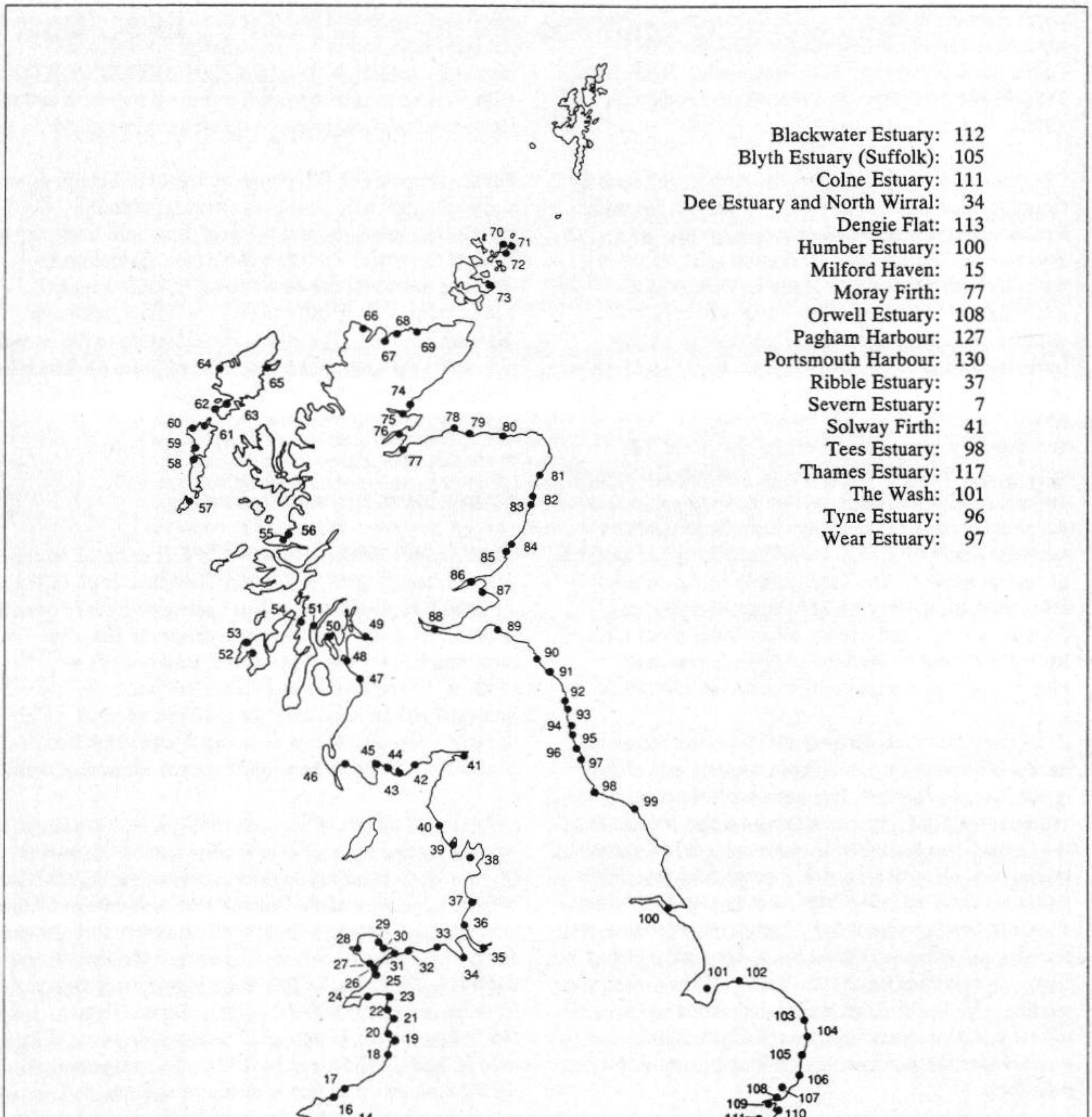

Figure 1 The location of the 155 estuarine sites in Britain covered by the Nature Conservancy Council's Estuaries Review (Davidson *et al.* 1991). Each estuary is marked by its central grid reference, with site numbers being those used in the Estuaries Review and the *Inventory of UK estuaries* (Buck 1993). Only estuaries mentioned in the text are named.

Their utilisation by wildlife is as a linked resource so that many individuals of, for example, migrant birds depend on the presence of a network of different estuaries and habitats for their survival through the year (e.g. Pienkowski & Pienkowski 1983; Davidson & Stroud in press).

This abundant wildlife makes estuaries amongst Britain's most important wildlife conservation features. Their conservation likewise requires a network approach. This has been recognised by the identification under domestic legislation of over 330 Sites of Scientific Interest (SSSIs) associated with British estuaries. These are 6% of the total number of SSSIs in Britain, but because many are large these estuarine sites cover 23% of the British SSSI area. Many parts of the resource, including both large and small estuaries, contribute to this conservation network, and large parts of the area of British estuaries are within the SSSI series (see e.g. Burd 1989).

Already, 69 British estuaries have been identified as falling within the network of sites of international importance for their wetland habitats and the wildfowl which depend on them. This importance is recognised under the Convention on the Conservation of Wetlands especially as Waterfowl Habitat (the 'Ramsar' Convention), and the EEC Directive on the conservation of wild birds (Directive 79/409). As signatory to the Convention and as an EC Member State, Britain is required to designate wetlands of international importance and Special Protection Area (SPAs) for birds, to promote the 'wide use' of wetlands, and to maintain and enhance the populations of scarce and migratory birds in their areas of distribution through the safeguarding of their habitats. These and other international measures are further described in Stroud *et al.* (1990) and Davidson *et al.* (1991). In addition the future designation of Special Areas of Conservation (SACs) under the 1992 EC Directive on the conservation of natural habitats and flora and fauna (the 'Habitats and Species' Directive) will lead to further identification of the international importance of parts of Britain's estuaries (see for example Davidson & Stroud in press).

The 'estuarine squeeze'

The land-claim summarised in this paper is just one, albeit important, component of an estuarine 'intertidal squeeze' (Figure 2). This squeeze works on both the existing high water and low water marks, and on the balance of habitats within the intertidal zone.

On the one hand there are factors affecting the low water mark, notably those such as dredging that deepens and widens the subtidal channels for shipping. This can directly remove parts of the subtidal and lower intertidal areas of estuaries, and can cause loss and damage by the

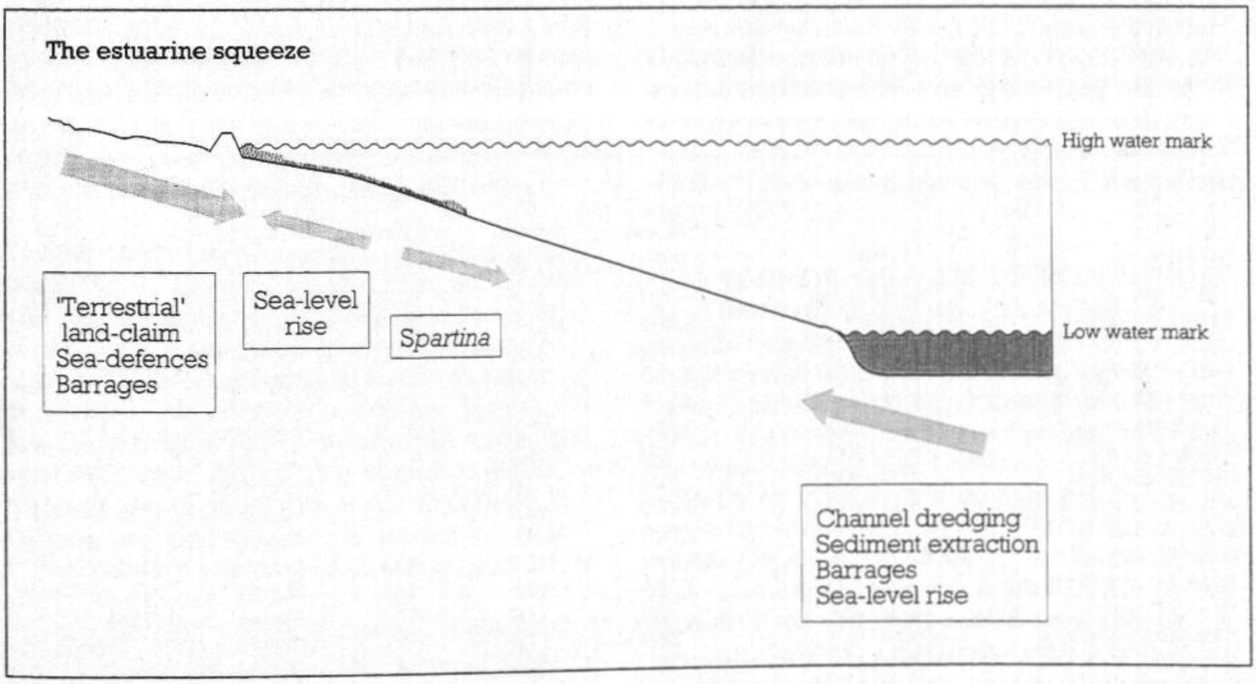

Figure 2 The 'estuarine squeeze'. The direction of pressure on estuaries tends, for a variety of reasons, to inundate or remove the lower intertidal levels and to depress or restrict upper tidal levels. Some effects, such as the encroachment of cord-grass *Spartina*, tend to increase the proportion of saltmarsh at the expense of open tidal flats within this already restricted intertidal zone.

indirect effects of increased scour and erosion on artificially steepened shores and in some places from intertidal dumping of dredging spoil.

Barrages and barriers constructed for a variety of purposes at the least restrict tidal amplitude, effectively raising low water mark. In extreme cases, such as the 'leisure barrage' planned for Cardiff Bay, the tidal influence is entirely removed, so that low water is effectively raised to equal high water. This 'water-claim' has the same effect as land-claim in destroying all existing intertidal estuarine habitats within the barrage, and also tends to remove the brackish or sea-water influence on subtidal aquatic communities.

In addition, rising sea-level, especially if accelerated by global warming, will inundate existing lower tidal flats (Carter 1989; Davidson *et al.* 1991). The continued presence of any intertidal area in the face of such rising sea-levels then depends on rates of sediment deposition keeping pace with the rise in water level, but this may not always occur. Burd (1992) has shown that rising relative sea-levels in south-east England have led to substantial erosion of saltmarshes, resulting in a reduction of about 20% of saltmarsh area in a 15-year period. Added to this was a land-claim of 2% of the area over the same period.

There is also a squeeze at the high water mark. Tidal and storm surge barriers tend to depress the high water mark, so that existing upper saltmarshes become dry and subsequently develop terrestrial vegetation. Sea level rise would naturally move the high water mark gradually further inland and altitudinally higher. But the construction and maintenance of sea-walls for sea defence around many parts of Britain's estuaries greatly restricts this. The impacts of this are complex, depending on patterns of sediment redistribution. They can be to increase periods of tidal inundation at upper tidal levels, or to alter deposition patterns and steepen the shore.

The 'terrestrial land-claims' described below are an extension of this sea defence impact since the effect of the land-claims are to shift the sea defence line further downshore so as to use the land behind it for another purpose. The spread of cord-grass *Spartina* acts in an analogous way to the other habitat loss impacts described here, since it raises marsh levels and moves the saltmarsh coverage further downshore, so claiming and reducing areas of intertidal mudflats and sand flats (see Doody 1984).

The overall impact of all these squeezes and land-claim is a general one of narrowing and steepening of the shore profile, and in many cases particularly reducing areas of intertidal flats. There are some key points of impact that emerge from the land-claim patterns described below. First, on each British estuary land has been claimed at different times and for different purposes. Thus although overall the impacts have been broadly similar no two patterns are the same. Second, in most estuaries the loss of areas and habitats has occurred, and continues to occur, in a piecemeal way in both time and space. The patterning is complex, with often a mixture of very small and very large land-claim episodes for a wide variety of purposes. Thirdly, land-claim on British estuaries has been reducing the overall

Table 1 Examples of areas of historical land-claim on some estuaries around Britain

estuary	*period*	*area lost (ha)*	*% former total area*	*% former intertidal area*	*source(s)*
The Wash	since Roman	47,000	41.4	61.2	Dalby (1957), Doody (1987)
Severn Estuary	since Roman	c. 8,000	c.12.6	c.32.1	Doody (in prep.)
Dee Estuary/North Wirral	since 1730	6,000	27.1	31.6	Davidson *et al.* (1991)
Humber Estuary	1600–1850	4,600	13.2	25.4	Stickney (1908)
Greater Thames Estuary	mostly pre-1800	4,340	8.9	11.9	Macey (1974)
Tees Estuary	since 1720	3,300	71.0	87.5	Davidson *et al.* (1991)
Ribble Estuary	since 1800	2,320	16.3	17.9	Davidson *et al.* (1991)
Morecambe Bay	1200–1900	1,320	2.8	3.7	Gray (1972)
Ore/Alde/Butley Estuary	since 1200	3,640	66.7	73.2	Beardall *et al.* (1988)
Deben Estuary	since 1200	2,240	69.0	76.5	Beardall *et al.* (1988)
Stour Estuary	since 1200	1,600	38.7	49.4	Beardall *et al.* (1988)
Blyth Estuary (Suffolk)	since 1200	1,280	80.5	84.5	Beardall *et al.* (1988)
Orwell Estuary	since 1200	980	35.4	63.0	Beardall *et al.* (1988)
Southampton Water	since 1830	690	14.8	33.4	Coughlan (1979), Tubbs (1984)
Poole Harbour	since 1807	530	12.2	20.5	May (1969)
Portsmouth Harbour	since 1540	490	23.5	33.7	Davidson *et al.* (1991)
Mersey Estuary	19th century	490	5.2	8.0	Doody (in prep.)
Tay Estuary	19th century	150	1.2	2.6	Doody (in prep.)
Total		88,970			

size of the resource for many centuries and seems to continue largely unabated (see also Pye & French 1992).

Historical estuarine land-claim in Britain

Land-claim on estuaries and coasts has been practised since at least Roman times. Much of the early land-claim was for the enclosure of saltmarshes with earth banks so as to restrict and control tidal inundation and provide better grazing conditions for stock. These agricultural land-claims have been very extensive and have been progressive over many centuries on many estuaries. They account for much of the overall area of estuarine habitat lost to man. In general it can be assumed that almost all of the low-lying land protected by sea-walls surrounding estuaries was originally enclosed for agricultural purposes, although sea-walls have often later been enlarged or strengthened to protect the housing and industry that expanded subsequently onto the enclosed marshes. On this basis of presence of linear sea defences parts of at least 132 of the 155 British estuaries (85% of the total number) have lost intertidal habitat to agricultural land-claim in the past (Davidson *et al.* 1991).

In some estuaries such as the Wash and the Ribble Estuary the main reason for loss of estuarine habitats has continued to be for agricultural purposes. These places have remained largely rural. On other estuaries the more recent intertidal land-claims have been for a wider variety of industrial, urban and, very recently, recreational purposes. Such places as the Tees, Tyne and Wear Estuaries are now largely urban and as well as further intertidal land-claims much of the land formerly claimed for agricultural purposes has now been secondarily 'claimed' for urban and industrial uses.

The full extent of the historical land-claim has not been established for all British estuaries, but it is clear from many cases that progressive land-claim is the typical pattern, and that such claims have been occurring for many centuries. Many British estuaries have lost between one quarter and one half (or more) of their intertidal habitats (Table 1), and often this amount has been claimed over the last 150-200 years. Rates and timings of loss vary for different estuaries (Figure 3), but rates have generally averaged between 0.2% and 0.7% of intertidal area per year (Davidson *et al.* 1991).

Figure 3 illustrates the different timings of piecemeal land-claim over the centuries. The purposes for land-claim also differ between estuaries. In Portsmouth Harbour small areas of land-claim for dock and other urban uses have occurred progressively since at least 1540 around many parts of the tidal basin, but land-claim was particularly rapid for dock development during the late 19th century, and more recently for road construction and rubbish tipping. In contrast, largely agricultural land-claim on the Dee Estuary in north-west England was at its most rapid during the 18th century, whereas on Fagbury Flats in the lower reaches of the Orwell Estuary the rate of claim accelerated during the second half of the 20th century. This illustrates a common pattern of the development of new docks closer to the mouths of estuaries so as to accommodate larger ships, a phenomenon that is continuing.

The Tees Estuary is widely known as a prime example in Britain of progressive industrial and dock-related land-claim (see for example Davidson & Evans 1986a). Here progressive land-claim has been almost continuous since at least the early 18th century and almost 90% of the intertidal area has been lost since 1720; despite its industrial reputation some of the most rapid and extensive land-claims were made initially for agriculture prior to 1890 (Figure 3, Figure 4). Since then progressive downstream encroachment, often speculatively, for a variety of industrial purposes has removed a similarly large area of intertidal flats. This culminated in the largest single area being claimed for industry and port-related developments as recently as 1973/74 (Figure 4).

Agricultural land-claim elsewhere has also generally occurred progressively out over saltmarshes from the shore. The largest single area of land-claim on any British estuary is thought to be the progressive claim of some 47,000 ha of the Wash since Roman times. Here and on some other estuaries, the effect of this repeated sea-wall construction has been to encourage further accretion of saltmarsh (at the expense of tidal flats) outside each new wall. This then permits further land-claims onto each newly developed area of marsh (Doody 1987).

Minimum areas of historical land-claim are known for the 18 British estuaries in Table 1. Although these estuaries form only 14% of the British total, they do include most of the largest estuaries so that they presently form almost half the estuarine intertidal area in Britain. Known areas lost on these estuaries total almost 89,000 ha. Assuming that there have been no extensive compensatory expansions of intertidal areas during this period, this means that at least 22.4% of British intertidal estuarine habitat has been claimed since Roman times. Since there are other estuaries on which extensive agricultural and urban land-claim is known to have taken place, it is likely that direct human actions have removed between one-quarter and one-third of estuarine intertidal area in Britain during the last 2,000 years.

Although there are no precise figures for the total losses of estuarine area in Britain, estimates have been made

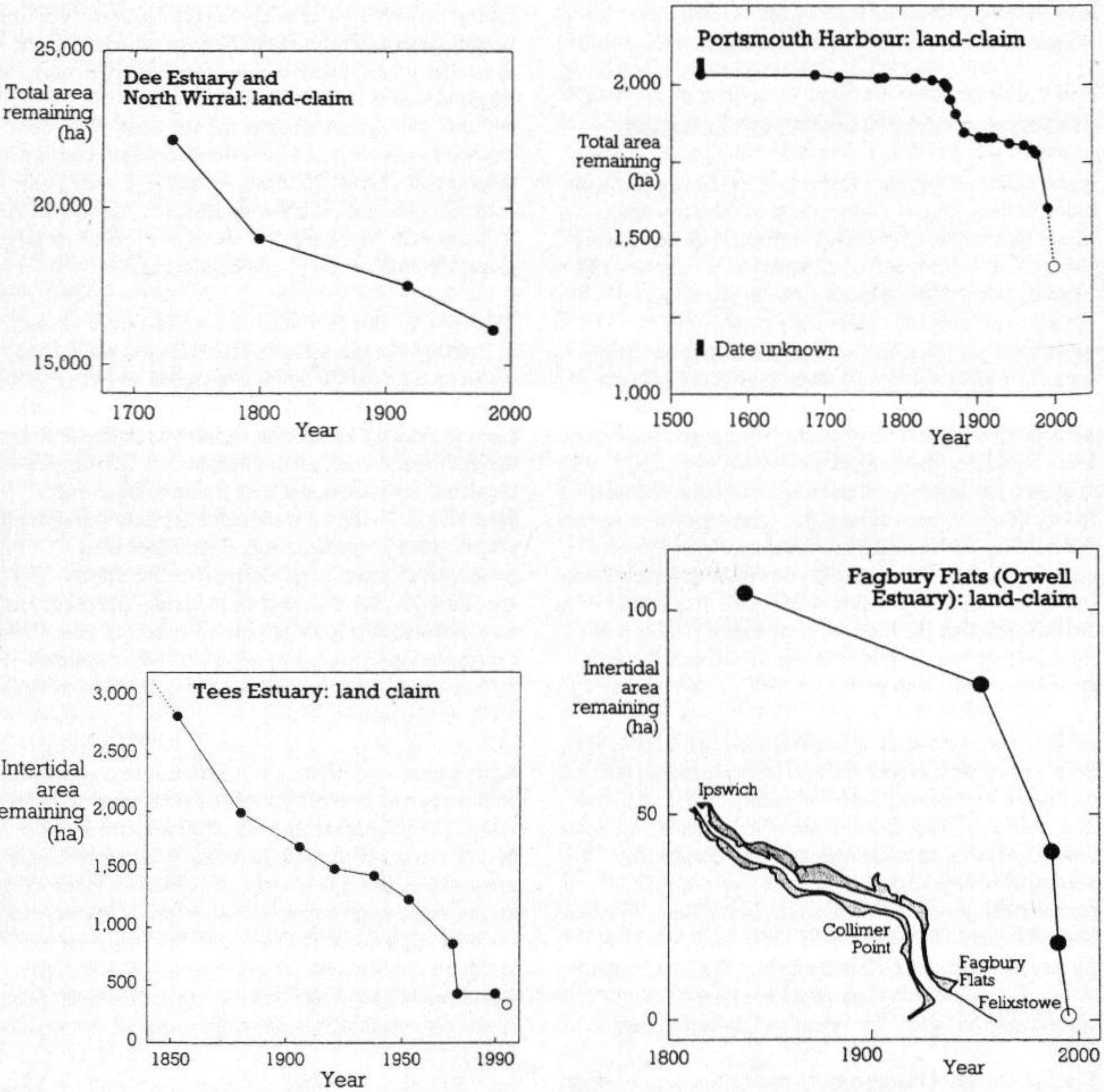

Figure 3 Four examples of the historical decline in estuarine area resulting from land-claim. Note that time-scales differ, and that for the Orwell Estuary only the progressive loss of Fagbury Flats (close to the estuary mouth) are shown. Open symbols show the further reductions in area that would occur if land-claim proposals extant in 1989 were implemented.

for some coastal habitats (see also Pye & French 1992). Doody (in prep.) estimates that approximately 57,000 ha of saltmarsh has been claimed for agriculture alone in Britain. This represents about a 50% decline in saltmarsh habitat. In addition, other pressures affect the surrounding terrestrial habitats, both through total loss of these habitats and damage to them. Doody (in prep.) describes the major pressures on sand dunes, shingle and other coastal habitats both adjacent to estuaries and elsewhere on the coasts of Britain. For example, many dune systems have been extensively afforested, often in an attempt to stabilise mobile dunes, and on some areas such as the Culbin Forest and Findhorn adjacent to the Moray Firth afforestation covers in excess of 90% of the dune system. Other dune pressures include intensive agriculture, recreation (notably trampling and damage from off-road vehicles), golf-courses and a variety of other developments.

Much of the agricultural land-claim of saltmarshes prior to the late 20th century created areas of semi-permanent wet lowland grasslands, with various degrees of freshwater and brackish influences. Many of these coastal grazing marshes have developed considerable

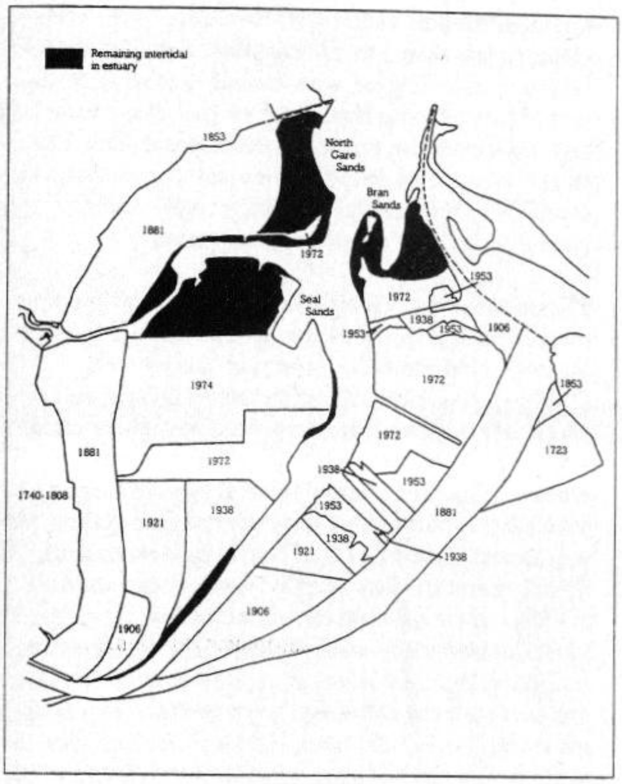

Figure 4 The progressive piecemeal land-claim on the Tees Estuary. Dates within each block are those by when the land-claim is known to have occurred.

wildlife conservation interest in their own right, notably in their ditch flora and fauna, and their breeding and wintering bird assemblages. Coastal grazing marshes have, however, themselves been under considerable pressure, particularly in south-east England and East Anglia. Losses of these grazing marshes through this 'secondary land-claim' have been very extensive. Substantial declines have occurred in the remaining areas associated with estuaries in the Norfolk Broads, Essex, the Thames Estuary and north Kent (Williams & Hall 1987; Thornton & Kite 1990). In the Thames Estuary only about 30% of grazing marsh present in the 1930s now remains, and proportional losses in some other areas such as in East Essex (around the Colne and Blackwater Estuaries and Dengie) have been even greater, with an 82% decline in the last 50 years. Rates of loss have been generally 1%–2% per year, even higher than most intertidal loss rates.

As for land-claim on intertidal areas, grazing marshes have been claimed for a variety of purposes. Although much of the area has been destroyed by agricultural intensification – the ploughing and conversion of the grassland to arable land – the major loss of grazing marsh in the Thames Estuary, particularly in the inner part of the estuary, has been due to urbanisation, which overall accounts for 35% of the losses (Thornton & Kite 1990). This urbanisation covers a variety of purposes, including housing, industry, leisure facilities and rubbish tipping and spoil disposal.

The overall impact of this land-claim of grazing marshes has been to fragment immensely what was formerly, even as recently as 1935, a largely intact continuous spread of marshland alongside the shores and estuaries. This loss of integrity of grazing marsh systems, which often depend critically on a coherent water level control system, further diminishes the effective losses of the resource beyond the area losses alone. This fragmentation and destruction of grazing marsh continues – for example the last remaining large area of grazing marsh in the inner Thames Estuary, Rainham Marshes, is currently threatened by major infrastructure and leisure developments. These losses to coastal grazing marshes are just part of a widespread intensification of agricultural practice that has destroyed all but 3% of the area of semi-natural grassland in England and Wales during the last 50 years (Fuller 1987).

Unlike many 'hard' land-claims for urban and industrial use, agricultural land-claim is not entirely irreversible. Failure to maintain earth banks and sea-walls leads to their erosion and the re-inundation by the tide of former mudflats and saltmarshes. This has occurred in several places, particularly in southern England where sea-levels rising relative to the land mean that former intertidal areas are often well below the current high water mark and so readily become reflooded. Pagham Harbour was bunded and claimed for agriculture during the 19th century but sea-wall breaches have now permitted the bay to revert to intertidal mudflats and saltmarshes.

By the middle of the 20th century effectively all the intertidal flats and marshes of the Blyth Estuary in Suffolk had been claimed for agriculture (chiefly grazing marsh), leaving only a narrow tidal channel. Since the 1940s, however, the progressive breakdown of unmaintained sea-walls has resulted in the incursion of tidal water over 235 ha (about 15% of the original tidal area) of agricultural land, returning it to tidal mudflats. Here, in contrast to the major agricultural land-claims described above, there has been very little redevelopment of saltmarsh on these tidal flats. These reversions to tidally-influenced habitats provide a model for the development of the 'managed retreat' approach to coping with rising relative sea-levels, especially on eroding coastlines.

Current and proposed estuarine land-claim in Britain

The past pattern of progressive and piecemeal losses of estuarine habitats continues. The Estuaries Review surveys in 1989 identified widespread current human activity causing further losses of intertidal and surrounding terrestrial habitats. These are summarised below, and are likely to provide minimum estimates of the extent and variety of land-claims.

Land-claim in 1989

The 1989 surveys found 123 cases of current land-claim on British estuaries. These are land-claims that were either in progress, or had been consented. Of these, 72 cases (59% of the total) affected intertidal and subtidal parts of estuaries (and often also the surrounding terrestrial habitats), and 51 cases were entirely on these associated terrestrial areas.

In 1989 land-claim was causing loss of intertidal/subtidal habitats on 32 of the 155 British estuaries (over 20% of the total number). In addition a further 13 estuaries were being affected by terrestrial land-claim, so overall 45 (29%) estuaries were currently losing further areas of habitat (Figure 5).

Twenty-six of these estuaries (57% of those undergoing land-claim) had more than one concurrent instance of land-claim. It appears that in 1989 there were particular land-claim 'hot-spots', notably the Severn Estuary, the Greater Thames Estuary (particularly in Essex), the Solway Firth and the Moray Firth.

The 1989 land-claims were occurring for a great variety of reasons and purposes. Overall the most frequent cause of current habitat loss was rubbish tipping (including domestic refuse, spoil disposal and power station ash disposal), accounting for one-third of all current cases. Transport schemes (chiefly road-building), housing, car-parks and marinas together accounted for a further one-third of land-claims. Many other purposes contributed to the remaining third of cases, but it is notable that as yet barrage schemes feature as only a small cause of current claims. The pattern of current land-claim causes for England, Scotland and Wales suggests some regional differences in the main causes of habitat loss. Scotland has relatively few current cases of land-claim for the number of estuaries around its shores but almost half of these (i.e. more than the national average) were rubbish tipping and spoil disposal. There were a higher proportion of marina developments causing land-claim in England than elsewhere, whereas in Wales major causes were housing and car-parks, sea-defence and transport schemes as well as rubbish tipping (Davidson *et al.* 1991).

Individual cases of current land-claim varied greatly in size. Of the 56%–59% of known cases for which area data was available, many were individually small: 43 intertidal/subtidal and 31 terrestrial cases were each claiming less than 5 ha of land (Figure 6). Others were, however, much bigger, with several exceeding 50 ha each. Many of these large areas of land-claim were long-term active rubbish tips and spoil disposal sites. The largest land-claims tended to be spoil disposal sites on the associated terrestrial habitats, notably on the remaining areas of coastal grazing marsh.

The prevalence of rubbish and spoil disposal amongst these large land-claim sites results in this being by far the largest current cause of habitat loss by area (Table 2). Overall 61.5% of the intertidal area and 95.5% of the surrounding terrestrial area under claim was for rubbish and spoil disposal. As a minimum estimate of areas currently under active land-claim or scheduled as part of long-term current land-claims, there was almost 1,100 ha of land below high water mark (chiefly intertidal flats and saltmarshes) and almost 1,300 ha of the adjacent terrestrial land under claim. The intertidal land-claim areas represent 0.35% of the total remaining intertidal resource on British estuaries. Since this area has been calculated from less than two-thirds of the known land-claim cases, the total area under current claim is likely to be closer to 0.5% of the resource. Some of these land-claims are taking place over periods of longer than one year, but nevertheless this amount of current overall loss implied that there has been little or no recent decrease in the annual rate of estuarine habitat loss compared to the long-term historical rates.

Proposed estuarine land-claim

Land-claim pressure on British estuaries continues. In early 1989 the Estuaries Review survey found 135 proposals involving land-claim. These would, if all were carried through to development, affect 55 estuaries (35% of the British total). As for the current land-claims, proposals were scattered around almost the whole coastline of Britain, with the exception of north-west Scotland (Figure 7). There appear to be particular 'hot-spots' of future land-claim pressure in the Greater

Table 2 The percentages of estuarine habitat under land-claim for different purposes in 1989, and the minimum total areas being claimed (from Davidson *et al.* 1991)

activity	*% current land-claim area* *intertidal*	*adjacent terrestrial*
Rubbish and spoil disposal	61.5	95.5
Docks and industry	15.2	2.2
Marinas, housing and car-parks	7.9	1.6
Others	15.4	0.7
Minimum area of land-claim (ha)	1,074	1,297
% of known cases included	59	56

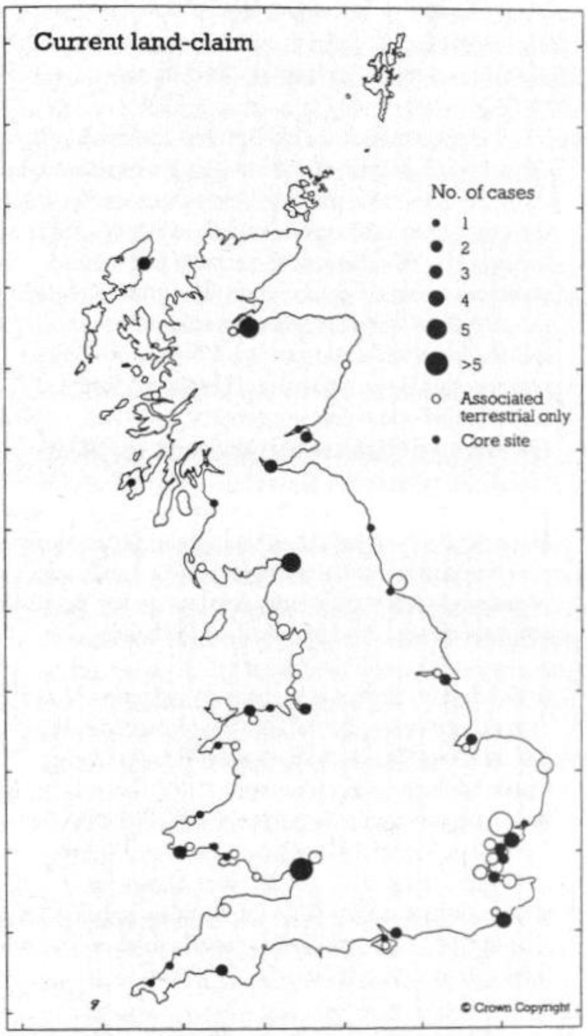

Figure 5 The number and distribution of current (in 1989) land-claim cases on British estuaries. Filled symbols are cases causing intertidal/subtidal habitat loss, and open circles are cases affecting surrounding terrestrial habitats only.

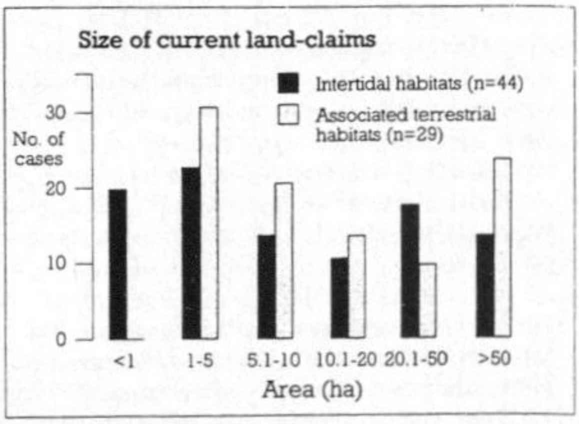

Figure 6 The frequency of current land-claims of different sizes on intertidal habitats and associated habitats on British estuaries

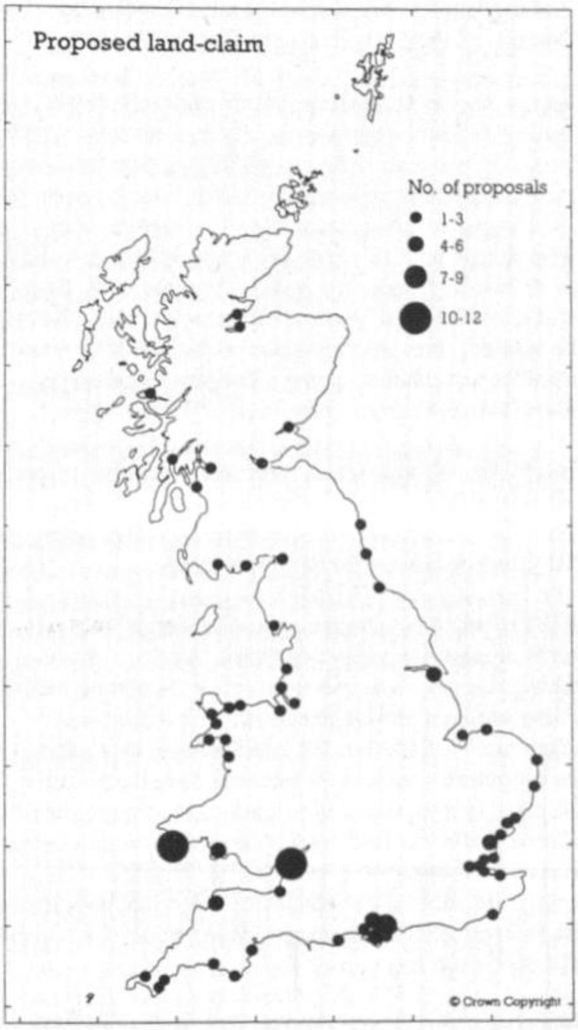

Figure 7 The number and distribution of proposed (in 1989) land-claims on British estuaries

Thames Estuary, around the Solent, in the Severn Estuary, Milford Haven and the Humber Estuary.

As for current land-claims, proposed developments leading to further estuarine habitat loss were for a wide variety of purposes. The pattern differs, however, from current activities: proposals involve higher proportions for housing schemes, marinas and barrage schemes. These three activity categories accounted for over 50% of known land-claim proposals. There are indications that the 'wasteland' approach to estuaries may, however, be waning, since the proportion of proposals for waste disposal and rubbish tipping affecting estuaries was lower than the current proportion of rubbish tipping.

In addition to these active proposals at the time of our surveys, at least another 48 proposals for estuarine land-claim had been made during the previous five years, but had become defunct for various reasons.

It is not possible to derive an assessment of the total areas of estuarine habitat threatened by these active land-claim proposals, since for many the precise nature of the proposed development and their impact was uncertain. Nevertheless the implication is clear: that development proposals on estuaries have the potential for causing a substantial further loss of estuarine habitat. Although the pressure for rubbish tipping, which forms some of the major areas of estuarine habitat currently being lost, seem to be decreasing, other activities are on the increase. Notable amongst these are barrage schemes of various kinds.

Whatever their precise impacts, the various proposed barrages would, if constructed, have a very much greater overall habitat loss impact than any other single type of land-claim proposal. If only half the intertidal areas was lost or damaged within barrages, this would amount to some 25,000 ha of intertidal flats and marshes from tidal power barrages alone – about 8% of the remaining intertidal area in British estuaries. Added to this would be the piecemeal habitat loss on other estuaries for the variety of purposes described above, so the area under future threat of land-claim may approach 10% of the remaining intertidal resource on British estuaries.

Conclusions: some conservation implications of estuarine land-claim

There has been progressive long-term reduction of estuarine wildlife habitats in Britain. This reduction is continuing: there was little evidence in 1989 that rates of habitat loss and damage on estuaries were diminishing. The impacts of piecemeal land-claim on naturally-functioning estuarine systems are many and interrelated such that it is difficult to establish the precise implications of any individual land-claim. Some general implications of land-claim for estuarine function and for wildlife conservation are summarised below.

- Land-claims reduce the tidal prism and tidal volume of estuaries. A land-claim involving a block of land down to low water mark or below can readily reduce the cross-sectional area of estuary by one-quarter or more. This will affect tidal currents and hence sediment transport throughout the estuary, but the magnitude of these impacts has seldom been assessed in determining consent for a land-claim scheme, nor is the downstream impact of any elevation of tide levels consequent on tidal volume reduction within the estuary generally assessed.

- Where land-claim is for docks, subtidal dredging to maintain deep-water channels in naturally shallow sediment-filled estuaries leads to the overall removal of sediment from the estuary system and to possible erosion impacts on other parts of the shore.

- Reduction in estuary area leads to reduction in the overall biomass and productivity of benthic plants and animals. This in turn, other influences being equal, tends to reduce the numbers of fish and birds that the estuary can support (e.g. Moser 1988; Goss-Custard & Moser 1988; Goss-Custard & Durell 1990; McLusky *et al.* 1992). Such losses have implications not only for wildlife conservation but also for the viability of other sustainable uses such as fisheries and shell-fisheries.

- Land-claim can lead to reduced numbers of wintering waterfowl (for which Britain has international responsibilities) able to use an estuary, since numbers are strongly related to estuary area, although there is seldom a reduction in direct proportion to area loss (Evans 1981; Evans *et al.* 1981; Evans & Pienkowski 1983; Davidson & Evans 1986a, 1986b; Lambeck *et al.* 1989; McLusky *et al.* 1992). For feeding behaviour reasons birds cannot just continue to pack in at ever higher densities (see *e.g.* Moser 1988; Goss-Custard & Durell 1990). With land-claim occurring on many estuaries simultaneously opportunities for birds to change location are increasingly limited, and so widespread piecemeal land-claim can ultimately lead to reduced population sizes.

- Land-claims often remove more of the upper than lower levels of intertidal shore (i.e. losses are greatest for saltmarshes and upper tidal flats). Where sediment supply is abundant, e.g. the Wash, new saltmarshes may form seaward of the new edge, so steepening and further reducing areas of intertidal flats. Saltmarshes that do develop under such conditions have a generally low diversity of plant communities and lack the natural transitions to terrestrial habitats - saltmarsh features of substantial conservation importance (Doody 1987; Burd 1989). Where sediment supply is scarce saltmarsh does not reform and remaining tidal flats stay at a low tidal level, reducing estuarine habitat diversity. Loss of upper tidal flats can interfere with macrobenthos populations, since these zones are depended on by some for settlement and survival of spat before later redistribution through the estuary.

- Removal of higher tidal levels of waterfowl feeding grounds increases the risk of mortality for waterfowl during severe weather when Britain acts also as a refuge for many birds from continental Europe. Increased risk occurs because low tidal flats are uncovered for insufficient time for feeding birds to meet their raised daily energy requirements (Davidson & Evans 1986a). Similar difficulties arise during other periods of raised energy requirements, notably during spring migration of waterfowl when fat and protein are being stored not just for migratory flight but also survival and breeding on Arctic breeding grounds. Restriction of feeding opportunities then jeopardises the birds' ability to take sufficient reserves so as to breed successfully, and may even increase their mortality (Evans *et al.* 1991).

- Increasingly there are attempts to develop compensatory habitat provision for major intertidal land-claims. Such attempts do not compensate for loss of estuarine function and seldom provide for the wildlife features lost under the land-claim, even where considerable attempts are made to minimise impact such as by the reinstatement of tidal flow (Davidson & Evans 1986a, 1986b; Hötker in press). There may, however, be opportunity to direct compensatory habitat provision towards creating areas of coastal habitats, such as natural landward transitions, made scarce overall by human interventions. Such an approach would fit with the increasing interest in using managed retreat as an option for coping with rising sea-levels and at the same time minimising the overall decreases in areas of estuarine habitat. Large areas are, however, involved: Pye & French (1992) suggested that minimum areas needed to compensate for habitat losses over the next 20 years in England were 10,000 ha of intertidal flats, 2,750 ha of saltmarshes, 240 ha of sand dunes and 200 ha of shingle. This represents an intertidal area equivalent to that of the existing Dee Estuary, which has the sixth largest intertidal area of any British estuary.

In conclusion, land-claim contributes substantially to the coastal squeeze, especially where relative sea-level is rising in southern and eastern England. It not only fixes the high water mark but moves it seawards. Habitat loss through land-claim is additive to that occurring through the effects rising sea levels and erosion, and prevents the natural movement of the shoreline and estuarine habitats in response to long-term changes in sea level. Land-claim, especially where property of high capital value is involved, often severely restricts options for future sustainable management of a changing coastline based on naturally functioning estuarine systems.

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References

Beardall, C.H., Dryden, R.C., & Holzer, T.J. 1988. *The Suffolk Estuaries.* Saxmundham, Suffolk Wildlife Trust.

Buck, A.L. 1993. *An inventory of UK estuaries. Volume 2. South-west Britain.* Peterborough, Joint Nature Conservation Committee.

Burd, F. 1989. *The saltmarsh survey of Great Britain.* Peterborough, Nature Conservancy Council. (Research & survey in nature conservation, No. 17.)

Burd, F. 1992. *Erosion and vegetation change on the saltmarshes of Essex and north Kent between 1973 and 1988.* Peterborough, Nature Conservancy Council. (Research & survey in nature conservation, No. 42.)

Carter, R.W.G. 1989. Rising sea-level. *Geology Today*: 63-67.

Coughlan, J. 1979. Aspects of reclamation in Southampton Water. *In*: *Estuarine and coastal land reclamation and water storage*, ed. by B. Knights & A.J. Phillips, 99–124. Farnborough, Saxon House.

Cowardin, L.M., Carter, V., Golet, F.C., & LaRoe, E.T. 1979. *Classification of wetlands and deepwater habitats of the United States. FWS/OBS-79/31.* Washington DC, US Department of the Interior – Fish and Wildlife Service.

Dalby, R. 1957. Problems of land reclamation. 5. Saltmarsh in the Wash. *Agricultural Review, London 2*: 31–37.

Davidson, N.C. 1990. The conservation of British North Sea estuaries. *Hydrobiologia 195*: 145–162.

Davidson, N.C., & Evans, P.R. 1986a. The role and potential of man-made and man-modified wetlands in the enhancement of the survival of overwintering shorebirds. *Col. Waterbirds 9*: 176–188.

Davidson, N.C., & Evans, P.R. 1986b. *Implications for nature conservation of the proposed Felixstowe Dock expansion.* Report to the Nature Conservancy Council.

Davidson, N.C., Laffoley, D.d'A., Doody, J.P., Way, L., Drake, C.M., Gordon, J., Key, R., Mitchell, R., Pienkowski, M.W., & Duff, K.L. 1991. *Nature conservation and estuaries in Great Britain.* Nature Conservancy Council, Peterborough.

Davidson, N.C., & Rothwell, P.I., *eds.* 1993. Disturbance to waterfowl on estuaries. *Wader Study Group Bulletin 69 (Special Issue).*

Davidson, N.C. & Stroud, D.A. In press. International coastal conservation: conserving coastal habitat networks on migratory waterfowl flyways. *Proc. 4th EUCC Congress, Marathon, Greece, April 1993.*

Doody, J.P., *ed.* 1984. Spartina anglica *in Great Britain.* Peterborough, Nature Conservancy Council. (Focus on nature conservation, No. 5.)

Doody, J.P. 1987. The impact of 'reclamation' on the natural environment of the Wash. *In*: *The Wash and its environment,* ed. by J.P. Doody & B. Barnett. Peterborough, Nature Conservancy Council. (Research & survey in nature conservation, No. 7.)

Doody, J.P., Johnston, C., & Smith, B., *eds.* 1993. *Directory of the North Sea coastal margin.* Peterborough, Joint Nature Conservation Committee.

Doody, J.P. 1995. Coastal change and conservation in Great Britain. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 81–89.

Doody, J.P. In preparation. Coastal habitat loss: an historical review of man's impact on the coastline of Great Britain.

Dugan, J.P., *ed.* 1993. *Wetlands in danger.* London, Mitchell Beazley.

Evans, P.R. 1981. Reclamation of intertidal land: some effects of Shelduck and wader populations in the Tees Estuary. *Verh. orn. Ges. Bayern (1978/79) 23*: 147–168.

Evans, P.R., Davidson, N.C., Piersma, T., & Pienkowski, M.W. 1991. Implications of habitat loss at migration staging posts for shorebird populations. *Acta XX Int. Orn. Congr.*: 2228–2235.

Evans, P.R., Herdson, D.M., Knights, P.J., & Pienkowski, M.W. 1981. Short-term effects of reclamation of part of Seal Sands, Teesmouth, on wintering waders and Shelduck. *Oecologia 41*: 183–256.

Evans, P.R., & Pienkowski, M.W. 1983. Implications for coastal engineering projects of studies, at the Tees estuary, on the effects of reclamation of intertidal land on shorebird populations. *Wat. Sci. Tech. 16*: 347–354.

Frayer, W.E., Monaham, T.J., Bowden, D.C., & Graybill, F.A. 1983. *Status and trends of wetlands and deep-water habitats in the coterminus United States, 1950s to 1970s.* Fort Collins, Colorado State University Department of Forest and Wood Services.

Fuller, R.M. 1987. The changing extent and conservation interest of lowland grasslands in England and Wales: a review of grassland survey 1930–84. *Biological Conservation 40*: 281–300.

Goss-Custard, J.D., & Durell, S.E.A. le V. 1990. Bird behaviour and environmental planning: approaches in the study of wader populations. *Ibis 132*: 273–289.

Goss-Custard, J.D., & Moser, M.E. 1988. Rates of change in the numbers of dunlin, *Calidris alpina*, wintering in British estuaries in relation to the spread of *Spartina anglica*. *Journal of Applied Ecology, 25*: 95–109.

Gosselink, J.G., & Baumann, R.H. 1980. Wetland inventories: wetland loss along the United States coast. *Z. Geomorph. N. F. Suppl. Bd. 34*: 173–187.

Gray, A.J. 1972. The ecology of Morecambe Bay. V. The saltmarshes of Morecambe Bay. *J. App. Ecol. 9*: 207–220.

Hötker, H., & Kölsch, G. 1993. *Die Vogelwelt des Beltringharder Kooges. Corax 15*: 1–145 (Special Issue).

Hötker, H. In press. Wadden Sea birds and embankments – can artificial wetlands compensate for the losses due to land-claims? *Proc. 8th scientific Wadden Sea conference, Esbjerg, 1993. Ophelia (Special Issue).*

Howe, M.A. 1987. Wetlands and waterbird conservation. *Am. Birds 41*: 204–209.

Lambeck, R.H.D., Sandee, A.J.J., & de Wolf, L. 1989. Long-term patterns in the wader usage of an intertidal flat in the Oosterschelde (SW Netherlands) and the impact of the closure of an adjacent estuary. *J. App. Ecol. 26*: 419–431.

Macey, M.A. 1974. Report 1d. Survey of semi-natural reclaimed marshes. *In*: *Aspects of the ecology of the coastal area of the Outer Thames Estuary and the impact of the proposed Maplin Airport.* Report to Department of the Environment. Natural Environment Research Council.

May, V.J. 1969. Reclamation and shoreline changes in Poole Harbour, Dorset. *Proc. Dorset Nat. Hist. & Arch. Soc. 90*: 141–154.

McLusky, D.S., Bryant, D.M., & Elliott, M. 1992. The impact of land-claim on the invertebrates, fish and birds of the Forth estuary. *Aquatic Conservation: Marine & Freshwater Ecosystems* 2: 211–222.

Moser, M.E. 1988. Limits to the numbers of grey plovers *Pluvialis squatarola* wintering on British estuaries: an analysis of long-term population trends. *J. App. Ecol. 25*: 473–486.

Natural Environment Research Council. 1975. *Estuaries research.* Natural Environment Research Council Publications series 'B', No. 9.

Pienkowski, M.W. & Pienkowski, A.E. 1983. WSG project on the movements of wader populations in western Europe. Eighth progress report. *Wader Study Group Bull.* 38: 13–22.

Pye, K. & French, P.W. 1992. Targets for coastal habitat recreation. Report to English Nature.

Ratcliffe, D.A. 1977. *A nature conservation review.* Cambridge, Cambridge University Press.

Stickney, W.M. 1908. Plan showing reclamation from the R. Humber to Holderness. *Appendix to evidence given to the Royal Commission on Coastal Erosion and Afforestation Second Report.*

Stroud, D.A., Mudge, G.P., & Pienkowski, M.W. 1990. *Protecting internationally important bird sites.* Peterborough, Nature Conservancy Council.

Thornton, D., & Kite, D.J. 1990. *Changes in the extent of the Thames Estuary grazing marshes.* London, Nature Conservancy Council.

Tiner, R.W., Jr. 1984. *Wetlands of the United States: current status and recent trends.* US Fish and Wildlife Service, National Wetlands Inventory.

Tubbs, C.R., 1984. Conserving the Solent. *Ecos 5*: 33–34.

Williams, G., & Hall, M. 1987. The loss of coastal grazing marshes in South and East England, with special reference to East Essex, England. *Biological Conservation 39*: 243–253.

Coastal change and conservation in Great Britain

J.P. Doody

Doody, J.P. 1995. Coastal change and conservation in Great Britain. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 81–89.

The conventional view of terrestrial nature conservation has as its cornerstone the protection of the most important sites representative of the remaining natural and semi-natural habitats and important species concentrations. The massive changes in land-use and habitat loss which have occurred historically mean the remaining areas are considered to be particularly precious, and nature conservation organisations in Great Britain have expended considerable time and money in an attempt to prevent the worst excesses of human destruction. Despite these efforts loss of habitat continues.

Habitats and sites within the coastal zone have similarly been affected. However, here coastal erosion in some areas exacerbated by rising sea-level, has caused further loss particularly where the landward transition is truncated by a sea wall. Opportunities for full successional development under these circumstances are therefore rare. In order to reverse this process, a more dynamic approach to nature conservation is suggested, which accepts change as being of positive value. This may require a reappraisal of the nature conservationists' protectionist philosophy. However, in so doing they will be challenging all those concerned with coastal management to review their own prejudices. This may be most apparent in relation to sea defence and coast protection where the maintenance of the existing line of defence, at all costs, may no longer be viable.

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Introduction

During the formative years of the conservation movement protection of species was an important consideration. The early Acts of Parliament were mainly concerned with bird protection. [The Sea Birds Protection Act, 1869, The Wild Birds Protection Act, 1872, and The Wildfowl Protection Act 1876.] Species protection remains an important component of the legislation and prominent among the Acts are the protection of Birds Act, 1954 (amended in 1967), and the Conservation of Wild Creatures and Wild Plants Act, 1975. In the early post-war period considerable interest was expressed by the leading ecologists of the day in the wider concepts of wildlife conservation. This is perhaps best described in A.G. Tansley's book *Our Heritage of Wild Nature*, published in 1945. The thesis of this book was that "the preservation of rural beauty" must involve the conservation of much of our native vegetation. At that time it was recognised that the planning of this work must "be balanced and harmonised with land utilisation and for agriculture and forestry". However, despite this statement of the wider needs of nature conservation, the National Parks and Access to the Countryside Act, 1949, made the establishment and management of nature reserves and the designation of Sites of Special Scientific Interest (SSSI) major components of the legislation.

The cornerstone of the nature conservation effort of both the Government Countryside Agencies and the voluntary conservation movement is today, as in the past, the protection of the best examples of particular habitats and species concentrations. Despite attempts to influence land-use management in the wider countryside, there have been few positive achievements. During the post-war period the needs of agricultural production and forestry were frequently cited as the reason for the drainage of a fen or bog in the lowlands, and afforestation in the uplands. With no legislative powers to prevent these activities, many of our precious wildlife areas were destroyed and in many areas survive only as small 'islands' in an intensive agricultural landscape. In the face of this it is perhaps not surprising that the conservation movement concentrated on protecting those places with special habitats and species.

This was achieved by the establishment of nature reserves, both statutory National Nature Reserves (NNRs) administered by the Nature Conservancy Council (NCC)* and by the voluntary conservation movement, including the National Trust (NT), the Royal Society for the Protection of Birds (RSPB) and local Wildlife Trusts, where nature conservation is the primary consideration in management. In addition, sites designated as Sites of Special Scientific Interest were given protection from developments requiring planning consent under the 1949 Act. More recently the Wildlife and Countryside Act, 1981, strengthened this

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* Responsibility for these areas passed to the Countryside Council for Wales, Scottish Natural Heritage and English Nature, which from 1 April 1991 replaced the Nature Conservancy Council.

designation by bringing owners and occupiers into the consultation process over a wide variety of 'damaging operations' including agricultural developments not covered by planning law. These legislative arrangements also apply to the coastal zone down to mean low water.

In considering the future of conservation in a changing coastal environment this paper will examine the successes and failures in protecting habitats and sites. It will review the role of change as a natural component of coastal systems and examine the way in which this is viewed by the conservationist, the developer and those concerned with sea defence and coast protection.

The historical perspective

Change is a natural phenomenon in the coastal zone, with the movement of barrier islands, sand dunes, tidal marshes and swamps, adjusting to the forces of the sea. The balance between erosion and accretion of the flat low-lying landscapes, around our estuaries, has proved a flexible response to changing sea-levels since the last ice-age. In Britain it would appear that these coastal areas were amongst the first to be colonised by early settlers. No doubt this was partly due to the more open nature of the landscape and the abundant source of food, both in the inshore waters and tidal lands.

Historically the coastline was also the zone where the first attempts to 'control' nature were made. The enclosure of the flat alluvial and tidal landscapes of East Anglia, Somerset levels and around many of our estuaries centuries ago, have helped create extensive areas of some of the most fertile land in Britain. Together with the building of ports and associated industrial and housing development, this has resulted in major change in many of our estuaries such that today they represent substantially smaller remnants of their former size (Davidson *et al.* 1991).

Whilst many sedimentary coastal areas were successfully 'tamed' by taking advantage of natural accretion, by contrast prevention of the erosion of soft cliffs was not. Along the Holderness Coast, for example, 29 towns and villages have fallen into the North Sea (Steers 1969). This loss of land can provide sediment for deposition elsewhere, leading to the creation of new land. Indeed this process may be partially responsible for the balance of accretion over erosion reported by the Director General of the Ordnance Survey in his evidence to the Royal Commission on Coastal Erosion (1911). He showed that between 1863 and 1896, whilst 4,692 ha of land had been lost, 3,544 ha had been gained.

However, the spectre of villages and individual houses falling into the sea, elicited a predictable response. Today many coastal cliffs are protected by a variety of structures, including revetments, groynes and concrete sea walls. Thus, in flat alluvial landscapes land has been created at the expense of the inter-tidal zone by the erection of embankments. At the same time, the landward movement of the sea on cliffed coasts has been prevented. Today, in south-east and south England these combined activities have resulted in some 50% of the coastline being artificially protected, resulting in much of its natural mobility being lost.

The coastal resource and protected status

Human activities are not always negative for nature conservation, and in many instances they have helped to create some important wildlife habitat. These include the grazing marshes of Essex and Kent, which were created by the enclosure of tidal land, notably saltmarshes, which was subsequently left unimproved. In addition, the long history of grazing on sand dunes by rabbits and domestic stock prevented the development of woodland, and helped to create the dune grasslands and heath so highly prized today. Similarly, saltmarshes are in many areas important for large numbers of over-wintering grazing ducks and geese, only because they are heavily stocked by sheep. Cliff tops and slopes are also influenced by grazing animals, which have helped extend the species-rich turf and heathland, beyond the confines of the maritime zone. It is perhaps only on the most inaccessible cliffs that the natural vegetation is more or less free from human interference.

This combination of modified habitats together with the remaining areas of relatively undisturbed land forms the basis for the recognition of a series of natural and semi-natural coastal habitats which are important for nature conservation. In the coastal zone these include: tidal flats and saltmarshes; sand dunes; shingle beaches and structures; sea cliffs (soft and hard rocks); grazing marsh; lagoons and estuaries.

The coastline of Britain is nearly 20,000 km long, including offshore islands and inland to the first artificial barrier upstream in estuaries. This figure and the data on the distribution and scale of the main coastal formations given in Table 1 are derived from existing survey and inspection of 1:50,000 OS maps. Saltmarsh figures are derived from Burd (1989).

There are some 764 Sites of Special Scientific Interest (including geological sites) on the coastline of Britain. Some habitats are well represented by this designation. For example, 80% of saltmarshes are included within Sites of Special Scientific Interest (Burd 1989). Comparable figures for sand dunes are not available,

Table 1 Length and area of the main coastal habitats in Britain

habitat	*length (km)*	*area (ha)*
Soft sedimentary shore	6,437	235,267
Shingle shore	3,634	4,219 *
Rocky shore	6,714	-
Saltmarsh	1,967	44,370
Sand dune	1,368	56,346
Cliff >20 m	2,708	-

*vegetated shingle structure

though there are approximately 120 SSSIs where they form a major component of the site.

Nature reserves protect a large number of the more important examples of coastal habitats and 15% of the total number of 234 National Nature Reserves owned or managed by the Government Countryside Agencies are coastal. These include part of all of nearly half the 42 nationally important sand dunes (Doody 1989a). Equally important are the areas protected by the RSPB, the National Trust and the Local Nature Conservation Trusts.

Achievements (protecting sites)

Conserving the coast by adopting the variety of protective measures described above has been a major plank of the nature conservationist's effort over the last 40 years or so. However, the number of designated sites in itself does not give a complete picture of the extent of protection nor of its effectiveness. In a few isolated areas these protective measures have reinforced each other to provide adequate safeguards. This is the case on the North Norfolk Coast, where most of the internationally important area of shingle, sand dune and saltmarsh is protected as a nature reserve (NCC, RSPB and NT). However, elsewhere the achievements are less obvious. A large number of the main estuaries of Britain are notified as SSSIs and in addition there are several international designations under the Ramsar Convention and as European Community Special Protection Areas (SPAs). However, despite these protective measures the current scale and rate of destruction is far worse than originally thought when the NCC's Estuaries Review was conceived (Davidson *et al.* 1991). Saltmarshes in particular are vulnerable because they develop at the upper levels of the tidal range and are thus the first habitats to be destroyed by land claim. It is also clear that the pace of development has not ameliorated and today there are proposals for ever more grandiose and potentially destructive schemes, such as tidal barrage developments. This situation has been highlighted by the RSPB in their publication *Turning the Tide* (RSPB 1990).

Recent developments elsewhere on the coast reveal an equally sorry story. For example, sand dunes have been used for a variety of purposes including golf links, an activity which may in the past have prevented more damaging development. However, intensively managed greens, tees and fairways represent a loss of habitat no less important than from other forms of destruction (Doody 1989a). Within recent years a new golf course has been established on a most sensitive part of the internationally important area of blown sand on the Durness limestone in north-west Sutherland, despite opposition from the Nature Conservancy Council. Proposals to develop two new golf courses on Machrihanish dunes in south-west Scotland would continue the process of destruction from this activity. Similar initiatives have been promoted elsewhere in Europe, in recent years, as the interest in golf has grown.

Morrich More, another internationally important site, has suffered partial destruction by the development of a site for the fabrication of bundles of oil pipelines. The initial contract, let by Shell UK, involves the destruction and disturbance of habitat along a corridor which crosses the most sensitive part of this 7,000 year old system. This will affect transitions between saltmarsh and sand dune which are, botanically, among the richest in Scotland (see Figure 1).

Developments are also difficult to oppose when major schemes are involved. The Channel Tunnel, for example, will cause unpredictable changes in the vicinity of the nationally important Folkestone Warren SSSI. An amenity barrage which will destroy the Cardiff Bay SSSI is claimed to be crucial to the economic revival of the docklands area of Cardiff City.

On a smaller scale many more very important coastal sites have been adversely affected in recent years, despite the protection of the SSSI legislation. These are often presented as relatively small incursions into large sites, such as car parks on sand dunes and sea cliffs. Individually they may be small; however, they represent a cumulative loss of habitat which may ultimately reach considerable proportions.

Achievements (managing nature reserves)

Nature Reserves may be owned by a conservation body, or at least have some form of agreement, the primary aim of which is the protection of wildlife. Direct destruction of habitat is, therefore, less frequent than with other protective measures. However, their conservation still requires adequate and appropriate management. Some of the problems which can arise will be highlighted by reference to two examples, both involving National Nature Reserves on sand dunes.

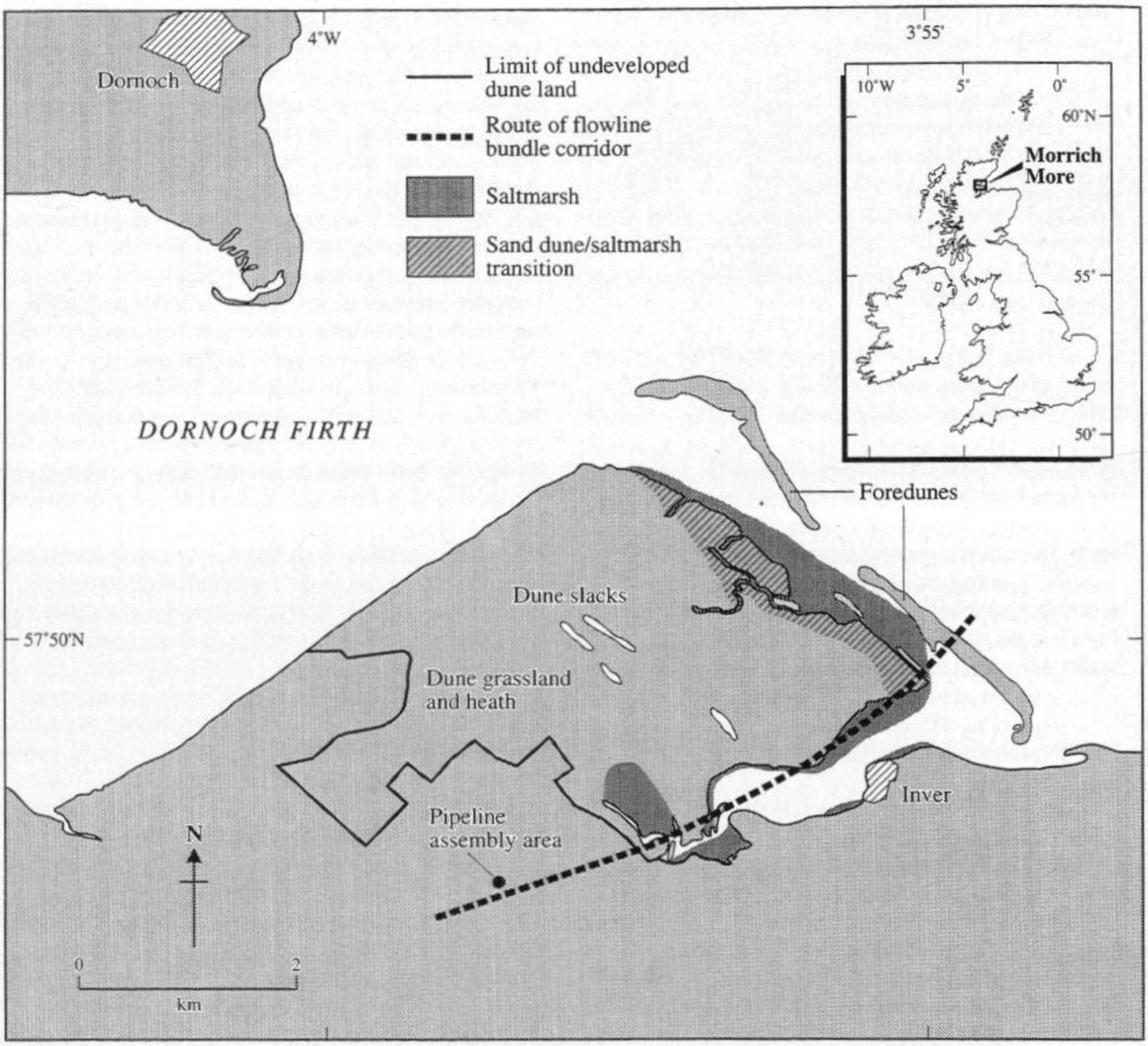

Figure 1 Morrich More: location of the corridor for the construction of bundles of pipes for the oil industry, in relation to saltmarsh and dune vegetation

The protection of sand dunes has been a concern for centuries and the human protectionist philosophy has continued to the present day, with a preoccupation for sand stabilisation in the face of 'threats' from recreational and other 'destructive' activities. However, on examination it appears that in some of the more important sites the conservation manager may have overlooked the potentially more serious problem of over-stabilisation and scrub development.

At **Braunton Burrows** in Somerset, for example, the major conservation problem today is one of loss of species-rich dune grassland as the site has become invaded with scrub. This was brought about by the combined affects of a reduction in rabbit grazing following myxamatosis, and planting *Ammophila arenaria* and *Hippophae rhamnoides*, to prevent sand movement, and conserve the dune system in the face of erosion in the early 1950s. A complete reappraisal of the management requirements have been carried out and the reintroduction of grazing together with some instability is now seen as an important prerequisite for the conservation of the site (Doody 1989b).

At **Tentsmuir**, a sand dune system on the north-east coast of Scotland, the management aim of the NNR centred on the natural development of this, one of the few actively prograding systems in Britain. Implicit in this strategy was a non-intervention policy within the boundary of the reserve. However, the management plan overlooked the fact that a large proportion of the system had been planted with an artificial forest (see Figure 2).

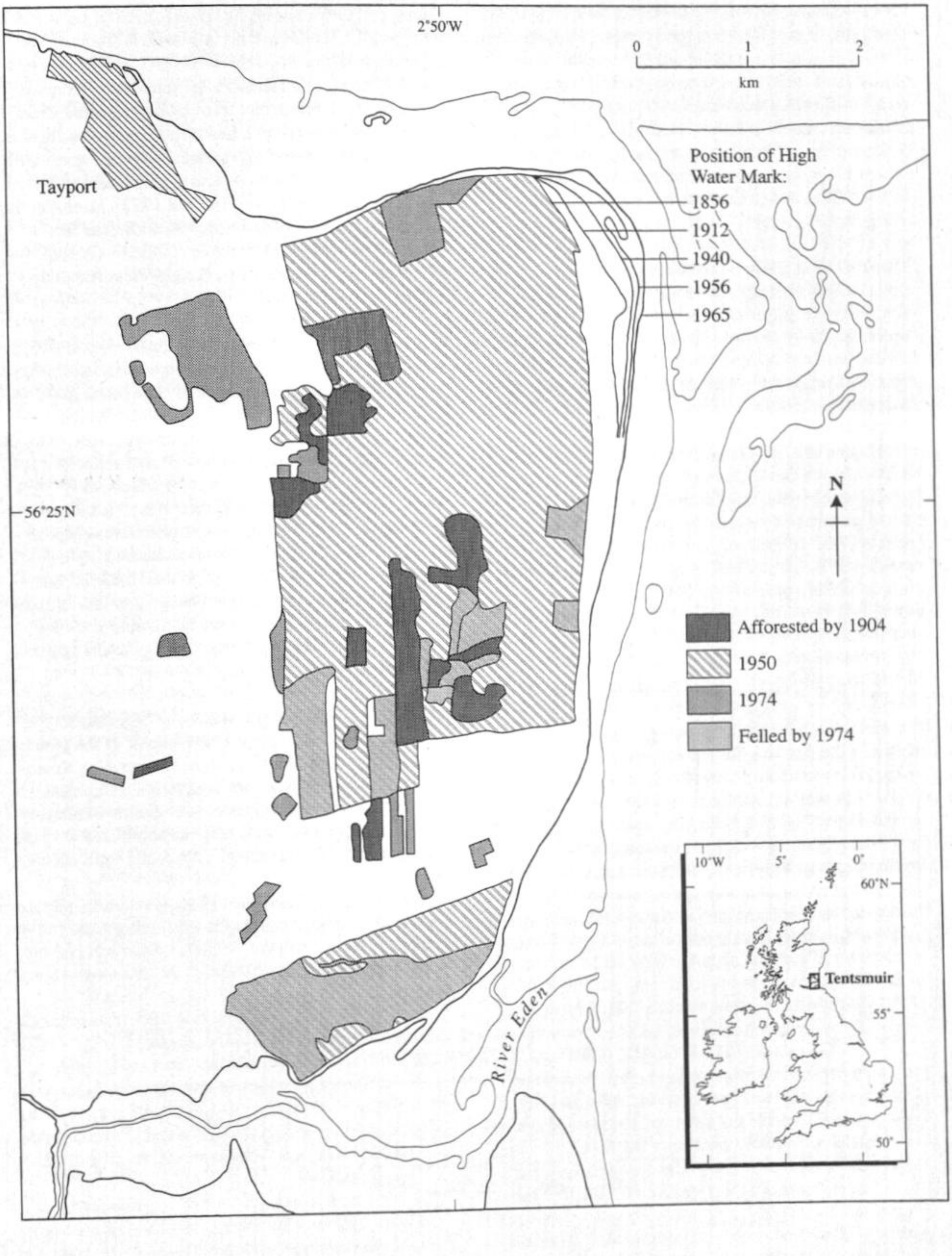

Figure 2 Tentsmuir, Fife: afforestation and coastal change

The combined effects of the shelter and lowering of the water table caused by the growing trees, and a reduction in the high rabbit population, quickly resulted in the expansion of scrub and self-sown pine. This affected some of the richest dune slacks on the reserve, which developed around the 1940s, on the prograding, Tentsmuir Point. Today goats are being used in an attempt to reverse the scrub encroachment (Bullock & Kinnear 1988) and pine removal is part of the management strategy.

These two cases illustrate the difficulty in devising and implementing a management strategy. Firstly, it is essential to recognise than an adverse change is occurring. For those sites showing retrogressive succession this may be obvious, though as we have seen above, this is not always the case. Secondly, even if the management problem is correctly diagnosed it may not be easy to initiate some forms of management, such as encouraging sand movement, which have traditionally been viewed as damaging. Conservation managers may be hampered by their own innate prejudices and a reluctance to take a more active part in initiating change because of the uncertainty of the outcome. In order to consider the issues further the ecological basis for the decision making process will be discussed in more detail.

Managing change – the ecological basis

The ecological basis for the development of coastal habitats is the first stop for the conservation manager. In looking at the textbooks he will invariably find a diagram showing successional processes on saltmarshes or sand dunes. Figure 3 shows a typical example (after Burd 1989). This view suggests that successional development can be inferred from the spatial relationships of the vegetation at any one time. However, a more critical look at the natural world shows this to be too simple a picture.

Dunes certainly develop by the accretion of sand, aided by specialist plants (notably *Ammophila arenaria* and *Elymus farctus* in Britain). However, once the main body of the dune is formed other processes come into play and the development from mobile foredunes and yellow dune to grassland, heath, scrub and woodland is not a straightforward progression. Blowouts occur with or without the intervention of man and can be the precursors of dune slacks (Ranwell 1972). Similarly the reprofiling of dune ridges under the influence of changing wind patterns bring an infinitely variable topography, the origins of which may be difficult or impossible to unravel.

Saltmarsh represents an equally complex system, which is often depicted as a straight forward succession, with accretion resulting in ever more diverse forms and transitions to terrestrial vegetation being the most complex. However, as with dunes this apparently simple picture hides a sometimes complex relationship (Gray 1992). Sequences of erosion may be followed by regrowth as estuary channels change their course. A series of steps can form as new saltmarsh develops to seaward of the eroding cliff as has occurred on the Solway in south-west Scotland (Marshall 1962). Salt pans can introduce yet another change into the complex mosaic and deposits of seaweed on the tide-line may smother the surface vegetation creating further spatial variation as the strandline deposits rot.

Sea cliffs composed of soft rocks also rely, in many instances, on mobility as an essential part of the process of retaining their biological nature conservation interest. A degree of instability is essential to the maintenance of open, species-rich communities at a number of sites. Slippages, especially where they are associated with the freshwater run-off, may also be important for a wide variety of invertebrates. Even the Axemouth to Lyme Regis undercliffs, which have extensive woodland, rely on occasional massive landslips to rejuvenate their habitats.

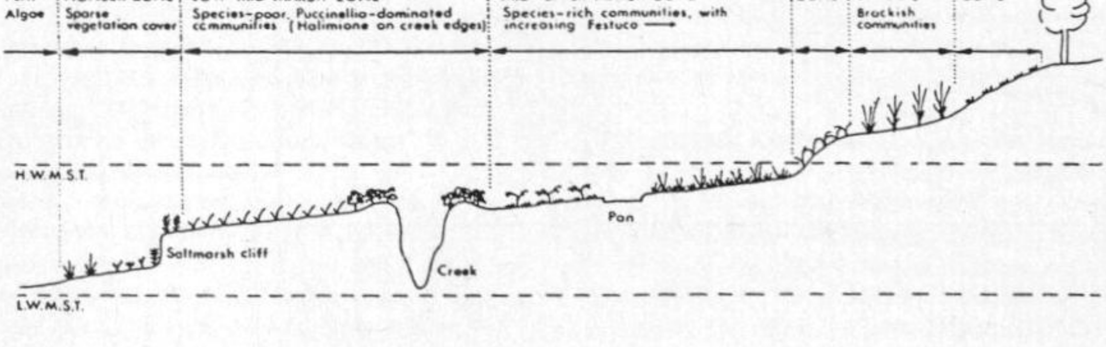

Figure 3 Typical successional diagram for saltmarsh, after Burd (1989)

Thus it can be seen that far from being static or progressing in an orderly fashion from one successional stage to another, coastal habitat development is much more complex and change is an integral part of that complexity.

Discussion

The nature conservationist may readily embrace concepts of change, as part of the normal course of habitat development, particularly in the early stages of succession. The prevention of change, in the sense of arresting succession in older more stable habitats, may be acceptable where change is seen to be detrimental to the nature conservation interest of a particular site. It has been less easy to consider change as a positive force in conservation management.

Typically, erosion of sand dunes will elicit a desire to 'protect' them and considerable energy will be expended on building sand fences and controlling people. Continuous disturbance around car parks and access points can result in the destruction of vegetation. However, if the system is given time to heal and the damaging vector removed, vegetation can become re-established without further interference. Some forms of physical damage can cause a change which becomes a permanent feature, such as 'footprints' on mature shingle. However, these should be separated from the more natural forms of change which may be an essential component of the proper development of the system.

Not only have human activities helped to stabilise mobile systems, but also there appears to be a reduction in the availability of sediment such that many habitats are fossilised. Thus, very few sand dunes or shingle structures in Britain are actively prograding today and as a consequence the earlier stages of development are absent. However, if we take our cue from nature the management options may be much more diverse than the text books or the collective experience suggest.

Thus, conservation managers may consider a more adventurous approach involving positive management for change. However, their own protectionist philosophy may make the more radical solution difficult to reconcile, especially if their actions are constrained by management objectives based on a more traditional understanding of the dynamics of coastal systems.

If conservationists' philosophy is based on the protection of sites and a restricted view of dynamic management, what of the role of engineers in coastal management? As we have seen, throughout history, by dint of his ingenuity and by virtue of ever improving techniques of sea defence, they have been successful in creating vast areas of land which have been 'won from the sea.' In the same way coastal cliffs, which formerly fell to the erosive forces acting on them, including wave attack, have progressively been protected and continue to be so. Having established this line of defence it is perhaps understandable that engineers should wish to see it retained.

Historically, the position of sea-level has moved relative to the land depending on the interplay of isostatic and eustatic changes. In the Wash (Shennan 1987) and the Severn (Allen 1992) for example, this resulted in periods of landward or seaward movement of the shoreline depending on the balance of sedimentary over erosive forces. Today the ability of the coastline to provide a flexible response to changing patterns of sea-level rise and storms is prevented in many areas by the creation of a straight inflexible barrier. In a zone which is recognised as being subject to the forces of nature, it could be argued that man has devoted too much effort to his 'battle with the sea'. In so doing he has squeezed the coastline into an ever narrowing zone.

Given this scenario it is right to consider whether any lessons can be learned from the natural world which might allow a more enlightened approach to the management of the coast. If it can be shown that the conservationist and the engineer have more in common in developing strategies for managing the coastline than previously realised, then a powerful alliance could be developed to effect positive change in the coastal zone.

Conclusion – the lessons

It is the thesis of this paper that a different approach to coastal conservation is required if the current trend in habitat and site destruction is to be reversed. In developing their own strategies for managing individual sites nature conservation managers may be ready to accept change as part of the natural process. However, in the face of the continuing destruction of precious wildlife habitat, they will continue to see protection through wildlife legislation and the establishment of nature reserves as a prime objective.

However, even if this approach is more successful than in the past, what are the prospects if the current predictions of sea-level rise are correct? In the south-east of England the combined effects of isostatic adjustments and the current relative rate of rise of sea-level, suggest that further loss of habitat can be expected, as the shore becomes steeper and narrower (Boorman, this volume). In Essex and North Kent where there are substantial lengths of artificial sea walls, saltmarshes are eroding (Burd 1992). The land behind the sea wall was formerly part of the tidal system and it could be argued that the sea is attempting to 'reclaim' some of the low-lying land.

Attempts should continue to be made to prevent damaging development on designated sites and by the establishment of nature reserves. However a more positive approach must be sought if we are to protect coastal habitats and to reverse some of the losses which have already occurred. Tidal motion and the movement of sand and shingle are important precursors to the development of beaches, sand dunes and shingle structures. In more sheltered conditions saltmarshes grow, and in each case the role of specialist plants is a major factor in accommodating change. The instability of cliffs and the importance of eroded material to the growth of coastal structures is also accepted.

Increasingly recognition is being given to more natural forms of sea defence, the so-called 'softer option'. For example, saltmarshes form a natural wave break as they absorb the energy of the incoming tide. Where their loss, through erosion, has exposed the toe of the sea bank, attempts have been made to recreate them. Similarly the fact that a beach forms the most effective form of coast protection is not lost on the engineer and beach feeding is now a well-established practice.

All these activities, including the 'softer' engineering option, still revolve around efforts largely designed to reinforce current lines of defence. Thus even the most enlightened attempts at recreating natural forms of protection are equated with a static view of where the coastline should be.

It appears increasingly likely, however, that the cost of maintaining the current line of defence may be prohibitive, especially in those areas where reduced land values make the cost benefit analysis more difficult to achieve. In addition, given the assessment of current sea-level rise of between 1 and 2 mm annually, and the predictions based on the impact of global warming (Houghton *et al.* 1990), it is becoming clear that if conservationists continue to adopt a 'protect at all costs' philosophy then in the long term they will be presiding over an ever diminishing resource. This will be particularly acute in areas where the landward limit is fixed by a sea wall and isostatic change adds to the rate of relative sea-level rise, as occurs in the south and south-east of England (Shennan 1989). Under these circumstances the conservationist and the engineer may find something in common if they are both willing to learn the lessons of the natural world. This will require recognition that change brings flexibility which may provide a more lasting response to the threat of sea-level rise. It will also require engineers to accept that the current line of sea defence is not sacrosanct and that retreat to a more defensible line may, in some circumstances, be the best long term option. Equally conservationists may be asked to give up some of the areas they consider important.

Difficult choices will have to be made, which may include the abandonment of some areas, both in the form of low-lying farmland and cliffs with housing and other developments on them. It should be possible to devise strategies which put resources into those areas where sea defence and coast protection requirements are paramount, provide more opportunities for 'soft' engineering options and increase the size of the zone in which the natural and more flexible sea defences can operate. In order to achieve this a more complete understanding of the mechanisms of coastal processes will be required. It will also be essential to set a framework in which decisions can be taken. This will require management to be considered in the context of a wider coastal zone strategy than is possible under current legislation.

References

Allen, J.R.L. 1992. Tidally influenced marshes in the Severn Estuary. *In*: *Saltmarshes: Morphodynamic Conservation and Engineering Significance* ed. by J.R.L. Allen & K. Pye. Cambridge, Cambridge University Press.

Boorman, L.A. 1995. Sea-level rise and the future of the British coast. *In*: *Coastal Zone Topics: Process, Ecology & Management 1*: 10–13.

Bullock, D.J., & Kinnear, P.K. 1988. The use of goats to control birch in dune systems: an experimental study. *Aspects of Appl. Biol. 16*: 163–168.

Burd, F. 1989. The saltmarsh survey of Great Britain – an inventory of British Saltmarshes. Peterborough, Nature Conservancy Council. (Research & survey in nature conservation, No. 17.)

Burd, F. 1992. Erosion and vegetation change on the saltmarshes of Essex and north Kent between 1973 and 1988. Peterborough, Nature Conservancy Council. (Research & survey in nature conservation, No. 42.)

Davidson, N.C., Laffoley, D.d'A., Doody, J.P., Way, L., Gordon, J., Key, R., Mitchell, R., Pienkowski, M.W., & Duff, K.L. 1991. *Nature conservation and estuaries in Great Britain.* Nature Conservancy Council, Peterborough.

Doody, P. 1989a. Conservation and development of coastal dunes in Great Britain. *In*: *Perspectives in Coastal Dune Management,* ed. by F. van Meulen, P.D. Jungerius & J.H. Visser. The Hague.

Doody, J.P. 1989b. Management for nature conservation. *In*: *Proceedings of the Royal Society of Edinburgh,* ed. by C.H. Gimmingham, W. Ritchie, B.B. Willetts & A.J. Willis, 96B, 247–265.

Gray, A.J. 1992. Saltmarsh plant ecology. *In*: *Saltmarshes, Morphodynamic Conservation and Engineering Significance,* ed. by J.R.L. Allen & K. Pye. Cambridge, Cambridge University Press.

Houghton, J.T., Jenkins, G.J., & Ephraums, J.J., *eds.* 1990. *Climate Change: the IPCC Scientific Assessment.* Cambridge, Cambridge University Press.

Marshall, J.R. 1962. The physiographic development of Caerlaverock merse. *Trans. Dumfries and Galloway Nat. Hist. and Antiq. Soc., 39*: 102.

Ranwell, D.S. 1972. *Ecology of Saltmarshes and Sand Dunes.* London, Chapman and Hall.

Ratcliffe, D.A., *ed.* 1977. *A Nature Conservation Review. The selection of biological sites of national important for nature conservation in Great Britain.* Cambridge, Cambridge University Press.

Royal Commission on Coast Erosion and Afforestation. 1911. *Coast Erosion – the reclamation of tidal lands, and afforestation in the United Kingdom.* Third (and final) report, Vol. 3, Part 1. London, HMSO.

Royal Society for the Protection of Birds. 1990. *Turning the Tide.* Sandy, RSPB.

Shennan, I. 1987. Impacts on the Wash of sea-level rise. *In*: *The Wash and its environment* ed. by P. Doody & B. Barnett. Peterborough, Nature Conservancy Council. (Research and survey in nature conservation, No. 7.)

Shennan, I. 1989. Holocene crustal movements and sea-level changes in Great Britain. *J. Quaut. Sc., 4*: 77–89.

Steers, J.A. 1969. *The Coastline of England and Wales.* 2nd edition. Cambridge, Cambridge University Press.

Tansley, A.G. 1945. *Our Heritage of Wild Nature, a plea for organised nature conservation.* Cambridge, Cambridge University Press.

Coastal changes: causes and effects

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Introduction

This contribution is offered as a discussion paper against which the scientific presentations in the volume might be set. It is hoped that it will bring together some of the thoughts expressed during the symposium that gave rise to the volume, although in a different context. It is very much a personal view based on attempts to simplify the diversity of aspects and judgements that are involved in the subject.

Coastal changes and their causes

Coastlines are dynamic features and, as such, changes are perfectly natural and continuous. The causes of the changes, their nature and their scale are diverse. Changes can be caused by the expansion and retreat of glaciers as well as the possibilities resulting from changes in the earth's atmosphere caused by man's activities (Shennan, this volume). These are changes on a large scale whereas others described in this volume are much more localised. It might, therefore, be useful to classify the changes on the basis of their cause and scale as follows:

Natural changes

Small scale	(Cliff or beach erosion)
Medium scale	(Coastal erosion)
Large scale	(Sea-level changes; large geographical and temporal scales)

Anthropogenic changes

Small scale	Seawalls, infilling – depending on size
Medium scale	barrages, ports
Large scale	(Sea-level changes due to 'greenhouse gases')

The use of the relative terms for scales is clearly very subjective, but they are meant to be practical expedients for discussion purposes and are not rigid definitions. Each of these categories of coastal changes will have a story, both in terms of their causes and consequences, and examples of all of them are included in this volume.

Coastal changes and their effects

Humans do not usually cause environmental damage deliberately. Their intention is to produce improvements to some aspect of their own lives but some consequences are deemed to be harmful to another aspect or to damage the natural environment. The difference between natural and man-induced changes are often those of scale and speed of change. Judgements of the changes are also made by man and can be at different levels as well as degrees.

Changes can be harmful or beneficial to:
- an individual
- a population
- a community
- an ecosystem.

The concept of harm is easily applied to the individual and possibly to the population as the result may be annihilation or reduction of vigour or numbers. At the community and ecosystem levels changes are usually of a different nature, as it is rare for life to be lost completely; instead one set of organisms is replaced by another. The one group has gained at the expense of the other and the judgement on whether or not the change is deleterious depends on the value afforded to the different groups by man. This judgmental factor is clearly very important in discussions of coastal change and features in several of the papers given in this volume, e.g. Doody, who describes changes in managerial practices with changes in attitudes and understanding.

Relationships between coastal changes and their effects

The causes and effects discussed above can be linked and represented schematically (Figure 1).

The effects of changes on the coastal environments can be either direct or indirect and these in turn affect the living systems so that the biological effects are mostly

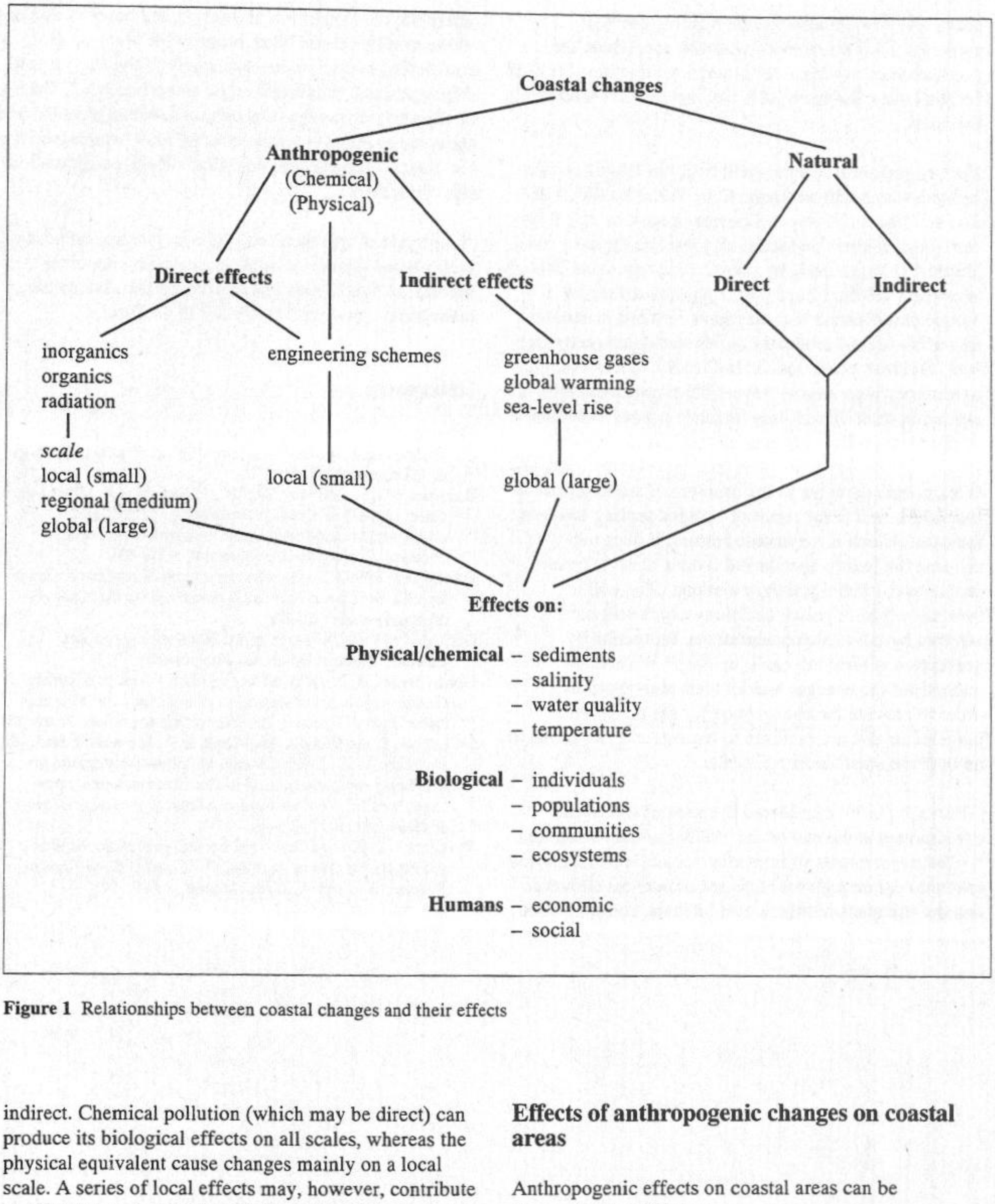

Figure 1 Relationships between coastal changes and their effects

indirect. Chemical pollution (which may be direct) can produce its biological effects on all scales, whereas the physical equivalent cause changes mainly on a local scale. A series of local effects may, however, contribute to effects at a considerable distance from the cause. An example is the loss of the European winter feeding grounds of birds that breed in the Arctic regions.

The relationships between coastal changes and their effects are complex because they involve the interaction of physical, chemical and biological aspects as well as crossing the media boundaries on various geographical and temporal scales.

Effects of anthropogenic changes on coastal areas

Anthropogenic effects on coastal areas can be summarised as follows:

- Changes to sea/land boundary – sea-level changes, loss of habitat
- Changes to water quality
- Changes to biological communities.

Sea-level changes will affect local water quality in that short sea outfalls will become effectively longer ones but this, of course, will be on a relatively long timescale. They would also reduce flushing times in estuaries. Rising sea-levels will also necessitate the production of more coastal protection structures (or head for the hills – Boorman, this volume), further modifying the coast.

Such engineering schemes will result in loss of certain habitats which will be irreversible. Water quality will also be affected by impoundments (Leeuw *et al.* , Seys *et al.* this volume). In fact, it may well be the case that physical changes made by man in land claim and coastal protection schemes have produced such extensive habitat modification that they have resulted in serious loss of biological production to the estuarine ecosystem (see Davidson *et al.,* and Goss-Custard, in this volume). In estuaries where such irreversible physical changes are extensive, their effects may be more serious than those of chemical pollution (much of which is reversible).

What people perceive as the problem of sea-level rise is inevitable, as it is the result of humans settling too near the coast, which is a dynamic feature. It does not threaten the human species but it does mean serious problems for individuals/populations. This will inevitably lead to policy decisions which will be affected by political considerations. Protection by prevention is ideal but can it be done? We need to understand the changes and all their repercussions in order to provide the background for the many value judgements that are required to manage our environment on both the small and large scales.

GESAMP (1990) considered the state of the marine environment at the end of the 1980s and concluded that "... the major causes of immediate concern in the marine environment on a global basis are coastal development and the attendant destruction of habitats, eutrophication, microbial contamination of seafood and beaches, fouling of the seas by plastic litter, progressive build-up of chlorinated hydrocarbons, especially in the tropics and subtropics, and accumulation of tar on beaches." This neatly summarises the relationship between physical and chemical changes brought about by man, which produce the coastal changes and biological effects considered in this discussion.

An integrated approach to these ecosystems, including their natural features as well as the socio-economic aspects, is clearly required to produce the sustainable, managerial compromises needed in the future.

References

Boorman, L.A. 1995. Sea-level rise and the future of the British coast. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 10–13.

Davidson, C.D., Laffoley, D.d'A., & Doody, J.P. 1995. Land-claim on British estuaries: changing patterns and conservation implications. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 68–80.

Doody, J.P. 1995. Coastal change and conservation in Great Britain. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 81–89.

GESAMP. 1990. *The state of the marine environment.* London, Blackwell Science Publications.

Goss-Custard, J.D. 1995. Effect of habitat loss and habitat change on estuarine shorebird populations. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 61–67.

De Leeuw, J., de Munck, W., Apon, L.P., Herman, P.M.J., & Beeftink, W.G. 1995. Change of saltmarsh vegetation following the construction of the Oosterschelde storm surge barrier. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 35–40.

Shennan, I. 1995. Sea-level and coastal evolution: holocene analogues for future changes. *In*: *Coastal Zone Topics: Process, Ecology & Management, 1*: 1–9.