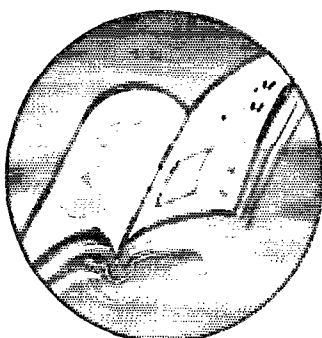
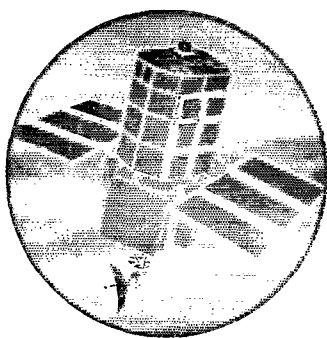


Impact of Nutrients in Estuaries - Phase 2



Research and Development

Project Record
P2/i639/1



All pulps used in production of this paper is sourced from sustainable managed forests and are elemental chlorine free and wood free

Impact of Nutrients in Estuaries - Phase 2

Project Record P2/i639/1

C R Scott, K L Hemingway, M Elliot, V N de Honge, J S Pentthick, S Malcolm
and M Wilkinson

Research Contractor:
CCRU & CEFAS

Further copies of this report are available from:
Environment Agency R&D Dissemination Centre, c/o
WRc, Frankland Road, Swindon, Wilts SN5 8YF



tel: 01793-865000 fax: 01793-514562 e-mail: publications@wrcplc.co.uk

Publishing Organisation:

Environment Agency
Rio House
Waterside Drive
Aztec West
Almondsbury
Bristol BS32 4UD

Tel: 01454 624400

Fax: 01454 624409

ISBN:1 85705 153 X

This document was produced with the support of the European Life-Nature Programme.

© Environment Agency 1999

All rights reserved. No part of this document may be reproduced, stored in a retrieval system, or transmitted, in any form or by any means, electronic, mechanical, photocopying, recording or otherwise without the prior permission of the Environment Agency.

The views expressed in this document are not necessarily those of the Environment Agency. Its officers, servant or agents accept no liability whatsoever for any loss or damage arising from the interpretation or use of the information, or reliance upon views contained herein.

Dissemination status

Internal: Released to Regions
External: Released to the Public Domain

Statement of use

This report will be of use to Environment Agency, and other protection agency, staff involved, either strategically or operationally, in the monitoring and management of estuaries.

Research contractor

This document was produced under R&D Project i639 by:

Cambridge Coastal Research Unit (CCRU)
University of Cambridge
Department of Geography
Downing Place
Cambridge
CB2 3EN

The Centre for Environment, Fisheries &
Aquaculture Science (CEFAS)
Pakefield Road
Lowestoft
Suffolk
NR33 0HT

Tel: 01223 333399

Tel: 01502 562244

Fax: 01223 333392

Fax: 01502 513865

Environment Agency Project Leader

The Environment Agency's Project Leader for R&D Project i639 was:
Robin Chatterjee, Environment Agency, Anglian Region

CONTENTS

	Page
LIST OF FIGURES	iii
LIST OF TABLES	v
GLOSSARY	vii
EXECUTIVE SUMMARY	xi
KEYWORDS	xiii
1 INTRODUCTION	1
1.1 Project Background	1
1.2 Project Consultants	3
1.3 Project Approach and Outline	4
2 EFFECT OF PHYSICO-CHEMICAL FACTORS	5
2.1 Introduction – The Nature of Estuaries	6
2.2 Nutrient Inputs	8
2.3 Nutrient Retention	14
2.4 Nutrient Transformations	16
2.5 Biological Responses	23
2.6 Conclusions	30
3 BIOLOGICAL IMPACT OF NUTRIENTS	33
3.1 Introduction – Characteristics of Nutrient Enrichment	34
3.2 Autotrophic Groups and Bacteria	35
3.3 Benthic Communities	52
3.4 Higher Trophic Levels and Food Webs	57
3.5 Recovery from Enrichment Impacts	65
3.6 Summary and Conclusions	68
4 AVAILABLE TOOLS FOR MONITORING & MANAGEMENT	71
4.1 Data Adequacy	72
4.2 Monitoring Tools	76
4.3 Approaches to Management of Estuaries	88
4.4 Hindcasting Nutrient Status	99
4.5 Summary and Conclusions	107
5 RECOMMENDED MONITORING & MANAGEMENT APPROACH	111
5.1 Proposed Estuary Classification Scheme	112
5.2 Proposed Monitoring and Management Protocol	119
5.3 Further Studies	125
5.4 Summary and Conclusions	129
6 REFERENCES	131

APPENDICES

Appendix 1: 'Impact of Nutrients in Estuaries – Phase 2' Objectives	149
Appendix 2: Researchers and Organisations contacted during the Nutrients in Estuaries review.	151
Appendix 3: Estuary definitions and characteristics.	155
Appendix 4: Biological features of estuaries.	159
Appendix 5: Examples of species with varying sensitivities to hypoxia (Diaz and Rosenberg 1995).	165
Appendix 6: Review of studies which have compared nutrient impacts in different estuary systems.	167
Appendix 7: Summary details of the Environment Agency's proposed GQA Scheme for Benthic Communities	171
Appendix 8: Details of the Comprehensive Studies developed by the Comprehensive Studies Task Team (CSTT 1993).	173
Appendix 9: DoE criteria for defining eutrophication in estuarine and coastal waters.	175
Appendix 10: Draft monitoring guidelines presented by the UK to the OSPAR Conventions for the prevention of marine pollution ad hoc working group on monitoring (1995).	177
Appendix 11: Tabulated framework for the assessment of biological information to be carried out during Stage 2 of the monitoring protocol.	179
Appendix 12: Tabulated framework for the assessment of physical information to be carried out during Stage 3 of the monitoring protocol.	180
Appendix 13: Tabulated framework for the assessment of nutrient input information to be carried out during Stage 3 of the monitoring protocol.	181
Appendix 14: Figures	183

LIST OF FIGURES

Figure 1:	Nutrient and Sediment Pollution entering Chesapeake Bay from areas of the catchment with from Different Land-uses (Adapted from Magnien 1995).	183
Figure 2:	A schematic description of the nitrogen cycle (Libes 1992).	184
Figure 3:	A schematic description of the biologically processes of the marine phosphorus cycle (Tait 1981).	185
Figure 4:	Conceptual model of the spatial changes along the salinity gradient of deep slow flushing estuaries (Fisher <i>et al.</i> , 1988).	186
Figure 5:	A conceptual model showing the factors which influence phytoplankton production (from Boynton <i>et al.</i> , 1982).	187
Figure 6:	A conceptual model showing the factors which influence phytoplankton production (from Wilkinson unpubl.).	188
Figure 7:	Changes in abundance of macroalgae in Laholm Bay (Kattegat) attributed to nutrient enrichment (Baden <i>et al.</i> , 1990).	189
Figure 8:	Schematic description of the self-perpetuating process by which nutrient enrichment results in enhanced algal growth (adapted from Duarte 1995).	190
Figure 9:	Schematic description of the changes to primary producers and related parameters during successive phases of eutrophication (Schramm 1996).	191
Figure 10:	Schematic description of the self-perpetuating processes by which an uncoupling of nitrification and denitrification drives further nutrient enrichment (Harding <i>et al.</i> , 1992).	192
Figure 11:	Influence of changes in the severity and periodicity of hypoxic events on benthic fauna (Diaz and Rosenberg 1995).	193
Figure 12:	Comparative rates of phosphate flux from five different intertidal benthic communities (Asmus <i>et al.</i> , 1995).	194
Figure 13:	An estuarine food web for a typical northern hemisphere estuary (McLusky, 1989)	195
Figure 14:	Energy and Pollutant Transport in Estuarine and Coastal Systems (Elliott unpubl.).	196
Figure 15:	Description of the relationship between nutrient input and primary production for a range of aquatic systems Nixon (1992).	197
Figure 16:	Schematic description of the relationship between nutrient availability, phytoplankton sizes and planktonic food webs (Legendre and Rassoulzadegan 1995).	198
Figure 17:	An overview of the potential effects of eutrophication (NOAA 1993).	199
Figure 18:	Conceptual model used to develop nutrient budgets for Chesapeake Bay (Boynton <i>et al.</i> , 1995).	200
Figure 19:	A biological model of eutrophication in Vilaine bay, France (Chapelle <i>et al.</i> , 1994).	201

Figure 20:	Summary of existing conditions in estuaries on the USA mid-Atlantic coast based on values of eutrophication indicators that suggest problem conditions expressed as cumulative percentage surface area for each salinity band for all estuaries combined (NOAA 1995).	202
Figure 21:	Summary of existing conditions in estuaries on the USA mid-Atlantic coast based on values of eutrophication indicators in the mixing zone (identified as the most heavily impacted region) of estuaries (NOAA 1995).	203
Figure 22:	Summary of existing conditions in estuaries on the USA mid-Atlantic coast based on the areal coverage of eutrophication indicators showing problematic conditions in each estuary and in each salinity band (NOAA 1995).	204
Figure 23:	Characteristics of the four estuary types identified by Perdon (1995) showing the: water depth, presence of stratified layer and primary productivity as a function of nutrient input.	205
Figure 24:	The four estuary classes identified by Perdon (1995) described in a single plot which separates stratified, non-stratified, slow flushing and fast flushing estuaries on the basis of primary productivity levels for given nutrient inputs.	206
Figure 25:	NOAA (1993) classification approach using measure of the capacity of estuaries for retaining nutrients (Dissolved Concentration Potential) plotted against nutrient loading.	207
Figure 26:	Graphical presentation of the estuarine nutrient classification scheme developed by Gunby <i>et al.</i> , (1995).	208
Figure 27:	Procedure for modelling nutrient export from catchments (from Johnes 1994).	209
Figure 28:	Sedimentary profiles from Chesapeake Bay showing characteristics of diatom populations as stratigraphic indicators of environmental change in the estuary over time from Cooper and Brush (1993).	210
Figure 29 (a and b):	Proposed approach for classifying estuaries based on: flushing time, light extinction and nutrient input using histogram plots (a = individual factors plotted, b = factors plotted cumulatively).	211
Figure 30:	Proposed approach for classifying estuaries based on: flushing time, light extinction and nutrient input using an XYZ scatter plot.	212
Figure 31:	Proposed protocol for driving the monitoring and management of nutrient enrichment in estuaries in the UK.	213
Figure 32:	A possible approach for determining the response characteristics in estuaries and tailoring monitoring programmes to take account of site-specific responses to nutrient input.	214
Figure 33:	Proposed protocol for carrying out Phase 3 of the 'Impact of Nutrients in Estuaries' Project.	215
Figure 34:	Analysis of harmonised monitoring Data (1995 and 1996) for 96 UK estuaries (Nedwell pers. com.)	216

LIST OF TABLES

Table 1:	List of the factors governing rates of nutrient export from catchments (Johnes 1994).	10
Table 2:	The parameters collected by Perdon (1995) to compare 47 estuaries in western Europe on the basis of their susceptibility to eutrophication.	74
Table 3:	Indicators and response ranges for describing the scale of eutrophication in estuarine systems and information collected to determine the existing conditions and trends in these indicators. NOAA Draft Report (1995).	89
Table 4:	Nitrogen loads to five estuarine systems with biological responses that are either observed, or predicted from comparisons against critical loading values (Lord and Hillman 1995)	94
Table 5:	Proposed WQO class boundaries for nutrients estuaries from Rees <i>et al.</i> , (1994) and Gunby <i>et al.</i> , (1995).	95
Table 6:	Proposed WQO class boundaries for DO and ammonia in estuaries from Nixon <i>et al.</i> , (1995b).	95
Table 7:	Tabulated framework for classifying estuaries on the basis of their susceptibility to nutrient loading based on eight key factors	115
Table 8:	Hypothetical application of tabulated classification scheme showing high risk and low risk estuaries.	116
Table 9:	Proposed EcoQOs to describe the 'symptoms' of eutrophication in estuaries	121

GLOSSARY

- Algae:** Large group of autotrophic organisms that range in size from microscopic to several metres long.
- Allochthonous:** Material imported into a system *e.g.* organic matter.
- Ammonification:** The formation of ammonia from decomposing nitrogenous organic compounds.
- Anoxia:** Total lack of oxygen in either the water column or sediment.
- Autochthonous:** Organic matter produced within an environment.
- Autolysis:** Process of cellular dissolution following death which occurs due to the action of the organisms own enzymes.
- Autotrophs:** Organisms which use light as an energy source.
- Benthic:** referring to the sea bed.
- Biodeposition:** Transfer of particulate material to sediment due to feeding activities of benthic and epibenthic organisms.
- Biogenic:** Derived from biological sources *e.g.* biogenic silicon derived from dissolution of diatom frustule.
- Bioturbation:** A term derived from biological perturbation of soil and sediment and applied for activities of benthic organisms which result in reworked sediments.
- CEWP:** Classification of Estuaries Working Party.
- Cyanobacteria:** Blue-green algae, which are prokaryotic like bacteria.
- Denitrification:** The conversion of nitrate by bacteria into nitrogen gas, resulting in the loss of nitrogen from ecosystems. This process occurs mainly in anaerobic systems, but is also possible in aerobic systems.
- Diagenesis/Diagenetic:** Post-depositional changes of sediments resulting from physical, chemical and biological processes.
- EcoQO:** Ecological Quality Objective, a statement of an ecological goal.
- EcoQS:** Ecological Quality Standard, a numerical ecological target value.
- Ekman Transport:** The total wind-driven transport of water in the Ekman solution of the equations of motion is found to be at right angles to the direction of the wind..
- EQO:** Environmental Quality Objective, a statement of an ecological goal.
- EQS:** Environmental Quality Standard, numerical environmental target value.
- Estuary:** A semi-enclosed body of water which has a free connection with the open sea, within which seawater is mixed with freshwater flowing into the estuary from rivers.
- Eutrophication:** A process whereby natural waters are enriched with inorganic nutrients (ammonia, nitrate, phosphate).

GQA:	General Quality Assessment: Schemes developed by NRA/Environment Agency to monitor trends of water quality in the rivers, estuaries and coastal waters of England and Wales. These form one part of the Environment Agency's proposal for setting SWQOs.
Halocline:	Layer in which the salinity changes rapidly with depth.
Heterotrophs:	Organisms taking organic matter from their environment, because unlike autotrophs, they are not able to synthesise organic matter themselves.
Hindcast:	Estimate historical conditions.
Hydrolysis:	Chemical reaction in which H ₂ O is added to the original molecule. Due to hydrolysis organic polymers such as polysaccharides, proteins and lipids are split into their basic units.
Hypolimnion:	In lakes, the water layer below the thermocline.
Hypoxia:	Oxygen depletion of either the water column or sediment.
Infauna:	Organisms which live within the sediment for the greater part of their lives.
Macrobenthos:	Bottom-living organisms larger than 0.5 mm.
Macrophytes:	The larger, macroscopic, multicellular plants.
Macroplankton:	Large plankton visible to the unaided eye, retained by a coarse net, with a mesh aperture of approximately 1 mm.
Meiofauna:	Group of small animals (between 0.1 and 1.0 mm) which live in the spaces between the sediment grains.
Meroplankton:	Planktonic organisms passing through a pelagic phase which is only part of the total life-span; for example, planktonic spores, eggs or larvae of nektonic or benthic organisms.
Mesocosms:	Experimental ecosystems. on a mesoscale, <i>i.e.</i> , having a volume of between 1 and 10 ⁴ m ³ in pelagic systems, and a surface area of >1 m ² in benthic systems.
Microbenthos/Microbenthic:	Benthos consisting of the smallest animals. The arbitrary upper boundary of the animal size is generally set at about 100 µm, which is the lower limit for meiobenthos.
Microbial decomposition:	The return of constituents of organic matter to ecological cycles by the activities of micro-organisms.
Microbial loop:	Term describing the complex of organisms and processes thought to be responsible for transferring back into the traditional grazing food chain dissolved organic matter 'lost' from phytoplankton either by exudation, autolysis, or due to herbivore feeding.
Microcosms:	Experimental ecosystems smaller than mesocosms that are generally laboratory based.
Microphytobenthos:	Small (usually unicellular) plants living on the (shallow) sea floor. Particularly abundant on tidal flats.

Microplankton:	Planktonic organisms less than 1 mm in maximum dimension, but retained by a fine-mesh plankton net with a mesh aperture approximately 0.06 mm.
Mineralization:	The process by which organic matter is degraded.
Nanoplankton:	Organisms too small to be retained by fine-mesh bolting silk (less than 60 µm) but larger than 5 µm maximum dimension.
Nitrate:	Inorganic nitrogen compound in which the nitrogen atom is in its highest oxidation state. Chemical symbol NO ₃ ⁻ .
Nitrification:	The process in which ammonia is oxidised by bacteria to nitrate.
Nitrogen fixation:	The process by which nitrogen gas is reduced to ammonia.
Nutrients:	Term that refers to inorganic compounds which are essential for phytoplankton growth.
Palaeolimnology:	The study of past freshwater, saline and brackish environments.
Pelagic:	Marine life belonging to the upper layers of the open sea.
Phanerogams:	Flowering plants
Phytoplankton:	The whole group of (usually microscopic) floating plants, mainly algae. The term refers to a functional group (floating plants) not to a systematic entity, and is used for the autotrophic part of the plankton.
Picoplankton:	Small (between 0.2 and 2 µm) planktonic organisms.
Plankton:	Floating organisms of many different phyla, living in the sea or in freshwater, to a large extent subjected to water movements.
Protozoa:	General term for a wide variety of microscopic unicellular organisms, most of which are motile and heterotrophic.
Pycnocline:	Layer in which the density of the water rapidly increases with depth, because of the presence of a halocline or a thermocline or both.
SAC:	Special Area of Conservation.
Salinity:	Measurement of the concentration of salt in seawater.
Secchi disk:	A white disk of standard size (30 cm) used for estimation of the turbidity of the water.
Seston:	All biotic and abiotic particles in the water column whose movement is principally determined by the movement of the water mass.
Silicates:	Salts of silicic acid H ₄ SiO ₄ .
SMA:	Special Marine Area.
SPA:	Special Protection Area.
SSSI:	Site of Special Scientific Interest.
Stratification:	The occurrence of more or less horizontal water layers in the water body as a consequence of differences in density (<i>i.e.</i> salinity or temperature).

- SWQO:** Statutory Water Quality Objectives: Statutory means of maintaining and improving water quality in controlled waters required by the Water Resources Act 1991.
- Terrigenous sediments:** Sediments consisting of material eroded from the land surface. This material is liberated from igneous, metamorphic, and pre-existing sedimentary rocks by chemical and physical weathering, and transported by water, wind, or ice.
- Thermocline:** Layer at some distance below the surface where the temperature rapidly decreases with depth.
- Tidal Flats:** An extensive near-horizontal, marshy or barren, sandy (sand flat) or muddy (mud flat) area lying between the levels of mean high and mean low tide.
- Tidal Prism:** The amount of water in a semi-enclosed area between the high and the low water level.
- Trace elements:** Chemical elements occurring in seawater in concentrations typically below the micromolar level.
- Turbidity Maximum:** Phenomenon characteristic of many estuaries where freshwater mixes with seawater. Due to the difference in density of freshwater and seawater, a counter-current of seawater occurs underneath the outflowing freshwater from river through the estuary to the sea. Settling sedimentary particles from the freshwater layer are transported back up the estuary by the bottom counter-current, causing an accumulation of suspended matter in the low brackish zone of the estuary. This is known as the turbidity maximum zone.
- Turbidity:** The degree to which water contains particles that cause backscattering and extinction of light. High turbidity is caused by a high content of silt or, sometimes, of organic particles.
- Upwelling:** Upward movement of deeper (may be nutrient rich) water under the influence of divergence of water at the surface, usually caused by differences in Ekman transport.
- UWWTD:** Urban Waste Water Treatment Directive, EU legislation.
- WQO:** Water Quality Objectives.

EXECUTIVE SUMMARY

Nutrient enrichment may have obvious, subtle or no biological impact on the aquatic environment, depending on the influence of a number of physical, chemical and biotic factors. This influence is never more difficult to predict than in the dynamic and variable estuarine ecosystem. The complexity of estuaries hinders a clear understanding of cause and effect. This, in turn, makes it difficult for the Environment Agency and English Nature to manage estuarine water quality and conservation status, as required by EC Directives on Urban Waste Water Treatment, Nitrates and Habitats, and other elements of the regulatory framework.

The Agency and English Nature commissioned this review of literature and expert opinion on the impact of nutrients in estuaries as an important step towards delivering the sound scientific foundations on which effective management strategies must be based.

A number of factors influence the nature of nutrient inputs to estuaries. While regulatory control has tended to focus on point-sources, diffuse inputs represent a major challenge. It is notable that nutrient inputs from catchments characterised by diffuse sources, tend to be dominated by nitrogen. Furthermore, it is evident that the physical characteristics of an estuary, particularly freshwater flushing time, determine the extent to which nutrients are available for utilisation by any process. The processes in question, both biological and chemical, play key roles in nutrient cycling, and consequently mediate the cycles of growth and decay in estuaries. In non-light-limited situations the classic estuarine response to nutrient enrichment is an increase in plant growth. However, when light is not available to drive primary production (and where consequently hypereutrophication may not lead to eutrophication), physical factors such as hydrodynamics and geomorphology become dominant in determining the extent of other processes, such as denitrification.

Estuaries dominated by phytoplankton (eg. diatom and dinoflagellate blooms), in terms of biomass, are regarded as demonstrating the final stage in a succession of changes driven by excessive and chronic nutrient inputs to estuaries, however there are other, often more subtle, but important, primary impacts. Benthic microalgal communities, compete with phytoplankton for nutrients in shallow waters, and more widely when suspended by turbulent mixing. Macroalgae may also compete with phytoplankton for water-column-nutrients, and, like benthic microalgae, when forming dense mats, may obstruct the exchange of nutrients across the sediment-water interface, and restrict the development of infauna. A change from a low to a high nutrient environment will tend to drive a stepwise shift in community structure from vascular plants, such as seagrasses and saltmarsh vegetation, which tend to have a competitive advantage in low nutrient environments, to phytoplankton. As well as plants, bacteria, bound to sediments or associated with particulate matter in the water column, utilise inorganic nutrients, particularly nitrogen species, and play an important role in the estuarine response to nutrient input.

In addition to the primary (autotrophic) response to nutrient enrichment, there are secondary effects. The most prominent of these are the impacts of oxygen depletion, caused by the increase in organic loading that accompanies the accentuated primary production and subsequent decay of plant material in nutrient-enriched waters. This can

have major deleterious effects on benthic macrofauna, but a lesser impact on the more tolerant meiofauna. The primary and secondary impacts of nutrient enrichment can have knock-on tertiary effects, higher in the food-chain. In most cases, however, the complexities of food-web structure and the varying interactions between physical and biological processes in estuaries (which limit the scope to extrapolate findings between different systems), make it extremely difficult to predict the biological response at these higher trophic levels. Estuarine morphology, and, indirectly, the production of toxic sulphide in sediments, may shape the community at these levels.

It appears that, once underway, community changes triggered by nutrient enrichment may be self-perpetuating, and hence not completely reversible without ecological intervention. The lack of field data means that this conclusion can only be provisional, however, the implementation of nutrient controls driven by EU legislation will offer opportunities to further examine the extent to which eutrophication can be reversed purely by reducing the nutrient loading.

With good scientific understanding and relevant monitoring data it would be possible to assess the current nutrient/trophic status of an estuary, set meaningful and realistic objectives, formulate and implement actions to meet those objectives, and measure the success of those actions. However, we currently only have a partial, qualitative understanding of cause and effect, with respect to the impacts of nutrients in estuaries, and this, together with the inadequacy of available data (often collected for other purposes), makes it difficult to set meaningful chemical or biological monitoring criteria, or management targets.

It is therefore not surprising that there do not appear to be any examples of complete management strategies on estuarine eutrophication, although certain elements of such strategies do exist, for example, assessment and classification schemes. These qualitatively, or semi-quantitatively, describe the nutrient or trophic status of estuaries, or distinguish their differing ecological susceptibilities to nutrient impacts (allowing priorities to be identified). A scheme is proposed, in this latter category, as a generic framework for screening the eutrophication-risk of estuaries. The classification criteria are physico-chemical parameters (nutrient input, turbidity, flushing time, etc.). As a broad-brush approach, this would not be applicable in issue-driven situations (eg. SACs, SPAs, etc.), where detailed information has already been collected.

A further fundamental challenge to management is the setting of relevant targets, and the most likely way forward is seen in the development of Ecological Quality Objectives (EcoQO's), as established, conceptually, in OSPAR and EU discussions on environmental protection. Key estuarine communities are identified, and high level objectives established for them, eg. for saltmarshes, it is suggested that there should be no persistent decline in areal coverage or productivity, relative to all other estuarine vegetation, nor increase in plant tissue nutrient levels, over specified timescales. Such nascent EcoQOs clearly need further development and trialling.

The classification and EcoQOs are two key elements of a proposed four-stage process for managing nutrient impacts on water quality and conservation interest, in the estuaries of England and Wales:

- screening to identify priorities;

- assessing, in detail, the status of those priority sites;
- setting specific objectives for the sites;
- taking appropriate action and monitoring to assess progress towards the objectives.

This proposal extends the approach already used to examine estuarine nutrient impacts, in the context of the EC Urban Waste Water Treatment, Nitrate and Habitats Directives, to a more generic and widely applicable process.

The Environment Agency and English Nature are invited to trial this management approach, and to consider recommendations for further research to fill important gaps in their understanding.

KEYWORDS

estuaries, nutrient, conservation, ecosystem, eutrophication, phytoplankton, microalgae, macroalgae, macrophyte, denitrification, flushing time, turbidity, estuarine monitoring, estuarine classification, ecological quality objectives.

1. INTRODUCTION

1.1 Project Background

The recent increasing concern about the ecological effect of nutrients in estuaries and nearshore waters has been accompanied by an increase in legislation aimed at both controlling nutrient levels and assessing the potential for biological impact from increases in nutrient loading. This includes the following EC Directives: the Urban Waste Water Treatment (UWWT), Nitrates, Habitats and Species, Bathing Waters, and the Framework Directive on water resources.

Despite this large interest and a large amount of associated scientific research and environmental monitoring, many questions regarding the ways in which nutrients affect estuarine communities still remain unanswered. However, the Environment Agency and English Nature both need to have a clear indication of the biological effects of nutrients and their dynamics in estuarine systems particularly with respect to assessments of environmental quality and the conservation of habitats and species respectively.

The Environment Agency is responsible for the protection of 175 estuaries in England and Wales and so require to clarify present knowledge on the dynamics of nutrients within estuarine systems, and to develop effective approaches for the monitoring and management of nutrient levels. In particular, the Environment Agency is obliged to ensure that estuaries meet statutory objectives and standards (derived from EC directives) and non-statutory water quality planning and assessment targets.

In contrast, English Nature (EN) need to gain a clearer understanding of the effect of nutrients on communities in order to manage effectively and conserve marine and estuarine resources. In particular there is a requirement for English Nature to manage Special Areas of Conservation (SAC) as well as other statutory areas such as Sites of Special Scientific Interest (SSSI), and non-statutory areas such as Special Marine Area (SMA) in respect of the potential impacts from water quality changes.

To pursue these objectives, the Environment Agency, in collaboration with English Nature, have instigated a Research and Development Study on the 'Impact of Nutrients in Estuaries' (i639). The first phase of this project took the form of a workshop, which was arranged by both Environment Agency and English Nature and was attended by several academics, policy makers and representatives from environmental monitoring agencies throughout the UK.

At this workshop the problems associated with high nutrients in estuaries and the implications for environmental management were discussed and the following conclusions were reached (Nedwell 1995):

- There is a lack of basic information in key areas required to define baselines; not enough information to tell us what we need to monitor and why.
- Particularly in estuaries, our understanding of the levels, fluctuations and impacts of different nutrients is unclear. Part of this difficulty is distinguishing the effects of nutrient elevation from natural variations due to tide or season. Relating nutrient concentration to salinity may be helpful, and nutrient ratios should be taken into account.

- There is a lack of knowledge as to what are key, limiting nutrients in estuaries, originating from a lack of understanding of the estuarine systems involved.
- The relative importance to nutrient budgets of subsystems within estuaries is unclear. Particularly, the role of benthic biofilms may be important, but is poorly understood.
- There is no objective basis presently available for defining quality standards with respect to nutrient concentrations or loading in estuaries, either in terms of the impact of increased loading on the estuarine system or in respect of limits for conservation.
- There is a general agreement that there is a fundamental difference between oligotrophic and eutrophic estuaries, but the definition of this difference is not clear. It may be in terms of tidal effects, residence time, or underlying geology, and it is unlikely to influence the capacity of the estuaries to respond to elevated nutrient load. Future work should concentrate on a comparison between these two major estuary types.
- There may be biological groups or species which are indicators of nutrient elevated estuaries, but further work is required to define these in any useful way.

From these conclusions it is evident that the complexity of the estuarine environments and the limited amount of available information on many aspects of nutrient dynamics and the biological responses in different estuarine systems present major obstacles to understanding the effects of elevated nutrient levels within estuaries.

Following this workshop, Phase 2 of the 'Impact of Nutrients in Estuaries' project was initiated. The aim of this second phase was to pursue the recommendations made during the Phase 1, to review the current status of scientific knowledge on the subject, and to propose possible approaches for the monitoring and management of estuaries. The complete specification document for this project is shown in Appendix 1.

1.2 Project Consultants

The Cambridge Coastal Research Unit (CCRU) was commissioned to undertake Phase 2 of the 'Impact of Nutrients in Estuaries' project. In collaboration with CCRU were four associated consultants who provided expert advice and opinions on the topics under discussion. These five project consultants were:

Dr M. Elliott (University of Hull). Director of the Institute of Estuarine and Coastal Studies and a senior lecturer and consultant in applied marine biology based at the University of Hull. He has a wide experience in the assessment of eutrophication. Together with Dr V.N. de Jonge, he recently completed a guideline study for "Considerations in Determining Potential Eutrophication' in response to the EU Urban Waste Water Directive and Nitrates Directive". They have also undertaken a study of the eutrophic status of the following estuaries: Deben, Ore/Alde, Stour and Colne for the Environment Agency (Anglian Region) and with Department of Agriculture (Northern Ireland) on the eutrophication status and nutrients in Strangford Lough, NI.

Dr S. Malcolm (Centre for Environment, Fisheries and Aquacultural Services, Lowestoft Laboratory). The manager of the JoNuS (Joint Nutrient Survey) project which is investigating the transport of nutrients through estuaries, examining many of the key nutrient transformation processes and also considering the impact of nutrients on coastal waters. In addition, Dr Malcolm has been involved in the development of national and international policy through the EU Directive, OSPARCOM (Oslo and Paris Commission) and CSTT (Comprehensive Studies Task Team of Marine Pollution Monitoring Management Group) working groups. He provided guidance to this contract as an independent advisor.

Dr M. Wilkinson (Heriot-Watt University) A senior lecturer and researcher in marine biology based with extensive experience of consultancy work in Environmental Impact Assessments, water quality evaluations and marine conservation. In recent years he has been involved in an EIA of the Channel Tunnel and Trans-Manche Link developments. He supervised a four year study of the entire intertidal shoreline of Northern Ireland to recommend sites for marine nature reserves. He specialises in the field of marine phycology and has been involved in studies of nuisance algal growths and in the use of effluent toxicity testing using marine algae. He also advised companies and organisations on the implementation of the EU UWWT and Nitrates directives.

Prof. V.N. de Jonge (RIKZ, Netherlands) A Senior Researcher at for the National Institute for Coastal and Marine Management (RIKZ) based at Groningen in the Netherlands. He is a researcher, lecturer and government advisor in a range of fields including: marine chemistry, chemical methods, sedimentology, marine ecology and environmental management. He has conducted extensive research on nutrient dynamics in estuarine and coastal environments and has published extensively on these topics.

1.3 Project Approach and Outline

This study involved a detailed review of literature on the subject of nutrient dynamics and their biological impacts in estuaries. To obtain the latest information for this review, a large number of relevant researchers and organisations (from the UK and abroad), were contacted and asked for the most recent published and unpublished literature. A list of these contacts is shown in Appendix 2 and all the information collated from these sources was incorporated into the review. Together with this review, meetings were also held between the five project consultants to discuss the issues detailed in the project specification document and to obtain their opinions and recommendations for the monitoring and management of estuaries with respect to nutrient enrichment.

The report is divided into four sections: the first three sections are reviews of literature and the final section presents the overall conclusions reached by project consultants. The contents of these sections are:

Section 2: The Effects of Physico-Chemical Factors:

This section is a detailed review of the effects of physical and chemical factors on nutrient dynamics and biological responses in estuaries.

Section 3: Biological Impact of Nutrients:

This section is a detailed review of the impacts of nutrients on estuarine communities as well the effects that estuarine species have on nutrient dynamics in estuarine systems.

Section 4: Previous Approaches for Assessing Nutrient Impacts:

This section is a review of the tools, models and techniques that have been used for monitoring and managing nutrient impacts in estuaries. It includes a discussion of the adequacy of available data for describing nutrient impacts, and a review of methodologies for determining 'background' nutrient levels within estuaries.

Section 5: Proposed Approaches for Assessing Impacts:

This section describes an approach for the monitoring and management of impacts in UK estuaries which has been developed by the project consultants based on their experience and the information contained in the three previous reviews. Based on the information contained in Section 2, a list is presented of the physical factors which have the most significant influence on the susceptibility of estuaries to impacts from nutrient enrichment and a scheme is proposed for using these factors to classifying estuaries on this basis. Information contained in Section 3 is used to give a list of the potential biological impacts of nutrients and a series of preliminary Ecological Quality Objectives (EcoQO) are proposed for use in the management of nutrient impacts in estuaries. This leads to a protocol for the monitoring and management of estuaries being proposed which incorporates the classification scheme and EcoQOs. Finally, recommendations are made for future studies to fill gaps in knowledge about nutrient effects and the future direction of the 'Impact of Nutrients in Estuaries' project.

2. EFFECT OF PHYSICO-CHEMICAL FACTORS

Section Aim

- To review the effects of physical and chemical factors on nutrient dynamics and biological responses.

Section Contents

Section 2.1 Introduction - The Nature of Estuaries:

A brief overview of the characteristics of estuarine environments and the role of nutrients in aquatic environments.

Section 2.2 Input of Nutrients:

A review of the effects of physico-chemical factors on the scale and balance of nutrient input to estuaries.

Section 2.3 Nutrient Retention:

A review of the physico-chemical factors which influence the retention of nutrients within estuarine systems.

Section 2.4 Nutrient Transformations:

A review of the processes which drive the cycling of nutrients within estuaries and the effects of physico-chemical factors on these processes.

Section 2.5 Biological Responses:

A review of the physico-chemical factors which affect the responses of estuarine organisms to available nutrients.

Section 2.6 Conclusions:

The effects of physico-chemical factors on the impact of nutrients are summarised and conclusions drawn.

2.1 Introduction - The Nature of Estuaries

Estuaries are interface environments between the highly contrasting conditions of riverine and marine waters and, as a result, they are characterised by complex and dynamic environmental conditions. Physical and chemical parameters exhibit a high degree of temporal and spatial variability in estuaries driven by changes in: river discharge, tides, weather and climatic conditions, which result in aperiodic changes in factors such as current speed, temperature, turbidity level, oxygen concentration, salinity and pH.

As with any environment, the nature of the communities in estuaries are dependant upon environmental forcing variables, as well as the influences of biotic processes such as competition and recruitment, and as a consequence of the dynamic and unstable conditions the communities in estuaries typically have a low diversity and stability. The single most influential factor is in the salinity regime which varies spatially along the length estuaries, and temporally at individual locations and has a large effect on the abundance and distribution of species by imposing osmotic stresses on organisms. As the majority of estuarine species are of marine origin there is typically a decrease in diversity of species that accompanies the decrease in the mean salinity levels up an estuary (McLusky 1989).

Estuaries also often receive a high input of allochthonous organic materials and nutrients from the surrounding catchment. This often leads to a high autochthonous production of organic material and a high degree of nutrient recycling within the system. Therefore, while the diversity of species is typically low, estuaries often have a greater abundance of organisms than adjacent aquatic environments because there is often a rich supply of food available to those species which can adapt to the estuarine environment. For the same reason, with the decline in diversity up an estuary due to the influence of changing salinity levels, there is often an accompanying increase in the abundance of individual species as salinity-tolerant species avoid competition from marine or freshwater species and exploit the abundant food resource (Remane and Schleiper 1951)¹.

Nutrient-rich conditions are typical of many estuaries and there is now world-wide concern about the potential consequences of sustained increases to the nutrient inputs that occur as a result of anthropogenic activities. Elevated nutrient inputs may cause adverse ecological impacts by promoting elevated levels of autotrophic growth and bacterial activity which can then cause environmental impacts at higher trophic levels (see Section 3). However, the response of communities to nutrients in estuaries is also dictated by the physico-chemical conditions which are not only dynamic and variable within individual estuaries, as identified above, but also vary greatly between different estuaries due to differences in the geomorphology and catchment characteristics of different systems (a review of different estuary types is presented in Appendix 3). This variability of environmental conditions within and between estuary systems makes it particularly difficult to clearly define the cause and effects relationships between nutrient increases and biological responses in estuaries.

This presents major problems for the Environment Agency and English Nature who need to identify the best strategies for monitoring nutrient levels and assessing the risk of impacts to

¹ Further details about the differences in the community types in different estuarine regions are contained in Appendix 4 along with descriptions of food web types and a review of the major habitats, based on the JNCC review of British estuaries (Davidson *et al.*, 1991).

estuarine communities. A good understanding of how conditions within estuary systems influence biological responses to nutrients is required in order for the Environment Agency and English Nature to develop an effective national strategy for estuary management respectively in relation to estuarine quality management and conservation management. This section therefore reviews the role and significance of physical and chemical factors in the impact of nutrients on estuarine communities.

Due to the significant degree of interdependence between physical and chemical factors in estuaries, the review is given in the context of their effects on each of the following processes:

- the input of allochthonous nutrients (Section 2.2);
- the retention of nutrients within the systems (Section 2.3);
- nutrient transformations (Section 2.4)
- biological responses (Section 2.5).

2.2 Nutrient Inputs

One of the primary considerations when assessing the potential for nutrient impact is the scale and relative balance of the nutrients input and several physical, chemical as well as socio-economic factors affect this input. Two categories of anthropogenic nutrient input are defined according to the two routes by which they enter the system, these are:

- Point source inputs arise from the outfall discharges of industrial plants and sewage treatment works and are therefore exclusively of anthropogenic origin.
- Diffuse source inputs are those which originate from non site-specific sources namely; drainage of the catchment, atmospheric deposition and groundwater seepage and through exchange with the coastal waters.

These two sources may be regarded as supplying allochthonous or 'new' nutrients as distinguishable from autochthonous or 'old' nutrients which are produced within the estuary by nutrient cycling processes (Barnes and Mann 1991). The relative nutrient inputs of these two sources are dependant upon several physical, chemical as well as socio-economic factors. For point source nutrients (excluding tributaries as point-sources), which are entirely anthropogenically-derived, the rate of input is determined by catchment socio-economic factors such as the population density, inputs from sewage outfalls and industrial discharges and wastewater treatments. The delivery of nutrients from diffuse source inputs are dependant upon a combination of anthropogenic activities such as the nature and extent of agricultural practices as well as natural factors such as the physical (*e.g.* soil type and geomorphology) and biological (*i.e.* habitat types) characteristics of the catchment and the climatic conditions.

The relationship between land use and nutrient export from the catchment was demonstrated by Magnien *et al.*, (1995) who described the relative proportions of nutrients and sediment exported to Chesapeake Bay from four different habitats within its catchment. This indicates that urban and agricultural areas leach a far greater amount of nutrients than forests or wetland areas (Figure 1).

Lyons *et al.*, (1993) studied the fluxes of nutrients from 39 rivers on the east coast of Scotland and identified catchment size, geomorphology and soil type as important determinants of the amount of nutrients reaching the estuary. They also showed that concentrations of nitrate and phosphate were greater in the lowland rivers than in the highland areas, a feature attributed to the contrasting soil types of these catchment areas. In the lowland rivers, there was also a greater variability of nutrient inputs because of the differences in the intensity of the arable production in different catchments. Gerritse (in prep) showed that soil type had an important effect on nutrient export to the Peel-Harvey estuary (Australia). They found that most of the nitrogen and phosphorus inputs comes from sandy areas of the catchment which represent only 35% of the catchment area.

Balls (1994) also demonstrated the relationship between land use and the nutrient inputs to nine Scottish east coast rivers. He showed that river catchments with intensive agriculture and low freshwater input, such as the Don and Ythan, were found to have enhanced nitrate (up to 600µM) and phosphate (up to 5µM) concentrations in their estuaries. In contrast, highland river catchments with mineral-poor soils, low populations and low agricultural intensity (Inverness, Cromarty and Dornoch Firths) in general had lower nutrient concentrations in river water than in coastal seawater.

Macdonald *et al.*, (1994) also attributed the very high concentrations of nitrate in the River Ythan to the highly agricultural nature of its catchment. This study analysed nutrient export rates from the Don, Dee and Ythan catchments and found that nitrate and suspended sediment export from catchments reflect the percentage area of agriculture in the catchments. Similarly, in the Ythan catchment, a change towards greater cereal production and an associated increase in rate of fertiliser application that is also believed to have led to the observed long term increase in estuary nitrate concentrations (Raffaelli *et al.*, 1989; Conway 1993; Balls *et al.*, 1995). The interpretation of phosphate data in the Ythan was more complex but concentrations in the estuary also appear to have increased.

Even where catchments have similar land uses and climates, differences in management practices may affect the nutrient loads. For instance the Ythan showed mean spring peaks of nitrate which were not observed in the adjacent estuaries of the Dee and Don, a feature presumed to be due to differences management practices such as agricultural ploughing which appears to causes the most significant increase in nitrate load (Macdonald 1995). Macdonald (1995) studied the nutrient inputs to the Ythan in greater detail and observed increased mean annual nitrate concentrations over the period 1980-1992 particularly during the spring months and concluded that cropping was responsible for the increase in winter, rather than spring. Addiscott (1988) proposed that the majority of nutrient run-off from agricultural sites occurs in autumn when bacteria mineralise organic matter in the topsoil and that certain management practices such as sowing crops in autumn could limit diffuse run-off.

The location of the different land use types within a catchment will also affect the rate of nutrient export in freshwater run-off. For example, Magnien (1995) notes that riparian forests, which act as a filter for run-off sediments and nutrients whilst also limiting shoreline erosion, have a greater value than forests set at the outer limits of the catchment. Equally, if heavily fertilised arable land, which is a major exporter of nutrients, is set adjacent to major drainage routes then this site is likely to cause an increase in nutrient inputs to the catchment run-off. For the same environment set at the periphery of the catchment, the degree of influence will depend drainage characteristics of the catchment, the geological characteristics, land uses between it and the run-off channels (Johnes and Heathwaite 1996).

The relationship between land-use and the rate nutrient export through catchment drainage has been closely examined by Johnes (1994) who developed an approach for modelling nutrient export in order to 'hindcast' nutrient loads and identify good practices for the management of catchments (see Section 4.4). To develop this model it was necessary to consider all the natural and anthropogenic factors which govern the rate of nutrient export from catchments and a list of these factors is shown in Table 1.

Table 1: List of the factors governing rates of nutrient export from catchments (Johnes 1994).

Primary Factor	Secondary Factors
1. Climate of Catchment	Mean Annual rainfall - seasonality, rainfall intensity
	Temperature, wind speed and direction
2. Catchment Geology	Presence and Lag-time of aquifer (also key determinant for factors 3 & 5)
3 Catchment Topography	Incline of hills
	Presence of flood plain
4. Soil Type & Distribution	Soil structure and intrinsic stability
	Intrinsic nutrient retention capacity
5. Catchment Hydrology	Hydrological pathways linking land to stream network
	Seasonal variations in routes
	Connections to aquifers (if present) to surface water
	Human manipulation of catchment hydrology (<i>e.g.</i> irrigation)
6. Land Management	Timing of fertiliser application
	Application of sludge and manure to land as slurry
	Cultivation of Legumes
	Cultivation of Crops with High Residues (<i>e.g.</i> oilseed rape)
	Fodder Cropping
	Production intensity (<i>e.g.</i> number of crops/year, livestock size & density)
7. Sewage Treatment	Proportion of human population on septic tanks
	Degree of sewage treatment
	Use of phosphorus stripping
8. Location of nutrient sources	Proximity of nutrient sources to catchment hydrological pathways

In some systems, seepage from groundwater reservoirs may be another source of diffuse nutrient inputs. Gerritse (in prep) suggests that this source can be particularly rich in nutrients so that, even where it represents a small proportion of the overall volume of freshwater input, it can be a significant contributor to the estuary nutrient load. For instance, Gerritse (in prep) showed that in the Peel-Harvey estuary (Australia) the volume of groundwater inputs is only 0.5% of the river flow but inputs from this source were implicated in benthic algal blooms.

Coastal waters have also been shown to provide an important source of nitrogen and phosphorus (Nixon 1995a; Nixon, 1996). Nixon et al. (1995a) showed that in Narragansett Bay, inputs of nitrogen from offshore constitute 10-20% of the total inputs and they believed that this input may be the most important contributor to estuarine productivity when the DIN is depleted. The flux of DP from offshore was found to be equivalent to that from land drainage such that the direction of the coastal flow may also be an important consideration with respect to nutrient imports. The landward flow of bottom water across the continental

shelf of the north-east US is considered to have an important effect on the productivity of eastern seaboard estuaries such as Narragansett Bay and Chesapeake Bay (Nixon, 1996).

Nutrients also enter the estuary via atmospheric deposition which can occur either as wet-deposition (rainfall) or dry-deposition (aerosols,). Atmospheric-deposition consists principally of nitrogen, the phosphorus content being small (Fanning 1989 cited in Lyons *et al.*, 1993). Paerl (1995) identifies the following three components of atmospheric nitrogen:

- Fossil fuel combustion products. Fossil fuel is a source of nitrogen oxides (NO_x) that dissolve in water (such as rainfall) and produce either nitrite (NO_2^-) or nitrate (NO_3^-) anions.
- Agricultural (atmospheric) discharges. These are often dominated by gaseous or ionised ammonia (NH_3 and NH_4^+) that are readily soluble in receiving waters.
- Organic nitrogen (ON). Detectable concentrations of amino acids have been found in rainfall along coastal Florida.

Atmospheric inputs are considered by some authors to be highly significant and are expected to increase substantially in the future. Barnes and Mann (1991) for example, suggest that N_2O emissions have increased continuously since 1900 and are projected to increase a further 40-60% over the next 40 years. Nixon (1996) estimated that atmospheric inputs of nitrogen to Narragansett Bay has increased 15 times over the 'natural background' level. Paerl (1995) reviewed the role of Atmospheric Nitrogen Deposition (AND) and concluded that among loading sources, atmospheric nitrogen deposition was of high and increasing importance. He also considered that accelerating primary production, or eutrophication of coastal water appeared to be linked to increasing atmospheric nitrogen loading. He calculated that the current contribution of AND to total external nitrogen loading ranges from 10 to over 50% and that this might play a central role in mediating "new" production (*i.e.* production resulting of from allochthonous inputs) in coastal oceans downwind of emissions.

The contribution of atmospheric input appears to be dependant on the size of the estuary. In a large system like Chesapeake Bay the precipitation-borne nitrogen inputs represent an estimated 39% of the total input (Barnes and Mann 1991) and in the Baltic Sea acid rain has contributed significantly to the nitrogen load (Larsson *et al.*, 1985). In smaller estuaries such as Narragansett bay, which as studied by Nixon, *et al.*, (1995b), atmospheric nitrogen deposition represents only 4% of the total input. Nowicki and Oviatt (1990) also studied atmospheric inputs to Narragansett Bay and observed that while dry deposition rates were difficult to assess, an estimated 5-10% of nitrogen and 5% of phosphorus enters the system via wet deposition. In the much smaller Colne estuary atmospheric inputs are considered to be insignificant (Ogilvie *et al.*, in prep.).

Atmospheric deposition must also supplement nutrient levels within the catchment and this may eventually affect the inputs to the estuary through catchment drainage although, during drainage, the nutrients are likely to be heavily modified. In turn, the scale of inputs to an estuary will be dependant on factors such as catchment size, soil type and land use. For instance Barnes and Mann (1991) note that the felling of forests reduces the capacity of a catchment to intercept precipitation-borne nutrients.

The relative contribution of diffuse and point source nutrient inputs to a systems has an effect on the balance of the nutrient load that enters the system. In general nitrogen is the most common nutrient in diffuse sources whereas in point source inputs concentrations of

phosphorus are often highest. The effect of different nutrient sources on the composition of the nutrient inputs was demonstrated by Boynton *et al.*, (1995) who compared the loading rates and ratios of nitrogen to phosphorus in the 18 different systems connected to the mainstem of Chesapeake Bay. They found that systems with mainly diffuse inputs have high TN:TP ratios while those with point sources have correspondingly lower ratios.

The input of silicate is derived from the weathering of geological material and, unlike nitrogen and phosphorus, is not derived directly from anthropogenic sources related to diffuse inputs. Human activities can influence the export of silicate through activities such as the working of soil during agricultural activity which affects the rate of weathering (Lyons *et al.*, 1993).

The relative contributions of diffuse and point source inputs also has important implications for the management of estuaries. Several authors have recognised that in systems where diffuse inputs are large then the management of nutrient inputs estuaries is likely to be most difficult and costly (Raffaelli *et al.*, 1989; Wulff and Niemi 1992; Malone *et al.*, 1993). This is because the control of diffuse inputs may require large-scale social changes, such as alterations in agricultural practices, urban developments and land usage. In Chesapeake Bay for example there has been an increasing realisation by both the scientists and managers of the importance of diffuse inputs as opposed to point source which were the only concern 20 years ago (Malone *et al.*, 1993). The need to address the problems of diffuse nutrient inputs has also been raised for the Baltic (Wulff and Niemi 1992) and the Ythan (Raffaelli *et al.*, 1989). Under the OSPAR Convention there has been a successful attempt to reduce the input of phosphorus to the aquatic environment of the north east Atlantic by 50% based on a 1985 baseline. A similar OSPAR commitment to reduce the input of nitrogen has yet to be effective because of the problem of managing diffuse inputs.

The rate of freshwater flow into estuaries has a important influence on the scale and composition of nutrient inputs. As the freshwater flow rate is clearly a reflection of the amount of rainfall and run-off it is directly related to the amount of nutrients entering the system from catchment drainage (*e.g.* nitrate from agricultural land). Alternatively, changes in freshwater flow do not affect the total nutrient input from point sources (*e.g.* ammonia and phosphate from sewage treatment works) but instead alter the level of dilution and the concentration of nutrient that enter the estuary from this source. Therefore, for any given estuary the effect of changes in river discharge on the nutrient inputs will be dependant on the relative contribution of diffuse and point source inputs. In Narragansett Bay, for instance, the riverine inputs of nitrogen vary by a factor of two between wet and dry years while DIP inputs however vary little reflecting the importance of diffuse sources for nitrogen and point source discharges for phosphorus (Nixon *et al.*, 1995a). The effect of river flow on the composition of the nutrient input was also demonstrated by Fisher *et al.*, (1988) who compared the effects of river discharges to Chesapeake Bay and Delaware Bay. They found that for the former estuary nitrate input was directly correlated with run-off while for the latter, where wastewater outfalls are an important nutrient, there was an inverse relationship.

Similarly, Macdonald *et al.*, (1994) showed that loads of both nitrate and phosphate from the agricultural catchments of the Don, Dee and Ythan were closely associated with rainfall and river flow rates while flow was positively correlated with nitrate, there was an inverse relationship with phosphate. Lebo *et al.*, (1994) observed a similar inverse relationship between river flow and phosphorus in Delaware estuary (USA). In the upper region of the Itchen estuary Franklin *et al.*, (1991) observed higher dissolved inorganic phosphorus (DIP)

and dissolved organic phosphorus (DOP) concentrations in summer than winter and attributed this to low rainfall and a net decrease in riverine water input prior to the summer survey. Franklin *et al.*, (1991) also showed that the relative importance of different DOP sources changed from summer to winter in the Itchen. In winter the main source of DOP was riverine inputs whereas in summer, under low riverflow conditions, inputs from treated sewage was more important.

Freshwater input is subject to a high degree of temporal variability due to changes in climatic conditions over both the short term (weekly, monthly) and the long term (seasonal/annual). These temporal variations will result in corresponding fluctuations in the scale and composition of the nutrient inputs. This temporal variability is largest in estuaries where nutrient inputs are largely from diffuse sources. For instance, in the Odense Fjord in Denmark where 86% of the nitrogen input is from agricultural run-off and 14% from waste waters sources, 10,000Kg N/day enters in February compared with <2000Kg N/day in June to August (Frederiksen 1987 in Conway 1993). In estuaries dominated by point-source inputs, the most severe impacts are likely to occur during periods of low-flow (*e.g.* dry summers) when nutrient concentrations are high. In the Forth, for example, which is subject to large inputs of domestic and industrial wastes, problems arise under low-flow conditions in summer months (Balls *et al.*, 1996).

The importance of river flow to nutrient inputs as well as the complexity of the relationship between these two factors was emphasised by Balls (1994). He compared the riverine nutrient inputs to nine Scottish estuaries and found that the magnitude of nitrogen and Si inputs were more greatly affected by the patterns of freshwater flow than by the nutrient concentrations in individual rivers. As a consequence, the highest nutrient inputs were recorded in the Tay and the Tweed which have high river discharge rates rather than the Don and Ythan which have high concentrations of riverine nutrient inputs. The relationship between phosphorus and river discharge was slightly different to that observed for nitrogen and Si. Fluxes of phosphorus were mainly from the rivers in the Tay and Tweed estuaries where the river discharge was high but, in the Forth and Don, phosphorus inputs were relatively more important compared with other nutrients. For the Forth, where there is a low flow, this was believed to be because of the release of phosphate from particulate material and for the Don this was attributed to the large number of wastewater discharges and industrial inputs.

2.3 Nutrient Retention

One of the factors which dictates the impact of nutrients is the length of time that they remain within an estuary which in turn affects their availability to communities and the potential for biological utilisation. The flushing time of an estuary has a major effect on nutrient retention in being the time that freshwater takes to pass through an estuary and is, therefore, an indication of the amount of time that freshwater-borne inputs of nutrients remain within the system before they are exported to coastal waters. The flushing time is influenced by the physical characteristics of an estuary such as its size, bathymetry, river discharge rate and tidal exchange. Helder and Ruurdij (1982), for instance, found that flushing times depended on a combination of the length of an estuary and the dispersion coefficient which is a function of river discharge and bathymetry.

The flushing time can also be affected by water column stratification which changes the pattern of water flow through the estuary. Stratification retains imported nutrients within the surface layers which then pass more rapidly through the estuary because of the restricted exchange with the deeper waters. The factors influencing water column stratification include the bathymetry, freshwater flow, tidal regime and climate. There is, for instance, a greater likelihood of a stable halocline in deep, poorly-mixed estuaries which are less susceptible to riverine-input and wind and tide-driven water circulation which can break down the stratified layer.

The presence of shallow banks or sills at the seaward margins of some systems can also influence the development of a stratified layer because they favour the faster exchange and circulation of surface water at the expense of the deep water areas. Sills can also play an important role in trapping nutrients particularly if they are shallower than the photic depth as is the case in the large semi-enclosed basins of the Mediterranean and Baltic Sea (Wulff *et al.*, 1990). Under these conditions the bottom water nutrients cannot leave the system without first entering the illuminated surface trophogenic layers. Once in the surface waters, these nutrients are then more likely to be assimilated by autotrophs and then returned to the deeper waters through the settlement of organic matter. This effect is seen in the Clyde estuary where a sill is present and in the summer months a strong halocline forms and nutrients can become trapped (Jones *et al.*, 1995).

The water circulation in an estuary and the strength of the currents can also affect nutrient retention by influencing the rate at which particulate nutrients or nutrients that are bound to sedimentary particles settle to the substratum. The deposition of these nutrients will be greater under slower flow rates. Elliott *et al.*, (1994) identify a relationship between the energy of the environment, the level of deposition and the extent to which an estuary retains organic matter. Similarly, Ogilvie *et al.*, (in prep.) demonstrated that the Colne estuary is a trap for organic nitrogen because of the deposition of organically bound nitrogen (from both terrigenous sources and from primary production within the estuary) in the turbidity maximum region.

The characteristics described above all influence the physical transport of allochthonous nutrients through an estuary which affects the availability of nutrients and the potential for them be transformed through biological utilisation or physico-chemical reactions. These transformations can themselves also promote the retention of nutrients so that physical transport and nutrient transformations have a synergistic effect on nutrient retention. In addition, the combined effects of transport and biological utilisation of nutrients can

influence the extent to which nutrients are recycled and can, therefore, affect the scale of autochthonous nutrient inputs to a system or the rates of nitrogen removal through denitrification.

This combined effect was demonstrated by Balls (1994) who examined the relationship between flushing time and changes in nutrient concentrations on several Scottish estuaries. This study showed a link between the duration of the flushing time and the degree to which internal processes (biological and geochemical) modify the nutrient input. In rapidly flushed estuaries such as the Tweed, Don and Ythan, it was shown that nutrients tend to behave conservatively showing relatively little evidence of being affected by biological or physico-chemical processes. However, in the large, slowly flushed estuaries such as the Forth, Tay and Dornoch Firth, internal processes had a significant effect. Balls (1994) concluded that as there is a greater degree of nutrient cycling in slower flushing systems, then estimates of riverine inputs to these estuary may not provide a good estimate of the load that eventually enters the coastal zone.

2.4 Nutrient Transformations

In estuaries, as in all other aquatic environments, nutrients are subject to continual 'cycling' as a consequence of biological, physical and chemical processes. These cycles involve changes in the chemical composition and spatial distribution of nutrient species which occur through processes such as assimilation by autotrophs or bacterial activities. The physical translocation of nutrients occurs through the circulation of water which transports water-borne nutrients around the estuary or through deposition and resuspension which move nutrients between the sediment and water column.

The cycles of nitrogen, phosphorus and silicon are driven by different transformation processes. The transformations of nitrogen are mediated almost exclusively by biological processes (e.g. assimilation by autotrophs and processing by bacteria) while those of the phosphorus and silica cycles are mediated by a combination of physical, chemical and biological processes. The processes involved in the cycling of these three nutrients are described in the following sections. The direct effects of physico-chemical factors on the phosphorus and silicon cycles are identified in the relevant sections and the indirect effects of physico-chemical factors on the nitrogen cycles, through their effects on bacterial activities, are considered in Section 2.4.5. Although physico-chemical factors will also indirectly affect nitrogen transformations through their effects on the responses of autotrophs this subject is considered separately in:

- Sections 2.5: discusses the effects of physico-chemical factors on the responses of autotrophs;
- Section 3.2: includes discussions about the effects of autotrophs and bacteria on nutrients.

2.4.1 The Nitrogen Cycle

The nitrogen cycle is largely driven by biological processes which convert nitrogen species between three states: elemental N_2 , inorganic and organic compounds. The nitrogen chemical cycle is depicted in Figure 2 (from Libes 1992) and the individual processes which make up the overall pathway are as follows:

- **Fixation:** This is the process by which gaseous dinitrogen is combined with free hydrogen molecules to produce ammonium ($N_2 \Rightarrow NH_4^+$) and is carried out by both free living and symbiotic bacteria (including cyanobacteria). This process requires anoxic conditions because nitrogenase, the enzyme which mediates this reaction, is sensitive to oxygen. Consequently, the rate of fixation tends to be highest in benthic environments where anoxic conditions occur. Nitrogen fixing bacteria are often found in the sediments of saltmarsh and seagrass beds where they can provide a significant contribution to the total nitrogen input of the ecosystem. However, in the majority of estuarine ecosystems nitrogen fixation makes a small contribution to available NH_4^+ compared with other sources (Barnes and Mann 1991) but can contribute significantly in nitrogen poor environments.
- **Assimilation:** This involves the uptake of inorganic nitrogen species by plants and their conversion to larger organic molecules (NH_4^+ , NO_2^- and $NO_3^- \Rightarrow$ amino acids \Rightarrow proteins).

- **Remineralisation/Ammonification:** This is the conversion of complex organic nitrogen molecules back to inorganic ammonia (amino acids and proteins \Rightarrow NH_4^+). Libes (1992) describes this as two step process with remineralisation being the conversion of organic materials from the particulate to the dissolve state ($\text{POP} \Rightarrow \text{DOP}$) and ammonification the conversion of DOP to ammonium ($\text{DOP} \Rightarrow \text{NH}_4^+$). These two processes occur through the decay of plants and animals mediated by heterotrophic bacteria or through excretion from animals. They occur both in sediments and in the water column although in sediments in shallow coastal environments, a larger proportion of the mineralisation takes place as particles settle more rapidly to the substratum.
- **Nitrification:** Autotrophic bacteria oxidise ammonium to nitrate via nitrite ($\text{NH}_4^+ \Rightarrow \text{NO}_2^- \Rightarrow \text{NO}_3^-$) such that nitrification is an oxygen requiring process which can occur in both the water column and oxygenated surface regions of sediments. **De-nitrification:** This involves the conversion of nitrate to gaseous forms of nitrogen ($\text{NO}_3^- \Rightarrow \text{N}_2$ or N_2O). This is carried out under anoxic conditions by heterotrophic micro-organisms which utilise nitrate as an electron acceptor in the oxidation of organic matter. The associated reactions result in the production N_2 in anoxic conditions, or N_2O in the presence of a small amount of oxygen. These gaseous forms are then unavailable to primary producers. In addition to hypoxic/anoxic conditions, large amounts of organic matter are required to fuel this process. As reducing conditions are prevalent in coastal and estuarine sediments anaerobic processes such as de-nitrification are important in these areas (Barnes and Mann 1991). Nitrification and de-nitrification can become tightly coupled in the sediment because the utilisation of oxygen during nitrification can lead to the anoxic conditions required to promote de-nitrification (Harding 1992).

2.4.2 The Phosphorus Cycle

Unlike the nitrogen cycle, the transformations and translocations of phosphorus are also mediated by physico-chemical factors as well as biological processes. This is because phosphorus is not only used by autotrophic organisms but phosphate ions also react with the metal oxides and clay minerals on the surface of sedimentary particles. Tait (1981) describes the effects of biological processes in the cycling of the phosphorus and identifies the pathways of the cycle in marine waters where the impact from sediment interactions is negligible. Figure 3 shows how biological processes convert phosphorus species between the following four states:

- dissolved inorganic phosphorus (DIP);
- dissolved organic phosphate (DOP);
- particulate organic phosphorus (POP);
- a component of living organisms (LOP).

Phosphorus is usually present as the form orthophosphate (PO_4^{3-}) either bound to a cation in insoluble inorganic compounds or as a component of organic molecules (O'Neill 1985). The following major processes are involved in the biological cycling of phosphorus between these organic and inorganic states:

- **Assimilation:** Phosphorus species are absorbed by plant species and incorporated into larger organic molecules. Most of the phosphorus that is assimilated by autotrophs is phosphate, although to some extent plants can also absorb certain dissolved organic phosphorus compounds ($\text{PO}_4^{3-} \Rightarrow \text{LOP}$) (Tait 1981).
- **Release:** Following the death of marine organisms much of the phosphorus in their tissues returns very quickly to the water mainly as phosphate, indicating that decomposition of organic phosphorus compounds is probably largely by autolysis and hydrolysis ($\text{LOP} \Rightarrow \text{Free POP, DOP and PO}_4^{3-}$).
- **Remineralisation:** Released particulate organic phosphorus is acted on by bacteria producing various solutes, *e.g.* glucose-phosphate, glycerophosphate, adenosine phosphate, some of which may be used by plants *via* their phosphatase enzymes, or further degraded by bacteria to phosphate ($\text{Free POP, DOP} \Rightarrow \text{PO}_4^{3-}$).

As discussed previously, physico-chemical factors act directly on the transformations of phosphorus as phosphate reacts with the metal oxides on particulates. Conley *et al.*, (1995) suggests that this process apparently occurs in the following two phases:

- initially there is a rapid adsorption of DIP onto metal oxides (occurring in days);
- secondly there is a longer term diagenetic change to a solid phase (over weeks to years).

This particle-reactivity of phosphorus means that the cycling of this nutrient is influenced by the sedimentary dynamics of estuaries. This process is particularly significant in turbid estuaries where there is both a greater surface area of sediments for phosphorus-sediment exchange and where, because light penetration is also reduced, the influence of autotrophic responses such as phytoplanktonic blooms on the cycling of phosphorus is restricted (Conley *et al.*, 1995).

The effect of suspended particles on phosphorus is often particularly important at the highly turbid Freshwater/Seawater Interface (FSI) region of estuaries. In areas where sedimentary deposition is responsible for the depletion of water column phosphorus concentrations then it is likely that a significant amount of phosphorus may be confined within the turbidity maximum region. This occurs for example in the St Lawrence (Lucotte and d'Anglejan 1988 in Conley *et al.*, 1995) and in the Ems Estuary (van Beusekom and de Jonge 1995).

The reactions between phosphate and suspended particulates are influenced by salinity levels. Low salinity levels favour the adsorption of phosphorus to metal oxides and increasing salinities and pH favour desorption processes as sea water anions, *e.g.* sulphates compete for adsorption sites (Froelich, 1988 cited in Balls 1992). The initial increase in salinity at the low salinity region (<5psu) of the upper Öre estuary was found to be the major cause of phosphorus adsorption. This aggregation to sediments and the subsequent rapid sedimentation resulted in the complete removal of phosphorus from the water column in this system (Forsgren and Jansson 1992).

As salinity increases further there may a desorption of phosphate and an increase in the concentration of DIP. This process was believed to be the reason observed mid-estuary peaks of phosphorus in the Tay and Forth estuaries (Balls 1992). As the release of phosphate from sediments is a slow and only approaches equilibrium on the timescale of days to weeks, Balls

(1992) notes that an increase phosphate water column concentrations would be most likely occur in slow flushing estuaries.

The reactions between particles and phosphates are also affected by the dissolved oxygen concentration. Under low oxygen conditions phosphorus species become desorbed from sedimentary particles such that changes in oxygen concentrations arising from biological process of respiration and nitrification can have significant effects on the sediment/water fluxes of phosphorus. The depletion of DO in bottom waters has been shown to coincide with high DIP in Chesapeake Bay (Conley *et al.*, 1995). Parr and Wheeler (1995) observed that, because of the relationship between DO and DIP release, phosphate release from sediment is enhanced in stratified systems where the bottom waters which are more susceptible to oxygen depletion. Whilst there can be phosphorus release from sediments due to the reduction and dissolution of metal oxide phases the process is rapidly reversed on encountering an oxic environment. This implies that in practice there is little net release of phosphorus from sediments via this mechanism.

Zwolsman (1994) described the combined effects of changing salinity, pH and dissolved oxygen (DO) level on the concentration of phosphorus along the length of the Scheldt estuary. It was found that there were increases in phosphorus in the low salinity region of the estuary where the anoxic conditions resulted in a release of phosphorus from the particles. In the summer further down the estuary there is a rapid removal of phosphates due to its co-precipitation with iron oxyhydroxides. With increasing pH down the estuary there is a desorption of phosphate from particles which results in a buffering of the phosphorus concentration during the winter months. In summer, however, this buffering capacity may be obscured by biological uptake.

In addition to the effects of salinity and DO, Franklin *et al.*, (1991) note that temperature can affect the adsorption of phosphate to particulates with higher temperatures favouring greater levels of adsorption. However, in their study of DIP level in the Itchen, they found no evidence for greater levels of DIP removal in summer months than winter which was expected given the clear temperature differences.

Studies in the Ems estuary indicate an alternative mechanism by which phosphorus can be released to estuarine waters (de Jonge and Villerius 1989). It is suggested that calcite acts as a carrier for adsorbed phosphate and that phosphate adsorbs to calcite in the most saline regions of the Ems estuary which is transported up the estuary during summer. On encountering the reduced pH associated with low salinity the calcite then dissolves and the absorbed phosphate is released.

Franklin *et al.*, (1991) reviewed the factors affecting DOP and DIP concentrations in estuaries and found that DIP concentrations varied greatly due to the differences in the levels of anthropogenic inputs from sewage treatment plants. They proposed that the following processes bring about a reduction DIP concentrations:

- adsorption to particulates;
- association with flocculated iron;
- biological uptake;
- physical transport (considered as potentially important);

- cycling in association with calcite (considered as potentially important).

Franklin *et al.*, (1991) also found that DOP levels were very similar in all the estuaries and that increases to the DOP concentrations were brought about by anthropogenic inputs and by *in situ* biological production. The processes which deplete DOP levels however, were not clearly identified, but were believed to include:

- biological uptake;
- physico-chemical factors such as adsorption to particulates;
- oxidation to DIP.

2.4.3 The Silica Cycle

In contrast to nitrogen and phosphorus, silicon, which is present in seawater chiefly as silicate (SiO_4), is derived from the weathering of terrigenous rocks and is not an anthropogenically introduced element. Silicate exhibits no specific chemical behaviour and its cycling occurs through assimilation and regeneration exclusively by diatoms because it is an important constituent of the frustule of these species. The concentration of dissolved silicate (DSi) has been shown to control the magnitude of diatom production during the spring bloom and depleted DSi levels prompt collapse of these blooms (Conley and Malone 1992). The collapse and deposition of the spring bloom can then bring about a flux of silicates to the benthos (Conley and Malone 1992).

Following the death of diatom species the remineralisation of silicon occurs through a process of dissolution. This is a relatively slow process so that in shallow waters the bulk of silicon mineralisation occurs in the sediments after the diatoms have been deposited on the substratum.

2.4.4 Comparison of Cycles

The previous sections have illustrated the processes involved in the cycling of nitrogen, phosphorus and silicon. From this review it is possible to identify the following major differences between the cycles:

- All three nutrients are assimilated by autotrophic organisms although nitrogen and phosphorus are taken up by all species, silicon is specifically taken up by diatom species as it is an essential component of their frustules.
- The remineralisation processes of the three nutrients are different. Nitrogen regeneration occurs entirely through heterotrophic grazing and bacterial respiration. These two processes also mediate the remineralisation of phosphorus, but phosphorus release this can also occur through cell autolysis. The release of silicon occurs through the pH-dependant dissolution of diatom frustules.
- The release of silicon occurs at a much slower rate than nitrogen and phosphorus so, both nitrogen and phosphorus will be recycled faster and reused on shorter timescales than Si. Phosphorus exhibits the fastest cycling rate because its remineralisation can occur through cell autolysis and unlike nitrogen it is not dependent solely upon biological processes.

- There are differences in the fate of these nutrients as silicon and phosphorus entering the system can either be exported to the open sea or retained within the system through either deposition in estuarine sediment or incorporation into biota. For nitrogen there is a third fate which is release into the atmosphere due to the effects of bacterial de-nitrification which produces gaseous nitrogenous compounds.

2.4.5 Bacterial Activity

Two important factors affecting bacteria activity are the oxygen concentration and the sedimentary dynamics. Sedimentary dynamics affect bacterial activity because the majority of bacteria survive attached to the surfaces of particulates such that the highest bacterial activity is in the sediments and in turbid waters (Owens, 1986; Balls, 1992). Bacterial activity is also high in turbid regions because the lower light levels under such conditions reduces the growth of autotrophic organisms and increases the availability of nutrients.

Oxygen concentration dictates the location and characteristics of the bacterial processes because bacterial processes such as nitrification are oxygen-requiring and can only be carried in the water column and surface oxygenated layers of the sediment. Alternatively, processes such as de-nitrification and nitrogen fixation are anaerobic and are, therefore, largely confined to the oxygen-depleted layers of the sediment.

Due to these two factors Balls (1992) notes that bacterial processes such as the mineralisation of Particulate Organic Nitrogen (PON) to ammonia and subsequent nitrification to nitrate is highest in turbid maxima regions of estuaries (Balls 1992). Balls (1992) concluded that this was the reason why there was a higher input of nitrate to the turbid Forth estuary (up to 600 mg/l in the turbidity maximum in summer) as compared to the relatively clear Tay estuary (<100 mg/l). Balls (1992) also notes that because extent of bacterial activities is dependent on the surface area of particulates the Biological Oxygen Demand (BOD) is a linear function of the SPM (Suspended Particulate Matter).

In sediments, bacterial activity is influenced by the particle size that affects both the surface area available for bacterial growth as well as the permeability of oxygen. Fine sediments which have a relatively high percentage surface area for the attachment of bacteria which means that bacterial activity is greater but also that the penetration of oxygen is low so that processes which occur in the absence of oxygen (such as de-nitrification) are prevalent.

The effect of sedimentary composition on transformations of nitrogen was demonstrated by Ogilvie *et al.*, (in prep.) in the Colne estuary. Even though this is a macrotidal system with a relatively short flushing time (0.9 days), there was significant attenuation of nitrogen due to bacterial processes (ammonification, nitrification and de-nitrification) in the muddy and anoxic sediments of the upper estuary. These processes were insignificant in the water column and in the lower estuary, where the sediments were coarser and more aerated and where an export, rather than an import, of nitrates were observed. They concluded that, due to the nitrifying and denitrifying activities of bacteria, sediments are major attenuators of nitrate flux in turbid muddy estuaries.

Another aspect of the sedimentary dynamics which affects the bacterial activities is the degree to which the surface sediments are reworked. This is noted Nixon (1988) who

observes that where currents produce a high level of surface sediment disturbance, there is in a greater level of bacterial activity and a greater consumption of organic matter.

In addition to the effect of sediment dynamics, bacterial activity is also influenced by temperature (Nixon, 1988; Balls, 1992; Parr and Wheeler 1995). Higher temperatures produce higher levels of bacterial activity and with every 8-10°C rise in temperature cause a doubling of bacterial growth (Parr & Wheeler 1995). Therefore with increased temperature, Nixon (1988) noted that the levels of nitrogen mineralisation and the consumption of organic matter increase and, as Parr and Wheeler (1995) observe, organic nitrogen becomes more important in determining algal productivity.

Balls (1994) identifies a relationship between the mechanisms which drive turbidity levels in estuaries and the potential impact of increased temperatures on bacterial activity. For example, in the Tay where wind induced resuspension was shown to be the most important process governing turbidity therefore turbidity levels is generally highest in winter when temperatures are lower. In the Forth however, turbidity is highest in the summer during low freshwater flow conditions and there is a coupling between high temperatures and high turbidity and therefore, high bacterial numbers. This coupling promotes high levels of bacterial mineralisation and an increased risk of oxygen depletion.

Parr and Wheeler (1995) conclude that, because of the effects of temperature on bacteria, bacteria-mediated processes are likely to be more important in the shallow subtidal and intertidal regions and in estuaries characterised by these habitats because they are generally warmer. Other studies have also observed a link between water depth and the relative contributions of benthic and pelagic bacterial processes to the cycling of nutrients. Nedwell and Trimmer (1996) for instance identified a relationship between water depth and the extent of de-nitrification. They showed that because the level of de-nitrification, which occurs exclusively in the bottom sediments, is related to the area of sediment as well as the availability of nitrate in the overlying water.

Nedwell and Trimmer (1996) demonstrated that the significance of de-nitrification as a sink for nitrogen in the Wash is dependant upon on the relationship between the sediment surface area and the flushing time. For example, they found that in the upper region of the Great Ouse, where there is a small surface area in relation to the freshwater flow, the importance of de-nitrification is small.

Furthermore, Kemp *et al.*, (1992) found a strong relationship between water depth and the contributions of benthic and pelagic processes to oxygen consumption in Chesapeake Bay. They showed that at depth greater than 5m or in the bottom layers of stratified water columns which are thicker than 5m, planktonic respiration is the major cause of oxygen depletion. At depths of less than 5m however, benthic respiration was found to be more important.

2.5 Biological Responses

This section reviews the factors affecting biological responses to nutrients and highlights the influences of the physico-chemical factors in estuaries. Section 3 details the effects of physico-chemical factors on specific taxonomic groups and the response of these individual taxa to nutrients.

Freshwater discharge is a physico-chemical factor with a major influence on the response of estuarine species to nutrients. It not only influences the rate of nutrient inputs but also the environmental conditions in estuaries and the potential for communities to respond to the available nutrients by changing characteristics such as the flushing time, turbidity levels and the density stratification of the water column (the individual effects of these are discussed below). Several studies have shown correlations between rates of river discharge and observed biological responses in estuaries and coastal waters over varying time scales. For example, daily variations in the plankton community of a shallow coastal embayment have been linked to the increased input of nutrients from estuaries following rainfall events (Garcia-Soto *et al.*, 1990 cited in Balls 1992). Other studies have identified freshwater flow as a key influence in the changes in phytoplankton growth on a seasonal scale, and in particular have shown relationships between the timing of the spring increase in freshwater flow ('freshet') and winter/spring blooms of phytoplankton. In South San Francisco Bay Estuary, Cloern (1991) showed a strong correlation between the scale of the annual spring increases in phytoplankton biomass and the magnitude of the river flow. This link was believed to be largely due to the development of a stratified layer in the water column that is brought about by the spring increase in freshwater run-off. In the Wadden Sea similar correlations have been recorded between freshwater discharge, phosphate loads and annual primary production (de Jonge and Essink 1991).

Harding (1994) also found that subtle inter-annual differences in the abundance of phytoplankton during spring blooms were related to the magnitude of the spring freshet in Chesapeake Bay. He proposed that phytoplankton abundance is affected by the complex interplay between the effects of light and nutrient levels under different freshwater flow conditions. For instance under conditions of high freshwater flow there is a high nutrient load which should promote a high level of phytoplanktonic growth. However, under these conditions, the suspended particulate load is also high and this reduces light penetration and creates conditions that are less conducive to phytoplankton growth. The counteracting effects of nutrient loads and turbidity levels means the relationship between freshwater flow rate and phytoplankton abundance is not linear. This means that the largest blooms do not necessarily occur under the highest freshwater flow conditions, but instead occur with lower freshwater flows where the nutrient load is still sufficient to promote a bloom and where turbidity is low. Harding (1994) also notes that the influence of freshwater flow on turbidity affects the position of the blooms within the estuary with blooms occurring further down the estuary, when there is a high freshwater flow and a high turbidity in the upper estuary. In large estuarine systems a lag can be expected between the freshwater flow and the response of phytoplankton. For instance, Malone *et al.*, (1988) observed a direct correlation between monthly mean freshwater flows and the monthly mean chlorophyll levels in the mesohaline region of Chesapeake Bay but there was a lag of one month between these two factors.

In addition to the effects of freshwater flow on the responses of phytoplankton, correlations have also been obtained between river discharges and other biological responses that are secondary consequences of changing levels in phytoplankton productivity. Kemp *et al.*,

(1992) for instance, observed that inter-annual variations in the input of freshwater to Chesapeake Bay promote corresponding changes in the severity of seasonal hypoxia in the bottom waters. They found that the level of bottom water oxygen depletion is dictated by the rates of respiration and oxygen replenishment by physical transport and these two processes were shown to be coupled. This coupling occurs because biological respiration is partly responsible for driving the physical transport of oxygen through the creation of an oxygen gradient. Under conditions when freshwater inputs are low, there are correspondingly low levels of nutrient-generated algal biomass which means that less organic matter is available to fuel bottom water respiration. The coupling causes physical oxygen replenishment to also be reduced and consequently less oxygen reaches the bottom waters to drive consumption.

Josefson (1990) examined biological responses further up the trophic chain, and obtained evidence for a link between the annual mean river water flow in western Sweden and Denmark, and the mean biomass of infaunal species in the Skagerrak. He found the best linear correlation between these two factors was obtained when a time lag of two years was incorporated into the comparison, so that benthic biomass corresponded best with the river water flow rates that occurred two years previously. This two year lag was considered to be the amount of time required for nutrient inputs to increase benthic biomass through the processes of plankton production, and sedimentation, the enrichment of the benthos, and finally, the recruitment and growth of benthic animals.

There is also evidence that the timing of changes to the freshwater input influences the biological response. In Chesapeake Bay Harding (1994) observed that periods of high freshwater flow could occur too early in the season to support a phytoplankton bloom. He proposed that the 'memory' of the estuary did not extend freshwater events that occurred more than 4 months prior to the period from March to May when blooms usually occurred. There can also be a difference between the biological responses that occur during the spring and summer months. This was demonstrated again by Harding (1994) who showed that while there was a relationship between freshwater flow and phytoplanktonic blooms during the spring freshet there was no such relationship during the summer. The summer blooms were shown to be out of phase with freshwater flow and this was believed to be because they are driven by recycled autochthonous nitrogen rather than riverine inputs.

According to Malone (pers. comm.) the timing of the spring peak in freshwater flow also has an important effect on the overall level of enrichment in Chesapeake Bay. Typically the spring freshet results in large diatom blooms and the large scale accumulation of phytoplankton biomass and attendant problems of organic loading. During the summer bloom however, there is a biomass-specific increase in productivity which causes less adverse impacts to the system because it was more readily transferred through the food web. Consequently, when the freshet occurred 1-2 months late as it did in 1989 in Chesapeake Bay, then the allochthonous nutrient inputs support the summer bloom and under these conditions the problems associated with over-enrichment were less severe than they are during a typical year (see Section 3.2.1 for a further details of this study).

Stratification is often an important factor in the formation of phytoplankton blooms (Cloern 1991 Koseff *et al.*, 1993; Hickel *et al.*, 1993). Hickel *et al.*, (1993) for instance showed that the start of phytoplankton growth in the Elbe estuary was not related to the nutrient loads in the River Elbe, but that the dominant factor in the instigation of blooms was the occurrence of a stratified layer. Koseff *et al.*, (1993) and Cloern (1991) also demonstrated that water column stratification prompts the development of blooms in South San Francisco Bay. This

occurs because, under stratified conditions, phytoplanktonic cells are confined to the illuminated surface waters and are exposed to higher light levels. Another reason identified by Cloern (1991) is that the confinement of algal cells to the surface layers of the water column removes the phytoplankton from contact with the substratum and therefore reduces the grazing pressure of benthic suspension feeders.

In direct contrast to these observations, there is also evidence that water column stratification can in certain cases reduce phytoplankton development by inhibiting the release of nutrients from bottom waters to the illuminated surface waters. This is seen in Chesapeake Bay in the summer where the benthos is a principal source of nutrients to surface waters and the vertical flux is inversely proportional to the strength of the density stratification Malone (in prep.). Consequently an inverse relationship was found between summer productivity and the mean density gradient across the pycnocline.

The major risk to communities from the formation of a stratified layer is that sub-pycnocline water is susceptible to oxygen depletion. This occurs because of the reduced circulation of oxygenated water, the reduced supply of oxygen and oxygen removal by biological respiration in the bottom waters. The biological processes are both accelerated by the warming waters and the extra supply of organic matter from the settlement of phytoplankton blooms Kemp *et al.*, (1990). The development of a strong halocline and the accompanying depletion of oxygen is seen as the principle manifestation of eutrophication problems in several regions *e.g.* the Kattegat (Baden 1990). The impacts of oxygen depletion are discussed in greater detail in Sections 3.3 and 3.4.

The tidal regime within an estuary also influences biological response. The strength of tidal-driven water circulation can influence the stability of the sediments and the level of water column turbidity and therefore, affect the potential for growth of benthic and pelagic algae. It has been shown for instance, that tidally energetic estuaries limit the development of phytoplanktonic blooms. The relationship between tidal strength and the responses of phytoplankton was studied by Monbet (1992) who compared inorganic nitrogen and chlorophyll-*a* concentration in 40 microtidal and macrotidal estuaries. This study showed that macrotidal systems which are characterised by high tidal energy, generally exhibited lower chlorophyll-*a* concentrations (lower phytoplankton abundance) than systems with lower tidal energy. Villate *et al.*, (1991) also found that, in the five Spanish estuaries which they investigated, a high tidal exchange (spring-tides) corresponded with low level of phytoplanktonic activity.

Current strength can also influence stability and sedimentary composition of the substratum which in turn influences the potential for the development of benthic macroalgal and microalgal mats. Some macroalgae require a fixed stable surfaces for attachment, while benthic diatoms, other microflora and some opportunistic macroalgae are capable of colonising softer substrata (Parr and Wheeler 1995). Sedimentary characteristics such as: grain size, sediment stability, and the amount of light penetration all affect the stability structure and thickness of the microbial mat community Sundback (1994). In coarse, non-cohesive sediments stable microbial mats can form whereas on muddy cohesive sediments transient 'biofilms' occur. Also, in the coarser sediments, the gross primary production has been found to be greater than in the cohesive sediments because the light attenuation is less severe (Yallop *et al.*, 1994). The disturbance of sediments by currents can also lead to the suspension of microphytobenthos which can then become a component of the pelagic algae. Under these conditions, the contribution of the microphytobenthos to the primary

productivity of the estuary and the role of this community in the utilisation of nutrients, can be very difficult to assess (de Jonge pers comm.). Recent work (Underwood pers comm) has highlighted the importance of micrphytobenthic primary production as a component of the overall production in shallow estuaries.

In Section 2.3 the importance of flushing time in the retention of nutrient was identified and the physical characteristics of estuaries which determine the flushing rate were described. It was also shown that with longer flushing times nutrients are available to communities for longer which means that the potential for a biological response is greater. This relationship is described by Conley *et al.*, (1995) who identified a clear correlation between flushing time, the size of phytoplankton blooms and the effects of these blooms on the dissolved inorganic phosphorus (DIP) concentration. In this study it was shown that estuaries with short residence times had small planktonic populations and as a result DIP remained relatively high throughout the estuary. In estuaries with a long residence time however, where there were large planktonic blooms, the DIP may be lowered by biological uptake.

Flushing time also influences the relative responses of benthic and pelagic algae to nutrients. In rapidly flushing estuaries for instance there may be insufficient time available for the development of phytoplanktonic blooms. In such cases, and if other conditions such as light availability are favourable, then benthic vegetation is more likely to utilise available nutrients. This appears to be the situation in the Ythan where there has been a long-term increase in nutrient loadings. This resulted in an abundant growth of *Enteromorpha* sp. but no corresponding increase in phytoplankton (chlorophyll-*a* levels) due to the rapid flushing time and/or the strong tidal regime in this estuary (Balls *et al.*, 1995).

In considering the relationship between the response of phytoplankton and flushing time however, it is necessary to consider the entire geomorphology of the estuary rather than the overall flushing time. This has been pointed out by Parr and Wheeler (1995) who noted that there may be regions within individual estuaries where there are different flow rates and that in the slow-flushing embayments plankton may grow and then 'inoculate' the remaining major part of the estuary with a population of phytoplankton.

The turbidity levels in an estuary affect community responses by influencing the light available to photosynthetic organisms. In some estuaries it is light rather than nutrients that limit the growth of primary producers, for instance, in the shallow Wadden Sea the ultimate effects of eutrophication are also strongly affected by the turbidity levels (de Jonge and Bakker 1994).

The growth of phytoplankton is controlled by the degree of light penetration through the water column (the photic depth) and the mixing depth of the water column which together determine the overall exposure of phytoplanktonic cells to light. For any given level of turbidity, the maximum depth to which plankton can be mixed and still have sufficient light for net photosynthesis is referred to as the 'critical depth' (Grobbelaar, 1985). In estuaries which are shallower than the critical depth there is scope for phytoplanktonic growth, whereas in estuaries that are deeper than the critical depth there can be no growth of phytoplankton (Parr and Wheeler, 1995). However, as described previously, in deep systems water column stratification can reduce the critical depth and therefore promote phytoplanktonic blooms. Fichez *et al.*, (1992) observed phytoplankton growth in the turbid Great Ouse estuary and concluded that phytoplankton development is possible in turbid and well-mixed estuaries as long as the critical depth is greater than water depth. Other factors

which are found to affect phytoplankton development include the daily sunlight exposure, salinity stress and nutrient availability.

Turbidity levels, as well as estuary bathymetry, also influence the relative responses of benthic and pelagic algal groups. Phytoplankton can still thrive in turbid environments because they have an ability to absorb light at lower light compensation depth than macrophytes which enables them to compete more effectively for available nutrients in deeper waters (Duarte 1995). In contrast benthic algae, which can survive and photosynthesise in the intertidal regions where they are periodically exposed to direct sunlight, have a competitive advantage in turbid systems which have a large intertidal area. In the tidally dynamic and highly turbid Chignecto Bay (Canada) for example, light penetration is reduced to 2m and primary productivity is low and principally carried out by salt marshes and intertidal microphytobenthos, rather than phytoplankton (Keizer and Gordon 1985).

2.5.1 Temporal and Spatial Variability of Responses

Many, if not all, of the factors which affect biological responses to nutrients are subject to temporal variations in response to short term (days or weeks) or the long term (months or seasons) climatic changes. Seasonal changes in climatic conditions for instance have a major influence on rates of: catchment drainage, freshwater flow, nutrient input, flushing times and water circulation patterns which results in changes to water column stratification, levels of turbidity and characteristically produces seasonal cycles of sedimentary erosion and deposition.

Periods of calm weather and/or bright conditions are often essential prerequisites to estuarine bloom formation and are responsible for the occurrence of seasonal blooms. Cloern (1991) considered that local weather conditions were a secondary mechanism contributing to the variability of phytoplankton biomass and production at a seasonal time scale and that freshwater flow is the primary mechanism (see above). Cloern (1991) demonstrated that seasonal and interannual rates of precipitation and run-off have a large influence on the stratification of the water column in the South San Francisco Bay estuary which, in turn, affects the pattern of phytoplankton dynamics.

However, even over much shorter timescales (daily, hourly) the weather conditions can change to affect the degree of light penetration and the associated photic and critical depths. Insulation plays an important role in determining both the light available to primary producers and also in regulating water temperature and promoting the stratification of the water column. Wind strength can be highly significant in determining the details of the water circulation and associated factors such as turbidity.

Pedersen *et al.*, (1995) found that wind induced turbulence exerts a major control on the productivity of shallow aquatic ecosystems. He observed an increase in the concentration of nutrients in a Danish Fjord in response to a persistent SW wind. This was caused by the resuspension of sediment and a greater sediment loss of nitrogen and phosphorus than would normally be observed from a standard rate of 'benthic diffusive flux'.

Perdon (1995) also considered that the wind strength and the degree of exposure of the estuary was of particular importance for shallow estuaries where high winds can markedly affect the degree of water column mixing. Wind-induced circulation of the water column

also dictates the extent to which phytoplankton are retained within the euphotic zone of deeper aquatic systems. Wind speed influences stratification and will affect the mixing depth. Koseff *et al.*, (1993) found that in shallow turbid estuaries with populations of benthic suspension feeders, stratification itself is not sufficient to ensure that a bloom will develop, minimal wind stirring was believed to be a further prerequisite. With increased sediment resuspension there is also a resuspension microphytobenthos and in the Ems estuary microphytobenthos resuspension was found to be a linear function of the wind stress (de Jonge van and Beusekom 1992).

Kemp *et al.*, (1992) also observed a noticeable effect of wind conditions in the long term (over decades) temporal and spatial dimension of the oxygen depletion within the bottom waters of Chesapeake Bay. They found that changing wind conditions alter the relative proportions of the stratified layers in Chesapeake Bay by varying the orientation of the pycnocline and this influences the scale of the impact from hypoxia.

Temperature influences the rate of biological activity temperature and regulates photosynthesis (Hobson 1985) and bacterial processes (see Section 2.4.5). Due to the shallowness of estuaries and the varied temperatures of freshwater inputs temperature regimes within estuaries are often more variable than adjacent waters.

As well as the temporal changes to physico-chemical factors there are often marked spatial changes within estuaries which can have a major effect the biological response to nutrients. The most notable change occurs in the salinity levels and the seaward increase in salinity in estuaries has a major influence on the distribution of species and, therefore, on the characteristics of the biological response to nutrients (a review of the effects of salinity on estuarine communities is contained in Appendix 4). Turbidity levels can also show significant changes along the length of an estuary (described above) as can other natural factors such as water circulation, temperature and pH.

The biological responses along the length of an estuary will also be influenced by the changing concentration and composition of nutrients which are altered by physico-chemical processes such as phosphate-particulate interactions as well as by biological activities such as autotrophic uptake and bacterial transformations (this is discussed in greater detail in Section 3).

There is now a widely held perception that freshwaters are generally phosphorus limited and that marine waters are typically nitrogen limited (Boynton *et al.*, 1982; Taylor 1995a) and that along the length of an estuary there is often a change in the limiting nutrient. However, the pattern of nutrient changes along estuary gradients will vary greatly between different estuaries due to differences in the physical characteristics and Taylor (1995a) identifies two estuary types which are exceptions to this rule:

- temperate estuaries and bays which receive large seasonal inflows of freshwater which show seasonal changes in the limiting nutrient;
- marine bays with long residence times and minimal freshwater inflows where phosphorus is apparently limiting.

In addition, anthropogenic factors such as the location of mid-estuary nutrient inputs can also have an effect on the changing balance of nutrients. Fisher *et al.*, (1988) examined spatial changes in the relationship between physico-chemical factors, biological responses and

nutrient levels along estuaries. They developed a conceptual model (Figure 4) of spatial changes in estuaries based on studies of phytoplankton, nutrient and turbidity levels in Chesapeake, Delaware and Hudson Bays.

According to this model, in the upper estuary, the first feature is the turbidity maximum, here there is a net heterotrophy as primary productivity is suppressed by limited light availability. Downstream of this area a chlorophyll maximum may occur as light ceases to be limiting. The key to the development of this chlorophyll maximum and the consumption of riverine nutrients is stable stratification to restrict the depth of mixing. Within the chlorophyll maximum there is a removal of nitrate, phosphate and silicate by the phytoplankton. Fisher *et al.*, (1988) speculated that a zooplankton maximum could occur downstream of the chlorophyll-*a* maximum in very large estuaries with long residence times. The authors believed that such a zooplankton area was present in Chesapeake Bay from the evidence of an Autumn survey which showed a limited accumulation of phytoplankton despite a low runoff, low turbidity and yet high nutrients. The scale of this spatial gradient is dependent on the freshwater discharge, morphology and flushing time of the estuary. It is important to note that this model was based on information from three large, deep slow flushing systems where the spatial changes are most extreme. In small faster flushing estuaries there are likely to be fewer zones because the residence time for nutrients is reduced and less time is available for biological utilisation.

2.6 Conclusions

Nutrient Inputs to Estuaries

1. The review has reviewed information from single and comparative studies and highlights the difficulties of making comparisons between studies.
2. Many studies consider the point source and diffuse inputs and indicate the net effect of the climatic-hydrographic characteristics of the area, especially the geomorphology and the catchment size.
3. More than many other features, the nutrient inputs have a high seasonal variability especially due to freshwater flows and catchment rainfall.
4. There is conflicting evidence of the importance of atmospheric inputs although they are likely to increase in importance.
5. The uses and users of the catchment, especially with respect to agriculture and forestry, will have a large effect on the nutrient inputs, especially the separation of hypernutrified estuaries from oligotrophic ones.
6. The management practices in the catchments, especially for agriculture, will influence inputs.

Nutrient Retention in Estuaries

1. Nutrient retention is primarily driven by physical processes although the resultant concentrations are the net affect of these coupled with internal kinetics.
2. The flushing time as determined by the estuary's size and bathymetry and the water exchange from each end of the estuary (freshwater flow and tidal prism) dictates the amount of time that nutrients are retained.
3. In addition, the production of stratified conditions, due to physical barriers such as a sill or due to the bathymetry, will influence retention and mixing between surface and deeper waters.
4. Finally, each of these factors in combination will influence the lateral and well as axial gradients.

Nutrient Transformations

1. The previous studies have indicated the good conceptual knowledge of transformation processes and nutrient chemistry although there are relatively poor quantitative data.

2. The importance of the autotrophic (both macro- and micro-) organisms and heterotrophic bacteria is illustrated as giving many of the processes; those processes are biological-mediated physico-chemical conditions.
3. The studies illustrate the importance and high dependence on surfaces for controlling the kinetics whereby those surfaces may be as suspended particles or bed surfaces. The kinetics are related to the patterns of the major forcing variables (salinity, temperature, pH, dissolved oxygen, and turbidity) especially as uptake and release processes.
4. It is concluded that the physico-chemical patterns create suitable conditions for the autotrophs to use nutrients and then biological interactions and cycles act; in contrast the heterotrophic processes are mainly dependent on the physico-chemical processes. The mechanisms affecting suspended solids levels and the types of bed surface will in turn dictate the strength of many of the transformations.

Biological Responses

1. The response by estuaries revolve around the delivery of nutrients and the time available for those nutrients to be used. There appears to be an adequate conceptual understanding of the processes for any system studied although there are poorly quantified processes which can be extrapolated between estuaries.
2. The conceptual basis indicates the main responses to flushing time, stratification, turbidity, and tidal and current strength although the temporal and spatial response to such changes is poorly defined.
3. The completed studies reinforce many of the consistent features between estuaries in their nutrient response to changes in physico-chemical factors, for example the controlling influence of turbidity and its link to hydrographic and bathymetric features.
4. However, some of the responses appear to be estuary-specific and thus the effect of a particular combination of conditions. For example, while in some estuaries, increased river flow delivers increased suspended solids whereby the higher turbidity controls primary production whereas in other systems, increased river flows deliver higher nutrients.
5. Some of the responses appear critical in the timing of the forcing factors, for example the delivery of nutrients via freshwaters has an optimal period in the spring but outside this period will result in unpredictable changes in the phytoplanktonic system.
6. The response time to inputs of nutrients requires further quantification but cases suggest that, depending on the size of the estuary and its ability to retain the nutrients, it could be up to two years with an eventual effect on the benthos.
7. On a spatial scale, the differing areas within the estuary respond differently to nutrient inputs and retention. For example, the autotrophy:heterotrophy ratio differs with proximity to the turbidity maximum.

8. It is concluded here that while the biological response to nutrient inputs may be predicted qualitatively from a good knowledge of the physico-chemical factors, few areas are understood sufficiently well to provide a quantitative prediction.
9. Furthermore, the wide variability between estuarine systems will make further generalisations across areas more difficult.

3. BIOLOGICAL IMPACT OF NUTRIENTS

Section Aims

- To review the effects of nutrients factors on communities and habitats in estuaries.
- To review the effects of communities on nutrient dynamics.

Section Contents

Section 3.1 Introduction:

A brief description of the characteristics of ecological impacts from nutrient enrichment and of the organisation of this section.

Section 3.2 Autotrophic Groups and Bacteria:

A review of the effects of nutrients on the responses of autotrophic taxa and bacteria as well as the effects that these organisms have on nutrient dynamics. This review is divided into:

- Four separate reviews of the responses to, and effects on, nutrients of the following autotrophic groups: Phytoplankton, Microphytobenthos, Macroalgae and Vascular Plants (Sections 3.2.1 to 3.2.4 respectively).
- A review of the comparative responses by different autotrophic taxa (Section 3.2.5).
- A review of the effects of enrichment on bacterial processes (Section 3.2.6).

Section 3.3 Benthic Communities:

Reviews of the effects of organic enrichment and oxygen depletion enrichment on benthic communities (Section 3.3.1) and of effect of benthic fauna on nutrient cycling (Section 3.3.2)

Section 3.4 Higher Trophic Levels/Food Webs:

A review of the effects of enrichment on the top trophic consumers and food webs including discussions of the role of phytoplankton in food webs, the effects on waterfowl and fish and changes to trophic interactions within sediments.

Section 3.5 Recovery from enrichment impacts:

A discussion of the potential recovery of ecosystems from the impacts of enrichment

Section 3.6 Summary and Conclusions:

3.1 Introduction – Characteristics of Nutrient Enrichment

The enrichment of aquatic systems with plant nutrients can result in ecological impacts at a range of scales from changes in the abundance of certain species to fundamental alterations in the entire trophic structure of an ecosystem. To describe how these impacts can occur, the process of nutrient enrichment can be divided at its most general into the following three stages.

- The primary impacts are upon bacteria and plants which directly utilise the imported inorganic nutrient. The initial impacts are the increased growth of plant if currently nutrient-limited (resulting in the conversion of inorganic nutrients to organic matter) and the increased activity of certain bacterial groups.
- Secondary impacts arise from organic enrichment due to the increase in autotrophically-produced organic materials. For example, the death and bacterial-mediated decay of the increased vegetative matter can result in the depletion of oxygen levels causing detrimental impacts to estuarine fauna.
- Finally there are resultant (tertiary) impacts to those species/communities, that may not be directly impacted by the changes in primary production and water quality but are indirectly affected by the depletion of, or reduced access to, prey species caused by either vegetative growth and/or organic enrichment.

Although a range of potential impacts can arise from enrichment, care should be taken not to view eutrophication exclusively as a detrimental process because, by promoting increased productivity enrichment can have ecological benefits. Indeed because high inputs of nutrients are typical of many estuaries these ecosystems are some of the most fertile and productive in the world (Davidson, 1991; Nixon, 1992). However, the Environment Agency and English Nature require to be aware of all the potential consequences of nutrient enrichment in estuaries in order to effectively monitor conditions within estuaries to assess the risks to habitats. This section, therefore, reviews the effects of increasing nutrient levels on the structure and functioning of estuarine communities. The responses are described in the context of the three stages of enrichment presented above and will be separated into the following sections:

- **Section 3.2** reviews the impact of nutrients on each of the major autotrophic taxa which includes the three algal groups: phytoplankton, phytobenthos, macroalgae and the macrophytes. This includes the ecology of each of these groups, the evidence of the responses of the different autotrophic groups to, and influence upon, nutrient concentration and nutrient cycling.
- **Section 3.3** reviews effects of organic enrichment and the associated problems of oxygen depletion on estuarine fauna, and benthic communities in particular. The effect that benthic fauna have on nutrient cycling will also be described
- **Section 3.4** reviews the impacts at the highest trophic levels (*i.e.* avifauna, fish) and the ecosystem-scale effects on food web structures.

As the biological response of species and communities to nutrient input is affected by environmental conditions, further details about the effects of physico-chemical factors on the responses of individual species/communities is included in the relevant sections. This is in addition to the information contained in Section 2.5 which reviewed the effects of physico-chemical factors on the general biological responses to nutrients.

3.2 Autotrophic Groups and Bacteria

3.1.1 Phytoplankton

Marine and estuarine phytoplankton include approximately 15 classes of algae (≈ 5000 species). All are small (1 to $> 200 \mu\text{m}$ in diameter) and single celled but show a great diversity in shape. The classes are distinguished on the basis of fine morphological structure and photosynthetic pigment content. Like all plants they derive energy from sunlight via chlorophyll and require inorganic (mainly) nutrients (Nitrogen, Phosphorous, Silica) to grow by cell division. Under favourable conditions rapid growth, greater than one cell division per day, may occur and if sustained can result in dense growth (bloom) leading to visible discolouration of the water.

Phytoplankton growth is dependent upon an adequate supply of light and inorganic plant nutrients. Consequently, at low irradiance phytoplankton may become light limited and at low nutrient concentration nutrient limitation will occur. The response of the phytoplankton community to nutrient enrichment is primarily dependant on light supply which itself is dependant on turbidity and on the depth of vertical mixing. The depth of mixing is important because light levels decline exponentially with depth. As cells move through the water column, as a result of turbulent mixing, they experience a range of light levels which may be less than the minimum required for growth.

Although the response of phytoplankton to nutrient input in the presence of an adequate light supply is to grow there may not be an observable increase in biomass in the water column if the rate of increase in biomass is less than the rate of biomass loss. The loss terms include grazing, sinking, respiration and advection (flushing). Consequently, nutrient enrichment may not lead increases in biomass despite increasing primary production.

The factors affecting phytoplankton production are shown in Figures 5 and 6 from Boynton *et al.*, (1982) and Wilkinson (unpubl.) respectively. The growth of phytoplankton shows a distinct seasonality which can generally be explained in terms of the availability of light, nutrient supply and other abiotic factors as outlined above. In winter phytoplankton growth is usually light limited. As well as shorter daylength increased run-off contributes to increased turbidity further reducing light levels. Nutrient input is at its highest during the winter, with maximal freshwater run-off, and uptake by plants at its lowest which generally results in the highest measured nutrient concentrations. During the spring increasing light levels allow phytoplankton growth to occur. This initial increase in phytoplankton biomass is termed the spring bloom. The classical view of phytoplankton exhibiting a spring bloom followed by a summer minimum (when nutrient are limiting for growth) and a second autumn bloom is derived from observation in shelf-seas which become thermally stratified during the summer. In the shallow turbulent waters of most estuaries seasonal stratification does not occur. Consequently, the mechanism leading to summer nutrient limitation does not exist and provided sufficient nutrients are available growth can occur throughout the summer months until light limitation sets in later in the year.

A water column will become stratified if the input of turbulent mixing energy is insufficient to overcome the input of buoyancy from heating and freshwater. Mixing energy is generated at the bottom of the water column from water movement created at the by tides and background flow and near the surface by winds. Although estuaries do not generally exhibit

seasonal (>1 month) thermal stratification short term (hours to days) may develop in response to increases in buoyancy input (heat and/or freshwater) or a reduction in mixing energy (neap tides, slack water) or both. The significance of stratification is that it may increase the level of irradiance experienced by phytoplankton in the buoyant surface layer. As a result previously light limited cells may now receive sufficient light energy to grow and therefore initiating a phytoplankton bloom.

Further phytoplankton growth can occur in response to the release of recycled nutrients that are exported from the sediment following the decay and remineralisation of organic matter deposited from the first bloom. Therefore, a distinction can be made between growth of phytoplankton using 'new' nutrients, which occur mostly earlier in the year and subsequent growth which uses 'old' regenerated nutrients (Boynton *et al.*, 1982; Malone *et al.*, 1988).

Different estuaries show differences in these observed patterns and in the proportions of the overall primary productivity that is attributable to the old and new nutrient inputs. This will depend on factors such as the sources and timing of nutrient inputs and the proportions of nutrients that are recycled which will in turn depend on factors such as the flushing time and water circulation. As well as the inter-estuary differences there will be inter-annual variations in the seasonal cycles of phytoplankton biomass and production and variability in the nature of the nutrient supply (new or recycled), due to differences in the seasonal climatic conditions.

In the open ocean the distinction between old and new nutrients is relatively clear because most of the new nitrogen is in the form of nitrate while recycled nitrogen in the euphotic zone occurs as ammonium. However, identifying the relative balance and impacts of old and new nutrients in estuaries can be particularly difficult because in estuaries much of the 'new' nutrient is ammonia so it is impractical to measure new-nutrient production (Nixon 1992). It is possible, however, that differences in the scale of seasonal phytoplankton blooms may explain the scale of new production. Boynton *et al.*, (1982) for example concluded that in Chesapeake Bay most of the total annual production is believed to be dependant upon the regenerated nutrients. This was because, in Chesapeake Bay, the bulk of the annual load of 'new' nutrient is delivered in late winter/early spring but maximum productivity does not occur until much later in the year.

The timing and composition of the nutrient input also affects the species composition of the phytoplankton. Boynton *et al.*, (1982) for instance showed that the annual phytoplankton bloom is better correlated with freshwater nitrogen loading than phosphorus loading. While nitrogen and phosphorus govern overall algal growth, the ratio of nutrients present and particularly the availability of dissolved silicate can regulate the species composition of phytoplankton assemblages. There are essentially two types of phytoplankton communities, those dominated by diatoms and those dominated by flagellates or non-diatomaceous forms (Doering *et al.*, 1989) and diatoms are often dominant during the spring bloom utilising new silicates.

In northern temperate aquatic ecosystems, which are characterised by a marked climatic seasonality, a spring increase in diatom biomass is a common response to nutrient loading (Conley *et al.*, 1993). Diatoms are not as abundant in blooms which occur later in the year as silicate recycling is very slow and comparatively little of this is present in the 'old' nutrient fraction. The annual cycle of phytoplankton in Chesapeake Bay follows this pattern (Conley and Malone 1992) with a transition from diatom-dominated microplankton blooms with a

high biomass to mixed non-diatom pico/nanoplankton with a high-productivity (Malone *et al.*, 1988).

The mechanisms of the spring bloom development, its subsequent collapse and the development of a summer plankton community are complex and therefore, still poorly understood even in a well studied estuary like Chesapeake Bay. Gilbert *et al.*, (1995) for instance showed that the collapse of the bloom in Chesapeake Bay was not a single event and not attributable to a single factor. One of the factors they identified was that diatoms may have a greater capacity for nitrate uptake which imparts a competitive advantage to this species during the bloom development when nitrate levels are high. Gilbert *et al.*, (1995) also identified temperature as another vital factor. They found that when temperatures remained low the spring bloom was prolonged and proposed that diatoms may be better able to exploit available nutrients at low temperatures. Similarly, Malone (1992) showed that taxonomic change in the phytoplankton blooms consistently occurred at 18°C.

Conley *et al.*, (1992) suggest that dissolved silicate controls the magnitude of diatom production during the spring bloom but also causes its collapse, and leads to changes in species composition. They proposed that the high sedimentation rates of phytoplankton observed during seasonal spring diatom bloom in Chesapeake Bay could be due to Si deficiency, suggesting that the supply of silicate may also control the flux of phytoplankton biomass to the benthos, an important parameter of seasonal oxygen depletion in the Bay.

The complexity of the processes which mediate the change in blooms was clearly demonstrated by the Forth River Purification Board who recorded marked oscillations in bloom densities in the inner Firth of Forth (Scotland) using a semi-continuous water quality sampler (FRPB 1995).

The seasonal cycle of phytoplankton blooms and its implications for estuarine enrichment have recently been discussed by Malone *et al.*, (in prep.) who reviewed, in depth, the mechanisms which drive the responses of phytoplankton growth and biomass to nutrients on the seasonal scale in Chesapeake Bay. Malone *et al.*, (in prep.) concluded that the response of phytoplankton biomass to nutrient inputs depends on two factors:

- the nutrient status of the plankton;
- the capacity of consumers to respond to increases in phytoplankton production (grazing).

They also note that phytoplankton growth rate is a function of the cellular nutrient concentration while increases in biomass are related to the ambient concentration of the limiting nutrient. Thus:

- productivity (which is the product of biomass and growth rate) is a function of the nutrient supply not the ambient nutrient concentration;
- productivity can increase just by an increase in the growth rate when growth is nutrient limited and biomass is nutrient deficient;
- when growth rate is nutrient saturated, productivity cannot increase without an increase in biomass;
- under growth rate limited conditions the response of phytoplankton productivity is rapid (hours) but when biomass is limited the response takes longer (weeks).

Based on these conclusions two extremes of response to nutrients were identified:

- rapid response where productivity is driven by the rate of nutrient uptake ('maximum response');
- slow response where productivity is driven by changes in biomass ('minimum response').

The problems of excessive nutrient input and associated declines in water quality occur in the latter case when growth rate is nutrient saturated and production increases because of increases in biomass.

Malone (in prep.) found that, in Chesapeake Bay, there is an oscillation between these response modes over an annual cycle. There is a transition from a high biomass, low growth rate phytoplankton assemblage in spring (minimum response mode), to a lower biomass high growth rate assemblage during the summer (maximum response mode). Therefore, the problems associated with enrichment in this system occurred following the spring bloom because the inputs of nutrients produced an accumulation of phytoplankton biomass at this time of year. During the summer, phytoplankton growth rates are generally limited to the maximum response as the high grazing pressure and low nitrogen availability relate to the light and temperature-driven phytoplankton demand.

The above examples describe the seasonal changes in the dynamics of phytoplankton communities although more lasting changes can occur in response to long term chronic nutrient loading concentration (Cooper and Brush 1993; Harding 1994). As cultural nutrient enrichment generally leads to increases in the water column nitrogen and phosphorus concentrations and not Si increases in biomass are confined to non-silicate requiring species such as flagellates leading to a shift in species composition as a result of enrichment. , Harding (1994) observed just this type of change in Chesapeake Bay where the species composition of phytoplanktonic blooms has changed over the past few decades from assemblages dominated by centric diatoms to ones made up primarily of flagellated species. He identified two further changes in the ecology of phytoplankton that are attributable to chronic nutrient loadings in the Bay. There has been a significant increase in the overall plankton biomass since the early 1950's, and in the 1970's there was the first occurrence of a winter-spring bloom. In addition there has been a change towards an abundance of smaller picoplankton (< 2.0 µm in diameter) and nanoplankton (2 – 20 µm in diameter).

Cooper and Brush (1993) used palaeolimnological techniques to describe changes in the environmental conditions long-term (pre-European settlement to present) by describing changes in the diatom community structure and composition within sedimentary profiles. They observed a progressive decrease in diatom diversity lasting 40-50 years indicating increasing stress on this community, together with an increase in the centric (pelagic):pennate (benthic) ratio that is indicative of an overall loss of habitats. Cooper and Brush (1993) cite other studies recording a decrease in diatom diversity in response to eutrophication: Raritan Bay estuary (Patten 1962), Lake Washington (Stockner and Benson 1967) and the north coast of Cornwall (Hendey 1976). Cooper and Brush (1993) note that this 'decrease in diversity of diatom communities parallels an overall decrease in diversity of larger organisms in the Chesapeake Bay, attributed to human influences'.

Phytoplankton blooms deplete water column nutrients through assimilation although there is some debate about whether some nutrient species are more readily assimilated than others.

Ammonium is widely believed to be the form of dissolved inorganic nitrogen that is preferred by phytoplankton (Libes 1992). Phytoplankton will preferentially assimilate the nitrogen species with the lowest oxidation state, so in order of decreasing preference, urea and ammonia → nitrite → nitrate (Libes 1992). Although it was originally believed that the presence of ammonium inhibits the uptake of nitrate by phytoplankton, Gilbert *et al.*, (1995) found many examples in the literature to show that nitrate is used preferentially even in the presence of ammonium.

Lipschultz (1995) observed nitrate uptake in the presence of ammonium but also showed that all size classes of phytoplankton took up more ammonium than nitrate (by an order of magnitude) and that growth rate was also found to be greater on ammonium (0.38 day^{-1}) than on nitrate (0.05 day^{-1}). Lipschultz (1995) also found that small cells accumulated ammonium faster than larger cells whereas the reverse was observed for growth on nitrate. When studying the nitrogen uptake of phytoplankton and bacteria together, 50% of the ammonium uptake was mediated by phytoplankton. Bacteria in the water column were not thought to compete with phytoplankton for nitrate but a small proportion (10%) of nitrate was tentatively attributed to bacterial uptake.

In a 'lagoon' mesocosm experiment the responses of phytoplankton were no different with the addition of either nitrate or ammonia either because ammonia was not preferentially taken up in these shallow systems or the temporal variability of phytoplankton biomass was too great (Taylor *et al.*, 1995b).

The assimilation of water column nutrients by phytoplankton clearly plays a major role in reducing the dissolved nutrient concentrations and therefore reducing the amount of nutrients that are exported to the coastal zone (Fichez *et al.*, 1992). The effect of phytoplankton on phosphate concentrations was demonstrated by Conley *et al.*, (1995) who found that in estuaries with long residence times and large plankton populations there is an uptake of DIP. In estuaries with short residence time and smaller phytoplankton populations the DIP concentrations are unaffected. Other authors have shown that, in small, shallow fast flowing estuaries and/or turbid estuaries where there is a low phytoplankton development, nutrients behave conservatively and are largely exported to coastal waters (Nowicki and Oviatt 1990; Villate *et al.*, 1991; Fichez *et al.*, 1992).

The removal of nutrients by phytoplankton within estuaries can also change the composition of the nutrients exported to the coastal zone and affect the species composition of offshore phytoplankton blooms. Fichez *et al.*, (1992) note that riverine and estuarine blooms occur before the offshore blooms in the North Sea and therefore, where the estuarine input is depleted in nutrients, particularly silicon, during the period of the offshore spring bloom, it may reduce the potential for diatom blooms in offshore waters. Despite this, according to these authors, the subsequent fate of the algal material produced within the estuary is often uncertain.

Barnes and Mann (1991) suggest that there are essentially two possible fates for the nutrients within an estuary once they have been assimilated by phytoplankton. Either:

- cells lysis resulting in the release of DON and its degradation by bacteria; or
- the plants are eaten by heterotrophs (grazing) in the water column and DON is released by excretion or the death and decay of these organisms.

These processes may occur in the water column or in the benthos if cells sink. Diatoms are heavier than water and in the absence of sufficient turbulent mixing will sink. In contrast flagellates have some limited motility and are generally smaller than diatoms and are less susceptible to sinking. Estuaries are physically energetic regimes, however, time varying inputs of turbulent mixing (e.g. over a tidal cycle, neap – spring cycle, wind) and buoyancy (heat and freshwater) will cause significant variability in turbulent mixing. Consequently, particle sinking rate will vary and if sufficient turbulence is generated may result in resuspension of previously sunk cells. Phytoplankton sinking often in large numbers, following the collapse of a bloom can significantly alter the nutrient conditions of the sediment by creating a huge influx of organic nutrients. This flux of nutrients to the sediment has been particularly observed following the collapse of spring diatom blooms (Conley *et al.*, 1993). When DSi (dissolved silicon) concentrations become low or limiting to diatoms, sinking rates can greatly increase leading to a sudden termination of a diatom bloom and a large scale, rapid settling of organic matter to the sediments (Conley *et al.*, 1993). This process of sedimentation of diatoms, may be a survival reaction response to the depletion of surface water nutrient concentrations, it can be precipitated by the mass flocculation of live diatoms and may represent a transition phase from pelagic to benthic stages in the life history of these diatom species (Lomstein *et al.*, 1990). According to Lomstein *et al.*, (1990) however, the role of the live fraction of the deposited diatoms in the sediment nitrogen cycle is poorly known. However, they note that in shallow estuaries where light is sufficient, inorganic nitrogen assimilation and photosynthetic O₂ production by benthic diatoms can continue and greatly affect the NH₄⁺ and NO₃⁻ flux, rates of nitrification and denitrification.

The accumulation of biogenic silicon in the sediments can lead to a depletion of the water column reservoir of DSi (Wulff *et al.*, 1990; Conley *et al.*, 1993; Rahm *et al.*, 1996). This has been observed in the Baltic where, over the last few decades, nitrogen and phosphorus have increased whereas silicate concentrations have declined. Wulff *et al.*, (1990) also described decreases in DSi and the DSi:DIN ratio in the Baltic Sea from 1970-1990 and predicted that this nutrient may become limiting if the decline continues (a ratio of 1:1 is required for diatom growth).

Conley *et al.*, (1993) discusses the sequence following such a decline in the DSi reservoir and propose that if the long term depletion of DSi reduces the scale of the spring diatom bloom then there will be a reduction in the settling of diatoms leading to a decrease in the amount of nutrients lost from the surface layers. The excess of nutrients remaining in the euphotic zone after the spring diatom bloom could then lead to larger summer phytoplankton blooms which otherwise would be limited by recycling processes within the upper water column and by the transfer of new nutrient inputs from below by the physical processes of mixing. The increased summer bloom would lead to decreasing water transparency, greater eutrophication, and perhaps an increased frequency of novel phytoplankton. However, it is likely that a reduction in the absolute amount of Si will result in a shift in the species composition of the spring bloom rather than a decrease in nutrient uptake. The fate of algal nutrients following the collapse of blooms is unclear (Fichez *et al.*, 1992) but it depends upon the rate of nitrogen and phosphorus mineralisation. It is important to note however, that the rate at which phytoplankton nutrients are mineralised is faster than that from macroalgal species because planktonic cells can be more rapidly broken down. Thus where phytoplankton dominate the algal biomass, algal productivity may remain high because of the rapid cycling of inorganic and bio-assimilable organics even in some waters even where available inorganic nitrogen resource is low in concentration.

The combined effects of assimilation and regeneration of nutrient during the seasonal cycle of algal blooms (as described earlier) can lead to complex changes in the nutrient concentrations. For example, in assessing the seasonal variations in the nutrient concentrations, Malone (in prep) describes the changes in the surface water concentrations of DIN, DIP and DSi along the length of the Chesapeake Bay mainstem and identifies the following changes:

- both DIN and DSi show a winter-spring maximum and summer minimum and decreasing concentrations from the riverine source to the estuary mouth;
- in spring DSi is depleted relative to DIN;
- DIP concentrations are slightly higher in summer than spring and the distribution pattern along the estuary shows no clear trend;
- in Summer DIN is depleted relative to both DSi and DIP;
- in contrast, the bottom waters show strong summer maxima for ammonium and DIP increases rapidly in June as the bottom water becomes anoxic;
- DSi distribution shows a second peak in the lower Bay which indicates that benthic recycling of silicate is occurring during the summer.

Finally, a study of enrichment in the Baltic by Rahm *et al.*, (1996) concluded that the decline in silicate due to nutrient enrichment has implications for the recycling of nutrients and the ecology of the system. They proposed that an observed decline in silicate and the increased growth of cyanobacteria at the expense of diatoms would hypothetically lead to higher rates of nutrient recycling. This is because, while diatoms are primarily deposited and are therefore a sink for nutrients, cyanobacteria (which have a buoyancy regulation mechanism) come to the surface towards the end of their bloom and are mainly degraded in the trophic layer and the released nutrients are readily taken up by the biota.

3.2.2 Microphytobenthos

The microphytobenthos, or benthic micro-algae, are found within the illuminated and oxygenated upper layers surficial layers of sediments. They include diatoms, flagellates which can under stable conditions form complex microbial communities or 'microbial mats' in association with bacterial chemoautotrophs and heterotrophic (e.g. ciliates) species. Most benthic diatoms species are pennate (having symmetrical lanceolate cells) whereas in the plankton the centric (radially symmetric cells) predominate. A broad sub-division recognises three groups. The first of these comprises both pennates and centric forms which live on intertidal and subtidal sediments but are readily transported upwards into the water column where they survive to return to the sediment under calm conditions. A second group comprises entirely pennate diatoms with well developed motility which allows migration into the sediment. This group may be referred to as the epipelon. A third group is comprised of largely immobile diatoms fixed firmly to their substrata i.e. sand grains. Of these three groups it is largely the first two that are of significance in estuarine environments. The epipellic algae which will typically a range of flagellates are primarily associated with fine-grained sediments. These benthic microalgae are able to utilise nutrients from the overlying water as well as those within the sediment and/or produced by bacterial nutrient cycling processes.

The factors which affect the response of these algae to nutrients include:

- bathymetry which dictates the area of illuminated sediments;
- turbulent mixing which influences the stability of substrata and delivery of nutrient;
- water column turbidity which affects the light availability;
- Permeability of the sediment which affects the rate of water and gas exchange.

A few studies have been undertaken on the impacts of nutrients on the microphytobenthic communities. Using nutrient addition bioassays and analysing the relative concentrations of photosynthetic pigments (*i.e.* chlorophylls and carotenoids) Pinckney *et al.*, (1995) examined the responses to nutrients changes of the phototrophic groups within a temperate cyanobacterial mat community. The community was composed primarily of filamentous cyanobacteria and diatoms at the surface and colourless sulphur bacteria (*Beggiatoa*). They found marked monthly and seasonal changes in community composition and mat biomass with clear changes in the relative abundance of diatoms and cyanobacteria. In spring and late autumn/winter there were blooms of benthic diatoms but following the spring bloom, cyanobacteria increased while diatoms showed a decline in abundance and in winter a decrease in cyanobacterial abundance occurred with a relative increase in diatoms.

These seasonal changes were believed to be mediated largely by nutrient availability. Microbial mats exposed to high nutrient levels showed a rapid response with changes in the relative balance of diatoms and cyanobacteria and in periods of low nitrogen, the cyanobacteria, as fixers of nitrogen, appeared most tolerant. As concentrations of only nitrate, or nitrate and phosphate together, increased there was a decrease in the cyanobacterial nitrogen fixation rates and a corresponding increase in the diatom biomass. Alternatively, an increase in phosphorus alone resulted in an increase in nitrogen fixation rates and cyanobacterial abundance. Diatoms appear competitive dominants at high nutrient concentrations and the diatoms and cyanobacteria were found to be nitrogen limited (even though the latter are capable of nitrogen fixation) and their response to increased nutrient levels was rapid, biomass doubled and nitrogen fixation tripled after three weeks in the nutrient bioassay.

Sundback (1994) found that elevated nutrient levels favoured the growth of the microalgae filamentous cyanobacteria, flagellates as well as diatoms in a the microbial mat of a shallow-water sandy-sediment site in the Skagerrak. Although benthic diatoms were not expected to flourish under these elevated nutrient conditions, that they did so was attributed to a possible supply of silicate within the sediment. In laboratory conditions where silicate availability was restricted, the predicted increase of flagellates and cyanobacteria, at the expense of diatoms, did occur with increased levels of nitrogen and P.

Similarly, Höpner and Wonneberger (1985) showed that the activity of epiphytobenthos in an intertidal region increased in response to artificial additions of ammonia or phosphate. Maximum stimulation was achieved when the N:P ratio of the efflux is the same as that in diatoms. They found that the mats mainly utilise the nutrients that flux from within the sediment and concluded that an observed natural patchiness in the distribution of these mats was due to spatial changes in the scale the nutrient efflux and its N:P molar ratio.

An impact of epiphytic mats is that by utilising the available nutrients they can cause a significant decline in nutrient concentration of the surface sediments and act as barriers preventing the exchange of nutrients across the sediment-water interface (Höpner and Wonneberger, 1985, Nilsson *et al.*, 1991). They may also affect the water column nutrient concentrations by removing nutrients from the overlying water. Nilsson *et al.*, (1991), for example, observed the benthic microalgae to be highly dependant on nutrients in the overlying waters and they may compete with phytoplankton in shallow waters. The extent to which the benthic microalgae utilise water column nutrients is dependant upon the bathymetry of the estuary and particularly the percentage area of shallow waters, the water column light availability and the pattern of water column circulation (Nilsson *et al.*, 1991).

With a sufficient input of turbulent mixing the microphytobenthos can become resuspended and then compete directly with phytoplankton for water column nutrients (de Jonge and van Beusekom 1992). Therefore, under more turbulent conditions, the contribution of the microphytobenthos to the primary productivity of the whole estuary and the role of this community in the utilisation of nutrients, is difficult to assess and the distinction between phytoplankton and benthic microalgae becomes blurred.

The contribution of microphytobenthos to estuarine production can also be complicated by other factors. Kromkamp *et al.*, (1995) studied the relative contributions of phytoplankton and benthic microalgae to the total primary productivity of a turbid estuary (Schelde). Although they expected that the microphytobenthic contribution would be relatively large because of the more favourable irradiance regime in the intertidal regions instead it was relatively small. This was largely because the photosynthetic rate rapidly declined when the intertidal flats were exposed and when the most light is available in this estuary. The authors proposed that CO₂ limitation was partially responsible for this observed decline.

3.2.3 Macroalgae

Benthic macroalgae can be separated into the rapidly growing opportunistic 'r' dominant species and the slower growing 'k' dominants. The former group, including green algal species such as *Ulva* spp., *Enteromorpha* spp. and *Chaetomorpha* spp., have a seasonally variable abundance whereby they develop during the summer months and decline in the autumn. The slower growing group, including brown algae such as *Fucus* spp. are perennial. The majority of macroalgae grow attached to stable surfaces such as boulders and bedrock although opportunistic species can be 'loose-lying' and survive on tidal flats attached to small stones (some fronds may also become detached and float freely in the water column). Macroalgae obtain nutrients solely from the water column but their close proximity to the substratum enables them to also utilise nutrients newly released from the sediment. The physico-chemical factors which affect macrofaunal growth and therefore the capacity of these species to respond to nutrients includes:

- bathymetry which dictates the area of sediments with sufficient light availability;
- turbulent mixing which influences the stability of substrata and delivery of nutrient;
- water column turbidity which also affects the light availability;
- substratum type which dictates whether algae can become attached;
- temperature which affects the rate of photosynthesis.

In response to increases in nutrients, macroalgae (particularly opportunistic species which have thinner tissues and a faster growth rate) exhibit an increase in growth and spatial coverage. There are numerous examples of opportunistic algae flourishing when nutrient levels increase:

- the Ythan estuary has had an increased coverage of weed mats on intertidal mudflats over the last 30 years (Raffaelli *et al.*, 1989; Balls *et al.*, 1995);
- Langstone Harbour has shown a greater coverage and persistence of algal mats (Nicholls *et al.*, 1981);
- Peel-Harvey estuary (Australia) has shown an increased coverage of opportunistic algae (Gerritse in prep.);
- Laholm Bay in the Kattegat has had a change in the species composition of macroalgae with *Fucus* spp. being replaced by filamentous green algae species such as *Cladophora* sp. and *Enteromorpha* sp. (Baden 1990) (see Figure 7);
- Under experimental conditions Sundback (1994) observed an increase in the growth of filamentous green macroalga *Enteromorpha clathrata* in response to nutrient enrichment.

The growth of macroalgae can affect nutrient level within estuaries either by depleting the water column nutrients, or intercepting nutrients fluxing across the sediment/water interface. Macroalgal mats can remove nutrients from the water column as shown by McBride (1992) where *Chondria* spp. mats stripped the water column of nitrogen although they only remove a small proportion of phosphorus. Similarly, Riisgard *et al.*, (1995) showed that a dense surface mat of filamentous algae (*Chaetomorpha linum*) absorbed large amounts of inorganic nitrogen in excess of its growth requirement and also acted as a filter and controlled the release of nutrients to the water column from the nutrient-rich sediment below. This was indicated by a steep nutrient profile within the algal mat with high concentration just above the sediment to zero concentration in the upper layer of the algal mat. In this nitrogen-limited system, the algae only removed a small proportion of the mineralised inorganic phosphorus which was consequently present in higher concentrations in the water column. When the algal mat was removed there was an uncontrolled flux of nutrients out of the sediment that resulted in the bay ecosystem becoming unbalanced. Finally, Asmus *et al.*, (1995) observed that the macroalgae present on natural mature mussel beds assimilated phosphate and controlled its release into the water column. In juvenile beds, where there was no algae, they found that most of the released phosphate does reach the water column.

Macroalgal mats can also affect nutrient dynamics by promoting anoxic conditions and affecting the bacterial process and macroinfaunal communities within the sediments underlying the algal mats (see also Section 3.4). McBride *et al.*, (1992) identified three reasons for the development of anoxic condition in the sediment covered by opportunistic algae:

- mats provide an abundant source of organic matter from plant die-off;
- the presence of mats inhibits the physical sediment/water exchange due to currents and wave action;
- the macrofauna in sediments below the algal mats, which would otherwise promote the exchange of gases and other material between sediment and water column, decline in abundance.

3.2.4 Vascular Plants

Two major groups of vascular plants are found in temperate estuaries these are seagrasses and saltmarshes although reed beds are found in the upper reaches of estuaries. As these are rooted plants they require stable substrata and are typically found in sheltered environments. Seagrasses are predominantly subtidal plants that are generally located in shallow sublittoral regions as they do not grow well in intertidal regions and they tend to be light limited at greater depths (Barnes and Mann 1991). Saltmarshes and reed beds in contrast occur in the supra-littoral zone at or above the high water mark. Unlike the algae, vascular plants are able to assimilate nutrients directly from the sediment via their root systems and some taxa (*e.g. Spartina*) also have nitrogen-fixing bacteria on the roots which provide supplementary nitrogen. For these reasons vascular plants often have a competitive advantage over algal species under low nutrient conditions.

Most research on the impacts of nutrients on vascular plants has been directed towards seagrasses or 'submerged aquatic vegetation'. Although they are competitive under low nutrient conditions, there is evidence that seagrasses thrive under elevated nutrient levels. For example, Eelgrass (*Zostera* sp.) growth increases in response to the addition of fertilisers (Orth 1977 cited in Short 1987). However, increases in nutrient levels have an adverse impact on seagrasses because, under these conditions, algal species often respond more rapidly and out compete seagrasses for available light (Taylor *et al.*, 1995b; Lin *et al.*, 1996; Kemp *et al.*, 1983 and 1984).

Taylor *et al.*, (1995b) for example, found that a decline in eelgrass (*Zostera marina*) occurred with the addition of nutrients because of the declining light levels following the blooming of phytoplankton in mesocosm experiments. The growth of eelgrass was favoured by the addition of nutrients only under conditions which precluded the development of phytoplankton blooms *i.e.* where there were shallower depths (favouring growth of benthic vegetation), faster flushing times (inhibiting phytoplankton bloom development), or where the nutrient were added directly to the sediment (favouring the growth of rooted plants).

Lin *et al.*, (1996) found increasing evidence that since the 1980's anthropogenic nutrient inputs have stimulated the growth of epiphytic algae on the leaves of seagrasses and caused of seagrass decline through shading.

Kemp *et al.*, (1983; 1984) has observed large scale declines in the coverage of Submerged Aquatic Vegetation (SAV) in Chesapeake Bay which has been attributed to the effects of shading by epiphytic algae and phytoplankton. Suspended particulate matter, which includes dead or decaying organic matter, living material and inorganic particles, all contribute to water column turbidity. Furthermore, settlement of particulates directly on the leaf surface and growth of epiphytic bacteria block the passage of nutrients across the leaves or promote the growth of animal fouling communities. The interactions between algal growth and the growth of vascular plants can, however, be complicated by the effects of herbivorous grazers. For example, Lin *et al.*, (1996) investigated the system-level responses of epiphytes to enrichment in mesocosm experiments and found a greater increase of epiphytes under low nutrient conditions rather than in the mesocosm enriched with nutrients. This result differed from previous studies and was attributed to the interactions between trophic levels. They were unclear as to the exact cause but presumed that 'top-down' controls of epiphytic growth by grazing species would be a significant factor. Conversely, McGlathery (1995) observed that nutrient enrichment caused an increase in the growth of the mat-forming macroalga

Spyridea hypnoides caused a decline in the seagrass *Thalassia testudinum*. However, it was concluded that, in this case, the decline was primarily due to enhanced grazing by herbivorous fishes and not the effects of shading.

Several studies have examined the impacts of seagrasses on the fluxes of nutrients (Kemp *et al.*, 1984; Caffrey and Kemp 1990). In experimental ponds, Kemp *et al.*, (1984) found that submerged vascular plants were an important seasonal sink for nutrients with a rapid uptake and then following senescence, death and decay these nutrients were relatively slowly released compared to algae. Thus the oxygen demand during decomposition of submerged vascular plants is also lower due to this factor and the fact that the mortality of these plants also occurs later in the year when it is the cooler (Autumn) they contribute less to oxygen depletion of the bottom waters. It was proposed that SAV play a particularly important role in buffering pulsed inputs of N and P and in maintaining water quality (Kemp *et al.*, 1984).

Caffrey and Kemp (1990) described the effects of seagrasses on bacterial nutrient transformations within the sediment. They compared the pattern of nitrogen transformations in bare estuarine sediments against those in areas vegetated by the *Potamogeton perfoliatus* (Redhead grass) and *Zostera marina* (eelgrass). They concluded that the following key microbial processes were enhanced by the presence of *P. perfoliatus* and *Z. marina*:

- ammonification by inputs of organic nitrogen from the plants;
- nitrification by release of O₂ by plant roots;
- denitrification by production of NO₃⁻ following nitrification.

The magnitude of these effects varied on a seasonal basis apparently with the growth patterns of the two plant populations.

In contrast to seagrasses, the interaction between nutrients and saltmarshes appear to be more complex and less well understood. Although the responses of saltmarshes have been well investigated, most studies have concentrated on describing the gross rates of nutrient flux to and from saltmarsh systems. Dame *et al.*, (1986) for instance proposed that marsh-estuarine systems are generally exporters of most nutrients and materials (referred to as the 'Outwelling' hypothesis). Dame *et al.*, (1990) in contrast, showed that an estuarine/marsh system in the North Inlet, South Carolina imported significant quantities of phosphate, particulate phosphorus and total phosphorus. A detailed review by Vernberg (1993) included the following studies of nutrient fluxes to and from saltmarshes which highlight the complexity of the interactions between the processes in tidal creeks and the saltmarsh:

- Marshes can be a statistically significant sink for nitrogen (dissolved nitrate plus nitrate) produced by the adjacent tidal creeks (Spurrier and Kjerfve 1988).
- The tidal marshes of the Rhode River estuary (Chesapeake Bay) trapped particulate nutrients and released dissolved nutrients, whereas subtidal mudflats trapped particulate phosphorus, released dissolved phosphate, and consumed nitrate (Correll *et al.* 1992). The net effect of these processes was a decrease in the ratio of dissolved inorganic nitrogen to phosphorus in the estuary.
- NH₄⁺ and PO₄³⁻ were exported to estuarine waters from a saltmarsh and 75% of this was derived from advective transport of interstitial water from subtidal creek sediments to

overlying creek water while 25% was through low tide run-off from the vegetated marsh flats (Whiting and Childers 1989):

- The addition of nitrate-poor water may produce a net export of nitrate from the marsh and a net uptake from the creeks whereas nitrate-rich water produces a large net uptake from both regions (Taylor 1992).

Other studies have identified the maturity and tidal height of the saltmarsh systems as factors which have an effect on the system-level flux of nutrients. Dame (1994) concluded that the maturity of the saltmarsh is important for determining the flux of material with mature systems exporting all materials, middle-aged systems importing just the particulates and immature systems importing all forms of materials. Dame and Lefeuvre (1994) observed that European saltmarsh systems are import-dominated because they are confined to the upper reaches of the intertidal zone while New World saltmarshes, which extend into the mid-shore region, are export-dominated.

There are few studies describing the ecological impacts of elevated nutrient levels on the functioning of saltmarsh systems. Morris (1991 in Vernberg 1993) concluded that primary productivity in marshes is generally nitrogen-limited and that an increase in nitrogen loading leads to significant increases in primary production. The interactions between nutrients and saltmarshes are therefore poorly understood, a feature mainly attributed by Vernberg (1993) to the absence of any long term studies on these interactions.

3.2.5 Comparative responses from autotrophic taxa :

As well as considering the responses of separate autotrophic groups to enrichment, examinations of the comparative responses of these groups have shown that enrichment can lead to a change in the relative abundance of autotrophic taxa, because of differences in their response to nutrients (Nielsen and Sand-Jensen, 1990; Sand-Jensen and Borum 1991; Duarte 1995; Pedersen 1995; Schramm 1996). An extensive review by Duarte (1995) found much historical and experimental evidence to demonstrate a link between increases in nutrient loads and changes in balance of primary producers. The review indicated that as 'eutrophication proceeds' the faster-growing opportunistic autotrophs become more dominant. This occurs because opportunists respond quickest under high nutrient conditions and their growth indirectly has a detrimental impact on slower growing species by restricting the light penetration through the water column. Therefore, a change occurs from a nutrient-limited to a light-limited condition with the process of eutrophication.

Duarte (1995) considered this process to occur as the progressive replacement of autotrophic groups in the following sequence: seagrasses → slow growing macroalgae → fast growing macroalga → phytoplankton, leading ultimately to a dominance of phytoplankton and, more specifically, bloom-forming phytoplankton. Duarte (1995) considered that the process of vegetation change during eutrophication is a self-propagating process (a 'domino effect') which once started is maintained and amplified until the final phase of eutrophication is reached (see Figure 8). Duarte (1995) also observed that this progression occurs in a stepwise manner rather than as a gradual change which indicates that there are indirect and feedback mechanisms involved.

Nielsen and Sand-Jensen (1990) consider this change of species dominance and the differences in the relative growth rate of plants to be related to the thickness of the plant with

the thickest plants having the lowest corresponding rate of growth. Slower-growing, thicker organisms such as seagrasses and macroalgae have a lower nutrient and higher light requirement than the faster growing phytoplankton and thin macroalgae, they therefore dominate under low nutrient conditions. Conversely, phytoplankton and thin macroalgae with a high surface area/volume ratio compete better for available nutrients when light is limiting. Under low nutrient conditions however the larger benthic plants are more competitive either because they have access to nutrients within the sediment *i.e.* seagrasses and certain macroalgal species (*e.g.* *Caulerpa* spp.), or because they can internally recycle nutrients.

Pedersen (1995) also noted that increased nutrient availability promotes the growth of phytoplankton and fast growing ephemeral macroalgae while suppressing large slow growing benthic plants but it does not alter the total system productivity. Pedersen (1995) studied the relationship between nitrogen availability and the growth rate of different marine plants in a Danish estuary by monitoring the changes in the phytoplankton, *Ulva lactuca* and *Chaetomorpha linum* (ephemerals), *Fucus vesiculosus* (macroalga) and *Zostera marina* (seagrass). Increased phosphorus tissue concentration was not associated with improved plant performance whereas an increase in tissue-N was followed by increased levels of chlorophyll-*a* + *b* in all benthic plants. The growth rate of thin, potentially fast-growing aquatic plants increased relatively more than that of the large, inherently slow-growing plants upon N-enrichment. It was predicted that N-enrichment would boost growth rates and possibly the abundance of phytoplankton and ephemeral macroalgae which could increase shading effects on perennial benthic plants.

The nature of the estuary is also important in determining which autotrophic group responds most successfully to available nutrients and the hierarchy of nutrient competitiveness may differ under different conditions. Fong *et al.*, (1993), for instance, studied the competition for nutrients among estuarine phytoplankton and algal mats in shallow experimental mesocosms. In high nutrient load conditions, attached algae were better competitors than phytoplankton and the following competitive hierarchy was identified: cyanobacteria > attached green macroalgae > floating green macroalgae > phytoplankton.

Several reviews of the responses by marine benthic vegetation to eutrophication in European coastal waters are contained in Schramm (1996) which shows that a common pattern of response to eutrophication was observed in all areas of Europe although there are qualitative and quantitative variations in the character of the response due to local differences in conditions. A general sequence of successive responses to eutrophication was described (Figure 9) and separated into the following four phases:

- The first phase (“un-eutrophicated”) is characterised by a low nutrient load and nutrient concentrations which may temporarily become limiting. Under these conditions the dominant primary producer communities typically comprise perennial benthic macrophytes, *e.g.* seagrasses and other phanerogams, with opportunistic seasonal macroalgae representing a smaller proportion of the biomass and production.
- The second phase represents slight to medium levels of eutrophication and is characterised by increasing blooms of ‘nutrient opportunists’ such as the fast-growing epiphytic macroalgae and bloom forming phytoplankton. At this stage perennial macroalgae and phanerogam communities gradually decline, along with their depth

distribution, and then disappear. In addition, an increase in free-floating macroalgae is also often observed.

- Further increases in the nutrient loads results in Phase 3 conditions where free floating macroalgae and particularly 'green-tide' forming taxa (*e.g. Ulva, Enteromorpha* spp.) alternating with uncontrolled phytoplankton blooms have completely replaced the perennial and slow-growing benthic macrophytes.
- At the fourth phase, conditions become hypertrophic with continuously high nutrient concentrations and phytoplankton dominate the primary producers.

Schramm (1996) considered the first step of eutrophication where there is an increasing occurrence of seasonal fast-growing nutrient opportunists to be 'without question a direct effect of increasing nutrient loads'. The subsequent changes in the sequence described above cannot, however, be attributed only to nutrient inputs but were influenced by other indirect, direct or secondary effects and feedback mechanisms, including:

- declining light levels due to the enhanced growth of phytoplankton and epiphytes which adversely affects benthic plants which are less competitive under low light conditions;
- the increased organic enrichment in sediments due to the breakdown of faster growing vegetation which causes anoxic conditions and accelerates the rate of nutrient release from sediments;
- the change in hydrodynamic conditions at the sediment-water interface and the increased level of sediment resuspension that accompanies the decline in seagrasses and benthic microalgal films;
- the feeding activities of herbivores which can result in negative or positive feedback mechanisms, *e.g.* the adverse impacts arising from the growth of epiphytes can be counteracted by grazing at low nutrient levels where the rate of epiphyte growth is too slow to compensate for grazing losses. Alternatively, the enhanced growth of epiphytes or phytoplankton can lead to an increase in grazing and filter feeding populations which may in turn accelerate eutrophication through the increased nutrient regeneration.

In agreement with Duarte (1995), Schramm (1996) concluded that the changes in benthic vegetation were self-propagating and were difficult to control once they have been initiated.

3.2.6 Bacteria

Bacteria, like the autotrophs, also directly utilise and transform inorganic nutrients especially nitrogen species. There are several functionally different bacterial groups present which utilise and transform different nitrogen species and, as a result, drive the cycling of nitrogen (see Section 2.4). Where environmental conditions favour a given bacterial response, then the response is generally resource driven (Billen *et al.*, 1985; Kemp *et al.*, 1990; Barnes and Mann 1991; Ogilvie *et al.*, in prep.) such that an increase in the availability of a particular nitrogen species promotes an increase in the activity of the functional group of bacteria (*e.g.* nitrifiers, denitrifiers etc.) which utilises that species. This resource-driven nature of bacterial responses and the fact that bacterial processes transform nutrients can produce a coupling between different bacterial processes whereby one bacterial response produces the substrate necessary to drive a subsequent one. An example of this coupling effect is nitrification which occurs in a step-wise fashion due to the actions of two different bacterial species *Nitrosomonas* and *Nitrobacter*. Initially, the presence of NH_4^+ stimulates growth of

Nitrosomonas which oxidise ammonia to nitrite, then the increased nitrite concentration stimulates the growth of *Nitrobacter* which oxidise nitrite to nitrate (Barnes and Mann 1991).

Due to the fact that there are several different bacterial groups responding to different nutrient species and, because of the variability of conditions within and between estuarine systems, it is difficult to generalise about the impacts of elevated nutrient levels on bacteria. However, the following series of key observations about the relationship between nutrient loading and bacterial activities have been highlighted:

- The bacterial-mediated conversion of organic nitrogen to ammonium involves oxygen demanding processes (mineralisation and nitrification) and therefore high organic loading leads to oxygen depletion (Diaz and Rosenberg 1995).
- A coupling between mineralisation and nitrification can occur and is commonly observed in the turbidity maxima of estuaries where particulate organic nitrogen from river or estuarine sources promotes mineralisation and the production of ammonium which then drives nitrification (Balls, 1992; Balls *et al.*, 1995). This process then contributes to the characteristic development of the oxygen sags associated within turbidity maxima regions.
- In some estuaries such as Port Philip Bay (Parr and Wheeler 1995), Peel-Harvey (Gerritse in prep.) or the Scheldt estuary (Billen *et al.*, 1985), nutrient enrichment problems are ameliorated by very high rates of denitrification which converts nitrate to gaseous nitrogen which is then lost to the atmosphere.
- Nowicki (1994), however, observed that there was an increase in denitrification with increased enrichment in mesocosm studies but that the increase in denitrification did not keep in step with enrichment such that denitrification 'may not necessarily provide a significant pathway for alleviating eutrophication effects in coastal waters'.
- A coupling can occur between nitrification and denitrification with the result that after decaying organic matter is mineralised to ammonia, it is rapidly converted to nitrate (nitrification) and then to gaseous nitrogen (denitrification) and then lost from the system. This can only take place in the sediments because these two functional groups need to be in close proximity and it is only in sediments that the necessary oxic and anoxic conditions alternate (Harding 1992; Boynton and Kemp 1985).
- When nitrification and denitrification are coupled, nitrification apparently controls denitrification by determining the rate of nitrate production (Kemp *et al.*, 1990). However, denitrification can be a significant process on its own where the inputs of nitrate to the system are high. For example, in the Colne estuary denitrification removes 50% of the nitrate while coupled nitrification-denitrification is less important and removes only 18-27% of the total nitrogen flux (Ogilvie *et al.*, in prep.).
- High levels of organic input to the sediment have a major impact on the bacterial processes and particularly the effectiveness of the nitrification/denitrification pathway. Large 'pulsed' inputs of organic matter to the sediment (e.g. following the collapse of large algal blooms) lead to an initial increase in microbial activity and denitrification but the increase in NH_4^+ remineralisation and oxygen consumption results in the development of anoxic conditions which temporarily stops nitrification (Conley and Johnstone 1995). This results in an uncoupling of the nitrification-denitrification processes because ammonia is not converted to nitrate and is not available to denitrifiers.

- The uncoupling of nitrification and denitrification causes nutrient enrichment to become self-accelerating because nitrogen is retained as nitrate (the product of nitrification), rather than being denitrified and removed from the system. As a result, nitrogen remains in the system and is cycled throughout the summer, producing additional metabolizable substrate and continuing to drive a high phytoplankton production (Baden 1990, Kemp *et al.*, 1990; Harding *et al.*, 1992; Jensen *et al.*, 1990). The self-perpetuating process, an uncoupling of nitrification and denitrification, then produces further enrichment (Figure 10).
- Bacterial activity can indirectly influence the physico-chemical reactions within the phosphorus cycle by affecting the concentration of dissolved oxygen. Under depleted oxygen conditions the fluxes of DIP from the sediment are increased. Boynton *et al.*, (1991 in Conley *et al.*, 1995) observed the seasonal changes that summer, bottom-water oxygen depletion in Chesapeake Bay largely depleted the fluxes of DIP from the sediment whereas during the rest of the year, they are either negligible or negative.

3.3 Benthic Communities

3.3.1 Impact of Enrichment on Benthic Communities

Nutrient enrichment effects on estuarine benthic fauna are mainly from organic enrichment arising from inputs or the elevated levels of local plant growth; more apparent are the accompanying effects of oxygen depletion from the increased bacterial-mediated decay of these organics. In general, the benthic fauna are most at risk from these impacts for the following reasons:

- sediments are often the site of organic deposition;
- the benthos is in intimate contact with the sediments;
- demands for oxygen arise from both the water column and from the sediment through respiratory demands of large populations of heterotrophs;
- benthic fauna are relatively sedentary and unable to move rapidly away from an area of enrichment and thus integrate changes over time.

The impact of increasing levels of organic enrichment on benthic communities is illustrated in the model developed by Pearson and Rosenberg (1978) (also called the Rhoads-Germano model in some literature). According to this model, at low levels of organic input, there is an initial increase in abundance and biomass due to the increased productivity and availability of food but as sediment organic matter increases, significant changes occur to the structure and species composition of the communities. There is a proliferation of small, fast-growing, short-lived opportunistic species at the expense of larger longer-lived species. Species diversity is reduced as the start of the anoxic layer becomes shallower and the fauna become increasingly squeezed in the surface layers of the sediment, particularly affecting deeper burrowing species. This model applies to spatial as well as temporal changes and the process of recovery occurs as a reversal of the above impact model. The Pearson and Rosenberg model is widely accepted as the best description of the impacts to the benthos arising from changes to the organic load and nutrient enrichment (e.g. Rosenberg *et al.*, 1987; Elmgren 1989; Josefson 1990; Rachor 1990; Heip 1995; Cederwall and Elmgren, 1980; Beukema and Cadée, 1986).

Incidents of oxygen depletion arising from organic enrichment are typical of many estuarine systems and often occur during the summer when biological activity is high and the rate of oxygen replenishment is low. A key determinant of the scale of the impact to the benthos from these incidents is the concentration of oxygen. In shallow estuarine environments the oxygen concentration can range from 100% Air Saturation Value (ASV) (at 8-10mg l⁻¹ oxygen to 2mg l⁻¹) or even super-saturation if high rate of photosynthesis is sufficient, with hypoxic conditions occurring between 2-0mg l⁻¹ (<30% ASV) and anoxia occurs at the zero oxygen level. The responses to hypoxia of communities in Laholm Bay in the Kattegat (Baden *et al.*, 1990) indicated that at below 2mg l⁻¹ (30%) the benthos was significantly affected and that at about 1mg l⁻¹ many species of zoobenthos and *Nephtys* die. Baden *et al.*, (1990) identified the following impacts to benthic fauna at decreasing levels of oxygen saturation:

- <40% fish disappeared and Norway lobsters emerged from burrows;
- <15% Norway Lobsters were immobilised and some benthic infauna emerge;

- <10% Norway Lobsters, *Abra alba* and *Echinocardium cordatum* were dead and species such as *Polyphisia crassa*, *Ophiura albida* and *Amphiura filiformis* are dying;
- 5-7% some hypoxia-tolerant benthic infaunal species can survive for several weeks;
- <5% most benthic macrofauna species eventually die although meiofaunal species survive.

Diaz and Rosenberg (1995), in further reviewing the effects of oxygen depletion events on benthic fauna, concluded that these hypoxic events are increasing on a global scale due to eutrophication. They made the following observations about the impacts of oxygen depletion on benthic communities:

- Hypoxic or anoxic conditions within the sediment can lead to the decline or removal of infaunal and epifaunal organisms either through a lack of oxygen itself or because of the build up of H_2S (and, under more severe circumstances, CH_4) which is toxic to benthic organisms.
- Different benthic major taxa exhibit different levels of tolerance to oxygen depletion so that declines in oxygen levels affects the community composition; evidence from *in situ* studies and experiments on natural communities, indicates that the hierarchy of hypoxia tolerance for the major taxonomic groups, from most to least tolerant, is; Polychaetes \Rightarrow Molluscs/Bivalves \Rightarrow Echinoderms and Crustacea. Bivalves are repeatedly found in hypoxia-stressed areas while echinoderms and crustacea are generally absent. This is a general trend and there are certain genera which prove to be exception to this rule, e.g. the isopod *Saduria entomon* can survive periods in hypoxic water.
- Different species also exhibit different levels of tolerance or sensitivity to oxygen depletion (see Appendix 5).
- There is also evidence of a hierarchy of oxygen tolerance within genera, e.g. in experimental conditions, the tolerance of four Nereids were compared and a sequentially lower tolerance was recorded from *Nereis diversicolor* \Rightarrow *N. virens* \Rightarrow *N. succinea* \Rightarrow *N. pelagica*. Surprisingly, there is no evidence of this discrimination between functional groups e.g. subsurface deposit feeders, filter feeders and predators.
- The depletion of oxygen can have an affect on the behaviour of infauna such that during hypoxia infauna can cease feeding, move to the surface of sediments and even emerge and begin swimming; these behavioural activities reduce the extent of bioturbation-mediated oxygen diffusion and exacerbate the problems of oxygen depletion.
- The duration of oxygen depletion events affects the level of impact. Elevated organic matter in the surface sediments increases the role of microbes in energy recycling and carbon remineralisation and for short hypoxic periods the transfer of energy to microbe still favours macrobenthic recolonisation. Over longer periods however, a larger proportion of the organic carbon will be remineralised by the microbes and less energy will be available to support benthic recruitment. Seasonal hypoxia can last for longer periods (up to several months) and can cause mass mortality irrespective of whether pre-conditioning has occurred. Recurring seasonal hypoxia/anoxia in the mesohaline region of Chesapeake Bay has been related to the decline in the abundance of large long-lived species and the dominance of smaller short-lived species.
- The response of the benthic communities can be influenced by the frequency of the hypoxic/anoxic events which may either occur regularly (*i.e.* on a seasonal basis) or can

be aperiodic (Figure 11). Persistent hypoxia disturbs communities and under severe persistent hypoxia/anoxia a total depauperation of the communities can occur. If systems experience an aperiodic hypoxia event for the first time without any pre-conditioning then there is high mortality of individuals and an elimination of species. However, if hypoxia is aperiodic then communities have an opportunity to recover although successive aperiodic events eventually lead to the development of successional stage hypoxia-resistant communities. These communities are pre-conditioned to hypoxia and they will show little change when experiencing periodic/seasonal hypoxia incidents. With periodic events then communities do not have an opportunity to recover and the level of community organisation shows a progressive decline (a community with a lower organisation is better able to capture energy during short intervals of normal oxygen conditions).

- The timing of hypoxic events also has major implications for the scale of the impact. Seasonal hypoxia events often occur in late summer to autumn when there should be a productivity peak, thus exacerbating the impact to the benthos and increasing the amount of organic matter resulting from the death and decay of benthic organisms as well as juveniles and larval stages. Such an event in relation to the annual settlement will also affect the rate of recolonisation.

Further details about the effects of oxygen depletion on fisheries are presented in Section 3.4.

3.3.2 Effect of Benthic Fauna on Nutrient Cycling

Benthic fauna play an important role in the cycling of nutrients and the biological effects of nutrients:

- By feeding on suspended and deposited material they promote the physical flux of water-borne nutrients (most notably phytoplankton) from the column into the sediment where they are then available to bacteria.
- By grazing they can also play important role in controlling the development of phytoplankton blooms (Cloern 1991; Officer 1982) and in doing so, can influence the trophic structure of the ecosystem (Doering *et al.*, 1989).
- By consuming and digesting organic material and then excreting inorganic nutrients, or simpler and more readily degradable organics, they accelerate the recycling or 'biomineralisation' of nutrients (Parr and Wheeler, 1995).
- By reworking the surface sediments through burrowing, tube formation and feeding activities they makes nutrients and oxygen more available to bacterial populations. This bioturbation results in an increase in the microbial biomass and therefore accelerates the biochemical and physical turn over of nutrient. (Heip 1995). In particular bioturbation promotes the coupling between the aerobic process of nitrification and anaerobic denitrification (see Section 3.2.6) such that in the absence of macrofauna nitrification and denitrification coupling would be restricted to the 'redox potential discontinuity' which separates the surface aerobic bacteria from deeper anaerobic ones. Macrofauna however, increase the associations between aerobic and anaerobic bacteria through bioirrigating activities, which produce oxic microzones in the anoxic lower layers, and the production of faecal pellets, which create anoxic microzones in the oxygen-containing surface layer. The resulting 'redox heterogeneity' brings a greater contact between nitrifiers and denitrifiers.

These combined macrofaunal activities promote both chemical cycling of nutrients and their exchange between the sediment and the water column and play a critical role in promoting 'benthic-pelagic coupling'. Several studies have assessed the role of particular species and/or habitats in the cycling of nutrients and benthic-pelagic coupling for example:

- the hard clam *Mercenaria mercenaria* increased the fluxes of oxygen and silicate by 20% and 86% respectively, and probably also increased inorganic nitrogen fluxes but did not affect those of phosphate and CO₂ (Doering 1987);
- benthic-pelagic fluxes of ammonium and silicate increased by up to two orders of magnitude in the presence of *Nereis diversicolor* in the Tamar (UK) (Davey and Watson 1995);
- under increasingly eutrophic conditions, animals such as the sea urchin *Echinocardium cordatum* play an important ecological role by stimulating aerobic respiration or chemical oxidation of reduced inorganic compounds and decreasing the accumulation of toxic sulphide compounds (Osinga *et al.*, 1995).

The role of mussel and oysters beds in the cycling of nutrients is well known (Dame and Dankers 1988; Dame *et al.*, 1989; Dame 1993; Asmus *et al.*, 1995). These epibenthic bivalve filter feeders are believed to be particularly important in the cycling and processing of nutrients by increasing sedimentation, retaining nutrients, controlling water column microbial biomass and increasing the rate at which materials are recycled. This is because they are filter feeders species which form dense aggregations, which means that they can remove large amounts of phytoplankton and particulate organics as well as absorbing organic compounds such as amino acids and glucose. However, because they are sessile epibenthic species they do not bioirrigate the sediment but conversely promote sedimentation of particulates and release faeces and pseudofaeces to the sediment (biodeposition). The sediment beneath these aggregations therefore play a major role in nutrient processing by accelerating the cycling of nutrients but can also act as nutrient sinks. Dense mussel beds, as biogenic reefs, however, may also prevent recycling of materials between the water column and bed sediments.

Dame and Dankers (1988) found significant uptake of chlorophyll-*a*, indicative of grazing, as well as suspended sediments, organic carbon, nitrate and nitrite by mussel beds and a release of large amounts of ammonium and phosphate. Mussel beds also release dissolved nutrients into the overlying water as the metabolic by-products of the bivalves and other fauna associated with the bed (Dame 1993).

Asmus *et al.*, (1995) monitored the flux of TP and DIP between an intertidal mussel bed and the water column using a *in-situ* flume in the northern Wadden Sea. Mussel beds were found to interact strongly with overlying water by filtering particles and excreting ammonium and phosphorus. Organic deposits collect beneath the mussels promoting mineralisation and, because the mussels limit the contact between sediment and water, this leads to oxygen depletion. Phosphorus release is promoted under anaerobic conditions as the sorption capacity of phosphate to iron oxides decreases. This study showed that mussel beds increase the flux of phosphorus in both directions, a net release of DIP and the net uptake of particulate phosphorus through filtration. Dame and Dankers (1988) showed that the directions of nutrient flux in oyster beds were similar to those in mussel beds although the magnitudes are greater in mussel beds where the biomass is generally higher and where there is often a substantial microbenthic community and the environmental setting is different. Dame *et al.*, (1989) also examined the nutrient fluxes across oyster beds and showed that,

although the fluxes were temporally variable due to changes in the tidal flow, there was a net removal of carbon nitrogen and phosphorus in oyster beds.

Asmus *et al.*, (1995) compared the fluxes of phosphorus from mussel beds with those that occur in other intertidal habitats. They found that, assuming a mean release of $870 \mu\text{m m}^{-2}\text{h}^{-1}$, mussel beds have a 5-9 times greater phosphate release than other intertidal communities. Phosphate release rates from bare sand flats and sand flats inhabited by lugworm were found to have a range of between $100\text{--}190 \mu\text{m m}^{-2}\text{h}^{-1}$. The flux from mudflats was found to be insignificant and seagrass beds exhibited a net uptake of phosphate. Asmus *et al.*, (1995) then used this information to determine the relative importance of each of these different communities to the total flux within a tidal basin (Sylt-Rømø) based on the spatial coverage of these habitats (Figure 12). Where mussel beds constitute 1% of the intertidal area their contribution to phosphate exchange is relatively low but is largest where they cover 10% of the area. However, the authors also observed that very little of the phosphate released from mussel beds may actually reach the water column because the macroalgae which cover natural mussel beds apparently store most of the released phosphate. This is not the case in juvenile or culture beds where no algae are present and the phosphate is released to the water column.

The strength of the coupling between benthos and plankton communities and therefore, its importance in terms of nutrient impact in estuaries, is evidently highly dependant on the characteristics of the environment particularly the depth of water and the flow regime. Doering *et al.*, (1989) concluded that the impact by macrofauna is likely to be greatest in shallow and energetic system where the water column is vertically mixed and where phytoplankton are transported through the entire water column by turbulence.. Similarly, Baden (1990) demonstrated the importance of environmental conditions and water circulation patterns on the grazing impact of suspension feeders in Laholm Bay (Kattegat) and calculated that suspension feeders could filter water equivalent to the entire volume of the bay in three days and that the residence time for the bay was 7 days. This suggested that the suspension feeders should, in theory, be able to restrict phytoplankton development. In practice however, it was found that suspension feeders consumed only 25% of the new production either because of environmental disturbance or because the water circulation did not bring sufficient amounts of phytoplankton into contact with the benthos.

The potential effects of nutrient enrichment on the benthic-pelagic nutrient dynamics are discussed by Heip (1995). On the basis of the assumption that there is an increase in primary production and therefore also an increase in the inputs of organic matter to the sediment, Heip (1995) concluded that mineralisation in the surface sediments would become more intense and that organic matter would become buried deeper in the sediment. This would lead to an upward migration of reducing conditions and anoxia in sediments and ultimately in the water column thus changing sediments from aerobic to anaerobic pathways. This may eventually cause the disappearance of fauna after following the well-defined progression (see Section 3.3.1). The process can then become self-propagating as the initial decline in fauna and particularly deep burrowing fauna alters the benthic-pelagic coupling processes, the degree of bioturbation and thus affects the nitrogen and phosphorus cycles. As bioturbation decreases, the volume of oxygenated sediment declines so does the rate of denitrification, less nitrogen is lost from the system and nitrogen flux to the water column increases and also anoxic sediment releases more phosphorus to the water column. The release of these nutrients then acts as a positive feedback mechanism prompting further primary production followed by decomposition and further oxygen depletion.

3.4 Higher Trophic Levels and Food Webs

Nutrient enrichment can lead to impacts at the highest trophic levels either directly through oxygen depletion or indirectly through alteration to the trophic structure of the ecosystem arising from changes at the lower trophic levels. For the former, hypoxic or anoxic events can have a major effect on the upper trophic levels and often commercially important species due to mortalities and behavioural changes. The following impacts to fish and shellfish species have been observed in response to depleted oxygen:

- Fishes, particularly demersal (bottom dwelling) species, can suffer mortalities or can migrate from an area depending on the nature of the hypoxic incident such as its speed and severity, thus reducing fish catches such as those observed in the Kattegat (Baden 1990; ICES 1995).
- The emergence (at $< 2\text{mg l}^{-1}$ dissolved oxygen) of species such as *Nephrops* from burrows increases the likelihood of their capture in fishing trawls which further reduces stocks (ICES 1995). This was the cause for peak landings of lobster in 1982 followed by a general decline in the *Nephrops* landings for the entire Kattegat (Baden 1990).
- Species tolerant of low oxygen levels (e.g. scavenger hagfish, *Myxine glutinosa*) remain in an hypoxic area, and other species return to a formerly oxygen depleted site possibly to exploit the increased food resource as benthic prey emerge from the sediments and become more available to mobile predators (Diaz and Rosenberg 1995).
- In estuaries, low DO regions can act as water quality barriers which prevent the migration of fish species to and through the area (Pomfret *et al.*, 1991).
- There is evidence that the susceptibility of fish to infectious diseases is greater under low oxygen conditions, for instance, lymphocystis and epidermal papillomas in Dab, *Limanda limanda*, are linked to hypoxia in parts of the North Sea and Kattegat (Dethlefsen, 1990).

As well as the impacts of oxygen depletion there is also the possibility that higher trophic levels may be indirectly affected by changes to specific trophic interactions or to the overall structure of food webs (Odum 1985). It should be noted however, that it is difficult to determine the impacts to food chain because there is a poor understanding of the interactions between trophic levels (Weston, 1990; Nilsson *et al.*, 1991; Costanza *et al.*, 1993; Parr and Wheeler; 1995). This problem is particularly evident in estuarine ecosystems where food webs are often complicated (see Figures 13 and 14) and where individual species, particularly the benthic invertebrates, are often generalists that are capable of altering their feeding strategies as an adaptation to survival in the variable environmental conditions (Costanza *et al.*, 1993). Therefore, a single species may exhibit different feeding strategies for different regions and/or over different time scales. The ragworm *Nereis diversicolor* for instance, can feed both as a deposit feeder and a carnivorous predator (Kiorbøe 1994). As Costanza *et al.*, (1993) point out, this absence of clear indications about the sources of food and feeding strategies of estuarine benthic organisms as well as the highly mobile nature of most organisms makes it difficult to monitor or measure impacts between trophic levels.

Despite these difficulties, several authors have investigated the impacts of nutrient enrichment to specific interactions between one or two trophic levels, or as wider assessments of holistic changes to the structure of food webs (Nixon 1986, 1988 and 1992; Cloern 1991;

Legendre and Rassoulzadegan 1995; Harding 1994). These studies indicate that in general terms, nutrient inputs can have:

- a beneficial effect by fuelling a system-wide increase in productivity (and/or)
- an adverse impact by changing the abundance and availability of prey species.

These are likely to be respective stages in a progression leading to environmental degradation rather than alternatives. Nixon (1986, 1988 and 1992) considered whether nutrients entering aquatic systems increase the overall productivity and concluded that there was sufficient evidence to show that a first-order relationship between nutrient supply and productivity for marine systems was 'generally correct' although the complexity of marine webs makes it difficult to predict responses at the highest trophic levels and that data are insufficient to quantify the relationships. For this, Nixon (1988) combined data from experiments in Narragansett Bay with other literature sources, and identified a linear relationship between primary production and fisheries yields for a range of aquatic systems (see Figure 15).

Other authors have also observed links between enrichment and the productivity of higher trophic levels and some, for instance, have found that increases in primary productivity were accompanied by increases in the biomass of benthic fauna (Beukema and Cadée, 1986; Josefson 1990; de Jonge and Bakker 1994; Rachor 1990; Elmgren 1989). Enrichment has been associated with increases in fisheries productivity, for example, Iverson (1990, cited by Nixon 1992) demonstrated a relationship between fish production and the amount of nitrogen incorporated into phytoplankton biomass in 'new' production in offshore waters. In coastal lagoon mesocosms, Taylor *et al.*, (1995b) also observed increases in the growth rate of juvenile winter flounder and killifish in response to nutrient enrichment.

Green (1992) reviewed the effects of organic matter on wildfowl and waders and showed that moderate increases of organic inputs enhanced the carrying capacity of intertidal areas for overwintering bird populations. Similarly, other studies reviewed showed declines in bird numbers following decreases in organic inputs although none of the studies confirmed a cause and effect relationship because of the existence of confounding factors in each case.

Boynton *et al.*, (1995) found that production of the top of the food web was only slightly enhanced by eutrophication and they concluded that the extent to which productivity is increased may only be limited. Similarly, Nixon (1986) showed from mesocosm experiments, that only a small fraction of the added nutrients appears as an increment in the production of higher trophic levels. Instead a large fraction of the primary production that is enhanced by nutrient addition is broken down by bacteria so that nutrient input and recycling increases.

Duarte (1995) considered that increased nutrient enrichment was not necessarily accompanied by a change in the overall levels of primary production. He concluded that ecosystem level effects of nutrient inputs were more likely to be associated with the qualitative changes in the structure and function of the primary producers rather than quantitative changes in the level of productivity. Benthic macrophyte communities which are adversely impacted by nutrient enrichment are important components of shallow water ecosystems which provide stable substrata for epiphytes, habitats for fauna and influence the hydrodynamic conditions by reducing currents and sediment resuspension.

Several other studies have described qualitative changes in structure of food webs in response to enrichment including, for example, the relationship between nutrient levels, the characteristics of phytoplankton blooms and the structure and functioning of food webs.

Legendre and Rassoulzadegan (1995) described the relationship between nitrogen availability, the size of phytoplankton, the type of grazers and the structure of marine food web and noted that two contrasting food webs which are generally identified:

- an 'open' and productive herbivorous web (phytoplankton-zooplankton-fish);
- an almost 'closed' microbial loop (heterotrophic bacteria-zooflagellates-DOM-bacteria).

However, the authors conclude that these two food webs represent extreme trophic pathways each with a relatively low stability and that between these extremes there is a continuum of webs from herbivorous web/chain → multivorous food web → microbial web → microbial loop. The characteristics of the food web are dependant upon the relative availability of ammonium or nitrate which affects the size of the phytoplankton. Nitrate is preferentially assimilated by large phytoplankton while ammonium benefits the production of small phytoplankton and it is preferentially assimilated when both nitrate and ammonium are present. The model, which is for marine waters, gives a clear distinction between nitrate is which is allochthonous and ammonium is autochthonous although this distinction is less clear in estuaries where ammonium inputs from external sources can be significant. The characteristics of the four web types and the influence of nutrients is described below:

- Herbivorous web/chain which occurs in nitrate-rich waters where the abundance of microbial heterotrophs is low and, therefore, nutrient recycling and concentrations of ammonium are also low. These condition are typified by blooms of large phytoplankton, large zooplankton and productive fisheries.
- Multivorous food web where herbivorous and microbial grazing modes both have significant roles, nitrate inputs are high but there is a possible feedback mechanism whereby ammonium produced by microbes reduces nitrate uptake and suppresses the production of large phytoplankton in favouring smaller phytoplankton.
- Microbial web is characteristic of a post-bloom phase where nitrate levels are low and therefore most of the available nitrogen is present as recycled ammonium; this drives the production of small phytoplankton which are preferentially consumed by protozoa rather than large zooplankton such that nutrients do not progress to the higher trophic levels.
- Microbial loop occurs under low nutrient conditions where large phytoplankton represent a small proportion of the total standing stock, zooplankton grazing is low and the bacterial biomass is possibly high; considered to be an inherently unstable condition where grazers eventually regain control of bacterial production creating the microbial web.

These features produce relationships and a conceptual model between nutrient availability, phytoplankton sizes and food web types (Figure 16).

The species composition of phytoplankton blooms, which changes from diatom to flagellate-dominated with enrichment (see Section 3.2.1), affects the eventual characteristics of the entire food web (Officer and Ryther, 1980; Doering *et al.*, 1989; Harding 1994). Officer and Ryther (1980) concluded that diatoms are the preferred food of most grazers, that they

promote transfer of energy to higher trophic levels and form the basis of those food chains which are 'beneficial to man'. They concluded that other flagellate-based food webs were undesirable either because species remained ungrazed, or because they fuel food webs that were economically undesirable.

Similar conclusions were reached by Doering *et al.*, (1989) who assessed the structure and function in a model coastal ecosystem (mesocosm). They found that an enhanced diatom flora (produced by lowering the N:Si ratio of nutrient inputs) did not reduce the extent of eutrophication, as judged by oxygen concentration and metabolism or by chlorophyll-*a* biomass. However, they did find an enhanced growth of Atlantic menhaden *Brevoortia tyrannus* under nutrient treatments which promote diatom blooms. Given the perceived benefits of diatom blooms, Doering *et al.*, (1989) suggested adding extra silicon in sewage effluent to lower the N:Si ratio of discharged nutrients and promote diatom growth over that of other phytoplankton species as a method for alleviating the problems associated with enrichment.

Harding (1994) also discussed the relationship between nutrients, the species composition of phytoplankton blooms and the food web structures in Chesapeake Bay. He proposed that the increases in phytoplankton biomass and changes in the balance of phytoplankton species would cause a change in trophic structure from a productive *Diatom-Zooplankton-Fish* food chain to a system dominated by the microbial loop. This would occur because an increase in the POM arising from greater algal biomass would stimulate microbial decomposition. Under conditions of a high biomass and net production of free-living bacteria, much of the primary productivity goes through the microbial web of bacteria and heterotrophic flagellates and this contributes to the oxygen depletion.

Malone (in prep.) highlights the speed of the response in relation to the potential impact of phytoplanktonic growth and cites the work of Caperton *et al.*, (1971) who developed a simple bottom-up food web numerical model according to which, in low-nutrient systems, productivity is driven by the rate of nutrient uptake (biomass specific) and the response to changes in nutrient supply is rapid. In high nutrient systems productivity is driven by changes in biomass and responses are slow (*i.e.* they depend on the generation time). Based on this model Malone (in prep.) concluded that the spring diatom blooms in Chesapeake bay are the primary cause of the over-enrichment problems as, during the spring survey, phytoplankton production is growth saturated and biomass limited. Therefore, allochthonous nutrient inputs arising from increased freshwater flow causes an increase in the phytoplankton biomass which leads to increased organic loading and anoxia of the bottom waters of the Bay's mesohaline region. In this case the assumption that diatom blooms *per se* are beneficial to the food webs (Doering *et al.*, 1989; Officer and Ryther, 1980) does not appear to apply.

Another mechanism by which the food webs can be affected is through the reduction in benthic fauna as a result of organic enrichment or oxygen depletion. The grazing activities of benthic fauna play an important role in modifying the structure of pelagic communities (Doering *et al.*, 1989; Koseff *et al.*, 1993; Riisgard *et al.*, 1995). Doering *et al.*, (1989) for example showed that in the presence of an intact benthos, the zooplankton had a larger meroplankton component and that the higher trophic levels were dominated by larvae of benthic adults (anemones, shrimp) or temporary planktonic adults (mysids). When the benthos was removed, the zooplankton were smaller (holoplanktonic) and higher trophic levels were dominated by ctenophores, medusae, chaetognaths and fish. According to Dame

(pers comm.), the reduction of benthic fauna resulted in nutrient enriched systems such as Chesapeake Bay and Marennes-Oleron (France), experiencing 'phase shifts' from a state where there is benthic-pelagic coupling based on filter feeding bivalves to a dominance of planktonic food webs.

In pelagic food webs, zooplankton grazing does not appear to control phytoplankton blooms as does benthic grazing. White and Roman (1992 cited in Gilbert *et al.*, 1995) found that zooplankton grazing was insignificant in the collapse of the bloom in Chesapeake Bay removing only about 5% of the phytoplankton biomass. They concluded that zooplankton grazing may actually promote phytoplankton growth through the release of urea which contributed over 80% of the nitrogen available and 50% of the nitrogen consumed in the South Bay.

The depletion of benthic fauna can also affect higher trophic consumers such as waterfowl and fish as they may be the dominant prey for these species. For example, the effects of declining prey resources on fish populations has been observed in the Baltic (Elmgren 1989) where the disappearance of the deep-water isopod *Saduria entomon* due to bottom water hypoxia may have affected the cod population for which this isopod was an important prey item. Another important prey species for the cod, the snake blenny (*Lumpenus lampetraeformis*) has become rare in the northern Baltic due to the increased hypoxic area (Larsson *et al.*, 1985). This means that alterations to the benthic communities in the Baltic will favour the survival of economically less-important fish species such as roach rather than perch, pike and salmonids.

With respect to impacts on wildfowl, it has been difficult to identify a clear cause and effect relationship with declines in benthic fauna because of the variability of wildfowl feeding activities. However, Goss-Custard *et al.*, (1991) have identified a correlation between the numbers of feeding waders and the abundance of their principal prey species which suggests that avifaunal species are susceptible to impacts at the lower trophic level.

In addition to the effects of declining oxygen levels, blooms of intertidal macroalgal mats may affect the feeding activities of waterfowl and fish species either because of the impacts that these mats have on the underlying benthic fauna or because the algae present a physical obstacle to feeding. Elliott and Taylor (1989) demonstrated that the intertidal areas of estuaries have a disproportionately greater importance for supporting invertebrate prey than the subtidal regions such that any impacts to prey or feeding activities in the intertidal regions have a greater likelihood of effect on higher trophic consumers.

In assessing impacts of macroalgal mats, Soulsby *et al.*, (1982) demonstrated that the development of macroalgal mats in Langstone harbour changed the underlying macrofauna. There was an overall increase in the biomass of some macrofauna largely because *Hydrobia* was present in larger numbers, but a decline in the surface feeding polychaete *Streblospio shrubsolii*. The changes to the macrofauna were observed when the algal mats covered between 25% and 75% of the mudflat and it was hypothesised that change occurred when coverage reached 50%. The study was unable, however, to provide conclusive evidence of a causal link between macroalgal growth and the numbers of feeding wild fowl and concluded that any major impact to the waterfowl numbers and fundamental changes to the ecology of the mudflat were only likely to arise when the algae covered the entire mudflat system.

Nicholls *et al.*, (1981) also examined the effects of algal mats in Langstone Harbour on invertebrate abundance and bird feeding activities. They observed a decrease in the biomass and diversity of mud dwelling infauna in areas covered by green algal mats. This was due to the anoxic conditions prevalent below the mat and as a result only *Tubificoides benedeni*, *Capitella capitata* and *Scolecopsis pinnatus*, all characteristic of stressed conditions, remained present in high numbers. For epifaunal species there was an increase in the abundance and biomass but despite this the algal mats were still avoided by feeding wildfowl and waders. This was attributed either to the physical difficulty of feeding in algal mats or the high levels of H₂S which adversely affects the palatability of the sediments and prey.

Raffaelli *et al.*, (1989) found that the polychaete *Capitella capitata* (a species indicative of enriched and anoxic environments) became more abundant under the mats while other species such as *Cerastoderma edule* declined. They concluded that major impacts from enrichment are likely if these algal mats become contiguous and, therefore, leave no weed-free refuges for the amphipod *Corophium*, a major wader prey species. This species may disappear from large areas of the estuary leading to detrimental impacts to fish and bird populations as observed in Langstone Harbour by Tubbs (1977).

Sundback (1994) tested the impact of macroalgal mats *Enteromorpha clathrata*, at low and high abundance's (0.9kg and 1.8kgm⁻² fresh weight respectively), on the benthos. There was no observable effect at the low abundance's under which conditions the benthic environment was autotrophic (*i.e.* photosynthesis exceeded respiration P>R). However, at the high abundance level there was decline in benthic productivity and in the abundance of microflora, ciliates and meiofauna but no change in bacterial numbers was recorded. The authors observed that this macroalga strongly affected the benthos due to the effects of shading and oxygen depletion. They also investigated whether a flow of carbon to the sediment could cause an impact but they did not record any such flow and concluded that the macroalgal mats can be considered to be an 'independent habitat'.

Hull (1987) also tested the effects on the benthos of different densities of *Enteromorpha* spp. (0, 0.3, 1 and 3 kg Fwm⁻²) and found that in summer the numbers of *Corophium* were reduced at all weed levels and a few species such as: *Macoma balthica*, *Nereis diversicolor* and *Capitella capitata* increased in abundance in sites with algal stands. In winter when the algal stands became covered with sediment the differences between sites was less clear. *Macoma*, *Nereis* and *Capitella* were significantly more abundant in the medium and high weed densities and the population structure of *Corophium* were different. The medium and high density plots also showed rapid anoxia and significant siltation.

In addition to the effect of macroalgae on the underlying benthos there is also the potential problem that these mats will physically interfere with the birds feeding behaviour. Tubbs (1977) examined the changes in the abundance of 13 waterfowl species in Langstone Harbour. Of these, nine species increased in numbers in line with populations internationally, but three species Redshank, Shelduck and Dunlin, had shown an 'anomalous decline' which was attributed to the blanketing growth of *Enteromorpha*. For Shelduck, which feeds by skimming their beaks along the surface, the weed may provide a significant obstacle to feeding. Although Redshank and Curlew do not feed in this manner both species largely avoided feeding on the weed mats. In the Ythan, while most shorebirds have also increased in abundance, with the general increase in productivity only Shelduck have declined (Raffaelli *et al.*, 1989).

Conway (1993) states that there is controversy over the evidence for a link between the spread of macroalgae and the decline in shorebird numbers, and that it would be unwise to assign a simple causal relationship in this case due to the many variables involved when examining bird population dynamics. She concludes that a more information is required before a relationship between nutrients, increased primary productivity, macrofaunal abundance and shorebird numbers can be confidently assumed. Kemp *et al.*, (1984) found a high correlation between the declines in abundance of both vascular plants and waterfowl species. They also predicted numerous other attendant impacts occurring through the loss of this habitat type such as declines in the abundance and diversity of fish and epifaunal species which are associated with vascular plant communities.

Estuarine fish are highly dependant on intertidal regions as sources of prey (McLusky *et al.*, 1992) which suggests that benthic macroalgal mats could have a major impact on fish feeding. However, little work has been carried out on the effects which macroalgal blooms have on fish populations probably because of the practical difficulties associated with monitoring fish species over mat areas and clearly identifying their feeding sites and consumption efficiency. Cullen (1992) cites the following authors who have identified possible connections between the macroalgal growth and fish populations:

- Fitzgerald *et al.*, (1978) observed that seasonal migrations of fish influenced the standing crop of *Enteromorpha* and presumably benefited from the increased supply of food that the algae represented.
- Raffaelli *et al.*, (1991) suggested that a decline in the abundance of *Corophium volutator* associated with an increase in macroalgal cover would significantly affect Flounder which are known to feed heavily on this amphipod species.
- Perkins and Abbot (1972) assessed the effect of macroalgal blooms in the Clyde Estuary and concluded that although fish may survive, a fishery may decline due to the mortality of suitable food organisms in the unfavourable conditions which result from the excessive addition of nutrients.
- Rosenberg (1985) identified a potential link between macroalgal growth, oxygen deficiency in the sediment and the mortality of benthic fauna and demersal fish.

Nutrient related impacts at the highest trophic levels may be shown by incidents where the transfer of toxins, derived from blooms of toxic algal, passes along the food chain. Most notably problems arise from the human consumption of shellfish species which have bioaccumulate algal PSP or DSP toxins and Armstrong *et al.*, (1978 cited by Lyons *et al.*, 1993), has also linked incidents of PSP with seabird fatalities. However, the extent to which enrichment prompts the blooming of toxic algae remains unclear (De Jonge *et al.*, 1995).

Some evidence for a relationship between these two factors is presented by Conley *et al.*, (1993) who concluded that silicate concentration may be an important component in the formation of toxic algal blooms through the production of the neurotoxin domoic acid. Conley *et al.*, (1993) cite studies correlating the production of domoic acid by the diatom *Nitzschia pungens f. multiseries* and a related diatom *Pseudonitzschia australis* with shellfish poisoning (Bates *et al.*, 1991) and brown pelican mortality (Buck *et al.*, 1992). Conley *et al.*, (1993) also cite Smayda (1990) who conclude that declining Si:N and Si:P ratios, which result from increasing inputs of nitrogen and phosphorus, are responsible for altering the species composition of phytoplankton bloom and increasing the incidents of blooms by novel

and toxic species. This was proposed as the mechanism which produced exceptional *Chrysochromulina polylepsis* blooms in Scandinavia (Maestrini and Graneli 1991).

In addition to the above examples which mainly describe the impacts to the highest trophic levels, studies have examined the effects of nutrient enrichment on the trophic interactions within the benthic communities (Widbom and Elmgren 1988; Nilsson *et al.*, 1991; Sundback 1994; Pinckney *et al.*, 1995).

Nilsson *et al.*, (1991) and Sundback (1994) found that the biomass of both the microalgal and meiofauna increased following nutrient enrichment and that oligochaetes and harpacticoids became increasingly dominant components of meiofauna. This may have benefits for the productivity of higher trophic levels because, as Nilsson *et al.*, (1991) points out, these two groups of meiofauna are the ones most commonly eaten by macroepifaunal species such as flatfish and crustacea. However, Widbom and Elmgren (1988) identified some potentially major changes to the benthic food web following a longer term study (over 2.4 years) which showed that, with prolonged exposure, nematodes and polychaetes increased whereas harpacticoid copepods decreased possibly due to predation pressure of macrofauna.

Another potentially adverse impact can occur if benthic fauna fail to control the growth of autotrophic species through grazing in which case organic material is not channelled through the higher trophic levels but instead decays and contributes to organic enrichment problems. . Nilsson *et al.*, (1991) and Sundback (1994) however have both demonstrated that meiofaunal grazing fails to control the growth of microalgae. Similarly, Sundback (1994) also showed that grazing by two macrofaunal species, the filter feeding cockles (*Cerastoderma edule*) and the predatory brown shrimp (*Crangon crangon*), did not control microalgal growth in sediments. Instead Sundback (1994) found that cockles, and possibly also the brown shrimp, stimulated the growth of benthic microalgae through the excretion of ammonia. The presence of brown shrimp in sediment was also found to contribute to growth of microalgae because the meiofaunal grazing rate (which has already been shown to have only a minimal impact on the microflora) decreased in their presence.

Sundback (1994) therefore, proposed that changes occurring to benthic food webs in response to nutrient enrichment would become self-accelerating because the increase in nutrients leads to an initial increase in productivity and a higher macrofaunal biomass. The greater macrofaunal biomass then increases nutrient cycling and further stimulates primary productivity. With these elevated levels of nutrient input, a rapid growth of filamentous algae occurs at the expense of macrofauna and with sustained nutrient loads there is an increase in organic matter loads and a decrease in macrofaunal abundance.

3.5 Recovery from Enrichment Impacts

In addition to understanding how estuarine communities are affected by nutrient enrichment, there is a need to understand how communities recover from nutrient impacts in order to effectively manage estuaries. This is particularly important for determining the potential effectiveness of measures designed to reduce nutrient inputs. However, there is a poor understanding of the way an estuary recovers from impact mainly as there are few examples of systems showing recovery, although there are studies of the response of communities to reductions in chronic and acute nutrient inputs. A major concern with respect to the recovery of communities is that many of the responses to nutrient enrichment have been found to be self-perpetuating *i.e.* once they begin to occur they create conditions which further create deterioration. Examples of this include:

- changes in vegetation type (Duarte 1995);
- the uncoupling of nitrification/denitrification (Harding *et al.*, 1992);
- reduction in benthic-pelagic coupling (Heip, 1995);
- changes to benthic food webs (Sundback, 1994).

Thus once threshold levels of impact are reached, then these processes and associated detrimental impacts will continue without necessarily any further increase in the nutrient inputs. What is unclear however, is the effect of seasonal changes in environmental conditions and biological responses in controlling these self-perpetuating processes. The natural seasonal cyclicality within temperate estuaries may allow some communities to recover and to halt the progression of some or all of these self-perpetuating processes (seasonal impacts). For example, in winter nitrification/denitrification coupling can recommence because strong currents reoxygenate the sediments and allow nitrification to proceed. However, some responses may only be briefly interrupted by the winter months with the process then continuing the following summer (long-term impacts), for instance the decline in seagrass coverage due to impacts of algal growth. It is unlikely that a seagrass bed will recover during the winter period and if no improvement in nutrient load is achieved and the same rates of algal growth occur the following year then a decline in seagrass is likely to continue the following summer. This may apply similarly to the decline in the coverage of saltmarsh species and perennial algae such as fucoids which may show a decline in coverage over successive years due to competition from opportunistic species in the summer, but with little or no winter regrowth.

The effect of seasonal changes is also unclear for other factors such as macroalgal coverage of bare mudflats. For the macroalgal growth, if holdfasts of algae are retained in the sediment over the winter then an alga which settled and flourished the previous summer may be sufficiently established. If strong winter currents cause the resuspension of sediments and the removal of the holdfasts then each year the process would need to recommence.

Several authors have reached differing conclusions regarding the abilities of estuaries to recover from chronic and acute nutrient inputs (Boynton *et al.*, 1982; Nixon 1987; Kemp and Boynton 1992; Costanza *et al.*, 1993; Harding 1994; Legendre and Rassoulzadegan 1995; Duarte 1995). An example of an estuary's response to acute nutrient loading is the response of Chesapeake Bay following Hurricane Agnes in June 1972 in which all of the Bay's 64,000 square mile catchment received 5 inches of rain in less than three days. This resulted in major

nutrient input to the estuary (Malone *et al.*, 1993). Boynton *et al.*, (1982) found that the majority of this nutrient load was retained within the sediment and progressively released over subsequent years resulting in unusually high phytoplankton production. Costanza *et al.*, (1993) concluded that the recovery of the Bay or its 'memory' of this event was less than five years. In another study, Harding (1994) examined the response of the Bay to seasonal peak nutrient inputs and found that it had short term memories to acute inputs. He found that periods of high freshwater flow could occur too early in the season to support a phytoplankton bloom and so proposed that the 'memory' of the Bay did not extend to freshwater events that occurred more than 4 months prior the period when blooms usually occurred (March to May).

Although these studies provide an indication of the responses to acute inputs, for management purposes it is probably more important to understand the impacts of long term chronic inputs of nutrients and the biological response to reductions in these inputs. Costanza *et al.*, (1993) described the changes which result from chronic nutrient loading to Chesapeake Bay. These include a decline in the:

- fisheries productivity and habitat diversity;
- clarity of the water column;
- submerged aquatic vegetation (SAV);
- oyster beds.

This estuary appears resilient to this long term chronic enrichment and workers observed that many plant species rapidly returned (within < 5 years) to the tributaries following reductions in the pollution load. Duarte (1995) however found that little information is available on the response of vegetation to reduced nutrient levels although he points out that the recovery of seagrasses, in particular, is likely to be slow and require periods of a few years to centuries. Nixon (1987) suggests that in order to understand the effects of chronic inputs it is necessary to know whether the system retains nutrients from year to year which will affect the value of managed nutrient reductions. To answer this question for Chesapeake Bay, Nixon (1987) produced a nutrient budget for this system and found that only 3-6% of nitrogen, 11-17% phosphorus and 33-83% silicon were retained in the system. He concluded that, if this is the case, the observed impacts of enrichment are a consequence of present nutrient inputs rather than a response to cumulative inputs over the longer term. Thus the management of nutrient loadings may lead to a more rapid improvement in environmental conditions than would otherwise be the case if a large pool of nutrients was retained within the estuary.

Kemp and Boynton (1992) believed that because benthic nutrient recycling is important for supporting primary production in Chesapeake Bay, then high nutrient inputs in one year may have biological responses in the following year. However, they still considered that these nutrients were not retained for long periods in the estuary and that ecological processes can be expected to respond rapidly to reductions in nutrient inputs. Harding (1994) also considered the relationship between phytoplankton blooms and nutrient loads in Chesapeake Bay and observed little carryover effect of nutrient loading between years. Harding (1994) concluded that 'significant reductions in nutrient inputs from the watershed can be expected to elicit fairly rapid decreases in the abundance of phytoplankton'. However, he expected a 40-50% reduction would be required to achieve small increases in summer levels of bottom water oxygen.

Legendre and Rassoulzadegan (1995) studied food web structures in the open ocean and considered that the microbial food web may be a continuous system that retains nutrients and links two successive spring phytoplankton blooms.

Kemp *et al.*, (1992) considered the potential effect of reduced nutrient inputs on the DO concentrations in Chesapeake Bay and found that plankton were the major contributors to oxygen depletion and that a reduction in nutrient load would rapidly produce improvements in the oxygen conditions by reducing phytoplankton abundance. However, they also observed a relationship between the physical conditions which replenish oxygen and the overall biological response so they suggested that any improvements in the response would be buffered by this interaction. Therefore, they concluded that 'any reduction in nutrient loading would result in rapid but proportionately smaller increases in oxygen'.

In highly polluted systems which are subject to excessive nutrient impact, it is possible that a reduction in nutrient loads would not be beneficial. This appears to be the case in the Scheldt for example, where any decreases in the waste input is not expected to result in improvements in water quality because of the natural phosphate-buffering processes in the estuary (Zwolsman, 1994) or because denitrification rates, and losses of nitrogen from the system, would be reduced (Billen *et al.*, 1985).

3.6 Summary and Conclusions

Impacts on Autotrophs and Bacteria

1. This review focuses upon evidence from experimental work and field observations to examine the response of autotrophs and bacteria to anthropogenic nutrient input and their role nutrient cycling. The interaction of the effects of nutrients on autotrophs and heterotrophs and vice versa are described with special attention given to uptake processes, and the primary and secondary responses. The central role of phytoplankton, in non-light limited environments, in the uptake of nutrients is identified. Food chain interactions resulting in bottom-up and top down controls are elucidated. Within each major taxon, the sub-groupings have differential responses to nutrient enrichment and again such responses are known qualitatively but quantitative information is poor and often non-predictable. The different species of nitrogen have differing bioavailabilities and use although the biogeochemistry is well understood.
2. The macrophytes play a central role, especially in mat formation, in sequestering nutrients and preventing transport to the coast. They may act to reduce recycling but they ensure that decomposing plant material is available to the estuarine detritivores. Nutrient recycling and hence re-liberation is moderated by sediment and water-column bacteria. There is a growing recognition of the importance of bacteria and microphytobenthos in recycling.
3. There is a good conceptual understanding of the features affecting each category of autotrophs and heterotrophs, of the role and importance of micro- and macrophytic mats, of the water-interface-sediment interactions, of the differing competitive advantages of certain producer groups under specific estuarine conditions and of the interactions between such groups, e.g. shading effects. Of greatest significance is the recognition that bloom-forming unicellular algae are at the end point in each sequence identified as a response to increasing nutrient inputs and recycling. However, as illustrated with many features, the quantitative prediction of the responses is poor and difficult even on a site-specific basis.

Impacts on Benthic communities

1. This component shows perhaps the most severe impact of organic enrichment as the result of excess nutrient and thus has implications for water quality through the production of sediment-derived toxins, in addition to having extensive effects on the conservation of sensitive species and habitats, for example by affecting prey availability and palatability and affecting the passage of migrants over the bed.
2. The macroinfauna and epifauna provide the dominant prey for demersal fishes and estuarine birds. The extensive literature indicates that the duration, frequency and timing of anoxic events are important and there is a good understanding at the conceptual and quantitative level.
3. The effects of increases in nutrients are difficult to separate from organic matter effects at the benthic community level although the responses differ only in their severity and the distance along the Pearson-Rosenberg model to which the community is changed.

4. Once changed, the community then affects the cycling of nutrients. The benthos has a large control on sediment dynamics and biogeochemistry and therefore the major role in the re-liberation of nutrients under certain conditions. The extent of benthic-pelagic coupling is shown by nutrients affecting the benthos and vice-versa but as two main components - the sessile epifauna, especially bivalve beds and biogenic reefs, which increase the flow of water-borne components to the bed including biodeposition, and the macroinfauna through both deposit and detritus feeding and bioturbation.
5. In addition, the macroinfauna can have a stabilising effect on sediments thus affecting diagenesis and nutrient recycling. The biogenic reefs may affect transport from sediment to water but this is likely to be outweighed by the excretion of by-products in a readily-assimilable form for primary producers.
6. In general, there is a well-defined benthic conceptual model of the response to organic enrichment although the nutrient input-organic enrichment links and effects on the benthos are less well quantified. The component has a major significance for environmental quality considerations, especially the bed and the water column, and nature conservation considerations.

Impacts on Higher Trophic Levels

1. There is conflicting evidence for the manner by which predators respond to changes further down the trophic structure although certain main features are accepted by workers.
2. The initial organic enrichment leads to an increase in production and hence benefits for predators but then over-enrichment causes deleterious change to the diversity, abundance and availability of prey. The latter are acceptable until enrichment progresses to form toxins as the result of the production of water column blooms or sediment H_2S or CH_4 or until barriers (of water quality in the water column or physically on the sediment surface) are formed.
3. The major response to organic enrichment of oxygen depletion will lead to toxin production as well as changes in prey type and quantity at lower trophic levels. There are then concomitant direct effects on fishes as poor health, including disease, or mortalities, migration behavioural changes as avoidance of sedimentary or water column areas, or the disruption of prey populations and feeding behaviour.
4. With respect to birds feeding, there is conflicting evidence but in general there is likely to be an initial increase in bird carrying capacity although most of the excess nutrients get recycled through lower trophic levels rather than assimilated to higher level production.
5. There appears to be a change of food web characteristics and dependence from herbivory to the microbial loop although it is considered that this should be less of a problem for detrital-based estuarine systems. However, the modification of food chains plus the diversion of material to lower levels is likely to have repercussions for fisheries.
6. The severe benthic changes resulting from anoxia of subtidal areas may lead to a reduction in grazing by suspension feeders and hence in top-down control on the microalgal system

but also the intertidal sediment anoxia resulting from algal mat production will provide less prey-availability for predators. It may also affect the palatability of prey. These effects may have wide consequences given the disproportionate importance of the intertidal area. However, some mat areas may encourage wildfowl rather than waders.

7. The complex nature of estuarine food webs dictates that prediction to higher levels of the results of change to a lower trophic level is difficult. Similarly, the tendency by many estuarine predators for opportunistic feeding may allow compensation for changes to certain prey species.
8. The information on the carrying-capacity of estuaries for predators is poorly defined such that it is not possible to define the amount of area, either in the water column through poor water quality, or as intertidal or subtidal bed, that could be lost to predators as the result of organic enrichment and yet the estuary remain to function as expected.
9. Hence, although there are sufficient case-studies for a conceptual model of change especially on a site-specific basis, it will be difficult to extrapolate across estuaries.

Recovery from impacts

1. There are few studied examples of estuaries recovering from nutrient enrichment such that there is conflicting evidence at best and at worst a poor understanding of the processes.
2. There may be self-perpetuating mechanisms, carried over from one year to the next, which indicates that nutrients are trapped suggesting the hypothesis that estuaries are sinks rather than sources.
3. There is little indication of the mechanisms necessary to trigger both the negative response and the recovery process, for example whether thresholds are required to cause changes in primary producers.
4. There is the need for better studies of the times for nutrient retention and for the recovery of areas, especially in order to separate the responses to short (acute) and long-term (chronic) inputs.
5. Of concern are the few cases indicating that even with management initiatives, there are doubts regarding the possibility and nature of recovery.
6. A useful summary of the key effects of nutrient enrichment in estuaries was produced by NOAA (NOAA 1993) and this is shown in Figure 17.

4. AVAILABLE TOOLS FOR MONITORING & MANAGEMENT

Section Aim

- To review available techniques for monitoring and managing nutrient impacts

Section Contents

Section 4.1 Data Adequacy:

A review of the value of available data for describing the impacts of nutrients based on the opinions of authors expressed in the published and unpublished literature.

Section 4.2 Tools:

A review of the techniques that are most commonly used by researchers and monitoring authorities:

- **Field survey techniques (Section 4.2.1):**
- **Analyses of Nutrient Dynamics (Section 4.2.2):** Techniques for identifying nutrient-limited plant growth, Nutrient/salinity plots and Nutrient budgets.
- **Models (Section 4.2.3):** A review of empirical and stochastic models that have been developed to describe nutrient inputs and biological responses to nutrients in estuaries.

Section 4.3 Approaches to Management of Estuaries:

A review of the most common approaches towards the management of nutrient impacts in estuaries which includes:

- an approach towards a national assessment of estuarine eutrophication (Section 4.3.1);
- estuary classification schemes and water quality targets (Section 4.3.2);
- environment Agency National Strategy For Eutrophication Control (Section 4.3.3);
- ecological quality targets.(Section 4.3.4).

Section 4.4 Hindcasting Nutrient Status:

A review of techniques for hindcasting nutrient concentrations in estuaries and evaluating the extent of anthropogenic interference:

- taking contemporary measurements (Section 4.4.1),
- modelling nutrient exports from catchments (Section 4.4.2),
- using palaeoecological techniques (Section 4.4.3),
- developing empirical models (Section 4.4.4).

4.1 Data Adequacy

The Phase 1 of the 'Impact of Nutrients in Estuaries' project workshop concluded that the understanding of nutrient impacts is constrained by a lack of knowledge about the biological effects of nutrients and that there are insufficient data to make sound judgements on their ecological impact in estuarine environments. This problem is identified by many authors studying the biological impact of nutrients in estuarine systems despite the fact that the relationship between nutrients and responses in aquatic environments has been the subject of extensive scientific research, and the focus of analyses by monitoring authorities, throughout the UK and elsewhere. To understand the reasons for this evident disparity, this section reviews data and discusses the current lack of understanding identified during the workshop.

4.1.1 Data Adequacy

Data collected in English and Welsh estuaries are often unsuitable for identifying nutrient impacts because the Environment Agency do not conduct surveys with the specific aim of describing the trophic status of estuaries. Therefore any data that are currently used to identify nutrient impacts are being used for a purpose for which they were not originally intended. To obtain the necessary data, a specifically designed and standardised sampling methodology is required, particularly in estuaries where there is a need to take account of the temporal and spatial variability of this environment. However, no such monitoring programme presently exists for estuarine systems. Instead, the present monitoring strategy involves separate independent nutrient monitoring programmes in each Environment Agency region along with some ad-hoc nutrient monitoring carried out in the major estuaries as part of the National Coastal Baseline Survey.

An example of the problems which can arise is provided by the investigation by Elliott *et al.*, (1994) on the trophic status of four Suffolk estuaries. This study used data from the NRA Anglian Region and although these estuaries had been studied extensively, specific studies of the trophic status had not been carried out and, as a result, the following difficulties were encountered:

- no turbidity measurements were available so a specific turbidity survey had to be undertaken;
- no information was available on silicate concentrations;
- the assessment of: diffuse nutrient inputs, estuary dynamics, and residence times required several assumptions;
- although chlorophyll-*a* measurements were available there were no data on primary productivity and only anecdotal evidence of phytoplankton blooms was available.

Several other studies have identified problems with available data collated from the Environment Agency and also from other sources across the UK and abroad. The findings from these studies are reviewed below.

Two major reviews have been made of the adequacy and availability of nutrient monitoring data for UK estuarine and coastal waters by Rees *et al.*, (1994) and Gunby *et al.*, (1995). Rees *et al.*, (1994) found very few useful data from published sources, so they concentrated on analysing unpublished data from NRA regions, universities and other research

establishments. From these organisations they requested data on nitrogen and phosphorus, together with 'interpretation parameters' such as DO, salinity and temperature. They found four main problems with collating and using the available data:

- most of the data were for large contaminated estuaries and very few data were from anthropogenically 'unaffected' estuaries;
- there was no standard sampling strategy and often factors such as tidal state were not considered such that the data were often incompatible;
- there were no standard analytical procedures, and nutrient levels were measured in a variety of fractions and units;
- many of the data were sent as hardcopy and data sent on diskette were in various different formats.

Gunby *et al.*, (1995) collated a large amount of information on nutrient levels (ammonia, nitrate, nitrite, and/or TON SRP) and environmental variables (salinity and temperature) from various sources including all the NRA regions, JoNuS and surveys. The authors found that the classification scheme was severely constrained by a lack of comparable data from UK estuaries and that very few 'appropriate data' (particularly winter nutrient measurements) were available for the purposes of their classification. They also found that samples were not necessarily taken under the standard required tidal conditions *i.e.* high tide during a spring and neap, or that the data were not accompanied by a record of the tidal state.

Data collation was also carried out by Parr and Wheeler (1995) who were studying the impacts in 12 proposed estuary Sensitive Marine Areas (SMAs) for English Nature. These authors searched published literature for the previous seven years together with unpublished 'grey literature' from the following sources: WRc Assassin Database, English Nature internal literature catalogue, 'Aqualine' and 'Biosis' commercial databases and the Plymouth Marine Laboratory's annual bibliography of tidal waters. They found that most of the grey literature available had not been included in these databases and they were also aware of other work that had been undertaken but which was difficult to access or which remained subject to client confidentiality. They found that the major obstacle to identifying the impact of nutrients was the availability and quality of the biological information. Their review examined changes in the community structures, concentrating on the species, which are believed to be the most sensitive to change but they found that it was difficult or not possible to determine whether a trend in species diversity and abundance had occurred. The main problem with the data was that records, when they were available, were often taken in differing areas of the estuaries and it was therefore difficult to collate the findings from separate studies.

Further problems with the collation of data from NRA monitoring programmes were also identified by Parr and Wheeler (1995) who did not approach the NRA for biological data because, based on their previous experience, they were aware of three potential problems:

- the length of time needed to collate the data;
- the difficulties of ensuring that a uniform taxonomic approach has been used between different NRA regions;
- the conversion of older manuscripts to a usable digital format.

Problems with data adequacy have also been observed in other European and USA estuaries. In the USA the National Oceanic and Atmospheric Administration (NOAA) gathered information to assess the status and trends of nutrient enrichment in estuaries on a national basis. This involved the collation of data from sources across the US and provides a good example of the problems associated with the collation and interpretation of data. Initially, data were requested from 400 sources but only 25% of the participants responded so a series of site visits and regional workshops were undertaken to collate more data (NOAA Draft Report 1995). Several problems were encountered with the data once it had been collected, particularly there was little information on water quality and biological responses over corresponding time-scales making it difficult to accurately identify the cause/effect relationships between nutrient loadings and primary productivity.

In Europe, Perdon (1995) collated data from 192 published papers on a range of physical, chemical and biological parameters (a list of these parameters is shown in Table 2) in 47 estuaries across Western Europe. He found that the major obstacle to identifying nutrient impacts in European estuaries was the lack of available biological information.

Table 2: The parameters collected by Perdon (1995) to compare 47 estuaries in western Europe on the basis of their susceptibility to eutrophication.

Physical characteristics	Chemical factors	Biological responses
Depth	Total P input at river-estuary transition	Phytoplankton Production
Surface Area	Total N input at river-estuary transition	Microphytobenthos Production
Volume	Nutrient Load (<i>mmol/m²/yr</i>) <i>calculated from above, surface area, river discharge and atomic mass of N and P.</i>	Chlorophyll- <i>a</i> in sediment
River Discharge		Chlorophyll- <i>a</i> in water
Sea Salinity		<i>As limited data were available</i>
Estuary Salinity		Primary Production <i>was also</i>
Flushing time <i>calculated from volume, salinity and river discharge</i>		<i>calculated from algal growth rate and flushing time</i>
NB Not all the above parameters were available for all of the 47 estuaries under investigation		

Additional to the problems of describing the contemporary nutrient/trophic status of estuaries there are further difficulties with describing the historical changes over time because of the lack of long term data (a list of the few studies which include long term datasets are shown in Appendix 3). Several authors, in the UK and overseas, have identified the paucity of long term data as the reason why there is a poor understanding of many aspects of estuarine ecology (Baird *et al.*, 1985; van Impe 1985; Cloern 1991; Nixon 1992; Heip 1995). However, even where historic datasets are available, as they are in well studied estuaries such as Chesapeake Bay (Cooper and Brush 1993; Harding 1994); the Tay (Balls *et al.*, 1996) or the Wadden Sea (de Jonge and Essink 1991), the trends can be obscured by the large seasonal and interannual variations in the environmental conditions and the biological responses.

In addition to the limited availability of data, problems of data compatibility can also arise because of a lack of clarity about the best approach for carrying out surveys and resulting differences in the techniques used for measurement and analysis of parameters.

Some examples of possible problems are:

- there is debate about the efficacy of using certain parameters, for instance, the value of chlorophyll-*a* as an indicator of planktonic productivity (Cartwright *et al.*, 1993; Elliott *et al.*, 1994; Parr and Wheeler 1995; Malone, in prep);
- there may be problems with experimental procedures for describing particular biological processes, for instance, the best approach for measuring denitrification (Kemp *et al.*, 1990; Nowicki 1994 and Ogilvie *et al.*, in prep.);
- there may be differences in the field and laboratory practices used by different organisations, for instance, Gunby *et al.*, (1995) observed different approaches towards the measurement of freshwater phosphorus levels by different Environment Agency regions and Mathieson and Atkins (1995) found that the seven Scottish RPB's analysed different nutrient species and used different units.

4.2 Monitoring Tools

This section will review the techniques that are most commonly used by researchers and monitoring authorities. Two categories are considered:

- **Field Survey Techniques:** This provides a general discussion about possible field survey techniques although detailed review of all available techniques has not been undertaken as details can be readily obtained from the cited literature, the relevant authors or from biological sampling manuals (e.g. Kramer *et al.*, 1994). However, key examples of approaches towards the measurement of nutrient concentrations and biological responses are taken from the literature.
- **Analyses of Nutrient Dynamics:** This is a review of some of the techniques that are most commonly used by researchers and monitoring authorities for either identifying whether plant growth is nutrient limited or for describing the behaviour of nutrients in estuaries.

4.2.1 Field Survey Techniques

Field measurements of nutrient levels and accompanying physical data (e.g. turbidity, currents) can be obtained by either:

- **individual water quality sampling at discrete locations; or**
- **the deployment of continuous recording equipment.**

The former allows the best spatial coverage while the latter provides a good temporal description. The majority of researchers have used the former approach with varying levels of temporal and spatial sampling intensities. The FRPB (now SEPA) have used the latter approach in the Firth of Forth by deploying a semi-continuous Multi-Parameter Water Quality Monitoring System (MPWQMS) which measures a range of parameters: chlorophyll-*a*, nitrate, light attenuation, salinity and water temperature at a predetermined variable rate. This technique provides detailed temporal information and has been useful in describing the complexity of changes in phytoplankton blooms in this area (FRPB 1995).

The individual water quality sampling approach requires several factors to be considered when designing the surveys because of the complexity of the estuarine environment, for example the number and location of sample positions and the timing of the surveys. An example of the criteria that can be taken into account in the selection of sites is provided by the Chesapeake Bay Monitoring Study:

- sites were selected to describe conditions in segments of estuary with distinct circulation, salinity and geomorphology characteristics;
- sites were selected with particular water quality problems e.g. with incidents of hypoxia and areas adjacent to point source discharges;
- sites were selected close to areas of ecological importance to accumulate evidence on the links between biological responses and nutrient levels;
- sites were selected if they already had available historical records to ensure that there will be a continuation of the long-term water quality analyses.

Gunby *et al.*, (1995) also recommended practices for *in situ* nutrient measurements to be used as part of their nutrients General Quality Assessment (GQA) scheme (see Appendix 7). Protocols for nutrient measurements are also contained in Strickland and Parsons (1968) and further references containing methodologies for the analyses of nutrients and biological responses, which have been taken from the UK contributions to the Oslo and Paris Conventions, are shown in Appendix 10. In addition, an extensive range of techniques could be used to identify the biological responses at all trophic levels including surveys of:

- **planktonic blooms**, to describe changes to the scale and species composition of blooms;
- **intertidal algal growth** to identify changes in the coverage of intertidal algae;
- **benthic communities** to assess impacts to benthic communities of organic enrichment and/or incidents of oxygen depletion;
- **fish populations** to describe the impacts of intertidal algal growth or DO sags on the feeding behaviour or migration of fish communities;
- **avifaunal populations** to describe impacts of intertidal algal growth on the behaviour and abundance of waterfowl species in intertidal areas.

The particular biological indicator studied will be dependant upon the characteristics of the system and the perceived impacts and the protocols for these surveys will also vary depending on the level of detail required. For example, to determine macroalgal growth in intertidal areas, several techniques could be used which vary greatly in terms of their coverage (temporal and spatial) and cost:

- **'Skilled-eye' *in situ* surveys** of algal coverage by an experienced surveyor are comparatively low cost such that several individual surveys can be carried out and, therefore, that the temporal coverage is good. This procedure also allows detailed analyses of parameters such as: species composition, mat thickness/wet weight but the spatial coverage may be poor.
- **Aeroplane Surveys** describes the spatial coverage of algae very well but does not provide detailed information of algal mats (species composition etc.) and, as this is a relatively costly procedure, the temporal coverage is likely to be poor.
- **Airborne remote sensing techniques**: This is the most costly procedure which offers an opportunity for obtaining detailed spatial coverage on a whole range of physical, chemical and biological characteristics. For example the CASI equipment, used as part of the Land-Ocean Interaction Study (LOIS 1994), could be used to survey the suspended and intertidal sediments, plant pigments (chlorophylls), DOC and land biota types. However, the temporal coverage is likely to be poor.

Investigations of biological response can also range in detail and cost from studies of responses by individual species or communities to broad-scale assessments of the whole trophic structure of a system. The latter, however, are costly and would require a large amount of data and an extensive range of biological studies. Sufficient data are available from the heavily studied Baltic and Chesapeake Bay systems and, using these data, Wulff and Ulanowicz (1989) were able to compare the ecosystem-level impacts of these two systems (see Appendix 6).

It is considered here that the most appropriate approach to investigations of marine ecosystem health, and the one recommended by Harding (1992), involves integrated assessments on varying scales from individual to ecosystem level. This integrated assessment approach has a very high cost but has been used on some systems in Australia such as Port Philip Bay and Perth Coastal Waters (Hillman *et al.*, 1995). The Perth Coastal Waters Study (PCWS) was designed to assess the environmental impact of changes to the level of wastewater-discharged nitrogen in Perth's coastal waters and involved studies at three levels:

- **laboratory experiments** to investigate the effect of increased nutrient loading on ecological processes;
- **detailed field measurements** of coastal circulation patterns, water and sediment quality, community structure of plants and animals;
- **development of an integrated ecological model** (COASEC, see Section 4.2.3).

4.2.2 Analyses of Nutrient Dynamics

In addition to field survey procedures, several analytical techniques have been used to assess nutrient impacts and may be useful tools for monitoring nutrients in estuaries. The following potentially useful techniques are reviewed:

- **techniques for identifying nutrient-limited autotrophic plant growth;**
- **nutrient/salinity plots to describe the behaviour of nutrients in estuaries;**
- **nutrient budgets to describe the amount of the nutrient input retained within, or exported from, an estuary.**

Identifying nutrient-limited growth

Biological impacts from nutrient enrichment are likely where the autotrophic growth increases with increased nutrient levels such that it is necessary to determine whether autotrophic growth is nutrient-limited. Therefore, previous studies indicate that the following five techniques have been used to identify nutrient-limited growth:

- **mesocosm experiments;**
- **analyses of algal tissue nutrients;**
- **analysis of molar nutrient ratios;**
- **comparison of nutrient concentrations against of half-saturation constants;**
- **bioassays.**

Mesocosms have been used to determine the relationships between environmental parameters, nutrient inputs and biological responses (Nixon 1992; Nixon *et al.*, 1995a; Taylor *et al.*, 1995b) and of these five techniques for identifying nutrient-limitation, this is probably the most effective but most expensive approach. Mesocosms are artificially reconstructed environments, which can be manipulated to allow variations in only single parameters and thereby determine the influence of that factor in the absence of any environmental noise. Therefore, it is possible to manipulate the input of nutrients and identify the biological

consequences. Taylor *et al.*, (1995b) considered mesocosms to be the most appropriate way to identify the limiting nutrients in aquatic systems because they may simulate nutrient impacts under of natural conditions. They can also be used to monitor the effect of natural seasonal variations in nutrient supply on community responses as has been shown by the use of Marine Ecosystems Research Laboratory (MERL) mesocosms in Narragansett Bay (Nixon 1992, Nixon *et al.*, 1995a).

Another possible indicator of nutrient limitation is the concentration of algal tissue nutrients. The following authors have either related tissue nutrients to the potential response of autotrophs to nutrient increases or have observed correlations between ambient nutrient and tissue nutrient concentrations:

- Lowthion *et al.*, (1985 cited by Conway (1993) concluded that macroalgal blooms in Langstone Harbour are not nutrient limited because the levels of nitrogen and phosphorus in the tissues of *Enteromorpha* were >2% of tissue saturation.
- Soulsby *et al.*, (1985) found high nitrogen concentrations and N:P ratios in macroalgae tissue showed that nutrients were stored to excess in Langstone Harbour and concluded that increases in sewage would not result in higher standing crops of algae.
- Kemp *et al.*, (1984) observed an increase in tissue nitrogen of *P. perfoliatus* with elevated nitrogen loading.
- Vidondo and Duarte (1995) observed a high tissue C:P (1712) in the slow-growing *Codium bursa* compared to the C:N ratio (31.8) indicating that the growth of this species was phosphorus limited.
- McGlathery (1995) also showed that the nutrient content for seagrasses increases with the increased availability of nutrients.
- Short (1987) found depleted nitrogen levels in seagrasses growing in nitrogen poor soils.
- Pedersen (1995) observed increases in tissue-nitrogen and chlorophyll content of phytoplankton and two ephemeral macroalgae *Ulva lactuca* and *Chaetomorpha linum* and two perennial macrophytes. Phosphorus enrichment resulted in increased tissue phosphorus in all plant types.

Many authors have assessed nutrient-limitation by the analysis of molar ratios of water column nutrients (O'Neill, 1985; Johnsson *et al.*, 1995; Malone in prep; Parr pers comm.). As organic matter has an atomic nutrient ratio of 106C:16N:1P, referred to as the Redfield Ratio (Redfield *et al.*, 1963), nutrients are required by autotrophs at these ratios. The influence of N:P ratios on plant growth were described by O'Neill (1985) who assumed that algae require one phosphorus atom for every 12-20 nitrogen atoms and that at high ratios (30:1) all the phosphorus is consumed before the nitrogen. However, at lower ratios (6:1), nitrogen would be assimilated faster and this would be the limiting nutrient species.

It is possible to use variations in the stoichiometry of dissolved nutrients to indicate which nutrient is most likely to limit autotrophic production:

- Johnsson *et al.*, (1995) concluded that phosphate was limiting in Loch Hyne where ratios of TN:TP, DIN:DIP and DSi:DIP were all substantially higher than the Redfield ratio (and that silicon is required in the ratio 1P:16Si:6N for diatom production).

- Malone (in prep) showed that there was a decline in the DIN:DIP ratios from above the Redfield ratio to below it along the length of Chesapeake Bay and also during the transition from spring to summer blooms. This indicated that nitrogen is limiting in the lower estuary and during the summer months.

Although nutrient ratios can provide information about the potential for plant growth, under certain conditions the interpretation of measured nutrient ratios could be misleading. If, for instance, nitrogen-fixing blue green algae are present then, where there is adequate phosphorus, there could still be a prolonged growth of algae beyond the limitations of the original N:P ratio (O'Neil 1985). Also where the measured ratios of nutrients indicate that phosphorus is limiting, Malone (in prep) found that there can still be available pools of phosphorus that have not been measured. Johnsson *et al.*, (1995) also suggest that differences in the elemental composition between benthic algae/seagrasses and that of phytoplankton means that the assumptions of Redfield stoichiometry is only reasonable in systems without extensive algal stands or large allochthonous loadings of organic matter.

Another means of indicating nutrient limitation is the use of 'half-saturation' nutrient concentration values, i.e. levels at which nutrient uptake is half of its maximum value. Concentrations of less than these half-saturation levels are associated with greatly reduced uptake rates (Fisher *et al.*, 1988). Fisher *et al.*, (1988) used this approach to compare the major estuaries on the eastern seaboard of the US, Chesapeake Bay, Delaware Bay and Hudson Bay. From the literature they obtained half saturation values for dissolved inorganic nitrogen, phosphate and silicate of <2, <0.5 and <5µm respectively. Using these values as indicators of nutrient limitation they concluded that Chesapeake Bay was nutrient limited and required management controls on freshwater nutrient inputs while the other two estuaries were considered to be less of a problem because they had a lower capacity for nutrient assimilation.

A final approach for assessing nutrient limitation is to collect water samples *in situ* and carry out subsequent laboratory nutrient bioassays. Parr (1993 cited in Parr and Wheeler 1995) state that the use of N:P ratios is complicated because only a proportion of the nutrients are bioavailable and these can only be measured using bioassays.

Nutrient/Salinity plots

Nutrient/salinity plots are produced by measuring nutrient concentration at several sites along an estuary and then plotting these values against salinity to indicate any conservative behaviour or the additions and losses of nutrients throughout the estuary. Where nutrients are subject only to physical mixing processes within an estuary, then a plot of nutrients against salinity will be a straight line (conservative behaviour). Departures from the linear pattern of nutrient reduction occur where there is either a net utilisation or a net production of nutrients. This is one of the most widely used techniques for describing the behaviour of nutrient in estuaries, e.g. by Fisher *et al.*, (1988) on Chesapeake Delaware and Hudson, Balls (1992) on the Forth and Tay, Christian *et al.*, (1991) on the Neuse, and Zwolsman (1994) on the Scheldt Estuary (Netherlands).

This technique, therefore, has proved to be a useful tool although several authors have highlighted its limitations. Firstly, it is not possible to make definitive conclusions about the nutrient cycling mechanisms that are taking place. Where nutrients are shown to be behaving conservatively this may indicate that there is no biological modification of the nutrients or

that rates of biologically mediated nutrient uptake and release are in balance. Ogilvie *et al.*, (in prep.) believe that these plots are only inaccurate indicators of nutrient behaviour and that significant processing may occur in the absence of statistical deviations from linearity.

Secondly, it may also be difficult to identify clearly which of the several possible biological, physical and chemical processes are responsible for changing nutrient concentrations. This was demonstrated by Pitkanen and Tamminen (1995) who attempted to identify the cause for a non-linear decline in both nitrogen and phosphorus concentrations in the Neva Estuary (Gulf of Finland) by reviewing studies which used nutrient/salinity plots. They identified the following six possible reasons for non-conservative behaviour of nitrogen and phosphorus:

- Denitrification removes nitrogen whilst the opposite process of nitrogen fixation is probably negligible due to small amounts of N_2 -fixing blue-green algae in the estuary.
- Increasing pH and decreasing DIP concentration along the salinity gradient favour release of adsorbed phosphorus from riverine particulate matter while a large fraction of riverine nitrogen is included by the stable DON which liberates inorganic nitrogen slowly.
- Saline water sediments are generally poorer at permanently retaining phosphorus than are freshwater sediments due to the formation of insoluble Fe(II)S in anoxic saline water sediments, which reduces the potentially available adsorption sites for P.
- Denitrification and liberation of DIP from suspended particulate matter and from the bottom sediment tend to keep the N:P ratio low.
- Several blue-green algal species accumulate extra phosphorus in their cells when they are in aphotic, nutrient-rich waters and transfer phosphorus to the euphotic layer via vertical mixing of the mixed surface layer.
- The decrease of the N:P ratio in the euphotic layer suggests that the net losses of N, via sedimentation of autochthonous POM are relatively greater than those of P. Therefore, circulation of bioavailable phosphorus in the planktonic food web of the euphotic layer is probably more effective than that of nitrogen, leading to the observed decrease in the ratio.

Furthermore, nutrient/salinity plots can be difficult to interpret when the rate of nutrient input from rivers is variable. Zwolsman (1994) cites several authors who observed problems with interpreting concentration-salinity plots because of short term variations in riverine concentrations and Lebo *et al.*, (1994) showed that a knowledge of the temporal variations in river DIP are required to interpret DIP reactivity in estuaries from DIP-salinity diagrams.

Some authors have used nutrient/salinity plots to predict the riverine inputs of nutrients by extrapolating the plot to a zero salinity level (Balls 1992; Balls *et al.*, 1995; Muller *et al.*, 1994). Using an adaptation of this principle Rees *et al.*, (1994) and Gunby *et al.*, (1995) developed a technique for standardising nutrient measurements taken in different regions of estuaries for a proposed GQA (see Section 4.3.2).

The technique of predicting nutrient inputs based on mid-estuary measurement was criticised by Balls *et al.*, (1996) who observed that Total Oxidised Nitrogen (TON) concentrations (*i.e.* nitrate and nitrite) observed in river water are always larger than those predicted by

extrapolating down-estuary TON/salinity relationships to zero with the greatest discrepancies occurring under low river flows.

Nutrient Budgets

The production of nutrient budgets or 'mass balances' is a useful technique for describing nutrient dynamics in an estuary. The calculation of inputs, retention and export of nutrients from an estuary gives an understanding of the overall nature of the estuary and specifically, whether it functions as a sink or a source of nutrients. Mass balances have been used in several systems including: the Scheldt estuary (Billen *et al.*, 1985), Chesapeake Bay (Nixon 1987; Boynton *et al.*, 1995), Narragansett Bay (Nixon *et al.*, 1995a), Gulf of Bothnia (Wulff *et al.*, 1994), Moresby estuary in Australia (Eyre 1995). Boynton *et al.*, (1995) for example, collated and analysed quantitative annual input-export budgets for total nitrogen (TN) and total phosphorus (TP) for Chesapeake Bay and three of its tributary estuaries (Potomac, Patuxent, and Choptank rivers). They used a conceptual model to develop these budgets which they believed represents a compromise between current understanding of major inputs, exports, storage, and cycling of TN and TP in Chesapeake Bay; they also considered the availability of data with which to evaluate model terms. The model (Figure 18) includes estimates of:

- TN and TP sources (point, diffuse and atmospheric);
- internal losses (burial in sediments, fisheries yields, and denitrification);
- storage in the water column and sediments;
- internal cycling rates (zooplankton excretion and net sediment-water flux);
- net downstream exchange.

The mass balance approach has a drawback is that there is often a high degree of inaccuracy because of the limited quality and quantity of available data, the variability of conditions and the difficulties of measuring the scale of all the nutrient fluxes. Even in Chesapeake Bay, despite extensive research, Boynton *et al.*, (1995) identified the need for better estimates of:

- atmospheric dry deposition;
- direct inputs of near-surface groundwater are needed;
- sedimentation rates used in calculating nutrient burial;
- the role of migratory fish.

The nutrient mass balance approach to smaller scale model estuarine systems or mesocosms (see above) avoids some of these criticisms. Nowicki and Oviatt (1990) used this approach to examine the role of the bottom sediments and the impact of eutrophication on nutrient trapping efficiency in Narragansett Bay. Control and nutrient-rich mesocosms were used with and without sediment communities and the measurements taken included:

- inputs of nitrogen and phosphorus in inflowing water from adjacent Narragansett Bay and in rain, snow, and atmospheric dry deposition ('dryfall');

- outputs of nitrogen and phosphorus in outflowing water, in losses to denitrification, to burial, and to increases in biomass.

4.2.3 Models

A potential useful tool for the analysis and assessment of nutrient impacts is the development of descriptive models and several authors have developed models including both:

- **empirical models**, where the underlying theory of the relevant processes is only poorly understood but the model is designed to explain or illustrate observed patterns;
- **stochastic models** where there is a good knowledge of the theoretical basis of the processes.

Those models which have been detailed in the reviewed literature are described here together with any observations or criticisms made by the authors. The models have been separated into the following two categories:

- models of nutrient input,
- models of biological responses to nutrients.

Models of nutrient input

The following models describe the rate at which nutrients are imported to estuaries from diffuse sources:

Johnes (1994) mentions two physically-based models which predict water quality changes following changes in land usage, the Thames Nitrate model (Whitehead; 1988) and the SHETRAN-UK model (Lunn *et al.*, 1992). Johnes' criticism of these models is that they only work well for small catchments or for the catchments for which they were created. They therefore are not widely applicable to other areas. In addition they only predict nitrate and orthophosphate whereas, according to Johnes, it would be of more value to model the total nitrogen and total phosphorus components (TN and TP respectively) rather than particular constituent nutrient components.

Mainstone *et al.*, (1995) mentions a new model that has recently been produced by SNIFFER (the Scottish and Northern Ireland Forum for Funding Environmental Research) which links hydrodynamics of river flows to phosphorus export rates to describe the riverine phosphorus concentrations. Using a similar approach to MINDER (a model designed to assess eutrophication risk in lakes model using export coefficients), this model describes river flow from digitised terrain maps and includes MORECS flow data from the Meteorological office (Woodrow *et al.*, 1994). It also includes a soil hydrology sub-model HOST (Hydrology of Soil Types), a classification scheme for soils and geological conditions. This model assumes SRP:TP to be constant along the length of the river and Mainstone *et al.*, (1995) recommend the development of a sub-model of SRP:TP concentrations to account for changes in this ratio. They concluded that these models require successive calibration and validation stages in order to change the original export coefficients to reflect real data from catchments.

The US Department of Agriculture have developed a physical model, CREAMS (Chemicals, Runoff, and Erosion from Agricultural Management Systems), which is designed

to predict the catchment-wide loadings of sediments and nutrients in runoff (Cooper, Smith and Bottcher 1992). This model was adapted to represent nitrogen and phosphorus cycling in grazed pastures and seasonal variations in the 'hydraulic conductivity' specifically in pastoral watersheds. According to the authors the predictions of daily surface runoff volumes were good although estimates of nutrient and sediment loss on a daily basis were less accurate and the model was considered more suitable for predicting loss over long time scales (*i.e.* seasons).

The North Carolina DEHNR (State Department of Environment, Health and Natural Resources) modelled the input of nutrients from catchment run-off to the Albemarle/Pamlico basins using data from a LANDSAT land use/cover study. LANDSAT provided information on the areal coverage of each land use/cover category and export coefficients were calculated for different land use areas in each sub-basin of the catchment. Multiplying land use areas by the appropriate export coefficients gave a total nitrogen and phosphorus loading from each land use for each sub-basin. Account was taken of:

- inputs (fertiliser, precipitation, livestock wastes and nitrogen fixation);
- outputs (harvested crops, soil fixation, denitrification, loss to swamp forests and river export);
- storage of nutrients in the system.

However, due to data limitations they were only able to assess agricultural land and for this numerous data sources were required and several assumption required.

Nixon (1996) notes that models are also available for determining the input of nutrients from atmospheric sources. These are sophisticated atmospheric chemistry transport models which can calculate present, future and past rates of atmospheric nitrogen deposition (Levy and Moxim 1989; Dentener and Crutzen 1994). They describe the global patterns of deposition but they have a high level of spatial resolution and Nixon (1996) was able to use these to calculate preindustrial ('background') levels of atmospheric deposition in Narragansett Bay.

Models of biological responses to nutrients

Several authors have either used, developed or discussed the usefulness of models that are designed to predict the response of communities to nutrients:

Asknes *et al.*, (1995) developed for coastal waters a phytoplankton model based on information in the literature and validated against enclosure experiments tested with limiting light, nitrogen, phosphorus and silicon conditions. Characteristics such as the balance between diatoms and flagellates were predicted and a 3D physics-based model for the North Sea was simulated and then forced against realistic topographic and meteorological data and freshwater riverine and nutrient inputs. The authors believe that this model needs improvement before being applied in a management context.

Chapelle *et al.*, (1994) combined a three-dimensional hydrodynamic model and a biological model to simulate eutrophication in Vilaine bay (France). The resulting ecological box model simulates seasonal changes of nutrients, phytoplankton and oxygen reasonably well and reproduces the bay's spatial heterogeneity (the factors incorporated in this model are described in Figure 19). Chapelle *et al.*, (1994) analysed the model's behaviour and found to

provide a good representation of the eutrophication of Vilaine bay, with a yearly simulation showing:

- high amounts of nutrients brought by the river Vilaine;
- spring and summer phytoplankton blooms induced by high nutrient levels, especially during calm weather;
- bottom hypoxia following the phytoplankton blooms.

The major purpose of this model was to understand the local eutrophication process and to test possibilities for restoring water quality and controlling eutrophication (e.g. by the reduction of land-based inputs). Chapelle *et al.*, (1994) comment that:

- the simple description used in the model for benthic phosphorus creates uncertainties about the response to nutrient reductions and that a more precise model should be developed with detailed benthic phosphorus variables, and processes such as mineralisation, adsorption-desorption, interstitial phosphorus diffusion;
- the benthic nitrogen cycling needs to be better described in order to quantify the possible role of denitrification;
- the model is appropriate for simulating low-frequency trends such as seasonal changes (comparisons between years are also possible but require more detailed spatial and temporal modelling and probably more complex biological processes to identify brief and localised episodes, such as acute anoxic events).

de Jonge *et al.*, (1996) used the BOEDE model (Baretta and Ruardij 1988) to assess the relative importance of nutrients and light conditions in the Ems estuary. For this model the Ems estuary was divided into 5 compartments each with three models (pelagic, epibenthic and benthic) which were interconnected with a water transport model.

de Jonge and Bakker (1994) developed a methodology for predicting blooms in estuaries based on the flushing time and relative growth rate of algae. They used an equation for flushing time which incorporates a function of the tidal exchange as well as freshwater input and estuary volume. The equation is $E = [(V^*/T) \times (1/V)]$ where:

- E is the 'relative exchange rate of freshwater'
- V^* = exchanged volume of freshwater
- V = volume of the estuary
- T = time

When E and the losses of phytoplankton (L) are greater than the relative growth rate of algae (μ) i.e. $\mu < E + L$, then it can be predicted that blooming of phytoplankton is likely to occur.

Duarte (1995) notes that useful but relatively simple models are available for simple predictions of the spatial loss of seagrasses and benthic macroalgae due to eutrophication-related reductions in water transparency. However, he found that non-linear models which account for the non-proportional relationship between nutrient loading and biological response, and provide quantitative predictions of nutrient loading impacts, are still unavailable. Even less attainable are models which predict the consequences of reduced nutrient inputs.

Forth River Purification Board (FRPB 1995), now SEPA, developed a Firth of Forth Water Quality Model to assist UWWTD studies in tidal waters. This model predicts nutrient and algal carbon concentrations in the lower estuary and firth in response to changing riverine and diffuse source nutrient inputs. It has been validated against monitoring results and incorporates:

- a salinity-dependant mortality (which compensates for a sensitivity that it has to high nutrient concentrations);
- a separate mortality coefficient is included to mimic zooplankton grazing.

A good correlation between predicted and observed data was obtained by the FRPB survey of the lower Forth estuary (FRPB 1995). Other monitoring authorities including the Environment Agency also have estuarine models for deriving water quality information.

Harding (1994) used a model designed by Wofsy (1983) which predicts extinction coefficients and phytoplankton biomass in eutrophic waters, examines the relationship between light and nutrient regulation of phytoplankton and determines which of the two factors are limiting phytoplankton growth.

Heip (1995) discussed the potential usefulness of models in the management of estuarine ecosystems. He concludes notes that a further effort is required in ecological modelling in order to develop viable models which combine information on both physical and biological processes and include the following aspects:

- sedimentation rate;
- water movement;
- diagenetic processes;
- the breakdown of organic material within sediments;
- the effect fauna on changes to the physical and chemical behaviour of sediments and sediment-water exchange.

Lord and Hillman (1995) used an integrated ecological model known as COASEC (COAStal ECology) as part of the Perth Coastal Waters Study (Australia). This was developed to describe the dominant physical, chemical and ecological features of coastal waters and the effects of increased nutrient loads on these processes. For this model, the area under investigation was divided into 250m² 'cells', a major habitat type was assigned to each cell and the responses of the appropriate biota for that habitat type to nutrient concentrations were calculated. The difficulty of incorporating ecologically realistic temporal variations were recognised together with the problems of relating nutrient changes to community responses. These were accommodated by assuming simple relationships between nutrients and response for certain biota. The authors considered that sufficient 'complexity and accuracy of processes' were included to produce rational relationships and that it could be used to forecast 'realistic' chlorophyll-*a* levels (which are comparable to measurements) but not the subsequent influences on light availability, seagrasses and benthic algae. According to Lord and Hillman (1995) similar models have been, or are being, developed for Port-Philip and Moreton Bays in Australia. However, the authors concluded that no model could

accurately make quantitative predictions of changes and emphasised the importance of continued monitoring.

Parr and Wheeler (1995) reviewed models which attempt to relate light to photosynthesis. They found that some information could be obtained about the potential phytoplankton growth based on:

- available general hydrodynamic models which can describe physical conditions such as: water column mixing, the breakdown of stratification and the effect of silicon dynamics.
- a model which separates the light spectrum into several wavebands and relates these to PAR and chlorophyll-specific absorption of algae at several depths.

However, they found that insufficient information was available to model the impacts of grazing on the phytoplankton stock or on the potential for the occurrence of toxic species.

Wulff et al., (1990) developed a model to describe the nitrogen cycling within the Baltic Basin by accounting for the vertical movements (but not horizontal) and incorporating a seasonal pycnocline and effects of primary production, light, temperature, nitrate, ammonia, critical depth, remineralisation, nitrification, denitrification, sulphate reduction, and sedimentation using simple functions. The model was driven by nitrogen inputs typical of recent decades and by the weather and saltwater inflows and, when run for a 20-year period, the findings were similar to field results. However, the authors concluded that a detailed ecosystem model is not required to explain large-scale distributions of nitrogen and oxygen and that only some physical circulation and biogeochemical processes need be considered.

4.3 Approaches to Management of Estuaries

From the literature, several approaches to the management of nutrient impacts in estuaries have been identified. Examples of the following approaches are described:

- a national assessment of estuarine eutrophication (USA);
- estuary classification schemes and Water Quality Targets;
- Environment Agency National Strategy For Eutrophication Control;
- Ecological Quality Objectives.

4.3.1 National Assessment (USA)

The US National Oceanic and Atmospheric Administration has a national assessment of estuarine eutrophication of which the methodology and preliminary results of this study are currently presented in the unpublished draft report (NOAA 1995). The study chose a series of indicators as the basis for assessing the status of estuaries (following advice via workshops). These indicators were selected to characterise accurately eutrophic conditions despite the high degree of variability between estuaries, in terms of their physical characteristics and the expression of eutrophication symptoms. The study aimed to separate and classify estuaries on the basis of the number of indicators observed, for example, estuaries that have high or low nutrient levels can be separated as can those with or without eutrophic symptoms.

Response ranges were determined for each of the indicators using nation-wide data and regional expertise. These ranges were designed to separate estuaries along a gradient of high, medium or low response, or alternatively for single event criteria such as toxic blooms to separate estuaries where specific incidents were observed and characterised as a 'problem', from those where they were not observed at all ('no problem') (Table 3).

Further details of these indicators and response ranges were collected to describe existing or 'typical indicator' conditions (*i.e.* during a normal flow year) as well as trends over time. These details included the magnitude and duration of extreme conditions for measurable indicators and the perceived level of detrimental impact for non-measurable indicators. The direction and scale of the observed changes were also recorded to determine whether conditions were improving or deteriorating. To ensure spatial compatibility between different estuary types information sources, the estuaries and the corresponding data were divided into three salinity-related regions: tidal/fresh (0-0.5psu), mixing zone (0.5-2psu) and seawater (>25psu). The approach and success of the data collection procedure is discussed in Section 4.1.

Table 3: Indicators and response ranges to describe the scale of eutrophication in estuarine systems and information collected to determine the existing conditions and trends in these indicators (NOAA Draft Report 1995).

Part 1: Algal Conditions		Part 2: Nutrients and DO		Part 3: Ecosystem/Community Response	
Indicator	Response Range	Indicator	Response Range	Indicator	Response Range
Chlorophyll l-a	Hyper >60 $\mu\text{g l}^{-1}$ High 20-60 $\mu\text{g l}^{-1}$ Med. 5-20 $\mu\text{g l}^{-1}$ Low 0-5 $\mu\text{g l}^{-1}$	Total Dissolved nitrogen	High >1 mg l^{-1} Med. 0.1-1 mg l^{-1} Low 0-0.1 mg l^{-1}	Benthic Community	Diverse Crustaceans Molluscs Annelids Other (specify)
Nuisance Blooms	Problem/ No Problem	Total Dissolved phosphorus	High >0.1 mg l^{-1} Med. 0.01- 0.1 mg l^{-1} Low 0-0.0.1 mg l^{-1}	Phytoplankton Community	Diverse, Diatoms, Flagellates, Blue/Greens Other (specify)
Toxic Blooms	Problem/ No Problem	Anoxia	Problem/ No Problem	System Productivity	Mixed, Pelagic Benthic
Macroalgal Abundance	Problem/ No Problem	Hypoxia	Problem/ No Problem	SAV Spatial Coverage	V. Low 0-10% Low 10-25% Med. 25-50% High 50-100%
Epiphyte Abundance	Problem/ No Problem	Biologically stressful DO	Problem/ No Problem		
Turbidity	High <1m Med. 1-3m Low >3m				
Suspended Solids	Problem/ No Problem				
Existing Conditions			Trends		
Information collected by salinity zone			Information collected by salinity zone		
Months of probable occurrence			Direction of observed change		
Duration of event			Magnitude of observed change		
Episodic or periodic occurrence			Contributing factors		

In order to characterise the estuaries with regard to the level of eutrophication, the worst case conditions were considered *i.e.* high or medium for most indicators, 'hypereutrophic' for chlorophyll-*a* and other non-measurable indicators which were classified as having 'problem' status. The tabulated and graphical outputs that were used to characterise the estuaries are shown in Figures 19 to 25. The existing conditions and trends in estuaries were characterised collating indicators either by:

- 1) **Salinity bands for all estuaries:** Data from all estuaries were combined to show the extent of problem conditions across the region by expressing the indicator values as a

cumulative percentage of the surface area for each salinity band. This information is then presented in a bar diagram format (Figure 20).

- 2) **Mixing zones for individual estuaries:** The mixing zone was identified as the 'most heavily impacted' region of an estuary and therefore in keeping with the aim to describe 'worst case' conditions, this approach involved the characterisation of estuaries on the basis of results from this region alone. This information is presented on annotated charts of the region (Figure 21).
- 3) **Salinity bands for individual estuaries:** The spatial coverage of the indicators of impacted or problematic conditions in each estuary and in each salinity band. This information is then tabulated (Figure 22).

These three approaches and types of presentation were also used to describe the temporal trends in the eutrophication indicators to demonstrate whether condition were improving, deteriorating or had remained unchanged.

The future direction of this nation-wide estuarine monitoring over the period 1995 to 2005 led to the following possible stages being proposed (NOAA 1993):

- an initial improvement of monitoring methods (1995 to 1996);
- the increased collaboration and transfer of technology between agencies and regional authorities (1995 to 1999);
- investigation of major processes affecting symptoms of eutrophication and the development of estuary classification procedure (1995 to 1999);
- a review of the effectiveness of mitigation measures in representative estuaries;
- development of an 'early warning capability' to predict the onset of nutrient-related problems (1995 to 2005);
- establishment of a national database on nutrient enrichment (1995 to 2005).

4.3.2 Classification Schemes & Water Quality Targets

As a means of assessing the potential impact from nutrients in different estuarine systems several authors have proposed schemes either:

- to classify estuaries on the basis of their susceptibility to enrichment based on physical characteristics; or
- to set nutrient concentration targets to indicate the potential biological response.

These two approaches overlap as in some instances nutrient concentration criteria have been incorporated into classification schemes therefore examples of both approaches is reviewed in this section.

Boynton et al., (1982) compared the response of the phytoplankton to nutrients in 63 estuaries. They reviewed data on physico-chemical variables, phytoplankton production and chlorophyll-*a* levels. Following a qualitative examination of these data they chose to separate estuaries into 4 groups which show differences in flushing time, light regime and nutrient

input as the three 'primary mechanisms' influencing phytoplankton production. The four groups were:

- **fjords:** deep estuaries with shallow sills;
- **lagoons:** shallow and well mixed estuaries with only a slight influence from riverine inputs;
- **embayments:** deeper systems that are occasionally stratified with only a slight influence from riverine inputs and a large exchange with open ocean;
- **river dominated:** diverse group of systems with seasonally lowered salinities and variable degrees of stratification.

In order to determine whether the quantitative criteria could be obtained in clearly defining these four groups, Boynton et al., (1982) carried out multivariate analyses to compare various physical, chemical and biological characteristics of these estuary types. The groups had differences in salinity, extinction coefficient, latitude and flushing rate but not in phytoplankton production or chlorophyll-*a*. They concluded that similar rates of phytoplankton production and algal biomass are observed in widely varying types of estuary systems and recommended that further analysis be conducted but incorporating more variables (*e.g.* insolation, euphotic depth, flushing rate and nutrient recycling rate) in an attempt to account for the variability in the nutrient loading-phytoplankton production relationship.

Jones and Gowen (1985) compared Scottish Sea Lochs on the basis of the potential for algal bloom formation. They identified three categories of fjordic estuary based on flushing times and the doubling time with the following characteristics:

- 1) A relatively rapid flushing time in which phytoplankton biomass and the abundance of phytoplankton is mainly determined by the processes in the coastal waters.
- 2) Residence time or 'retention' time is greater than the doubling rate of phytoplankton and there is potential for phytoplankton blooms.
- 3) Residence time is considerably longer than the doubling time to give greater potential for phytoplankton growth in the estuary than in the adjacent coastal waters. However, in these systems, other factors affecting algal growth in estuaries (*e.g.* turbidity) must also be favourable.

Perdon (1995) collated data from 47 estuaries throughout Western Europe in an attempt to classify estuaries on the basis of their susceptibility to eutrophication. Problems in collecting sufficient information on the biological responses of estuaries (see Section 2) dictated that he concentrated on grouping estuaries on the basis of the physical and chemical parameters including relative depth, surface area, flushing time and nutrient load. The occurrence of stratification and the depth of the estuary were considered to be the most important factors in his classification system. The scheme produced therefore involved the initial separation of all estuaries into stratified and non-stratified (including partially stratified) categories. Following this division, he then plotted surface area against depth for both systems and from this plot he separated the extremely deep Fjordic systems of Nordfjord and Hardangerford from the remainder. The non-stratified estuaries were similarly compared but none of these estuaries were found to have distinctly greater depths and, therefore, no subdivision of this group was made.

With limited biological information, the stratified and non-stratified estuaries were then further sub-divided on the basis of their 'expected biological response' to an increased nutrient input. Perdon (1995) related the expected biological response directly to the flushing times on the premise that the longer the flushing time, the greater the expectation of an algal bloom. He also estimated the level of primary productivity, by using the Redfield ratio (the atomic C:N:P of organic matter 106:6:1) and to convert the known levels of phosphorus (in mmol/m²/yr) to levels of carbon (in mmol/m²/yr), simply by multiplying by a factor of 106. The carbon load was converted to g C/m²/yr for an estimation of the primary productivity. Following these procedures, he finally separated estuaries into four classes in order of increasing susceptibility to eutrophication and oxygen depletion:

- non stratified estuaries with very short flushing times (25 days);
- non stratified estuaries with short flushing times (75 days);
- stratified estuaries with long flushing times (10-25 years);
- stratified estuaries with very long flushing times (75 years).

The characteristics of these four estuary classes showing the water column characteristics (depth and stratification) and the estimated primary productivity over a given nutrient input range can then be summarised (Figure 23) and condensed into a single plot (in Figure 24) of nutrient loading against the estimated primary productivity for all the estuaries.

Nixon (1995) proposed a trophic classification scheme for estuarine and coastal ecosystems using organic carbon supply. He identified the following categories:

- oligotrophic (< 100gCm⁻²y⁻¹);
- mesotrophic (100-300gCm⁻²y⁻¹);
- eutrophic (301-500g Cm⁻²y⁻¹);
- hypertrophic (> 500gCm⁻²y⁻¹).

Nixon (1995) recognises that the source of the organic carbon could either be fixation by primary producers within the system of concern or an input of organic matter from outside the system. He notes that there is poor information concerning the total amount of allochthonous carbon entering coastal marine systems and the fraction of that total that is sufficiently labile to be metabolised. Nixon (1995) suggests that most marine systems could be classified on the basis of primary production data alone, whilst remembering that the description of any particular system may change if allochthonous inputs are included.

NOAA (1993) used a measure of the capacity of estuaries for retaining nutrients. The classification index was the Dissolved Concentration Potential (DCP) which defines an estuary's ability to concentrate a dissolved pollutant based on its hydrological characteristics. The DCP is then plotted against nutrient loadings and the relative susceptibility of estuaries is predicted from the position of the plot (Figure 25). This procedure however does not include important factors such as the role of nutrient cycling, stratification and the variability of freshwater inflow, nutrient loadings and sediment fluxes.

NOAA (1993) are also developing an index of eutrophication for estuaries which is to be based on the following seven indicators of existing conditions: chlorophyll-*a*, turbidity, TDN, TDP, anoxia, hypoxia and biologically stressful DO levels. As a preliminary step in this

process, data have been obtained covering each of these parameters for seven estuaries to show the highest annual values for chlorophyll-*a*, turbidity, TDN, TDP and the observation of anoxia, hypoxia and biologically stressful DO. The average area showing 'problem' conditions for each of these factors was combined to obtain a value for the average area in each salinity band and in each estuary as a whole exhibiting problems. In instances where areal coverage was unknown, 10% was assumed. The resulting classification was compared against the scheme developed by Nixon (1995) which was based on organic carbon supply although there was a good correlation for only three of the seven systems which had been classified by both approaches. A separate index was developed to describe temporal trends in the status of estuaries based on the same approach that was derived to describe existing conditions. This approach identified whether conditions were worsening, improving or had remained unchanged in the estuaries. Changes over time of same seven indicators along with SAV coverage, and bottom water dissolved oxygen were used. An assessment of the usefulness and reliability of this second approach was made more difficult by the paucity of long-term data.

Lord and Hillman (1995) made a preliminary estimation of the potential for eutrophication of different areas of the Perth Coast (Australia), based on a measure which combines flushing time and nutrient load. They identified the following critical values of the nitrogen load per flush:

- 125mg/m²/flush as the value at which seagrasses would be affected (this was considered as a safe loading rate)
- 250mg/m²/flush as the point at which eutrophication effects may commence
- 1000mg/m²/flush as the level at which significant eutrophication effects will be observed.

Lord and Hillman (1995) used these critical values to compare loads in three areas of the Perth Coast against other estuarine systems, and also to predict future values, and biological responses, based on expected changes to nutrient loads in the year 2040. The values and biological responses (observed and predicted) are shown in Table 4.

Table 4: Nitrogen loads to five estuarine systems with biological responses that are either observed, or predicted from comparisons against critical loading values (Lord and Hillman 1995)

Location	Flushing Time (days)	N load (tonnes per annum)	N load (mg/m ² /flush)	Comments and predicted impacts
Whitfords Lagoon (Australia)	2	613 (1642)	72 (177)	water quality criteria met (no evidence of eutrophication <250mg/m/flush)
Cockburn Sound (Australia)	30	1095 (180)	766 (125)	loss of seagrass (estimated safe level for seagrass)
Port Philip Bay (Australia)	100	7500	688	possible localised eutrophication
Narragansett Bay (USA)	51	2200	6800	localised eutrophication
Tokyo Harbour (Japan)	40	12400	9770	localised eutrophication
anticipated 2040 nitrogen loading rates and predicted impacts shown in brackets				

Environment Agency GQA Schemes: The Environment Agency developed criteria for the classification of estuaries as tools for monitoring and assessment environmental conditions. The classification which is currently in place, was produced by the Classification of Estuaries Working Party (CEWP) and is based on an assessment of the biological, chemical and aesthetic conditions in an estuary. This approach was considered by the Environment Agency to be too subjective and therefore a series of General Quality Assessment (GQA) schemes were developed with a view to replacing it. It was intended that the GQA schemes would be designed to objectively categorise the water quality of estuaries and will contain the following components:

- nutrients in water (Rees *et al.*, 1994; Gunby *et al.*, 1995);
- dissolved oxygen and ammonia (Nixon *et al.*, (1995b);
- measures of ecological quality based on benthic invertebrates (Codling *et al.*, 1995).

The schemes developed for nutrients (N and P), DO and Ammonia are described here and a review of possible methods for classifying benthic invertebrates is outlined in Appendix 7.

Rees *et al.*, (1994) and Gunby *et al.*, (1995) devised the GQA scheme for nutrients by selecting the mid-winter levels of total inorganic nitrogen (TIN) and soluble reactive phosphate (SRP) as appropriate determinands for the following reasons:

- most of the available data are for TIN rather than TN (including organic nitrogen);
- organic nitrogen will be very low in midwinter compared to inorganic nitrogen;
- TIN has the greatest relevance to primary production.

They collected data on winter TIN and SRP from the Environment Agency regions, special surveys (e.g. JoNuS programme) and from surveys commissioned specifically for the testing

the proposed scheme. Using these data they identified a series of salinity-related (presuming a conservative behaviour of TIN and SRP) class thresholds as a percentage of maximum observed values for the most contaminated estuaries. The resulting class boundaries are described in Table 5.

The authors attempted to develop criteria which included the less contaminated estuaries in the range of nutrient levels, but this was rejected because of a lack of information on lower nutrient systems. Another approach was considered of setting boundaries as multiples of a 'background level', but no unimpacted systems were identified on which to base such a background level.

Table 5: Proposed WQO class boundaries for nutrients estuaries from Rees *et al.*, (1994) and Gunby *et al.*, (1995).

Class Boundary	TIN $\mu\text{g/l}$		SRP $\mu\text{g/l}$	
	Salinity 0-2 PSU	Salinity 32-34 PSU	Salinity 0-2 PSU	Salinity 32-34 PSU
1/2	3000	300	300	75
2/3	6000	600	600	150
3/4	9000	900	900	225

To produce single value Water Quality Criteria from these class boundaries that could be derived from samples taken from any position within estuaries (*i.e.* other than within the 0-2psu and 32-34psu bands), Gunby *et al.*, (1995) proposed graphically projecting the measured nutrient levels obtained at any particular salinity level to zero salinity (Figure 26). This classification procedure was based solely on nutrient concentrations rather than biological effects such as changes in trophic status or algal blooms and the authors recognise that they cannot be directly related to the assessment of eutrophication. They note that Parr *et al.*, (1993) proposed classifying the trophic status of estuaries and coastal waters in terms of algal productivity and recognised that a GQA scheme would be improved if a direct assessment of trophic status could be included with an assessment of nutrient levels. However, this work concluded that there was at present no-clearly defined best option for assessing trophic status in all tidal waters.

The development and preliminary testing of dissolved oxygen and total ammonia GQA schemes for estuaries was carried out by Nixon *et al.*, (1995b). As a result of a review on toxicity data, four class categories were identified for DO and Total Ammonia (Table 6).

Table 6: Proposed WQO class boundaries for DO and ammonia in estuaries from Nixon *et al.*, (1995b).

Response	DO Category	Threshold DO mg l^{-1}	DO Category	Threshold total ammonia
'no effects'	A	A/B threshold 8mg l^{-1}	1	1/2 threshold 0.86mg N l^{-1}
'low risk of effects'	B	B/C threshold 4mg l^{-1}	2	2/3 threshold 4.7mg N l^{-1}
'possible effects'	C	C/D threshold 2mg l^{-1}	3	3/4 threshold 8.6mg N l^{-1}
'probable effects'	D	N/A	4	N/A

4.3.3 Environment Agency National Strategy for Eutrophication Control

During the early development of the Environment Agency's proposed management strategy on aquatic eutrophication, Entec (1995) reviewed the principle considerations for setting targets to control eutrophication in a range of aquatic systems. They recognised that in order to set targets that are applicable to more than one estuary, it is necessary to consider whether estuaries can be classified into groups. They also suggest that different zones within the estuary may need to be classified separately because of the different potential biological response at different salinities. They considered that the following physical factors may be valuable in developing such a classification scheme:

- tidal range;
- percentage of intertidal area;
- stratification behaviour;
- retention time.

In considering the key factors for target setting Entec (1995) note the need to incorporate the following which can be regarded as relevant to at least some estuaries:

- adverse effects on maintenance of a normal, diverse ecosystem;
- adverse effects on special ecosystems (especially those of interest for conservation of birds);
- aesthetic degradation due to plankton blooms or attached algal mats reducing the value of the estuary for bathing and seaside pursuits;
- physical interference of excessive plant growth with water borne recreation;
- adverse aesthetic effects (visual, odour) of excess plant growth, particularly *Enteromorpha* spp. on mudflats, affecting riverside residents;
- degradation of sport fisheries and restriction of passage of migratory fish caused by dissolved oxygen or pH variations.

For these considerations three potential approaches to setting targets were identified:

High-level (use-related) targets: are desired outcomes which might relate to functional or utility aspects, e.g. the classification system being developed for the setting of Statutory Water Quality Objectives (Rees *et al.*, 1994) which is based on different 'user' requirements. These high-level targets can be used to monitor compliance and could be included in Local Environment Agency Plans (LEAPS, as successors to Catchment Management Plans) and the Estuary Management Plans. Entec (1995) recognised the need to clarify which of the targets are influenced by eutrophication *i.e.* are affected by nutrient inputs and as a first step, to determine which estuaries are appropriate subjects for eutrophication control.

Low-level targets: represent the criteria for achieving the high level targets. Two sets of such targets were suggested, one set designed to achieve a target trophic level for ecosystem conservation and another to avoidance nuisance. However, it was noted that, with the present state of knowledge, it is over-ambitious to seek different targets. As an example of possible low-level targets, Entec (1995) identify biological indicators such as a benthic diatom index or an index involving benthic algae as general measures of trophic level. Alternatively for

the SWQO the low level targets for water sport activity could include factors such as DO, BOD and pH.

Provisional Targets: are to be used in the interim and should be directed to estuaries where the links between nutrients and eutrophication effects can be made with most confidence (including oligotrophic estuaries). The suggested sources for possible interim targets were Cartwright and Painter (1991) who indicate annual means of $200\ \mu\text{g/l}$ total nitrogen and $30\ \mu\text{g/l}$ total phosphorus as suitable targets; the practical targets adopted as an indicator value by the Highland River Board of $300\ \mu\text{g/l}$ total oxidised nitrogen; the figures for hypereutrophication quoted by the Comprehensive Studies Task Team (CSTT 1993) of winter values of $168\ \mu\text{g/l}$ DAIN and the baseline figure suggested by R. Jones (CCW *pers. comm.*) of $150\ \mu\text{g/l}$ of total oxidised nitrogen in the oligotrophic estuaries of North Wales.

4.3.4 Ecological Quality Objectives

Another approach towards the assessment of impacts is the setting of ecological targets. These are used to identify the occurrence of detrimental biological impacts as opposed to water quality targets which identify the potential for biological response based on the nutrient availability. In any discussion, the value of the standards is dependent on the adequacy of the perceived or actual links between the standard being met and the objective being achieved (Elliott 1996). An example of ecological targets and water quality objectives are those developed by the Eco-Target Group (ETG) on the recommendation of the Trilateral Governmental Wadden Sea Conference (Common Wadden Sea Secretariat 1994). The ETG were commissioned with the tasks of selecting a set of ecosystem parameters, assigning reference values to these parameters and developing ecological targets for these parameters for 2010. The ETG identified the following preliminary targets:

- **Nutrients Target 1:** Winter (January/February) concentrations of Inorganic nitrogen and reactive phosphate in a fixed mixture of riverine and marine waters in sector [xx] should be reduced to [xx] $\mu\text{mol dm}^{-3}$ in the year 2010;
- **Nutrients Target 2:** The molar N:P ratio should be within the range 10 to 30;
- **Phytoplankton Blooms:** The length (duration) of *Phaeocystis* blooms must be reduced;
- **Macroalgal mats:** The average coverage with algal mats should not exceed [xx] (area to be determined).

These targets have not been finalised and still require further development and testing. A workshop on 'eutrophication and eutrophication phenomena' held in May 1994 reviewed these targets (Common Wadden Sea Secretariat 1994) and recommended that:

- nutrient concentrations (N, P and Si in combination) are used rather than N:P ratios;
- *Phaeocystis* blooms macroalgal coverage and Chlorophyll-*a* concentrations are the best indicators of eutrophication in the Wadden Sea;
- a continued examination of the relationship between these three parameters and eutrophication to assess their suitability as indicators.

The derivation of Ecological Quality Objectives (EcoQO) as statements against which testing and monitoring can be carried out have been discussed in relation to any quality assessment

and as a precursor to setting targets and standards (Elliott 1996). It is considered that objectives should be a necessary precursor to surveying any area and thus they can be regarded as 'null hypotheses' to be tested. These EcoQO's should be derived for symptoms of eutrophication.

4.4 Hindcasting Nutrient Status

Authorities charged with monitoring and managing nutrient levels have difficulty in assessing the extent to which contemporary biological responses are a natural characteristic of the systems or are the result of anthropogenic interference. In essence, the 'signal' of man-induced nutrient levels has to be separated from the 'noise' in the system. Estuarine nutrient inputs before the onset of human interference in estuarine catchments ('background' inputs or baseline conditions) will have varied depending on numerous natural catchment characteristics such as the soil composition, geology, topography and the habitats present. The subsequent response of estuaries to anthropogenic interference will also have varied depending upon the nature of human activities and the physico-chemical characteristics of the estuaries.

It is necessary to determine and maximise the 'signal to noise' ratio in order to detect change, i.e. background nutrient concentrations require to be quantified as a precursor to assessing the importance of human impacts to an estuary and to better understand the reason for current conditions in estuaries and the responses of estuaries to changes in nutrient inputs. This information would clarify the relationship between the physico-chemical characteristics of estuaries and biological responses, which would aid the development of a classification scheme for estuaries and assessments of risk to estuarine communities from nutrient enrichment. This section will therefore, review the techniques available for hindcasting nutrient concentrations.

An initial difficulty with identifying background levels is selecting a suitable date or period against which to compare present conditions. The historical human interference in catchments, with changes in the scale and patterns of activity especially of land-use, makes it difficult to identify clearly a period in which conditions can be considered to be truly natural. The level of this interference will depend upon the particular baseline date that is selected. Possible background dates have been considered by several authors (Johnes 1994; Mainstone *et al.*, 1995; Moss *et al.*, 1996; van Raaphorst *et al.*, in prep) who all recommended using the period prior to the end of the second world war. During this time, however, agriculture was widespread and the catchments cannot be considered as natural, but this period is considered to be the most suitable for the following reasons:

- it was prior to a significant intensification of agriculture;
- there was less use of inorganic fertilisers and an emphasis on nutrient cycling rather than external loading, thus leaching of nutrients was considerably lower than the present;
- historical data are available for factors such as land uses, agricultural practices and habitat changes (although, as mentioned above, there is a lack of long term nutrient data);
- if background levels were to be used as management targets then this period, in which there was at least a measure of human interference, is the most realistic.

Following the review the following two techniques have been identified for hindcasting nutrient inputs to estuaries:

- **Taking contemporary measurements:** This involves taking measurements of nutrient loads/inputs to estuaries which are at present largely free of anthropogenic impact and using these measurements as indicators of background nutrient levels.

- **Catchment nutrient export modelling:** This involves determining the relationship between land uses and the rates at which nutrients are leached from the catchment and then using historical information on changing land uses in the catchment to determine the past rate of nutrient export from the catchment.

In addition to these, the following techniques can be used to hindcast the biological response in estuaries:

- **Palaeoecology:** This involves the determination of historical environmental conditions from the composition of organisms (particularly diatoms) in the layers/lamina of sedimentary cores;
- **Developing empirical models** which by a series of multiple regression analyses can be used to describe the association between environmental variables which in turn predict unknown parameters (dependant variables) from other 'known' parameters (independent variables).

Further details about these techniques are contained in the following sections.

4.4.1 Contemporary Measurements

This relatively straightforward approach involves using nutrient loading values taken from estuaries that currently have low nutrient levels or are deemed to be largely unaffected by anthropogenic influences. This has been highlighted as a possible approach for identifying background levels in aquatic environments (Rees *et al.*, 1994; Mainstone *et al.*, 1995; Moss *et al.*, 1996; Mathieson and Atkins 1995). For UK estuaries, Rees *et al.*, (1994) used Scottish Lochs, from which the lowest nutrient inputs have been recorded, to obtain background levels. Here, the nutrient levels are an order of magnitude lower than at corresponding salinity levels within estuaries throughout England and Wales (the lowest nutrient levels in England and Wales were found in the Dart). Mathieson and Atkins (1995) proposed using the Shin, Carron and Conon rivers in North-east Scotland, where the winter TON levels were <0.15mg l⁻¹, as examples of background input levels.

However, there are essentially three main problems with this approach:

- Firstly, there is the difficulty of finding a genuinely unimpacted estuary as even the estuaries of highland Scotland will receive anthropogenically-derived nitrogen through wet deposition/precipitation. Moss *et al.*, (1996) when faced with the same problem with respect to standing waters, considered using Canadian mountain lakes where catchments are likely to be the most natural but they doubted that any pristine lakes could be found in any part of the developed world. Given that estuaries receive freshwaters and their high level of surrounding development, then there is little possibility of an unimpacted estuary.
- Secondly, because the few potential examples of estuaries with unimpacted catchments will all probably share similar catchments characteristics. (*i.e.* characteristics which make them unattractive to impact, for example, a high relief or a nutrient-poor soil type). Background values, therefore, cannot be obtained from a range of different catchment types, and it is unrealistic, given the range of physiographic conditions between different catchments, to presume that background nutrient levels obtained from estuaries with a specific catchment type will be widely applicable across all estuarine catchment types.

- Finally, as Rees *et al.*, (1994) also point out, even if this were a viable approach then its application is presently restricted because there is a lack of available data on these un-enriched estuaries, with the majority of currently available data being for contaminated estuaries. In addition, although oligotrophic estuaries are known, e.g. in north-mid Wales (Dr R Jones, CCW, pers com.), their characteristics dictate that they would not provide comparable baseline conditions for other estuaries.

4.4.2 Nutrient Export Modelling

As described in Section 2.2 the rate at which nutrients are exported from the catchment to rivers and estuaries depends on a range of factors such as agricultural activities, management practices, habitat types, soil composition and topography. The relationship between these factors and nutrient export has been extensively investigated so that measures of nutrient export ('nutrient export coefficients') are available for a range of different conditions. The nutrient export modelling approach to hindcasting involves identifying the nutrient export coefficients for all the different areas of a catchment and then calculating historical changes to the rate of nutrient export from catchment based on known changes to the land uses.

A nutrient export model was developed by Johnes (1994) to relate the impact of land use changes since the 1930's to the nutrient (as annual means of TP and TN) loads reaching the Ore/Alde and Deben estuaries. The modelling procedure used by Johnes (1994) (Figure 27) involves the following four stages:

- **Collection of data on present and past land uses.** Questionnaires were sent to farmers to obtain details of the contemporary agricultural activities on a field by field basis. Parish Summaries of the Annual Agricultural Census Returns provided general details of agricultural practices, and the Survey of Fertiliser Practice for 1985 provided information on the extent of fertiliser applications to crops in the catchment. Ward and Parish Population Censuses conducted every decade since 1931 provided data on the size of the human population in the catchment and data were obtained from the NRA on sewage treatments and the location of point source discharges.
- **Model development:** TN and TP inputs to the catchments were calculated from accumulated data. Several suitable export coefficients were derived to estimate rates of nutrient loss from land use types and the overall nutrient load to the Ore/Alde and Deben was calculated as the sum of the individual exports from each nutrient source.
- **Model validation.** The model was checked against recorded data. Ideally, long term data are required but in this case only a single year's data was available (the model was constructed for 1988 and data were only collected after 1987) so the validation was purely at a spatial level.
- **Model application:** The model was initially run forward to predict changes based on impending alterations to catchment land practices or the instigation of mitigating management strategies. The model was then run backwards to a baseline date of 1931.

4.4.3 Palaeoecology

Palaeoecology is the study of historical environmental changes through analyses of biological or chemical indicators preserved in the substrata. Where there has been continued depositional conditions in aquatic environments, then changes in the layers of sedimentary

profiles of particular biological indicators can be used to reflect the accompanying temporal changes in environmental conditions. Potential palaeoecological indicators include pollen, diatoms, dinoflagellate cysts, total organic carbon (TOC), nitrogen, total sulphur, the degree of iron pyritization and Chironomid larvae head capsules (in freshwater) profiles. If suitable conditions prevail then this technique can be used to describe changing conditions for up to thousands to millions of years. Palaeoecological techniques are being increasingly used to assess 'ecosystem health' in standing waters although the value of this procedure in estuarine systems is less clear.

The procedure was successfully used in Chesapeake Bay by Cooper and Brush (1993) who studied the abundance and species composition of diatoms (Figure 28) preserved in the layers of four deep sedimentary cores (114-160cm) retrieved from the mesohaline region (in the deepest channels) of the Bay. These profiles were used to describe changes to the environmental conditions from the present day to pre-European settlement when climate was the major factor influencing the ecology of the Bay. The chronology of the sedimentary layers was obtained by 'Carbon dating' the bottom sediments, and using pollen concentrations in the upper layers as indicators of sedimentation rate. The following parameters of diatom community structure were analysed abundance, diversity (Shannon-Wiener), ratio of centric (generally pelagic):pennate (generally benthic) species, and the abundance of certain species with time. They reviewed the results of this study in conjunction with previous analyses of three other 'palaeoecological indicators':

1. **Total Organic Carbon** (TOC) which indicates the rate of organic matter accumulation in relation to sedimentation (input) and benthic oxygen demand (bacterially mediated decomposition). Phytoplankton (all species) contribute to the total carbon in the bay, whilst an increase in nutrients may lead to a depletion of bottom water oxygen;
2. **Sulphur**, this reflects changes in rates of sedimentation and the concentration of TOC available to sulphur reducing bacteria. In anoxic conditions these anaerobic bacteria reduce sulphate to sulphides which either form H_2S and are lost from the sediment or combine with metals, *e.g.* iron, first as iron monosulphide then as pyrite which is retained;
3. The **degree of iron pyritization** (DoP) is an indicator of bottom water oxygenation of organic bearing sediments because in anoxic conditions a higher proportion of iron minerals should become converted to pyrite.

These analyses allowed descriptions of the changes in: sedimentation rate, diatom community structure and geochemistry in the estuary from approximately 550AD to the present. There was an increase in sedimentation associated with European settlement and land clearance in this area. The DoP and sulphur results showed that incidents of hypoxia and anoxia may have been more severe since 1940. Additionally, during the last 40-50 years there appears to have been a progressive decrease in diatom diversity indicating increasing stress on this community, together with an increase in the centric:pennate ratio that is indicative of an overall loss of habitats.

Although, Cooper and Brush (1993) have shown that palaeoecological analysis are possible in Chesapeake Bay, the extent to which this technique can be used in other estuaries is unclear. Chesapeake Bay is particularly suitable because it has a high and continuous sedimentation rate which, together with periods of hypoxia and anoxia, prevents the development of a diverse macrofauna in this part of the bay (Harding *et al.*, 1992). However, in areas where the sedimentation is low or the reworking of surface sediments, by physical or

biological processes, is high then the environment is likely to be unsuitable for palaeoecological studies. This was demonstrated by Nixon (1988) who noted that in contrast to lake sediments which often preserve an historical record of increasing eutrophication, the sediments in the coastal waters of Narragansett Bay and Chesapeake Bay were found to be only enriched in the surface 5-10cm where there is active mixing. Below this surface zone of the sediments the concentrations of C, N and P rapidly decline. Nixon (1996) also points out that the sediments in Narragansett Bay are unsuitable for palaeoecological studies because of this extensive bioturbation. Nixon (1996) however, used another approach, in analysing pollen in sedimentary cores of ponds adjacent to the larger Narragansett Bay, to demonstrate that the Narragansett Indians had been clearing land by about 1000 years BP.

Another technique was used successfully in the Australian Peel-Harvey and Swan-Canning estuaries by Gerritse *et al.*, (in prep) who analysed deep sediment cores to describe the changes in phosphorus levels over time. Here correlations were observed between Fe concentrations and phosphorus in older sediment and Fe levels in the sedimentary layers were used as a guide to the 'background' phosphorus levels. However, this procedure may not be very useful for temperate estuaries where nitrogen is often the limiting factor and the primary concern.

4.4.4 Empirical Models

A further technique for determining background biological responses is to derive empirical models which describe the relationship between physico-chemical characteristics (including nutrients) and biological responses in estuaries. For example, such an empirical model is a multiple regression analysis which describe the relationship between one or more physico-chemical factors (including nutrient levels), which are independent variables, and one of the biological responses as the dependant variable. Such regression equations can be used to predict biological responses from physico-chemical conditions (e.g. see Elliott and O'Reilly 1991). For example to predict the chlorophyll-*a* level from current physico-chemical conditions the following equation could be used:

$$\text{Chlorophyll-}a = (\text{flushing time})b + (\text{nitrogen})b_1 + (\text{phosphorus})b_2 + (\text{light extinction coefficient})b_3 \text{ etc. until all relevant independent variables are accounted for and where } b \text{ values are regression coefficients.}$$

However, this equation only describes responses to current physical-chemical characteristics and nutrient inputs and the equation's limits will only be as broad as the data used to construct the equation. If data from a wide range of estuaries could be obtained, including nutrient poor areas, then the model may be valuable in prediction. If this is not possible, it may be necessary to use other techniques to obtain a measure of the background conditions in order to hindcast nutrient levels (e.g. nutrient export modelling).

This approach has a drawback in that in estuaries it is difficult to make clear links between physico-chemical variables and biological responses because of the variability of conditions or at best the nature of the relationship (i.e. linear, curvilinear) is not known. This is clearly demonstrated by the work of Boynton *et al.*, (1982) who collected data for 63 estuaries to determine the principle factors affecting phytoplankton growth. They found that, across a range of estuary types there were 'few, if any, environmental variables which alone satisfactorily predict' phytoplankton production.

An empirical modelling approach has however, been used by Moss *et al.*, (1996) as part of a scheme for classifying standing (lake) waters on the basis of changes in the environmental conditions since a predetermined baseline date. To describe the baseline environmental conditions within standing waters, the statistical relationships were calculated between environmental variables for which historical data are available ('driving variables') and other environmental indicators ('derived variables'). Therefore, baseline environmental conditions could be determined by predicting the unknown derived variables from the known driving variables.

The following four factors were selected as independent 'driving variables':

- inflow concentrations of total phosphorus;
- inflow concentrations of total nitrogen;
- conductivity;
- retention time.

The baseline status of these factors can be calculated from available data using export coefficient modelling for nitrogen and phosphorus export from catchment, geological databases to estimate conductivity, and meteorological records with a knowledge of the lake bathymetry to calculate retention time. Against these calculated variables the following series of further environmental and biological factors or 'derived variables' were predicted (unless otherwise stated these were mean annual values):

- total lake P concentrations;
- total lake N concentrations;
- total alkalinity;
- calcium concentration;
- winter total DON (effectively NO_3^-);
- maximum chlorophyll-*a* concentration;
- N:P ratio in inflow water;
- mean annual pH;
- minimum potential Secchi depth;
- index of submerged and emerged plant communities ('plant ranking score' based on an NCC aquatic plant classification system).

Each of these derived variables (or dependant) variables were calculated on the basis of regression equations with either one of the three driving variables or a previously calculated derived variable acting as the independent variable. For example, the TP in the lake (P) was calculated from two driving variables:

- the total annual export from the catchment;
- the mean annual river discharge.

These derived variables were chosen because:

- they are signals of potential eutrophication, oligotrophication, acidification, salinisation and sedimentation;
- they are the subject of standardised monitoring protocols (not TN);
- they are biological indicators which do not require high taxonomic expertise and investment;
- they can be measured rapidly and they are well understood by the scientific community.

The following parameters which were excluded from the above list because they are not predictable from driving variables or because the monitoring procedures that were required to determine them were too costly:

- Hypolimnion oxygen concentration, which despite being an important indicator of eutrophication was excluded because it is unpredictable and not easily correlated with lake productivity. This is because the oxygen concentration is not only a function of productivity but also of many other characteristics such as depth, temperature, degree of wind exposure and hypolimnion:epilimnion volume ratio.
- The benthic fauna, because of the high variability due to differences in substrata and levels of predation on community structure. This variability therefore makes it difficult to compare the benthos with, or to predict them from, other variables.
- Plankton species composition: again because of the high variability of this parameter and because it is not possible to predict species from water chemistry and to account for the impacts of zooplankton and fish predation.
- Fish community, because there have often been changes to natural stocks through activities such as restocking or the interference of migration routes.

4.4.5 Other Approaches

The literature identifies two other approaches although these are not strictly hindcasting techniques as they both require historical data but they are potentially useful techniques for assessing changes over time. The two techniques involve:

- estimating background nutrient conditions in estuaries where historical nutrient data from freshwater and coastal waters are available;
- identifying changes to environmental conditions and nutrient impacts where historical biological data are available.

Where historical nutrient data from freshwater and coastal waters are available then background nutrient concentrations in estuaries can be estimated by producing estuary mixing models. Van Raaphorst *et al.*, (in prep) used mixing models to combine the historical nutrient data from marine and freshwater sources to produce a single background value for the tidal basins of the Wadden Sea and Ems estuary. They note however, that for the Ems estuary which, as opposed to the Wadden Sea basins, is a true estuary, a more detailed model would be required and that simple mixing models producing one single background value for whole estuary is unlikely to be sufficient. They therefore, recommended using models such as the BOEDE model (Baretta and Ruardij, 1988) or 'with modifications' the EcoWasp

ecosystem model (Brinkman, 1993). These models include physical mixing and advection processes as well as the most important chemical and biological transformations.

Changes to environmental conditions can be identified by charting the changes in abundance or spatial coverage of indicator species/communities which are sensitive to enrichment impacts. One technique for monitoring changes in biotic indicators is the AMOEBA approach. This is a graphical procedure for simultaneously describing changes to communities and/or species. The relevant biotic indicators are set around the circumference of circle and over time the circumference is moved out to demonstrate increases in abundance of this species/community or moved inwards to show a decrease. This technique has been used on the Rhine to demonstrate changes in communities over time.

4.5 Summary and Conclusions

Data adequacy

1. All attempts at synthesising data from several sources have difficulties resulting from either survey design incompatibility, in which data are being used for purposes that the original survey was not for, and inadequate Analytical Quality Control and Quality Assurance (AQC/QA).
2. The problem of quality assurance may be minimised in the future with the use of ICES or QUASIMEME for nutrient measurements or through the OSPARCOM working group under ASMO (Assessment and Monitoring Group) who have produced agreed standardised protocols for international comparisons. Where similar determinands have been measured in monitoring programmes it is often the case that methods and/or units have not been standardised.
3. In addition to the above, together with constraints on the effort spent on surveillance monitoring across more than one geographical area, there are few wide-scale monitoring programmes which provide adequate coverage and wide case-studies. However, through experience elsewhere (see Sections 2 and 3), it is possible to indicate the parameters for which data are required in order to make an evaluation of trophic status of an estuary.
4. The key chemical parameters are: nitrogen, (NO_2^- , NO_3^- and NH_4^+), phosphorus (PO_4^{3-}), silicon, salinity and pH.
5. The key physical parameters are: water volume, flow rate, tidal range, flushing time and turbidity.
6. The key biological parameters are: water quality barriers (DO sags) to fish migration; the presence of opportunistic plant and animal populations in relation to the level of organic enrichment and the denitrification rate.

Monitoring Tools

1. As the cost and value of field surveys can range greatly, it is necessary to identify the level of detail at which a system needs to be studied before funding investigations (i.e. resources need to be allocated according to requirements).
2. The parameters to be modelled have been easily identified and the methods for those parameters are available in standard texts. However, the degree of Analytical Quality Control and Quality Assurance between field and laboratory assessments requires to be developed further.
3. The spatial and temporal frequency of monitoring places/opportunities can only be derived on a site-specific basis and according to the questions being asked and the effort available. Despite this, it is recommended that the most intensive studies are directed towards the estuaries where the perceived risk of impact is greatest while at the same time covering a

range of geographical areas to give spatial coverage and to extend the types of estuary covered from oligotrophic to (perceived) eutrophic.

4. A proposed framework for determining monitoring requirements is proposed in Section 5.2 which requires further development by a workshop or task team.
5. Many modelling exercises have been completed and models developed. These indicate that while the background (conceptual) basis is sufficiently well known to derive the model, either the data are incomplete to validate and calibrate the model, or long-term processes are insufficiently known or the non-linear responses in the system cannot be accommodated sufficiently well.
6. The physical-process models are most well developed although these again require a high degree of site-specificity, and the biological components can be modelled to some extent, e.g. nutrient uptake kinetics. However, with increasing complexity in the system being modelled, from physical characteristics to nutrient kinetics to phytoplankton uptake to bloom formation and zooplankton response to benthic response, the degree of unexplained noise increases such that the models are less-good for prediction and thus management.

Approaches to the management of estuaries

1. The literature indicates many valuable approaches to the nutrient management of estuaries and wetland areas, where the examples are taken from studies worldwide.
2. Some of these approaches are thorough, e.g. that proposed by US NOAA, but they require a field validation. Such a national strategy would be worthwhile for UK estuaries.
3. Other approaches, such as European or Australian schemes, could be expanded for the UK areas and, in the case of Classification schemes, will provide a wider benefit in terms of overall quality assessment and conservation value. It is notable that such schemes may also be valuable in the management of specific sites, e.g. Special Areas of Conservation under the EU Habitats and Species Directive.
4. The multivariate statistical approach has merit given the complex nature of data provided in comparisons of different water bodies although techniques have improved since early attempts, e.g. the use of canonical correlation and correspondence analysis. Similarly, increases in computing power makes such techniques easier and more effective and allows a predictive approach within multivariate methods.
5. The several examples of a hierarchical approach, in which estuaries are successively separated according to characteristics, are also valuable although it is of note that in the examples, the class boundaries appear somewhat arbitrary. There is a need to merge the limits given in the literature to provide an overall classification as long as the same processes are considered to operate in all water bodies.
6. The European legislation relevant to nutrients, the Nitrates and Urban Waste Water Treatment Directives and the proposed Water Framework Directive will have an increasing control on estuarine quality. In addition, the implementation of the Habitats Directive, and the need to introduce management schemes for SACs, will also require

nutrient effects to be controlled. However, while each of these discusses quality in the broadest sense and they include Objectives, they do not give standards relevant to nutrients in estuaries. The UK implementation of the Water Resources Framework Directive has not yet been defined and so it is not known whether it will control nutrients and their effects in a quantitative manner.

7. The role of Ecological and Environmental Quality Objectives and Standards also requires to be considered further with respect to nutrients. While it may be difficult to set quantitative standards, as targets (although the literature suggests that other countries are considering this) it is possible to set the objectives as statements against which monitoring can be carried out.

Hindcasting nutrient status

1. Hindcasting nutrient loads or environmental conditions in estuaries is valuable for providing a better understand of the relationship between estuary-type and the susceptibility of estuary communities to nutrient impacts. However, hindcasting techniques are often difficult, have drawbacks or are costly.
2. The taking of contemporary measurements is simple and inexpensive but appears to be an impractical approach for several reasons: no contemporary estuary can be truly pristine, between-catchment variability is too great to use nutrient inputs into one estuary to assess loads at another, low-nutrient systems are confined to upland areas and unlikely to be indicative of lowland conditions and there is a poor data availability for low-nutrient systems.
3. Catchment nutrient export modelling appears to be an effective approach and has the benefit that export coefficients are widely available as are historical land-use data. However, it is very costly, time consuming and is only likely to be viable tool for hindcasting nutrients if an economically viable, and therefore, less accurate, approach could be designed. In addition, it does not provide information on the biological response to nutrients in freshwater or estuarine systems.
4. Palaeoecology can measure biological responses for all potential baseline periods but it is expensive, time consuming, only applicable in certain types of estuary (with depositional conditions) and does not determine nutrient levels or discriminate between nutrient-related impacts or other environmental changes. It is possible that palaeoecological work carried out in the freshwater environments upstream of estuaries may be valuable for estimating the historical rates of nutrient imports to estuaries from freshwaters sources.
5. The development of empirical models is considered to be a useful approach especially as sufficient data are likely to exist to produce models which describe the contemporary relationships between nutrients and biological responses. However, in the absence of long term datasets on nutrient concentrations another technique, probably nutrient export modelling, is required to determine historical changes in nutrients in order to use these empirical models to describe environmental changes over time. Multiple regression models will be valuable and accurate predictors if the suitable algorithms can be derived for the nature of the relationships between independent (physico-chemical) and dependent (biological response) variables.

6. The two other approaches (mixing model and AMOBA) both require historical nutrient and biological data and are therefore, not true hindcasting techniques. However, the mixing model approach presents a possible method for determining background nutrient concentrations in estuaries if information on freshwater nutrient inputs can be obtained from other hindcasting procedures (e.g. nutrient export modelling and freshwater palaeolimnology studies). The AMOEBA approach could be valuable for graphically procedure for describing future changes to sentinel communities and species.
7. It should be emphasised that all these techniques are limited to some extent by the adequacy of historical data with which to validate such approaches. It should also be noted that historical conditions in estuaries should not be thought of as targets for management purposes. This is because returning estuaries to a natural state is generally not a viable or achievable goal. Instead the most viable management targets should be the improvement of ecosystems in which problems have been identified by using procedures such as the setting of Ecological Quality Objectives.

5. RECOMMENDED MONITORING & MANAGEMENT APPROACH

Section Aims

- To propose approaches for the monitoring and management of estuaries based on the information obtained in the review sections (Section 2, 3 and 4) and experience.

Section Contents

Section 5.1 Proposed Estuary Classification Schemes:

Approaches for classifying estuaries with respect to nutrient impacts are recommended. Key physico-chemical factors are identified as classification criteria (based on Section 2 review) and six potential classification schemes are proposed which include quantitative, semi-quantitative and qualitative techniques.

Section 5.2 Proposed Monitoring and Management Protocol:

Protocols for monitoring and management of nutrient impacts in estuaries are proposed. This protocol includes including a series of EcoQOs which have been derived (based on Section 3) to clearly assess biological symptoms of eutrophication.

Section 5.3 Final Recommendations:

Recommendations for future studies to fill gaps in knowledge and a recommended protocol for Phase Three of the 'Impacts of Nutrients in Estuaries' project.

5.1 Proposed Estuary Classification Schemes

The response of communities in estuaries is dependent on a range of physico-chemical factors and that the relative influences of these factors and the associated risk of biological impact can vary greatly between different systems (see Section 2). The differences in response between systems based on a review of those studies which have compared responses in separate estuaries are shown in Appendix 6. This between-estuary variability as well the dynamic nature of individual systems makes it difficult to define clearly the cause and effect relationship between nutrients and biological response and therefore, to clearly identify the risk of impact in an estuary.

These characteristics present difficulties for the Environment Agency and English Nature who require a clear indication about the potential biological response in an estuary if they are to clearly identify the best strategies for monitoring nutrient levels and assessing the risk of impacts to communities. A scheme to classify estuaries on the basis of their susceptibility to nutrient impacts is, therefore, a valuable tool for making valid assessments on monitoring and management protocols.

The potential problem for the development of classification procedures lies with the level of detail which needs to be incorporated into them. Any classification procedure which considers all the numerous physical and chemical factors may be too detailed and likely to produce too many categories to be of any benefit as a useful monitoring tool. Conversely, any scheme that groups the estuaries too broadly is equally unlikely to be adequate. This section therefore aims to review the possibilities for developing a scheme that is simultaneously sufficiently robust to encompass the entire range of different estuary types, whilst being sufficiently efficient to be used as an effective monitoring tool. However, a viable scheme is considered to be achievable because, despite the lack of a good quantitative knowledge of the cause and effect relationships between physico-chemical factors, biological response and nutrient loads, there is sufficient knowledge to make informed judgements about the risks of ecological change based on key physico-chemical characteristics.

The following section lists these key physico-chemical characteristics and, based on these characteristics or 'classification criteria', a range of suitable classification schemes are then proposed. These schemes all aim to assess susceptibility of estuaries to biological change but they vary in terms of their complexity and the amount of data that are required.

5.1.1 Classification Criteria

It is recommended that the following physico-chemical characteristics should be used as the basis for a classification scheme to identify susceptibility to impact of estuaries:

- **Nutrient Input:** The amount of allochthonous nutrients entering a system clearly influence the potential impact to communities within a system. Low-nutrient 'oligotrophic' systems may prove to be an exception to this because in such estuaries, only small increases to already low nutrient inputs may be needed to have an adverse impact. However, as these systems are atypical and little is known of their responses to nutrient they should be considered separately from the majority of estuaries.
- **Turbidity:** Turbidity levels affect the light availability and the potential response of autotrophic species to available nutrients. It also has a significant effect on the

transformations of nutrients with higher numbers of bacteria in turbid areas promoting higher levels of mineralisation and nitrification and consequently DO depletion. Turbidity also has a major effect on the dynamics of phosphorus.

- **Flushing Time:** The flushing time influences the duration of nutrients within the estuaries and therefore the availability of allochthonous nutrients to autotrophs and bacteria as well as the potential for the recycling of nutrients within the estuary and therefore, the supply of autochthonous organic matter.
- **Tidal Range (relative to depth):** This influences the water circulation and flushing rate which have consequences for the stability of the substratum and the turbidity of the water column.
- **Risk of Stratification:** Stratification can be responsible for many biological impacts in estuaries including phytoplankton bloom formation and oxygen depletion in the bottom waters.
- **Freshwater Input (relative to volume):** Freshwater input affects the nutrient delivery rate from diffuse sources, the dilution of point source nutrients and the physical characteristics of the system such as the water circulation, turbidity levels and flushing time.
- **Width:Depth and Intertidal:Subtidal area ratios:** These two physical characteristics describe the bathymetry of an estuary. They influence both the potential for response and also the character of the response itself because they determine which autotrophic taxa are most likely to utilise the available nutrients. For instance the percentage area of intertidal and shallow subtidal zone determines the degree to which benthic micro and macro-algae can respond to nutrients.

Two other factors that have not been included in this list are sediment type and resuspension intensity which both influence bacterial processes and the nutrient cycling capacity of an estuary by affecting bacterial numbers and the nature of the benthic bacterial process. Sediment resuspension also affects the stability of substrata and the potential for the growth of macroalgal and microalgal mats. These two factors have not been included in the list above because other factors such as turbidity, bathymetry and tidal strength may all incorporate some measure of these sediment characteristics. For example, sediment resuspension is generally greater in shallow estuaries, turbidity levels clearly reflect the levels sedimentary disturbance and tidal strength affects the characteristics of sedimentary deposition.

Data are available or accessible for most of these criteria but to classify estuaries using them it will be necessary to obtain quantitative measurements from a range of estuary types and to then derive scales and define limits for each criteria (e.g. from highest to lowest flushing time).

It will then be necessary to define the boundaries or cut-off levels on these scales in order to separate estuaries into categories on the basis of individual criteria. These boundaries can then be used to determine, for instance, whether an estuary is relatively turbid or whether it has a comparatively long flushing time.

It will also be necessary to identify the relative importance of each criterion to ensure that a good overall indication of susceptibility is achieved when all the criteria are collectively compared. For example, if a long flushing time makes a greater contribution to the

susceptibility of estuaries than a low tidal range then the former characteristic must be assigned a higher level of importance.

Determining criterion importance is however, complicated by the relationships between criteria as each individual one is to some degree related to one or more others (*e.g.* bathymetry affects the potential for stratification). It will be important to identify the inter-relationships between factors to avoid overemphasis on certain estuary characteristics. For example deep estuaries are most likely to become stratified but if the occurrence of stratification is considered as a sign of susceptibility independently of Width:Depth there will be a corresponding bias towards a classification of 'susceptibility' for deep estuaries.

Attention also needs to be directed towards identifying the most suitable coefficients for describing each criterion. Where there is a relationship between criteria, these coefficients may need to describe this link. For instance, flushing time is likely to be longest in the deep estuaries but it will also be altered where stratification occurs and therefore, a measure of flushing time needs to be derived which incorporates a measure of the degree of stratification.

Finally, it will be necessary to test and continually reassess the criteria scales, cut-off levels, coefficients and any associated classification schemes in relation to information on biological responses in order to ensure that they are ecologically relevant. All assessments of biological response should be made against compliance with Ecological Quality Objectives which are described in Section 5.2.1.

Assuming that the above conditions have been met then the following three protocols have been identified for classify estuaries using these criteria:

- **Multivariate Analysis** (described in Section 5.1.2);
- **Tabulated representations** (described in Section 5.1.3);
- **'Decision tree'** (described in Section 5.1.4).

It is recognised that, in the short term, it will be difficult to meet all the requirements described above for describing the classification criteria. Therefore, several alternative classification schemes have also been identified which require less data or utilise qualitative information and can be developed very rapidly. These schemes are:

- **Three factor Approach** This approach uses what are considered to be the three most important criteria: flushing time, light availability and nutrient inputs (described in Section 5.1.5);
- **Nutrient Input/Output** This scheme evaluates the nutrient capacity of an estuary, or regions of an estuary, based on the input and output of nutrients (described in Section 5.1.6);
- **Best Expert Judgement** This is a largely qualitative approach in which a questionnaire is compiled and completed for an estuary and the 'best expert judgement' about estuary susceptibility is sought based on the answers to the questions (described in Section 5.1.7).

5.1.2 Multivariate Statistical Analysis

Multivariate statistics provides the most objective means of classifying estuaries based on the criteria cited above. These techniques are used to identify trends in complex data sets and are

widely used for instance to describe the similarities between benthic communities. They could be used to make comparisons between a large number of different estuary systems based on the classification criteria. However, to achieve this requires a very clear quantitative description of the criteria and it would be necessary to meet all the requirements detailed above such as describing the scales of the criteria, defining their limits and identifying their relative importance and the relationships between them.

These techniques produce graphical outputs (*e.g.* ordinations and dendrograms) and single value coefficients to describe the similarities between estuaries and to separate them into classification categories. In addition biotic variables could be included and, using joint ordination (*e.g.* CANOCO) techniques, estuary types could be correlated, and tested, against biological response information.

5.1.3 Tabulated Representations

This approach involves presenting the classification criteria values in a tabulated format. The classification of estuary susceptibility is then based on the visual interpretation of the resulting table. This approach requires that accurate scales are derived for each of the classification criteria although, unlike multivariate statistics, the identification of an estuary category is achieved more subjectively.

A possible format for this interpretation is shown in Table 7 where the eight criteria are shown and perceived susceptibility represented by the extremes of each criterion has been identified (*i.e.* low flushing time represents low susceptibility; high nutrient input represents high susceptibility). The extremes of the scales are aligned within the tabulated framework so that criteria values to left of the scale represent conditions which characterise an estuary that is 'unlikely to be susceptible' and values on the right represent conditions of estuaries that are 'likely to be susceptible'.

Table 7: Proposed framework for classifying estuaries on the basis of their susceptibility to nutrient loading based on eight key factors

Classification Criteria	Estuary Unlikely to be Susceptible	Scale								Estuary Likely to be Susceptible
Nutrient Input	Low									High
Turbidity	High									Low
Flushing Time	Short									Long
Tidal Range (relative to depth)	High									Low
Risk of Stratification	Low									High
Freshwater Input (relative to volume)	Low									High
Width:Depth	Low									High
Intertidal:Subtidal	Low									High

To illustrate how this table might appear when this approach is used to classify estuaries, hypothetical descriptions of a high risk (likely to be susceptible) and a low risk (unlikely to be

susceptible) estuary are shown in Table 8. These two estuary types have the following characteristics:

- **High risk estuary:** This is a deep very slow flushing (probably fjordic) system with high freshwater and nutrient inputs, low turbidity (high light availability), low tidal range relative to depth (limited exchange with coastal water and reduced water circulation), high risk of stratification (likelihood of phytoplankton blooms and bottom water oxygen depletion).
- **Low risk estuary:** This is a shallow rapidly flushed system with a low freshwater and nutrient input and a high turbidity (inhibiting photosynthesis), high tidal range (high exchange with coastal water and strong water circulation) and negligible risk of stratification.

Table 8: Hypothetical application of tabulated classification scheme showing high risk and low risk estuaries.

Criteria/Estuary Class	HIGH RISK ESTUARY										LOW RISK ESTUARY										
Nutrient Input	Low										High	Low									High
Turbidity	High										Low	High									Low
Flushing Time	Short										Long	Short									Long
Tidal Range	High										Low	High									Low
Risk of Stratification	Low										High	Low									High
Freshwater Input	Low										High	Low									High
Width:Depth	Low										High	Low									High
Intertidal:Subtidal	Low										High	Low									High

5.1.4 Decision Tree

For this approach the overall classification for estuaries would be obtained by successively dividing estuaries into groups on the basis of each criterion in turn. For instance, estuaries could initially be separated into high and low turbidity before considering whether they have high or low inputs *etc.* Then a successive set of divisions would then be made for each of the remaining criteria. The sequential divisions could be graphically laid out as a 'decision tree' and the classification of the estuary is determined by the category in which it is found at the final division in the sequence.

5.1.5 Three Factor Approach.

A relatively simple classification could be produced using just three major factors. The principle behind this approach is that the risk of eutrophication in estuaries is considered to be largely dictated by the flushing time, the nutrient input and light availability. In general, nutrient enrichment is unlikely to occur if there is a rapid or moderate flushing time and a low nutrient input. Conversely, if there is a long flushing time together with a high nutrient input then the risk of eutrophication will depend on the light regime. Although this approach is simplistic and not as effective as that described in the previous section, it is likely that the quality and quantity of available data that are presently available are only sufficient to support

such an approach. Two graphical methods have been identified for deriving classification categories from this limited number of factors; these are:

- **Histogram Charts:** For this approach the three factors are plotted against three independently scaled y-axes. To separate high risk from low risk estuaries a 'cut off' point is identified on the y-axis and the classification category depends on the position of the plotted values relative to this point. For example, where the values for these factors lie above the 'cut-off' point (*i.e.* where estuaries have high nutrient, high flushing time and high light) then the risk of eutrophication will be high. This scheme can be expressed either as individual histograms for each factor (Figure 29a) or as cumulative histograms for each estuary (Figure 29b).
- **XYZ Scatter Plots:** For this approach the same three factors are plotted on an XYZ (3-dimensional) scatter graph (see Figure 30). The technique for separating high risk from low risk estuaries would be the same as that described above although instead of identifying the classification category from the position of the plotted values relative to discrete cut-off values, it is identified from the areas of the graph. This approach would therefore be more qualitative and robust than the histogram plot.

5.1.5 Nutrient Input/Output

This involves treating estuarine systems as 'black boxes' and classifying them solely on the basis of the inputs and outputs of nutrients from the system. This would describe the estuary's 'storage capacity' *i.e.* the amount of nutrients retained or removed by it.

A 'single box' approach could be used which describes inputs and outputs from entire estuarine systems although the eventual aim should be to develop a 'multiple box' approach in which the inputs and outputs are measured for separate sections of individual estuaries. This approach would help to separate areas such as the turbidity maxima regions in the upper estuary from other areas, which typically have different light regimes and environmental conditions.

This procedure would require data on nutrient concentration as well as information on the flushing rate and tidal exchange characteristics. It is considered that sufficient information is now available to measure these criteria across a range of estuary types. The Environment Agency's Harmonised Monitoring Programme for instance will provide a good source of information on nutrient inputs to estuaries. The results from the Harmonised Monitoring Programme have for instance been effectively used by Isles (1997) who studied inputs to the Mersey estuary. For this programme nutrient levels are measured at the tidal limit, this is considered to be the best location to take samples just as measurements taken within the tidal area will be inconclusive due to the chemically dynamic and highly variable nature of the Freshwater Seawater Interface.

Although further information is clearly required about the physical characteristics of the system in order to fully assess its susceptibility, this technique could be a potential first step towards more detailed classification systems.

5.1.6 Best Expert Judgement (Qualitative Approach)

A further possible method could be a questionnaire approach whereby a series of questions need to be answered in order to identify the sensitivity of a system. This approach will allow for the inclusion of often subjective assessments, by experts, of the conditions within an estuary. It can also accommodate general 'qualitative' indicators of an estuary's susceptibility to nutrient elevation as well as the quantifiable criteria described above. For example, one of the major influences on the trophic status of an estuary is the number and location of point source discharges. If an estuary has point source discharges at several locations throughout its length then it is likely to be at risk of eutrophication. A qualitative measure cannot easily be assigned to such a criterion and so it would not be possible to include this criterion within the previous classification techniques.

Another advantage of this approach is that it can accommodate factors that are not measurable or others for which data are unavailable. For instance, in the absence of any datasets to describe the hydrography of an estuary, then it may be sufficient, in the short term, to ask more general questions such as, 'Is the estuary subject to seasonal stratification?' or 'Is there a sill present?' *etc.* The answers to these questions would help to give a qualitative indication of the extent of water circulation and residence time.

The capacity to incorporate qualitative criteria makes this the most robust approach and possibly the technique most suitable for management purposes. However this classification procedure will also require the most testing and the greatest degree of manipulation before a final working procedure can be produced. The local expertise of Environment Agency and English Nature officers dictates that they may be in the best position to assess qualitatively an estuary's features, particularly with respect to quality assessment and conservation status.

5.2 Proposed Monitoring and Management Protocol

The Environment Agency and English Nature both have a clear need to identify the most appropriate way of monitoring and managing estuaries with respect to nutrient-enrichment. Ecologically relevant protocols are required for the Environment Agency's monitoring programmes for quality assessment and to alert English Nature to conservation concerns and for the protection of habitats and species. The present study considered the most appropriate way forward for both the Environment Agency and English Nature and concluded that a common approach is required by both agencies.

For both the Environment Agency and English Nature, the most important element in any protocol needs to be an assessment of the present scale of the problem in English and Welsh estuaries² and in particular an identification of any 'signal' (response or stressor) against the background of natural and analytical 'noise'. The Environment Agency needs to identify those estuaries where the actual or perceived threat of ecological change from enrichment is greatest in order to ensure that their level of monitoring is most comprehensive in these critical systems. English Nature needs the same information to determine where the risk of change to habitats is greatest.

A proposal is presented here which comprises a set of decision steps that need to be taken in order to identify the estuaries which are susceptible to enrichment. In the development of this protocol particular consideration was given to the requirements of staff in each of the regional Environment Agency and English Nature offices who will be required to implement such a programme. It is recognised that for implementation to be effective any scheme must be simple, amenable to easily collected or already available data and that the output should be readily assimilable by environmental managers. It should also be universally applicable to all estuary systems but that the level of monitoring should relate to the degree of the perceived problem.

The protocol outlined in the following sections is designed to meet these criteria but at present it represents a preliminary framework and adaptations will be required in response to the availability of data and the specific requirements of the Environment Agency and English Nature. It is recommended therefore, that the development and application of this procedure should be carried out concomitantly. The protocol is divided into the following four stages:

- screen estuaries for symptoms of eutrophication;
- assess the severity of biological symptoms and the status of available biological data;
- identify the reasons for severe symptoms, determine whether nutrient enrichment is the causal factor and assess the status of available physical and chemical data;
- define the response by the agencies.

A flow diagram has been constructed to show the protocol (Figure 31).

² In recognition that the great deal of valuable work additional expertise and information that is available that available in Scotland from SEPA, and in Northern Ireland from the DOE(NI), it is strongly recommended that there is a close collaboration with these organisations for all future projects and that estuaries are studied on a U.K.-wide basis.

5.2.1 Stage 1 - Screening of Estuaries

The aim of the estuary screening phase is to undertake an initial overview of estuaries to determine the presence of symptoms of eutrophication or hypereutrophication. This will involve a broad review of available biological information and may, in instances where there is a paucity of such information, require site visits or small scale surveys. At this stage there is a need to make a clear distinction between a response which constitutes a 'symptom' of enrichment and one which constitutes a 'natural' biological characteristic or a 'natural' level of biological change. To help make this distinction, a set of Ecological Quality Objectives have been derived for all the major communities in estuaries following the recommendations for EcoQO development given by Elliott (1996) (Table 9). Any observed deviations of communities or habitats from these EcoQOs can be considered as 'changes symptomatic of eutrophication'. Estuaries which exhibit these symptoms are then examined in the second stage. Table 9 consists of draft EcoQOs that may be used as ecological quality indicators for the purposes of screening. They will need further detailed development to meet the specific needs at individual sites.

As this stage only requires an overview of biological data it may not be possible to fully determine whether the EcoQOs apply or not, especially as some EcoQOs involve a knowledge of biological changes over several years for which sufficient quantitative data may not be available. It is suggested therefore, that a worst-case view is taken when interpreting the data pending a more detailed investigation of the data in Stage 2.

5.2.2 Stage 2 - Assessment of Symptoms

The second stage in the protocol involves a more detailed examination of the biological responses in those estuaries where symptoms of eutrophication have been observed. The ultimate objective of this stage is to assess the severity of the observed symptoms in these estuaries and this will be achieved by again comparing observed biological responses against the EcoQOs outlined in Table 9. In contrast to the previous screening of estuaries stage, a greater effort will need to be directed towards the collation of biological data at this second stage and with this more detailed information it should be easier to determine whether there has been a departure from the EcoQOs.

As part of this biological assessment it will also be necessary to describe clearly the quality of the available data and identify the current levels monitoring in the estuaries under investigation. This description of the 'baseline' data will serve two main purposes: to provide a means of assessing the validity of the conclusions that are reached about symptom severity and also, by describing the current monitoring levels it can be used to determine what changes to these monitoring procedures are required to ensure that they adequately reflect the observed levels of symptom severity or the perceived level of risk to estuarine communities.

Table 9: Proposed EcoQOs to describe the ‘symptoms’ of eutrophication in estuaries

Community	Ecological Quality Objective
Micro/macrophankton	No deviation from structural and functional indices within the estuary
Macroalgae	Prevention of the regular occurrence of contiguous mats of opportunistic algae in the intertidal regions or an increase in the scale of blooms over a period of <i>n</i> years or occurrence of odour problems or shoreline debris following the decay of algal mats.
Seagrasses	No persistent decline in the aerial coverage or productivity relative to all estuarine vegetation over <i>n</i> years.
Saltmarshes	No persistent decline in the aerial coverage or productivity relative to all estuarine vegetation over <i>n</i> years. No persistent increase in plant tissue nutrient levels over a period of <i>n</i> years *
Benthic Macrofauna	No development of communities indicative of organic enrichment <i>i.e.</i> at the degraded conditions of the Pearson-Rosenberg Continuum. No persistent increase in biomass of polychaetes over a period of <i>n</i> years*
Fish	No creation of a water quality barrier to prevent fish migration, or fish kills from toxic blooms.
Birds	No inhibition of bird feeding of feeding areas by macroalgal bloom development and no toxic effect which may affect the palatability of prey
Holistic Ecosystem Function	NO deviation from holistic structural and functional indices
* These EcoQOs need to be developed further to ensure that they are accurate indicators of enrichment <i>n</i> - Number of years needs to be determined	

To make this assessment of the available data an approach has been developed which is adapted from that recently developed by the GCSDM Dredged Material Disposal Monitoring Task Team for the purposes of co-ordinating the monitoring of materials disposed at sea (Riddell *et al.*, in prep.). This describes the quality of the monitoring programmes according to the following four factors:

- the biological characteristics that were analysed;
- the parameters of the biological characteristics that were studied;
- the analytical technique that was used;
- the level of accuracy of the technique used.

As an example of one biological characteristic, taking the growth of macroalgal mats, depending on the extent to which it has been investigated, the relevant ‘parameters’ could range in detail from a simple assessment of whether particular species are present or absent, to studies of the areal coverage of species, or analyses of the thickness or the dry weight biomass/m² of the mat.

The ‘techniques’ used to study macroalgae could also range in detail from relatively straightforward ‘skilled eye’ studies to more expensive aerial photography or remote sensing analyses. Finally a note should also be made of the ‘accuracy’ of the studies undertaken to indicate whether they were qualitative, semi-quantitative or quantitative.

This assessment is presented in Appendix 3. Within this framework, the different parameters, techniques and levels of accuracy that are associated with analyses of responses in each of the major biological groups are identified. These are each listed in order of increasing detail and are related to the levels of symptom severity or perceived risk (*i.e.* Low, Intermediate, Medium, High). It is emphasised that the degree of potential problem should govern the extent of assessment and monitoring and the methods to be used. Therefore, with increasing symptom severity or perceived risk the associated level of detail in the monitoring programmes shows a corresponding increase.

5.2.3 Stage 3 - Identify Reason for Problem

Once the detailed assessment of the biological characteristics has been carried out and a decision has been taken that a 'severe' problem exists, the next step is to evaluate the reasons why severe symptoms have been observed and particularly, whether enrichment is the causative factor. To do this consideration needs to be taken of the physical features of the estuaries and the level of nutrient input. The approach here is similar to that used for the assessment of biological symptoms. For a range of the physical features and nutrient inputs the 'parameters', 'techniques' and 'levels of accuracy' of available information will be identified.

As in Stage 2, the information, when presented in this manner, readily shows the degree to which these factors have been studied and how much further work is required to increase the levels of monitoring or the understanding of the system. A preliminary set of characteristics, parameters, techniques and levels of accuracy for physical features and nutrient inputs are shown in Appendices 4 and 5 respectively. It is suggested that this approach needs further development in a workshop/task team.

If, at the end of Stage 2, it has been determined that the symptoms are not severe it will still be necessary to assess the potential risk of severe problems arising if nutrient inputs increased. To achieve this assessment of the 'potential response' of estuaries to changing nutrient levels it will be necessary to employ one of the estuaries classifying techniques described in Section 5.1.

As part of this assessment of risk, there may be a need to consider the impact in estuaries where the nutrient input is presently low. The habitats present in these so called 'oligotrophic' estuaries could be particularly sensitive to increases in nutrient levels. It is not possible however, to include these estuaries within a classification procedure because the category of 'oligotrophic' estuary is an entirely distinct generic category that is based only on nutrient input levels. There is a lack of any clear indication about the value of habitats within these estuaries and their response to changing nutrient levels and for this reason, a separate study on these estuaries is recommended (see later). It must be emphasised that while the input of nutrients to the majority of estuaries (which are typically organically enriched, detritus based systems) may lead to 'eutrophication' or at least hypernutrification, for low nutrient ('oligotrophic') systems it will result in conditions that are typical of many other estuarine environments. A study needs to be carried out to establish the implications of this for the habitats in these systems and to clarify whether enrichment would have a harmful or beneficial effect.

5.2.4 Stage 4 - Response by Agencies

The information obtained from the first three stages can be used to direct monitoring and management programmes. The Environment Agency needs to develop a monitoring protocol which reflects the scale of the problem in a given estuary. The most comprehensive levels of monitoring should be carried out in the estuaries where the actual or perceived risk of change from enrichment is greatest. The biological characteristic that need to be measured, the technique used and the intensity of the study will all be dependent upon the severity of the impact.

English Nature is required to consider whether there will be any impact to either the habitats recorded in the Habitats Directive or to species which drive the designation of proposed SACs under this directive. Most importantly, the conservation integrity of the estuary requires to be maintained and thus the protection of all features leading to other designations (e.g. SPA, SSSI, SMA). While it is recognised that many of these designations for estuaries relate to the health and size of the dominant predator populations, especially overwintering birds, the maintenance of conservation integrity requires a sound estuarine functioning at all ecosystem tiers (see Elliott and Lawrence, in press).

5.2.5 Response Characteristics

A further element that could be included within this protocol is an assessment of the response characteristics for different estuary systems. It is evident that biological responses to nutrient enrichment are highly dependant on the physical features of an estuary and it is important for both the Environment Agency and English Nature to understand which communities, species or habitats are most likely to respond to nutrient enrichment in any particular system.

This information will ensure that the biological responses which are considered by English Nature, or the parameters that are measured by the Environment Agency will reflect the type of estuary being studied and ensure that the sampling regime is both ecologically-relevant and cost-effective.

In terms of the monitoring procedures used by the Environment Agency it means that, if a response can be predicted then it will be unnecessary to have a blanket approach to monitoring which involves the measurement of all potential biological responses. For instance in fast-flowing estuaries (e.g. Colne) where phytoplanktonic blooms are unlikely to develop, a measurement of chlorophyll-*a* levels would not be of value.

This information would also be of value for English Nature as it will highlight the communities that are at greatest risk of change. For instance in deep slow flushing estuaries where phytoplankton blooms are more likely occur than macroalgal blooms the risk of interference to bird feeding in the intertidal region is going to be lower.

A good indication as to which communities are responding in different estuaries will be obtained during the biological assessment procedures in Stages 1 and 2 of the monitoring program. However, there may also be a separate role within the framework of the monitoring protocol for a separate classification procedure which characterises estuaries in terms of the community or species which are likely to respond to enrichment. This type of scheme could be used to assign a level importance to the EcoQOs which are considered during Stages 1 and

2 of the protocol by indicating which biological factors require the greatest attention in terms of the quality and/or conservation monitoring and management.

A preliminary a scheme for classifying estuaries in this manner is shown in Figure 32. This scheme involves a set of three sequential divisions of estuaries. Firstly the estuaries are separated on the basis of their flushing time in relation to plankton generation time, then in terms of the light availability and finally, according to bathymetric characteristics. These three factors influence the relative responses of benthic and pelagic vegetation. For example, in estuaries where the flushing is faster than the generation time of phytoplankton then it is unlikely that planktonic blooms will develop. Similarly, macroalgal mats are less likely to occur in deep slow flowing estuaries that have a limited area of intertidal or shallow subtidal regions with high light-penetration.

This approach differs from the classification procedures discussed in Section 5.1 which are designed to group estuaries in terms of the overall risk of change to all communities from enrichment. However the development of both schemes will require an input of the same data and the classification of 'response characteristics' could be developed and tested together with the general estuary classification procedures with the need for only minimal extra analyses or resources.

5.3 Further Studies

It is evident from the review of literature that despite the extensive nature of this review on nutrient impacts in estuaries, many aspects of the processes of nutrient enrichment and their effects on estuarine species and communities remain unclear. The complexity of the estuarine physical environment and the consequently robust and complex nature of the estuarine communities present major obstacles to understanding the causes and effects' relationships between nutrients and biological response and therefore, to the effective management of enrichment in estuarine systems. The present understanding of the subject is also clearly hampered by factors such as the lack of compatibility between studies carried out in different estuary systems, the lack of system-wide investigations and the absence of any standard approach to the monitoring of nutrients in estuaries throughout the UK.

The wealth of information that is available however, does provide a good conceptual indication of the potential problems arising from enrichment and identifies several key biological responses that can be expected to take place where suitable physical conditions prevail. Most notable among these is the classical concept of successional change in the balance of autotrophic species from k-dominant to r-dominant species, one of the few well-defined potential responses. However, none of the topics reviewed here has complete scientific understanding and thus would benefit greatly from further study; furthermore, while conceptual understanding of many features of enrichment is well developed, any quantitative extension of this is poor.

In terms of the future studies therefore, the Environment Agency and English Nature need to consider where their efforts and resources can most effectively be targeted. There are three separate avenues which could be pursued:

- the application of the monitoring programme detailed in this report in order to assess the scale of the problem in estuaries;
- the funding of a data collation programme designed to bring together the information required to test and develop the estuary classification procedure outlined in this report;
- the funding of specific projects in fields of research where there is a particular paucity of information.

It is recommended that the first two of these approaches are pursued. With regard to the third approach, while there are several useful research topics which could be pursued, and a review of these is contained the next section. There is a clear need for a greater understanding of the scale of the problem in UK estuaries before commitments are made to the funding of specific research topics. For this reason the recommended direction for the project is towards an evaluation of the estuary classification procedures (Section 5.1) and monitoring protocols (Section 5.2) produced by this study. These two goals are not mutually exclusive because a valid classification system is required within the monitoring protocol therefore these two studies should be carried out together.

It is of note at this stage, that the classification procedure will have a value beyond just the investigation of nutrient impacts. As it describes the capacity of estuaries to retain nutrients it may also indicate the potential capacity for the retention of other 'pollutants' such as heavy metals or phthalates.

It will first be necessary to ensure that these two projects are workable before resources are committed to their implementation on national scale (which should be the ultimate aim of this project). It is recommended that a preliminary testing approach is taken whereby the classification procedure, assessment of biological symptoms and the subsequent designation of protocols for monitoring and management should all be tested on a small subset of estuaries.

The procedure for this preliminary investigation (see Figure 33) involves the following three stages:

1. A preliminary test of the classification scheme using data both to test the efficacy of the scheme and ultimately to identify the absolutes or extremes of each factor within the scheme (e.g. the highest and lowest tidal ranges). As physical data are readily available for almost all UK estuaries, it should be relatively straightforward to collate the minimum data required to make a preliminary selection of 20-30 estuaries. The classification scheme could be applied on a subset of estuaries which should cover a range of areas around England and Wales and particularly all Environment Agency regions. Consideration should also be given to the next stage of the programme, the review of biological responses (see below), and estuaries with a range of symptoms should also be included (where these are known). There are sufficient data available on the physical environment of estuaries which can be used and much work has already been done by the JNCC in bringing together this information. This database would be a valuable starting point for the development and testing of the classification procedures and the refinement and further development of EcoQOs.
2. A review of biological responses in these selected estuaries should be undertaken. Data should be collected from each Environment Agency region and other agencies according to the protocol laid down in Section 5.2. This stage should test all aspects of the monitoring protocol including the 'Estuary Screening' Process, the conclusions about Symptom Severity and the assessment of reasons for, or risks of problem conditions occurring.
3. Finally, a decision should be reached about whether clear monitoring and management decisions can be made in the light of the information collected during the first two stages.

In addition to the application of estuary classification and monitoring procedures there are numerous studies that can be undertaken to answer specific questions about ecological impact of nutrients in estuaries. No aspect of this subject can be considered as fully understood and unlikely to benefit from increased scientific investigation but a few specific studies are recommended here which would be of particular interest to the Environment Agency or English Nature.

1) An investigation of responses by Saltmarshes to changing nutrient levels.

The paucity of information on the responses of saltmarshes to increased nutrients must be a major concern for English Nature. A separate review and study of this subject is recommended.

2) *A review of the status of low-nutrient estuaries in England and Wales.*

An assessment of the value of low-nutrient estuaries and an assessment of their potential responses to nutrient enrichment. Firstly the number of such estuaries should be calculated, the habitats reviewed through field studies and the potential response to nutrients estimated using the classification approach. This work needs to be undertaken to demonstrate the value of these sites but care should be taken not to over emphasise the importance of these estuaries in the absence of any detailed information.

3) *A review of the relationship between physical and chemical parameters in U.K. estuaries.*

To complement the work being carried out on the estuary classification scheme it will be necessary to carry out a geomorphological review to describe the relationship between influential physical and chemical factors in estuaries. For example it is necessary to clarify the relationship between freshwater flow, tidal movement and the flushing time of estuaries in order to better predict the response of an estuary to changing freshwater flows and accompanying changes in the nutrient input rate.

4) *An assessment of the relationship between plant tissue nutrient and the ambient water column nutrient concentrations*

This study would be designed to show whether plant tissue nutrient could be used as a viable indicator of the ambient nutrient levels within an estuary. It is possible that plant tissue nutrients will reflect the nutrient concentrations over the long term and therefore provide a much better measure of the ambient nutrient concentrations than individual water column samples which will be highly influenced by the variability of the conditions within estuaries. Such a study would involve the measurement of nutrient concentrations in the tissues of several plant types from a range of different estuary systems together with concomitant long-term water column nutrient monitoring.

5) *A study of the impact of macroalgal mats on fish and bird populations.*

Some evidence is available to suggest that the feeding of some bird species can be impaired by the excessive growth of macroalgal mats in the intertidal regions of estuaries. This effect needs to be studied in greater depth to obtain a quantifying measure. An effect on fish feeding is likely but has not been proven probably because of the obvious difficulties associated with the observation of fish feeding activities. As birds and fish are key components in the ecology of estuaries, a study is required into the effects of macroalgal growth and the change in prey-palatability following organic enrichment on their feeding activities. Such a study may require fish feeding experiments in laboratory flumes.

6) *Analysis of the effects of nutrients on the formation of water quality barriers to fish migration.*

Oxygen depletion in the upper region of estuaries is a characteristic phenomena and the effects of oxygen depletion on fish and benthic species is well documented, what remains unclear however is the effects that nutrient loading has on the creation of water quality barriers to fish migration. This requires a specific examination of the relationship between

nutrient and organic matter loading, the morphological characteristics of estuaries and the stability and duration of DO sags in estuaries.

7) *Identification of a viable approach to a national assessment of background nutrient in estuaries.*

A measure of the background nutrient concentrations in estuaries would be a useful component for inclusion within the estuary classification procedures because it would provide a context for assessing contemporary nutrient loadings to estuaries and would assist both Environment Agency and English Nature in assessing areas of potential risks from nutrient enrichment. It is considered that the nutrient export approach represents the most effective way of describing background nutrient inputs to estuaries although its usefulness on a nation-wide basis is constrained by the cost of this procedure. It is recommended that a study should be undertaken to assess the efficacy of applying a simpler and more economical approach. For instance could a few indicators such as catchment area, pre-World War II land use and changing agricultural practices be used to describe historical loads to estuaries.

8) *Broad-scale studies on a range of estuary types.*

Once the classification of estuaries procedure has been completed it will be possible to identify a range of different estuary types in respect of their potential response to nutrient enrichment. The instigation of ecosystem-level investigations of nutrient responses will greatly enhance understanding of the cause and effect relationships of enrichment in estuaries.

Among the research topics listed above it is suggested that the first two warrant particular study in the next phase of the Impact of Nutrients in Estuaries project. The investigation of low nutrient estuaries will compliment the work done on the estuary classification project and considering the value of saltmarsh systems a study, or at least a specific literature review, on the impacts of nutrients to these habitats would be useful.

5.4 Summary and Conclusions

Proposed Estuary Classification Schemes

It is considered that the ideal approach for estuary classification is the multivariate statistical approach described Section 5.1.2 because this is the most objective procedure. The ultimate goal for Environment Agency and English Nature should be to collate sufficient data to apply this approach. However, this technique will require an extensive collation of data, the development of highly accurate normalised scales for the eight classification criteria and very detailed understanding of the relationship between criteria. For these reasons this approach is impractical in the short term.

The most realistic approach, and the one that is recommended here, is the 'tabulated representation' procedure described in Section 5.1.3. It is believed that sufficient data are presently available to be to apply this technique although an extensive data collation exercise is still required. The 'decision tree' approach is simply a variant on this approach and could be tested when sufficient data are obtained.

The three-factor approaches may be used as a valuable first stage towards the development of the more comprehensive eight-factor techniques. Data are available for this technique and could be collated relatively quickly.

The qualitative 'best expert judgement' technique has the advantage that many details can be obtained through this approach that may be missed by applying a purely quantitative approach. Therefore, some form of qualitative analysis should be used in conjunction with the objective techniques described above. It is likely that other management schemes adopted by the Environment Agency and English nature, for example for the management of SAC, will require elements of this approach.

Proposed Monitoring and Management protocol

For the following reasons the protocol proposed in this section has been designed to concentrate on the symptoms of eutrophication and responses to nutrient inputs rather than the causes:

- the causes require an extensive database and analytical commitment in order to provide meaningful spatial and temporal comparisons;
- the symptoms of eutrophication and or organic enrichment are easily observed and recorded by experienced field biologists;
- the symptoms have greater value in public perception of problems and thus potential solutions are rather more easily communicated to policy-makers;
- the symptoms have greater relevance in the overall considerations of environmental quality assessments (by the Environment Agency) and conservation designation and integrity (by English Nature).

However, the concentration on the effect rather than the cause can be seen to deviate from the precautionary principle although it does fit within the spirit of the proposed EU Water Resources Framework Directive.

It is suggested that effort in monitoring and management is better directed towards determining which estuaries are the sites of adverse symptoms and then carrying out detailed assessments of nutrient inputs and management of those areas.

Further Studies

To summarise, the following recommendations are made for future projects:

1 - Classification testing

- Data collation from JNCC, Environment Agency regional offices and private consultancies. Written requests, site visits required, unit conversions.
- Test classification procedures.

2 - Monitoring protocol testing

- Follow Figure 33.
- Incorporate results of classification tests

3 - Investigation of low-nutrient systems.

- Assess value of habitats within these estuaries. This is likely to require a great deal of local knowledge and unpublished documents rather than published literature.
- Propose the likely response to changing nutrient.
- Predict whether enrichment would have a harmful or beneficial effect.

4 - Review of saltmarsh responses.

- Review of published and unpublished information on the subject of saltmarsh responses to nutrient enrichment.

6. REFERENCES

- Addiscott, T. (1988). Farmers, fertilisers and the nitrate flood. *New Scientist*, 8 October, No. 1633, 50-54.
- AMOEBa (1992) – Ten Brink, BJE, SH Hosper & F Colijn (1991). A quantitative method for description and assessment of ecosystems: the AMOEBa approach. *Mar. Poll. Bull.* 23 265-270.
- Armstrong, I. H., Coulson, J. G., Hawkey, P. & Hudson, M. J. (1978). Further mass seabird deaths from paralytic shellfish poisoning. *British Birds* 71: 51-68.
- Asknes, D. L., Ulvestad, K. B. *et al.*, (1995). Ecological modelling in coastal waters: towards predictive physical-chemical-biological simulation models. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 5-36. October 13-16, 1993, Denmark. Ophelia Publications.
- Asmus, H., Asmus, R. M. & Zubillaga, G. F. (1995). Do mussel beds intensify the phosphorus exchange between sediment and tidal waters? In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 37-56. October 13-16, 1993, Denmark. Ophelia Publications.
- Baden, S. P., Loo, L-O., Pihl, L. & Rosenberg, R. (1990). Effects of eutrophication on benthic communities including fish: Swedish West Coast. *Ambio*, 19, No. 3, 113-122.
- Baird, D., Evans, P. R., Milne, H. & Pienkowski, M. W. (1985). Utilization by shorebirds of benthic invertebrate production in intertidal areas. *Oceanogr. Mar. Ann. Rev.*, 23, 573-597
- Balls, P. W.; Brockie, N., Dobson, J. & Johnston, W. (1996). Dissolved oxygen and nitrification in the Upper Forth Estuary during summer (1982-92): Patterns and trends. *Estuarine, Coastal & Shelf Science*, 42, 117-134.
- Balls P. W., McDonald, A., Pugh, K. & Edwards, A. C. (1995) Long-term nutrient enrichment of an estuarine system: Ythan Scotland (1958-1993). *Environmental Pollution*, Vol. 90, No. 3, pp. 311-321.
- Balls, P. W. (1994). Nutrient inputs to estuaries from nine Scottish east coast rivers; Influence of estuarine processes on inputs to the North Sea. *Estuarine, Coastal and Shelf Science*, 39, pp. 329-352.
- Balls, P. W. (1992). Nutrient behaviour in two contrasting Scottish estuaries, the Forth and Tay. *Oceanologica Acta*, 15, No. 3, pp. 261-277.
- Baretta, J. W. Ruardij, P. (1988). Tidal flat estuaries: simulation and analysis of the Ems estuary, *Ecological Studies* 71. Springer-Verlag, Heidelberg.
- Barnes, R. S. K. & Mann, K. H. (1991). *Fundamentals of Aquatic Ecology*. Blackwell Scientific Publications.
- Bates, S. S., de Freitas, A. S. W., Milley, J. E., Pocklington, R., Quillam, M. A., Smith, J. C. & Worms, J. (1991). Controls on domoic acid production by the diatom

Nitzschia pungens f. multiseriis in culture: nutrients and irradiance. Can. J. Fish. Aquat. Sci. 28:1136-1144.

Beukema, J. J. & Cadée, G. C. (1986). Zoobenthos responses to eutrophication of the Dutch Wadden Sea. *Ophelia* 26:55-64.

Billen, G., Somville, M., De Becker, E. & Servais, P. (1985). A Nitrogen Budget of the Scheldt Hydrographical Basin. *Netherlands Journal of Sea Research*, 19 (3/4), 223-230.

Boynton, W. R., Garber, J. H., Summers, R. & Kemp, W. M. (1995). Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries*, Vol. 18, No. 18, p. 285-314.

Boynton, W. R., Kemp, M., Barnes, J., Stammerjohn, S., Matteson, L. & Rohland, F. (1991). Long-term characteristics and trends of benthic nutrient fluxes in the Maryland portion of Chesapeake Bay. In: *New Perspectives in the Chesapeake System: A research and management Partnership* (Mihursky, J. & Chaney, A. eds.). *Proceedings of the 2nd Chesapeake Bay Research Conference*, Baltimore, MD, 4-6 December 1990, pp. 339-354.

Boynton, W. R. & Kemp, W. M. (1985). Nutrient regeneration and oxygen consumption by sediments along an estuarine salinity gradient. *Mar. Ecol. Prog. Ser.* Vol. 23: 45-55.

Boynton, W. R., Kemp, W. M. & Keefe, C. W. (1982) A comparative analysis of nutrients and other factors influencing estuarine phytoplankton production, pp. 69-90. in V. S. Kennedy (ed.). *Estuarine Comparisons*. Academic Press, New York.

Brinkman, A. G. (1993). Biological processes in the EcoWasp ecosystem model IBN Research Report 93/6, Wageningen.

Buck, K. R., Uttal-Cooke, L., Pilska, C. H., Roelke, D. L., Villac, M. C., Fryxell, G. A., Cifuentes, L., Chavez, F. P. (1992). Autoecology of the diatom *Pseudonitzschia australis*. A domoic acid producer, from Monterey Bay, California. *Mar. Ecol. Prog. Ser.* 84: 293-302.

Caffrey, J. M. & Kemp, W. M. (1990). Nitrogen cycling in sediments with estuarine populations of *Potamogeton perfoliatus* and *Zostera marina*. *Marine Ecology Progress Series*, 66, 147-160.

Caperon, J. S., Cattell, S. A. & Krasnick G. (1971). Phytoplankton kinetics in a subtropical estuary: eutrophication. *Limnol. Oceanogr.* 16: 599-607.

Cartwright, N. G., Painter, H. & Parr, W. (WRc) (1993). An assessment of the environmental quality standards for inorganic nutrients necessary to prevent eutrophication (nuisance growth of algae). Report to National Rivers Authority. R & D Note 230.

Cartwright, N. G. & Painter, H. (1991). An assessment of the environmental quality standards for inorganic nutrients necessary to prevent eutrophication (nuisance growth of algae). WRc Report No. NR 2397 for NRA (Ref A19.53/7.2.1.a).

Cederwall, H. & Elmgren, R. (1980). Biomass increase of benthic macrofauna demonstrates eutrophication of the Baltic Sea. *Ophelia* Suppl. 1:287-296.

Chapelle, A.; Lazure, P. & Menesguen, A. (1994). Modelling eutrophication events in a coastal ecosystem. Sensitivity Analysis. *Estuarine, Coastal & Shelf Science*, 39, No. 6, 529-548.

Christian, R. R., Boyer, J. N. & Stanley, D. W. (1991). Multi-year distribution patterns of nutrients within the Neuse River Estuary, North Carolina. *Marine Ecology Progress Series*, 71, 259-274.

Cloern, J. E. (1991). Annual variations in river flow and primary production in the South San Francisco Bay Estuary (USA). In: Elliott, M & Ducrotoy, J-P (eds.). *Estuaries and Coasts: Spatial and Temporal Intercomparisons*, pp. 91-96. ECSA 19 Symposium. Olsen & Olsen.

Codling, I. D., Warwick, R. M., Clarke, K. R. & Ashley, S. J. (1995). Development and Testing of General Quality Assessment Schemes: Assessment of the feasibility of using macrobenthic community status to describe the General Quality of estuaries and coastal waters. Report by WRc to NRA. Report Number: 469/19/HO

Common Wadden Sea Secretariat (1994). Wilhelmshaven, FRG. Eutrophication and eutrophication phenomena workshop. Wilhelmshaven, 3-4 May 1994.

Conley, D. J. & Johnstone, R. W. (1995). Biogeochemistry of N, P and Si in Baltic Sea sediments: response to a simulated deposition of a spring diatom bloom. *Mar. Ecol. Prog. Ser.* Vol. 122: 265-276.

Conley, D. J., Smith, W. M., Cornwell, J. C. & Fisher, T. R. (1995). Transformation of particle-bound phosphorus at the land-sea interface. *Estuarine and Coastal Shelf Science*, 40, 161-176.

Conley, D. J., Schelske, C. L. & Stoermer, E. F. (1993). Modification of the biogeochemical cycle of silica with eutrophication. *Mar. Ecol. Prog. Ser.* Vol. 101: 179-192.

Conley, D. J. & Malone, T. C. (1992). Annual cycle of dissolved silicate in Chesapeake Bay: implications for the production and fate of phytoplankton biomass. *Mar. Ecol. Prog. Ser.* Vol. 81: 121-128.

Conway, L. (1993). A review of macroalgal blooming in estuarine and coastal waters, with particular reference to the Ythan estuary in Aberdeenshire. Institute of Offshore Engineering, Heriot-Watt University. pp.61.

Cooper, S. R. & Brush, G. S. (1993). A 2,500-year history of anoxia and eutrophication in Chesapeake Bay. Publication No. UM-SG-RS-93-20 In: *Estuaries*, Vol. 16, No. 38, p. 617-626.

Cooper, A. B., Smith, C. M. & Bottcher, A. B. (1992). Predicting runoff of water, sediment, and nutrients from a New Zealand grazed pasture using CREAMS. *American Society of Agricultural Engineers*, 35(1), 105-112.

Correll, D. L., Jordan, T. E. & Weller, D. E. (1992). Nutrient flux in a landscape: Effects of coastal land use and terrestrial community mosaic on nutrient transport to coastal waters. *Estuaries* 15:431-442.

Costanza, R., Kemp, W. M. & Boynton W.R. (1993). Predictability, scale and biodiversity in Coastal and Estuarine Ecosystems: Implications for Management. *Ambio* Vol. 22, No. 2-3, pp. 88-96.

CSTT (1993) Comprehensive studies for the purposes of article 6 of the Urban Waste Water Treatment Directive. MPMMG Comprehensive Studies Task Team.

Cullen, S. (1992). Blooming of nuisance macroalgae in estuarine and coastal waters - a review. Institute of Offshore Engineering, Heriot-Watt University. pp.62.

Daiber, F.C. (1986) Conservation of Tidal Marshes. Van Norstrand Reinhold, Melbourne.

Dame, R. F. & Lefeuvre, J. C. (1994). Tidal exchange: import-export of nutrients and organic matter in New and Old World Salt marshes: conclusions. Global Wetlands: Old World and New, Mitsch, W. J. (ed.), pp. 303-305.

Dame, R. F. (1994). The net flux of materials between marsh-estuarine systems and the sea: the Atlantic coast of the United States. Global Wetlands: Old World and New, Mitsch, W. J. (ed.), pp. 295-302. Elsevier Science

Dame, R. F. (1993). The role of bivalve filter feeder material fluxes in estuarine ecosystems. NATO ASI Series, Vol. G 33. 'Bivalve Filter Feeders', in Estuarine and Coastal Ecosystem Processes, edited by Richard F. Dame. © Springer-Verlag Berlin Heidelberg.

Dame, R. F., Wolaver, T. G., Williams, T. M., Spurrier, J. D. & Miller, A. B. (1990). The Bly Creek ecosystem study: Phosphorus transport within a euhaline salt marsh basin, North inlet, South Carolina. Netherlands Journal of Sea Research, 27(1), 73-80.

Dame, R. F. & Dankers, N. (1988). Uptake and release of materials by a Wadden Sea mussel bed. J. Exp. Mar. Biol. Ecol., 118, 207-216.

Dame, R. F., Spurrier, J. D. & Wolaver, T. G. (1989). Carbon, nitrogen and phosphorus processing by an oyster reef. Marine Ecology Progress Series, 59, 249-256.

Dame, R., Chrzanowski, T., Bildstein, K., Kjerfve, B., McKellar, H., Nelson, D., Spurrier, J., Stanczyk, S., Stevenson, H., Vernberg, J. & Zingmark, R. (1986). The Outwelling Hypothesis and North Inlet, South Carolina. Marine Ecology Progress Series, 33, 217-229.

Dauer, D. M., Luckenbach, M. W. & Rodi, A. J. (1993). Abundance Biomass Comparison (ABC Method) - Effects of an estuarine gradient, anoxic hypoxic events and contaminated sediments. Mar. Bio. Vol. 116, No. 3, pp.507-518.

Davey, J. T. & Watson, P. G. (1995). The activity of *Nereis diversicolor* (Polychaeta) and its impact on nutrient fluxes in estuarine waters. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 57-70. October 13-16, 1993, Denmark. Ophelia Publications.

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

Davis, J. L. (1973). Geographical variation in coastal development. Longman, London.

- De-Jonge, V.N.; Bakker, J.F. & Van-Stralen, M. (1996) Recent changes in the contributions of River Rhine and North Sea to the eutrophication of the western Dutch Wadden Sea. *Neth. J. Aquat. Ecol.*, Vol 30, No. 1, pp.27-39.
- Dentener, F. J. & Crutzen, P. J. (1994). A three-dimensional model of the global ammonia cycle. *J. Atmos. Chem.* 19: 331-369.
- Dethlefsen, (1990) 10 Years fish disease studies of the Institut-fur-Kusten-und-Binnenfischerei. *Archiv. Fur. Fischerewissenschaft* 40 (1-2), pp. 119-132.
- Diaz, R. J. & Rosenberg, R. (1995). Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: an Annual Review*, 33, 245-303.
- DoE (1993). Methodology for identifying Sensitive Areas (UWWTD) and methodology for designating vulnerable zones (Nitrates Directive) in England and Wales. DoE.
- Doering, P. H., Oviatt, C. A., Beatty, L. L., Banzon, V. F., Rice, R., Kelley, S. P., Sullivan, B. K. & Frithsen, J. B. (1989). Structure and function in a model coastal ecosystem: silicon, the benthos and eutrophication. *Marine Ecology Progress Series*, Vol. 52, 287-299.
- Duarte, C. M. (1995). Submerged aquatic vegetation in relation to different nutrient regimes. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 87-112. October 13-16, 1993; Denmark. Ophelia Publications.
- Elkaim, B., Desprez, M., Ducrotoy, J-P., Dupont, J-P., Lafite, R., Pichard, S., Rybarczyk, H., Syland, B. & Wilson, J. (1994). L'évaluation et le suivi de la qualité biologique des estuaires. *Bull. D'écologie*, 23(4): 185-200.
- Elliott, M. (1996). The derivation and value of ecological quality standards and objectives. *Mar. Poll. Bull.* 32 762-763.
- Elliott, M.; de Jonge, V. N.; Burrell, K. L., Johnson, M. W., Phillips, G. L. & Turner, T. M. (1994). Trophic status of the Ore/Alde, Deben, Stour and Colne estuaries. Volume 1. Reports in Applied Marine Biology. University of Hull
- Elliott, M. & O'Reilly, M. G. (1991). The variability and prediction of marine benthic community parameters. In: *Estuaries and Coasts: Spatial and temporal intercomparisons*. Eds. M. Elliott and J-P. Ducrotoy. Olsen and Olsen.
- Elliott and Taylor (1989) The production ecology of the subtidal benthos of the Forth Estuary, Scotland. *Scient. Mar.* 53 (2-3), pp. 531-541.
- Elmgren, R. (1989). Man's impact on the ecosystem of the Baltic Sea: Energy flows today and at the turn of the century. *Ambio.*, 18, No. 6, 326-332.
- ENTEC (1995). National strategy for eutrophication control: Targets for eutrophication control. Draft Report. Report to NRA - Anglian Region. Report No. 28066/3il.
- Eyre, B. (1995). A first-order nutrient budget for the tropical Moresby estuary and catchment, North Queensland, Australia. *Journal of Coastal Research*, 11 (3), 717-732.

- Fairbridge, R. W. (1980). The estuary: its definition and geodynamic role. In: Chemistry and biogeochemistry of estuaries, ed. by E. Olausson and I. Cato, 1-35. New York, Wiley.
- Fanning, K. A. (1989). Influence of atmospheric pollution on nutrient limitation in the ocean *Nature*. 339: 460-463.
- Fichez, R., Jickells, T. D. & Edmunds, H. M. (1992). Algal blooms in high turbidity, a result of the conflicting consequences of turbulence on nutrient cycling in a shallow water estuary. *Estuarine, Coastal and Shelf Science*, 35, 577-592.
- Fisher, T. R., Harding, L. W. r., Stanley, D. W. & Ward, L. G. (1988). Phytoplankton, nutrients and turbidity in the Chesapeake, Delaware, and Hudson estuaries. *Estuarine, Coastal and Shelf Science*, 27, 61-93.
- Fong, P., Donohoe, R. M & Zedler, J. B. (1993). Competition with macroalgae and benthic cyanobacterial mats limits phytoplankton abundance in experimental microcosms. *Mar Ecol Prog Ser*. Vol. 100, No. 1-2, pp 97-102.
- Forsgren, G. & Jansson, M. (1992). The turnover of river-transported iron, phosphorus and organic carbon in the Öre estuary, northern Sweden. *Hydrobiologia*, 235/236, 585-596.
- Forth River Purification Board (1995). Semi-continuous monitoring of phytoplankton blooms at Gunnet Ledge in the Firth of Forth, May to November, 1994. Report No. TW2/95.
- Franklin, I., Ormaza-Gonzalez, F. I. & Statham, P. J. (1991). The occurrence and behaviour of different forms of phosphorus in the waters of four English estuaries. In: Elliott, M & Ducrotoy, J-P (eds.). *Estuaries and Coasts: Spatial and Temporal Intercomparisons*, pp. 77-83. ECSA 19 Symposium. Olsen & Olsen.
- Frederiksen, O. T. (1987). The fight against eutrophication in the Inlet of 'Odense Fjord' by reaping of sea lettuce (*Ulva lactuca*). In: *The use of Macrophytes in Water Pollution Control*. Athie F. & Cerri, C. C., (eds.) Pergamon Press.
- Froelich, P. N. (1988). Kinetic control of dissolved phosphate in natural rivers and estuaries: a primer on the phosphate buffer mechanism. *Limnol. Oceanogr.*, 33, 649-668.
- Garcia-Soto C., de Madariaga I. , Villate F. & Orive E. (1990). Day-to-day variability in the plankton community of a coastal shallow embayment in response to changes in river run off and water turbulence. *Estuar. Coast. Shelf Sci.*, 31, 217-229.
- Gerritse, R. (in prep). Records of pollutants in sediment cores from the Peel-Harvey estuarine system in Western Australia, I: Phosphorus. 1st Draft and Records of pollutants in sediment cores from the Swan-Canning estuarine system in Western Australia, I: Phosphorus. 1st Draft. CSIRO Division of Water Resources, Australia.
- Gilbert, P. M., Conley, D. J., Fisher, T. R., Harding Jr, L. W. & Malone, T. C. (1995). Dynamics of the 1990 winter/spring bloom in Chesapeake Bay. *Marine Ecology Progress Series*, 122, 27-43.
- Goss-Custard, J. D., Warwick, R. M., Kirby, R., McGrorty, S., Clarke, R. T., Pearson, B., Rispin, W. E., Durrell, S. E. A. L. V. D. & Rose, R. J. (1991). Towards predicting wading bird densities from predicted prey densities in a post-barrage Severn estuary, *J. App. Ecol.* 28: 1004-1026.

- Grasshoff, K. (1983). *Methods of Seawater Analysis*.
- Gray, J. S., McIntyre, A. D. & Stirn, J. (1992). *Manual methods in aquatic environment research. Part II: Biological assessment of marine pollution with particular reference to benthos*. FAO Fish. Tech. Pap. No. 324. Rome: FAO, 49pp.
- Green, P. T., Hill, D. A. & Clark (1992). *The effects of organic inputs to estuaries on overwintering bird populations and communities*. BTO Research Report No.59. ETSU TID 4086.
- Grobbelaar, J. U. (1985). *Phytoplankton productivity in turbid waters*. J. Plankton Res. 7: 653-663.
- Gunby, A., Nixon, S. C. & Wheeler, M. A. (1995). *Development and Testing of General Quality Assessment Schemes: Nutrients in Estuaries and Coastal Waters*. WRc Report 469/14/HO for the National Rivers Authority.
- Harding, L. W. (1994). *Long-term trends in the distribution of phytoplankton in Chesapeake Bay: roles of light, nutrients and streamflow*. Publication No. UM-SG-RS-94-01. In: *Mar. Ecol. Prog. Ser.*, Vol. 104: 267-291.
- Harding, L. E. (1992). *Measures of Marine Environmental Quality*. Marine Pollution Bulletin, Volume 25, 1-4, pp. 23-27.
- Harding, L. W., Leffler, M. & Mackiernan, G. E. (1992). *Dissolved oxygen in the Chesapeake Bay: A Scientific Consensus*. Prepared by the Sea Grant College Programs of Maryland and Virginia.
- Heip, C. (1995). *Eutrophication and zoobenthos dynamics*. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. *Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments* pp. 113-136. October 13-16, 1993, Denmark. Ophelia Publications.
- Helder, W. P. and Ruurdij, P. (1982). *A one-dimensional mixing and flushing model of the Ems-Dollard estuary: calculations of time scales at different river discharges*. *Neth. J. Sea. Res.*, 15: 293-312
- Hendey, N. I. (1976). *The species diversity index of some inshore diatom communities and its use in assessing the degree of pollution insult on parts of the north coast of Cornwall*. *Nova Hedwigia* 54:355-378.
- Hickel W., Mangelsdorf P. and Berg J. (1993) *The human impact in the German Bight: Eutrophication during three decades (1962-1991)*: *Helgoländer Meeresunters* 47, 243-263.
- Hillman, K, Lord, D. A., McComb, A. J. & Lavery, P. (1995). *Perth Coastal Waters Study: Environmental Values*. Water Authority of Western Australia, ISBN 0 7309 7228 3.
- Hobson, L. A. (1985). *Seasonal variations and spatial gradients of nitrogenous nutrients in waters of South-eastern Vancouver Island, British Columbia, Canada*. *Netherlands Journal of Sea Research*, 19 (3/4), 251-256.
- Holme, N. A. & McIntyre, A. D. (1984). *Methods for the study of marine benthos*. IBP Handbook No. 16 Oxford: Blackwell Scientific Publications, 387pp.

- Höpner, T. & Wonneberger (1985). Examination of the connection between the patchiness of benthic nutrient efflux and epiphytobenthos patchiness on intertidal flats. *Netherlands Journal of Sea Research*, 19 (3/4), 277-285.
- Hull, S. C. (1987). *Macroalgal mats and species abundance: a field experiment*. Academic Press Ltd.
- ICES (1995). Report of the Study Group on Ecosystem Effects of Fishing Activities. ICES Coop. Res. Rep. No. 200, 120pp. Abstracts taken from Chapter 5. (seen in TMAP Rpt.)
- Isles, K. (1997) An analysis of the water quality in the Mersey estuary between 1975 and 1997. Unpub MSc thesis, (University of Hull).
- Iverson, R. L. (1990). Control of marine fish production. *Limnol. Oceanogr.* 35: 1593-1604.
- Jensen, L. M., Sand-Jensen, K., Marcher, S. & Hansen, M. (1990). Plankton community respiration along a nutrient gradient in a shallow Danish estuary. *Marine Ecology Progress Series*, 61, 75-85.
- Johnes, P. J. & Heathwaite, A. L. (1996). Modelling the impact of land use change on water quality in agricultural catchments. To be published in *Hydrological Processes*, November 1996.
- Johnes, P. J. (1994). Modelling nutrient loading on the Ore/Alde and Deben Estuaries, 1930-1990. NRA Research Contract.
- Johnson, M. P., Costello, M. J. & O'Donnell, D. (1995). The nutrient economy of a marine inlet: Lough Hyne, South-west Ireland. *Ophelia*, 41, 137-151.
- Jones, K. J. & Gowen, R. J. (1985). The influence of advective exchange on phytoplankton in Scottish fjordic estuaries. In: Anderson, D. M., White, A. W. & Baden, D. G. (eds), *Toxic Phytoplankton*, 207-2121. Elsevier.
- Jones, K. J., Grantham, B., Ezzi, I., Rippeth, T. & Simpson, J. (1995). Physical controls on phytoplankton and nutrient cycles in the Clyde Sea, a fjordic system on the west coast of Scotland. *Ecology of Fjords and Coastal Waters*, 93-104.
- Jonge, V. N. de, Boynton, W., D'Elia, C. F., Elmgren R. & Welsh, B. L. (1995) Responses to developments in eutrophication in four different North Atlantic estuarine systems. In: Dyer, K. R & Orth, R. J. (eds.). 'Changes in Fluxes in Estuaries'. ECSA 22/ERF Symposium. Olsen & Olsen.
- Jonge, V. N. de & Bakker, J. F. (1994). Responses to changes in nutrient loads and physical characteristics in the Wadden Sea. In *Eutrophication and Eutrophication Phenomena Workshop 3-4 May 1994*. Common Wadden Sea Secretariat. Wilhelmshaven, FRG.
- Jonge, V. N. de & Van Beusekom J.E.E. (1992). Contribution of resuspended microphytobenthos to total phytoplankton in the Ems Estuary and its possible role for grazers. *Neth. J. Sea. Res.*, 30, pp. 91-105.
- Jonge, V. N. de & Essink, K. (1991). Long-term changes in nutrient loads and primary and secondary production in the Dutch Wadden Sea - In: *Estuaries and Coasts: Spatial and Temporal Intercomparisons*, pp. 307-316. ECSA 19 Symposium. Eds. Elliott, M. & Ducrotoy, J. P. Olsen & Olsen.

- Jonge, V. N. de, & Villerius, L. A. (1989). Possible role of carbonate dissolution in estuarine phosphate dynamics. *Limnol. Oceanogr.*, 36, pp. 251-267.
- Josefson, A. B. (1990). Increase of benthic biomass in the Skagerrak-Kattegat during the 1970s and 1980s - effects of organic enrichment? *Marine Ecology Progress Series*, 66, 117-130.
- Keizer, P. D. & Gordon, D. C. (1985). Nutrient dynamics in Cumberland Basin - Chignecto Bay, a turbid macrotidal estuary in the Bay of Fundy, Canada. *Netherlands Journal of Sea Research*, 19 (3/4), 193-205.
- Kemp, W. M. & Boynton, W. R. (1992). Benthic-pelagic interactions: Nutrient and oxygen dynamics. In: Smith, D (ed.). *Oxygen dynamics in the Chesapeake Bay: A synthesis of recent research.* pp. 150-221.
- Kemp, W. M., Sampou, P. A., Garber, J., Tuttle, J. & Boynton, W. R. (1992). Seasonal depletion of oxygen from bottom waters of Chesapeake Bay: roles of benthic planktonic respiration and physical exchange processes. *Mar. Ecol. Prog. Ser.* Vol. 85: 137-152.
- Kemp, W. M., Sampou, P., Caffrey, J. & Mayer, M. (1990). Ammonium recycling versus denitrification in Chesapeake Bay sediments. *Limnol. Oceanogr.*, 35(7), pp. 1545-1563.
- Kemp, W. M., Boynton, W. R. & Twilley, R. R. (1984). Influences of submerged vascular plants on ecological processes in upper Chesapeake Bay, pp. 367-384. In : V. S. Kennedy (ed.) 'Estuaries as a Filter'. Academic Press, NY. 511p.
- Kemp, W. M., Twilley, R. R., Stevenson, J. C., Boynton, W. R. & Means J. C. (1983). The decline of submerged vascular plants in Chesapeake Bay: summary of results concerning possible causes. *Mar. Tech. Soc. J.* 17(2): 78-89.
- Kiorbøe 1994 Riisgard, HU (1994). Filter feeding in the polychaete *Nereis diversicolor*: a review. *Neth. J. Aquatic Ecology* 28(3-4) 453-458.
- Koseff, J. R., Holen, J. K., Monismith, S. G. & Cloern, J. E. (1993). Coupled effects of vertical mixing and benthic grazing on phytoplankton populations in shallow, turbid estuaries. *J. Mar Res.* Vol. 51, No. 4, pp.843-868.
- Kramer, K. J. M., Brockmann, U. H. & Warwick, R. M. (1994). Tidal estuaries manual of sampling and analytical procedures. Netherlands. A. A. Balkema, 340pp.
- Kromkamp J., Peene J., Vanrijswijk P., Sandee A. & Goosen N. (1995) Nutrients, light and primary production by phytoplankton and microphytobenthos in the eutrophic turbid Westerschelde estuary (the Netherlands) *Hydrobiologia* 1995 311 (1-3), pp. 9-19.
- Land-Ocean Interaction Study (1994). Implementation plan for a community research Project. NERC ISBN 1 85531 121 6.
- Lebo, M. E., Sharp, J. H. & Cifuentes, L. A. (1994). Contribution of river phosphate variations to apparent reactivity estimated from phosphate-salinity diagrams. *Estuarine, Coastal and Shelf Science*, 39, 583-594.
- Legendre, L. & Rassoulzadegan, F. (1995). Plankton and nutrient dynamics in marine waters. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics

- in Coastal and Estuarine Environments pp. 153-172. October 13-16, 1993, Denmark. Ophelia Publications.
- Levy, H., II & Moxim, W. J. (1989). Simulated global distribution and deposition of reactive nitrogen emitted by fossil fuel combustion. *Tellus*. 41B: 256-271.
- Libes, S. M. (1992). An introduction to marine biogeochemistry. John Wiley and Sons, inc. pp. 734.
- Lin, H. -J., Nixon, S. W., Taylor, D. I., Granger, S. L. & Buckley, B. A. (1996). Responses of epiphytes on eelgrass, *Zostera marina* L., to separate and combined nitrogen and phosphorus enrichment. *Aquatic Botany*, 52, 243-258.
- Lipschultz, F. (1995). Nitrogen-specific uptake rates of marine phytoplankton isolated from natural populations of particles by flow cytometry. *Marine Ecology Progress Series*, 123, 245-258.
- Lomstein, E., Jensen, M. H. & Sorensen, J. (1990). Intracellular NH_4^+ and NO_3^- pools associated with deposited phytoplankton in a marine sediment (Aarhus Bight, Denmark). *Marine Ecology Progress Series*, 61, 97-105.
- Lord, D. A. & Hillman, K. (1995). Water Authority of Western Australia Perth Coastal Waters Study: Summary Report. ISBN 0 7309 7213 5.
- Lowthion, D, Soulsby, P. G. & Houston, M. C. M. (1985). Investigation of a eutrophic tidal basin: Part 1 - factors affecting the distribution and biomass of macroalgae. *Marine Environmental Research*, 15, 263-284.
- Lucotte & d'Anglejan (1988) Seasonal changes in the phosphorus-iron geochemistry of the St Lawrence estuary. *J. Coast. Res.* 4: 339-349.
- Lunn et al., (1992) cited in Johnes (1994)
- Lyons, M. G.; Balls, P. W. & Turrell, W. R. (1993). A preliminary study of the relative importance of riverine nutrient inputs to the Scottish North Sea Coastal Zone. *Marine Pollution Bulletin*, Volume 26, No. 11, pp. 620-628.
- Macdonald, A. M., Edwards, A. C., Pugh, K. B. & Balls, P. (1995). Soluble nitrogen and phosphorus in the river Ythan system, U.K.: Annual and seasonal trends. *Wat. Res.* Vol. 29, No. 3, pp. 837-846.
- Macdonald, A. M., Edwards, A. C., Pugh, K. B. & Balls, P. W. (1994). The impact of land use on nutrient transport into and through three rivers in the north east of Scotland. In: Kirby, C & White, W. R. (eds.) 'Integrated River Basin Development' pp. 201-214. Wiley Press.
- Maestrini, S. Y. & Graneli, E. (1991). Environmental conditions and ecophysiological mechanisms which led to the 1988 *Chrysochromulina polylepsis* bloom: an hypothesis. *Oceanol. Acta* 14: 397-413.
- MAFF (1993). Analysis and interpretation of benthic community data at sewage-sludge disposal sites. *Aquat. Environ. Monit. Rep.*, MAFF Direct. Fish. Res., Lowestoft, No. 37, 80pp.
- Magnien, R, Boward, D. & Bieber, S. (eds.) (1995). Chesapeake Bay Program: The state of the Chesapeake Bay 1995.

Mainstone, C. P., Ashley, S., Gunby, A., Parr, W., Woodrow, D., Turton, P. & McAllen, Y. (1995): Development and Testing of General Quality Assessment Schemes: Nutrients in rivers and canals. Report by WRc to NRA. Project Record: 469/11/HO.

Malone, T. C., Conley, D. J., Fisher, T.R., Glibert, P. M. & Harding, L. W. (in prep). Scales of nutrient limited phytoplankton productivity in Chesapeake Bay.

Malone, T. C., Boynton, W. R., Horton, T. and Stevenson, C. (1993). Nutrient loadings to surface waters: Chesapeake Bay case study. In: Uman, M. F. (ed.). Keeping Pace with Science and Engineering. National Academy Press, Washington, D. C.

Malone, T. C. (1992). Effects of water column processes on dissolved oxygen, nutrients, phytoplankton and zooplankton. In Smith DA, Leffler M, Mackiernan G (eds.) Oxygen dynamics in the Chesapeake Bay. Maryland Sea Grant College, College Park, p61-112.

Malone, T. C., Crocker, L. H., Pike, S. E. & Windler, B. W. (1988). Influences of river flow on the dynamics of phytoplankton production in a partially stratified estuary. Mar. Ecol. Prog. Ser. 48: 235-249.

Marshall, S. and Elliott, M. (1998). Environmental influences on the fish assemblage of the Humber estuary, UK Est., Coast and Shelf Science 46 175-184.

Mathieson, S. & Atkins, S. M. (in press). A review of nutrient enrichment in the estuaries of Scotland: Implications for the natural heritage. Netherlands Journal of Aquatic Ecology.

McArthur, C. I & Johnson, M. W. (1992). A study of eutrophication in eastern area saline waters. Interim Report - August 1992.

McBride, G. B., Quinn, J.M. & Smith, R. K. (1992). Manawatu River Estuary: A Review of Water Quality Issues. Consultancy Report No. 6038/1. Prepared for Manawatu-Wanganui Regional Council, Palmerston North.

McGlathery, K. J. (1995). Nutrient and grazing influences on a subtropical seagrass community. Marine Ecology Progress Series, 122, 239-252.

McLusky (1989). The Estuarine Ecosystem. Second edition. Blackie.

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 and Freshwater Ecosystems 2, pp. 211-222.

Monbet, Y. (1992). Control of phytoplankton biomass in estuaries: a comparative analysis of microtidal and macrotidal estuaries, Estuaries, 15: 563-571.

Morris, J. T. (1991). Effects of nitrogen loading on wetland ecosystems with particular reference to atmospheric deposition. Annu. Rev. Ecol. Syst. 22:257-279.

Moss, B., Johnes, P. & Phillips, G. (1996). The monitoring of ecological quality and the classification of standing waters in temperate regions: A review and proposal based on a worked scheme for British waters. Biol. Rev., 71, 301-339.

Muller, F. L. L., Tranter, M. & Balls, P. W. (1994). Distribution and transport of chemical constituents in the Clyde estuary. Estuarine, Coastal and Shelf Science, 39, pp. 105-126.

- National Oceanic and Atmospheric Administration (1993). National assessment of nutrient enrichment conditions in estuaries: Program Plan - Spring 1993.
- National Oceanic and Atmospheric Administration (1995). Estuarine eutrophication survey. Volume 1: Mid-Atlantic Region. Draft.
- Nedwell, D. B. & Trimmer, M. (in press). Nitrogen fluxes through the upper estuary of the Great Ouse, England: the role of the bottom sediments.
- Nedwell, D. B. (1995). The impact of nutrients in estuaries - proceedings of a Workshop on 23 May 1995. R & D Project 639. Commissioned by National Rivers Authority/ English Nature.
- Nicholls, D. J., Tubbs, C. R. & Haynes, F. N. (1981). The effect of green algal mats on intertidal macrobenthic communities and their predators. *Kieler Meeresforsch., Sonderh.*, 5, 511-520.
- Nielsen, S. L. & Sand-Jensen, K. (1990). Allometric scaling of maximal photosynthetic growth rate to surface/volume ratio. *Limnol. Oceanogr.* 35:177-181.
- Nilsson, P., Jonsson, B., L., Swanberg, I. L. & Sundback, K. (1991). Response of a marine shallow-water sediment system to an increased load of inorganic nutrients. *Marine Ecology Progress Series*, 71, 275-290.
- Nixon, S. C., Gunby, A., Ashley, S. J., Lewis, S. & Naismith, I. (1995b). Development and Testing of General Quality Assessment Schemes: Dissolved oxygen and ammonia in estuaries. Report by WRC to NRA. Project Record: 469/15/HO.
- Nixon, S. W. (1996). Prehistoric nutrient inputs and productivity in Narragansett Bay. Submitted to Estuaries.
- Nixon, S. W., Granger, S. L. & Nowicki, B. L. (1995a). An assessment of the annual mass balance of carbon, nitrogen, and phosphorus in Narragansett Bay. *Biogeochemistry*, 31, 15-61.
- Nixon, S. W. (1995). Coastal marine eutrophication: a definition, social causes, and future concerns. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 199-220. October 13-16, 1993, Denmark. Ophelia Publications.
- Nixon, S. W. (1992). Quantifying the relationship between nitrogen input and the productivity of marine ecosystems. *Pro. Adv. Mar. Tech. Conf.*, 5, 57-83.
- Nixon, S. W. (1988). Physical energy inputs and the comparative ecology of lake and marine ecosystems. *Limnol. Oceanogr.*, 33 (4, part 2), 1005-1025.
- Nixon, S. W. (1987). Chesapeake Bay nutrient budgets - a reassessment. *Biogeochemistry*, 4, 77-90.
- Nixon, S. W., Oviatt, C. A., Frithsen, J. & Sullivan, B. (1986). Nutrients and the productivity of estuarine and coastal marine ecosystems. *J. Limnol. Soc. Sth. Afr.*, 12(1/2), 43-71.
- Nowicki, B. L. & Oviatt, C. A. (1990). Are estuaries traps for anthropogenic nutrients? Evidence from estuarine mesocosms. *Marine Ecology Progress Series*, 66, 131-146.

- Nowicki, B. L. (1994). The effect of temperature, oxygen, salinity, and nutrient enrichment on estuarine denitrification rates measured with a modified nitrogen gas flux technique. *Estuarine, Coastal and Shelf Science*, 38, 137-156.
- O'Neill, P. (1985). *Environmental Chemistry*. George Allen & Unwin Publishers Ltd. pp.211.
- Officer, C. B. & Ryther, J. H. (1980). The possible importance of silicon in marine eutrophication. *Mar. Ecol. Prog. Ser.* 3: 83-91.
- Officer, C. B., Smayda, T. J. & Mann, R. (1982). Benthic filter feeders: a natural eutrophication control. *Mar. Ecol. Prog. Ser.* 9: 203-210.
- Ogilvie, B, Nedwell, D. B., Harrison, R. M., Robinson, A. & Sage, A. (in prep). Muddy estuaries as nitrogen sinks: the nitrogen budget of the River Colne estuary.
- Orth, R. J. (1977). Effect of nutrient enrichment on growth of eelgrass *Zostera marina* in the Chesapeake Bay, Virginia, USA. *Mar. Biol.*, 44:187-194.
- Osinga, R., Lewis, W. E. *et al.*, (1995). Effects of the sea urchin *Echinocardium cordatum* on oxygen uptake and sulphate reduction in experimental benthic systems under increasing organic loading. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 221-236. October 13-16, 1993, Denmark. Ophelia Publications.
- OSPAR (1995). Draft monitoring guidelines for the JAMP issue 2.2 Eutrophication presented by the UK. Oslo and Paris Conventions for the prevention of marine pollution ad hoc working group on monitoring. Copenhagen (Denmark) 13-17 Nov. 1995 MON 95/8/1-Rev.1-E.
- Owens N. J. P. (1986). Estuarine nitrification: a naturally occurring fluidised bed reaction? *Estuar. Coast. Shelf Sci.*, 22, 31-44.
- Paerl, H. W. (1995). Coastal eutrophication in relation to atmospheric nitrogen deposition: current perspectives. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 237-260. October 13-16, 1993, Denmark. Ophelia Publications.
- Parr, W & Norton, R.L. (1994) Eutrophication of tidal waters: classification and screening tools. WRc Report no. SR3707.
- Parr, W & Wheeler, M. A. (1995). Trends in nutrient enrichment of sensitive marine areas in England. Report by WRc for English Nature. WRc Report No.: CO 4055.
- Parr, W. (1993). Algal bioassays: a review of their potential for monitoring and managing the trophic status of natural waters. WRc Report No SR 3439/1.
- Parr, W., Nixon, S. C., Ashley, S. & Woodrow, D. (1993). Classification options for trophic status R&D Note 248.
- Parr, W., Wheeler, M.A. & Codling, I. (1998) Trophic status of North Welsh estuaries: nutrient budgets, ecological sensitivity and monitoring recommendations. WRc draft report to CCW.
- Parr, W., Mainstone, C.P. & Morgan, N. (1998) Nutrient budget of the upper reaches of the Hampshire Avon. WRc Report No CO 4426/1.

- Patten, B. C. (1962). Species diversity in net phytoplankton of Raritan Bay. *Journal of Marine Research* 20: 57-75.
- 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.
- Pedersen, M. F. (1995). Nitrogen limitation of photosynthesis and growth: comparison across aquatic plant communities in a Danish estuary (Roskilde Fjord). In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 261-272. October 13-16, 1993, Denmark. Ophelia Publications.
- Pedersen, O. B., Christiansen, C. & Laursen, M. B. (1995). Wind-induced long term increase and short term fluctuations of shallow water suspended matter and nutrient concentrations, Ringkøbing Fjord, Denmark. *Ophelia*, 41, Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 273-287. October 13-16, 1993, Denmark. Ophelia Publications.
- Perdon, J. (1995). Part A: Vergelijken van estuaria: abiotische parameters mbt eutrofiering. Rijksinstituut voor kust en zee/ Rijkz Mei 1995. Report 9JMM02. Part B: Literatuuruithrehsels van estuariene systemen in west-Europa. Rijksinstituut voor kust en zee/ Rijkz Mei.
- Perkins, E. J & Abbot, O. J. (1972). Nutrient enrichment and sand flat fauna. *Mar. Poll. Bull.* 3: 70-72.
- Pinckney, J., Paerl, H. W. & Fitzpatrick, M. (1995). Impacts of seasonality and nutrients on microbial mat community structure and function. *Marine Ecology Progress Series*, 123, 207-216.
- Pitkanen, H. & Tamminen, T. (1995). Nitrogen and phosphorus as production limiting factors in the estuarine waters of the eastern Gulf of Finland. *Marine Ecology Progress Series*, Vol. 129, 283-294.
- Pomfret, JR, Elliott, M.G., O'Reilly & Phillips, S. (1991). Spatial and temporal patterns in the fish communities in 2 UK North Sea estuaries. In: Elliott, M. & Ducrotoy, J-P. (Eds.) *Estuaries and Coasts: spatial and temporal intercomparisons*. Olsen & Olsen, Fredensborg, Denmark, 277-284.
- Pritchard, D. W. (1967). Observations of circulation in coastal plain estuaries In Lauff, G. (ed.) *Estuaries*. AAAS Publication No. 83. Washington, DC. P 37-44.
- Rachor, E. (1990). Changes in sublittoral zoobenthos in the German Bight with regard to eutrophication. *Netherlands Journal of Sea Research*, 25 (1/2), 209-214.
- Raffaelli, D., Hull, S. & Milne, H. (1989). Long-term changes in nutrients, weed mats and shorebirds in an estuarine system. *Cahiers Biologique Marine*, 30, 259-270.
- Rahm, L., Conley, D., Sandén, P., Wulff, F. & Stålnacke, P. (1996). Time series analysis of nutrient inputs to the Baltic Sea and changing DSi:DIN ratios. *Marine Ecology Progress Series*, 130, 221-228.
- Redfield, A. C., Ketchum, B. H. & Richards, F. A. (1963). The influence of organisms on the composition of sea-water. In: Hill, M. N. (ed.) *The Sea*, 26-79. Wiley Interscience New York.

Rees, Y. J., Nixon, S. C., Parr, W. & van Dijk, P. (1994). Nutrient levels and statutory quality objectives for estuaries and coastal waters. Produced by WRc plc. for National Rivers Authority. R & D Note 70.

Rees, H. L., Heip, C., Vincx, M & Parker, M. M. (1991). Benthic Communities: use in monitoring point source discharges. ICES Techniques in Marine Environmental Sciences No. 16, 70pp.

Remane, A. and Schleiper, C. (1958). Die biologie des brackwassers. E. Schwiezerbart'sche verlagbuchhandlung. Stuttgart 348 pp.

Riisgard, H. U., Christensen, P. B., Olesen, N. J., Petersen, J. K., Moller, M. M. & Andersen, P. (1995). Biological structure in a shallow cove (Kertinge Nor, Denmark) - control by benthic nutrient fluxes and suspension-feeding ascidians and jellyfish. In: Muus, K. Gray, J. *et al.*, (eds.). *Ophelia International Journal of Marine Biology*, Vol. 41. Proceedings of the International Symposium on Nutrient Dynamics in Coastal and Estuarine Environments pp. 329-344. October 13-16, 1993, Denmark. Ophelia Publications.

Rosenberg, R., Gray, J. S., Josefson, A. B. & Pearson, T. H. (1987). Peterson's benthic stations revisited. II. Is the Oslofjord and eastern Skagerrak enriched? *J. Exp. Mar. Biol.* 105:219-251.

Rosenberg, T. (1985). Eutrophication. The future marine coastal nuisance. *Mar. Poll. Bull* 16 (6): 227-231.

Rumohr, H. (1990). Soft bottom macrofauna: collection and treatment of samples. ICES Techniques in Marine Environmental Sciences No. 8, 18pp.

Sanders, R., Jickells, T., Malcolm, S., Brown, J., Kirkwood, D., Reeve, A., Taylor, J., Horrobin, T & Ashcroft, C. (1997) Nutrient fluxes through the Humber estuary. *J. Sea Research* 37(1-2), 3 -23.

Sand-Jensen, K & Borum, J. (1991). Interactions among phytoplankton, periphyton, and macrophytes in temperate freshwaters and estuaries. *Aquatic Botany*, 41, 137-175.

Schimmel, S. C., Melzian, B. D., Campbell, D. E., Strobel, C. J., Benyi, S. J., Rosen, J. S. & Buffum, H. W. (1994). Statistical summary: EMAP-Estuaries Virginia Province -1991. U.S. Environmental Protection Agency, Office of Research and Development, Environmental Research Laboratory, Narragansett, RI. EPA/620/R-94/005.

Schramm W. (1996). Conclusions of Ecological Studies, Vol. 123 Marine Benthic Vegetation Recent Changes and the effects of Eutrophication Schramm W. and Neinhuis P. H. (eds.) pp 449-458.

Short, F. T. (1987). Effects of sediment nutrients on seagrasses: literature review and mesocosm experiment. *Aquatic Botany*, 27, 41-57.

Smayda, T. J. (1990). Novel and nuisance phytoplankton blooms in the sea: evidence for a global epidemic. In: Graneli, E., Sundstrom, B., Edler, L., Anderson, D. M. (eds.) *Toxic marine phytoplankton*. Elsevier, Inc., Amsterdam, p. 29-40.

Soulsby, P. G., Lowthion, D., Houston, M. & Montgomery, H. A. C. (1985). The role of sewage effluent in the accumulation of macroalgal mats on intertidal mudflats in

two basins in Southern England. *Netherlands Journal of Sea Research*, 19 (3/4), 257-263.

Soulsby, P. G., Lowthion, D. & Houston, M. (1982). Effects of Macroalgal mats on the ecology of intertidal mudflats. *Marine Pollution Bulletin*, 13, No. 5, 162-166.

Spurrier, J. D. & Kjerfve, B. (1988). Estimating the net flux of nutrients between a salt marsh and a tidal creek. *Estuaries* 11:10-14.

Steidinger, K.A. & Vargo, G.A. (1998) Marine dinoflagellate blooms: dynamics and impacts. In: C.A.Lembi and J.R. Waaland (eds) *Algae and Human Affairs*, 373 –401. Cambridge University Press.

Stockner, J. G. & Benson, W. W. (1967). The succession of diatom assemblages in the recent sediments of Lake Washington. *Limnol. And Oceanogr.* 12:513-532.

Strickland, J.D.R. & Parsons, T.R. (1968) A practical handbook of Sea Water Analysis, Fish. Res. Res. Bd. Can. Bull 167.

Sundback, K. (1994). The responses of shallow water sediment communities to environmental change. In Krumbein W. E., Paterson D. M. and Stal L. U. (Eds.) *Biostabilization of Sediments* ISBN 3-8142-0483-2.

Tait, R. V.(1981). *Elements of Marine Ecology: An introductory course*. Third edition. Cambridge University Press.

Taylor, D. I. (1992). The influence of upwelling and short-term changes in concentrations of nutrients in the water column on fluxes across the surface of a salt marsh. *Estuaries* 15:68-74.

Taylor, D., Nixon, S., Granger, S. & Buckley, B. (1995a). Nutrient limitation and the eutrophication of coastal lagoons. *Marine Ecology Progress Series*, 127, 235-244.

Taylor, D. I., Nixon, S. W., Granger, S. L., Buckley, B. A., McMahon, J. P. & Lin, H. -J. (1995b). Responses of coastal lagoon plant communities to different forms of nutrient enrichment - a mesocosm experiment. *Aquatic Botany*, 52, 19-34.

Tett, P (1987). Plankton In: Baker, J. M. & Wolff, W. J. (eds.) *Biological surveys of estuaries and coasts*. EBSA Handbook. Cambridge University Press, Cambridge, UK pp 280-341.

Trilateral Monitoring Expert Group (1993). *Integrated Monitoring Program of the Wadden Sea Ecosystem*. Annex 4.1. Issues of concern and lists of relevant parameters.

Tubbs, C. R. (1977). Wildfowl and waders in Langstone Harbour. *British Birds*, 70, 177-199.

van Beusekom, J. E. E & Jonge, V. N. de (1995). De filterfunctie van het Eems estuarium voor opgelost anorganisch stikstof (DIN), opgelost anorganisch fosfaat (DIP) en silicaat. Rapport RIKZ-95,015, Rijkswaterstaat, Haren.

Van Impe, J. (1985). Estuarine pollution as a probable cause of increase of estuarine birds. *Marine Pollution Bulletin*, 16, No. 7, 271-276.

Van Raaphorst, W., de Jonge, V. N., Dijkhuizen, D. & Frederiks, B. (in prep). Natural background concentrations of phosphorus and nitrogen in the Dutch Wadden Sea. RIKZ.

- Vernberg, F.J. (1993). Salt-Marsh processes - A review. *Environmental Toxicology and Chem* Vol. 12, No: 12, pp2167-2195.
- Vidondo, B. & Duarte, C. M. (1995). Seasonal growth of *Codium bursa*, a slow-growing Mediterranean macroalga: in situ experimental evidence of nutrient limitation. *Marine Ecology Progress Series*, 123, 185-191.
- Villate, F., Franco, J., Gonzalez, L., de Madariaga, I., Ruiz, A. & Orive, E. (1991). A comparative study of hydrography and seston in five estuarine systems of the Basque Country. In: Elliott, M & Ducrottoy, J-P (eds.). *Estuaries and Coasts: Spatial and Temporal Intercomparisons*, pp. 97-105. ECSA 19 Symposium. Olsen & Olsen.
- Vos, P.C. & de Wolf, H. (1993) Diatoms as a tool for reconstructing sedimentary environments in coastal wetlands; methodological aspects, *Hydrobiologia*, 269/270, 285-296.
- Warwick, R. M. & Clarke, K. R. (1993) Comparing the severity of disturbance: a meta-analysis of marine macrobenthic community data., *Mar. Ecol. Prog. Ser.* 92: 221-231.
- Weston, D. P. (1990). Quantitative examination of macrobenthic community changes along an organic enrichment gradient. *Marine Ecology Progress Series*, 61, 233-244.
- White, J. R. & Roman, M. R. (1992). Seasonal study of grazing by metazoan zooplankton in the mesohaline Chesapeake Bay. *Mar. Ecol. Prog. Ser.* 86: 251-261.
- Whitehead (1988) cited in Johnes (1994)
- Whiting, G. J. & Childers, D. L. (1989). Subtidal advective water flux as a potentially important nutrient input to south-eastern USA saltmarsh estuaries. *Estuar. Coast. Shelf Sci.* 28:417-431.
- Widbom, B., & Elmgren, R. (1988). Response of benthic meiofauna to nutrient enrichment of experimental marine ecosystems. *Mar. Ecol. Prog. Ser.* 42: 257-268.
- Wilson, J. G & Jeffrey, D. W. (1987): Europe-wide indices for monitoring estuarine quality. In: *Biological Indicators of Pollution*, eds. Richardson, D. H. S, Royal Irish Academy.
- Wofsy, S. D. (1983). A simple model to predict extinction coefficients and phytoplankton biomass in eutrophic waters. *Limnol. Oceanogr.* 28: 1144-1155.
- Woodrow, D., Parr, W., Norton, R. L. and Johnson, D. (1994). River eutrophication risk modelling: a GIS approach. WRC Report to SNIFFER, No. NR 3698. WRC, Medmenham.
- Wulff, F., Perttilä, M. & Rahm, L. (1994). Mass-balance calculations of nutrients and hydrochemical conditions in the Gulf of Bothnia, 1991. *Aqua Fennica*, 24, 12-140.
- Wulff, F. & Niemi, Å. (1992). Priorities for the restoration of the Baltic Sea - A scientific perspective. *Ambio*, 21, No. 2, 193-195.
- Wulff, F., Stigebrandt, A. & Rahm, L. (1990). Nutrient dynamics of the Baltic Sea. *Ambio*, 19, No. 3, 126-133.
- Wulff, F. & Ulanowicz, R. E. (1989). A comparative anatomy of the Baltic Sea and Chesapeake Bay Ecosystems (Chapter 11). pp 232-256 in: Wulff, F., Field, J. G. &

Mann, K. H. (eds.), Network Analysis in Marine Ecology - Methods and Applications, Coastal and Estuarine Studies, 32, Springer-Verlag, 1989.

Wulff, F (1985). Eutrophication and the Baltic Sea: causes and consequences. *Ambio*. 14 (1): 9-14.

Yallop, M. L., Dewinder, B., Paperson, D.M. & Stal, L. J. (1994). Comparative structure, primary production and biogenic stabilization of cohesive and non-cohesive marine-sediments inhabited by microphytobenthos. *Estuarine, Coastal & Shelf Science*, 39, No. 6, pp.565-582.

Zwolsman, J. J. G. (1994). Seasonal variability and biogeochemistry of phosphorus in the Scheldt Estuary, South-west Netherlands. *Estuarine, Coastal and Shelf Science*, 39, 227-248

APPENDIX 1: THE IMPACTS OF NUTRIENTS IN ESTUARIES: PHASE 2 OBJECTIVES

OVERALL OBJECTIVES

To provide an authoritative review of information necessary for monitoring and managing the impacts of nutrients on water quality and conservation in estuaries.

SPECIFIC OBJECTIVES

DESCRIBING AND CLASSIFYING ESTUARIES.

1. To review the information on nutrient loads and concentrations in estuaries, including an assessment of the role of the catchment,

by:-
 - a) Establishing the adequacy of available data-sets in defining the current nutrient status of estuaries;
 - b) Establishing the significance of the interaction of nutrients with other estuarine physico-chemical variables (eg. dilution, turbidity, residence time, and other characteristics that may relate to the catchment);
 - c) Classifying estuaries into groups (using the above criteria) that will facilitate optimisation of monitoring and management strategies;
 - d) Reviewing the relative importance of estuarine sub-communities (phytoplankton, benthic micro-algae and benthic bacteria) to nutrient fluxes in these different types of estuary;
 - e) To establish the existence of any historical baseline nutrient data, or methodologies to assess (hindcast) pristine nutrient levels, in the different categories of estuary.

IMPACT ASSESSMENT

2. To assess the impacts of both chronic and acute nutrient elevation on estuaries (as ecological communities and functioning biological systems), and to determine aspects that could lead to the derivation of quality objectives, to drive the management of estuarine eutrophication,

by:-

- a) identifying the impacts of elevated nutrients on the conservation of rare species, and the maintenance of biological diversity in estuarine communities, in the different types of estuary;
- b) and so suggesting components of estuarine communities that could be monitored as indicators of nutrient impact (criteria supporting the designation of "sensitive areas" under the Urban Waste Water Treatment Directive should be critically appraised);
- c) determining changes in the significance of the different sub-communities to the estuarine nutrient budget, under conditions of nutrient elevation;
- d) examining how elevated nutrients interact with each other (reviewing nutrient limitation, N:P ratios and synergistic effects along the estuarine gradient), and with other environmental variables (inc. significance of seasonal effects), to influence biological communities;
- e) and so suggesting what aspects of nutrient dynamics, could most usefully be measured in the different types of estuary, as indicators of nutrient impact.

TOWARDS MANAGEMENT

3. To summarise existing tools relevant to the management of estuarine nutrient impacts, and consolidate any monitoring approaches emerging from the review detailed above, in order to move towards a strategy for the management of eutrophication in estuaries,

by:-

- a) reviewing the use of estuarine nutrient models;
- b) identifying deficiencies in the knowledge of nutrient impacts on estuaries, and recommending studies to fill the gaps;
- c) making recommendations for ecologically-relevant monitoring of estuaries;
- d) considering how any ecological quality objectives that emerge, might relate to other "use-based" objectives, which would contribute to integrated estuarine management;
- e) and ultimately recommending the specific objectives for the next stage of the project (in the form of a Phase 3 PID), that will progress scientific understanding towards management.

APPENDIX 2: RESEARCHERS AND ORGANISATIONS CONTACTED DURING THE NUTRIENTS IN ESTUARIES PROJECT

United Kingdom

Balls, Phil	SOAEFD Aberdeen
Davis, Jill	Environmental Agency, Welsh SE Region
Davison, Bob	Environment Agency National Marine Survey Centre, Bath
Dobson, Judy	SEPA, Edinburgh
Doge, John	Department of Biology, University of London
Edwards, Tony	Macaulay Land Use Research Institute, Aberdeen
Gize, Irene	Environment Agency, Warrington
Hall, Sarah	University of Huddersfield, Huddersfield
Harland, Rex	'Dinodata' Services, Nottingham
Holden, Nick	Environment Agency National Marine Survey Centre, Bath
Hunt, Chris	University of Huddersfield, Huddersfield
Jones, Ken	Dunstaffnage Marine Laboratory Oban
Juggins, Steve	University of Newcastle
Mathieson, Scot	SNH Edinburgh
Morris, Alan	Plymouth Marine Laboratory
Nedwell, Dave	University of Essex
Neil, Colin	Institute of Hydrology Wallingford
Park, Richard	SEPA, Edinburgh
Raffaelli, Dave	Culterty Field Station, University of Aberdeen
Raven, John	Dundee University
Richardson, Chris	University College North Wales, Bangor
Service, Matthew	AESD
Shaw, P.	LOIS/RACS Southampton Oceanography Centre
Tett, Paul	Napier University, Edinburgh
Thomas, Juan	Environmental Agency, Bangor
Turl, Lyndsay	SOAEFD, Edinburgh
Watson, P.	LOIS Plymouth Marine Laboratory
Wither, A.	Environment Agency, Warrington

USA

Alber, Merryl	University of Georgia, Dept of Marine Science, Athens, GA 30602-2206
Baptista, Antonio	Oregon Graduate Institute of Oceanography, Portland, OR 97291-1000
Bledsoc, (Sam) L.J.	University of California, Davis Dept of Land, Air and Water Resources, Davis, CA
Bortone, Stephen	Institute for Coastal and Estuarine Research, University of West Florida, Florida 32514.
Boynton, Walter	Chesapeake Biological Laboratory Solomons, MD 20688
Bricker, Suzanne	National Oceanic and Atmospheric Administration (NOAA)
Chris, Madden	Horn Point Environmental Laboratory
Christopher, D'Elia	Maryland Sea Grant College
Cloern, James	U.S. Geological Survey, CA 94025.
Cordell, Jeff	University of Washington, School of Fisheries, Wetland Ecosystem Team, Seattle, WA 98195.
Costanza, Robert	University of Maryland System, Solomons, MD

Cumming, Brian	cummingb@biology.queensu.ca
Daniel, Conley	Horn Point Environmental Laboratory
DeCurtis, Chuck	Institute of Ecology, University of Georgia, Athens, GA 30602-2206
Erik, Smith	Horn Point Environmental Laboratory
Hagy, James	Chesapeake Biological Laboratory Solomons, MD 20688
Holland, A. Frederick	S.C. Marine Resources Research Institute, Charleston, SC 29422-2559
Hopkinson, Chuck	The Ecosystems Laboratory, Marine Biological Laboratory, Woods Hole, MA 02543
Houde, Edward	Chesapeake Biological Laboratory Solomons, MD 20688
Jay, David	University of Washington, Seattle, WA 98195.
Klima, Karen	Environmental Protection Agency, USA
Lawrence, Sanford	Horn Point Environmental Laboratory
Lee, Virginia	Coastal Resources Center, Graduate School of Oceanography, University of Rhode Island, Narragansett, RI 02882-1197
Malone, Thomas	Horn Point Environmental Laboratory, Cambridge, MD 21613
McKellar, Hank	Associate Department of Environmental Health Sciences, University of South Carolina, Columbia, SC 29208.
Bender, Michael	URI Sea Grant College Program, University of Rhode Island, Narragansett RI 02882.
Kemp, Michael W.	Horn Point Environmental Laboratory
Nixon, Scott	University of Rhode Island, Narragansett RI 02882
Nowicki, Barbara	University of Rhode Island, Narragansett RI 02882
Scaffner, Linda	VIMS Eastern Shore Laboratory Wachapreague
Taylor, David	University of Rhode Island, Narragansett RI 02882.
Tucker, Jane	The Ecosystems Centre, Marine Biological Laboratory, Woods Hole, MA 02543
Vernberg, John	Belle W. Baruch Institute for Marine Biology and Coastal Research, University of South Carolina, SC 29208
Webb- Kenneth, L.	VIMS Eastern Shore Laboratory Wachapreague
Webb, Kenneth	Virginia Institute of Marine Science, Gloucester Point, VA 23062-1346
Wetzel, Richard	VIMS Eastern Shore Laboratory Wachapreague

Australia

Brunskill, Gregg	Australian Institute of Marine Science (AIMS), Townsville MC, Queensland
Deely, David	Swan Canning Cleanup Programme, Water & Rivers Commission, Perth WA
Davies, Richard	CSIRO Co-ordinator of the National Eutrophication Management Program.
Environmental Protection Agency	40 Blackall Street, Barton Act 2600
Finnigan, John	Environmental Mechanics Group
Furnas, Miles	Australian Institute of Marine Science (AIMS), Townsville MC, Queensland
Gerritse, Robert	CSIRO Division of Water Resources, Wembley WA 6014
Harris, Graham	CSIRO Project Office University of Canberra
Hegge, Bruce	D A Lord & Associates Nedlands WA.
Jacoby, Charles	CSIRO Division of Fisheries North Beach Western 6020
Jernakoff, Peter	CSIRO, Perth
Jones, Gary	Resource Management Institute, Indooroopilly, Queensland 4068
Larkum, A.	University of Sydney, School of Biological Sciences
Lord, D.	D A Lord & Associates Nedlands WA.
Murray, D.	University of Sydney, School of Biological Sciences
Newell, Brian	CSIRO Port Phillip Bay Environmental Study, Melbourne Vic 3003

Saenger, Peter	Centre for Coastal Management, Southern Cross University Lismore NSW 2480
Schofield, Nick	Land and Water Resources Research and Development Corporation (LWRRDC), Canberra.
Thompson, Peter	CSIRO Division of Fisheries, Perth
Vernon, David	Environment Technology & Industry Section, 40 Blackall Street, Barton

Canada

Wells, Peter,	Environment Canada, Bedford Institute of Oceanography, Dartmouth, Nova Scotia
Georgia Strait Alliance	no address available
Environment Canada	Bedford Institute of Oceanography, Dartmouth, Nova Scotia

Europe

Finnish Environment Agency

Leppänen, Juha-Markku	Finnish Institute of Marine Research, Helsinki, Finland
Jansson, Mats	Department of Geography, Umeå University, Umeå, Sweden
Rosenberg, Rutger	Kristineberg Marine Research Station, S-450 34 Fiskebäckskil, Sweden
Dr. Arne Sjöqvist	Swedish Environmental Protection Agency, Stockholm, Sweden
Dr. Fredrik Wulff	Systems Ecology, Stockholm University, Stockholm, Sweden

APPENDIX 3: ESTUARY DEFINITIONS AND CHARACTERISTICS

A number of definitions of estuaries are available these are described and reviewed below:

The definition of an estuary that is most commonly used by coastal scientists is that given by Pritchard (1967):

an estuary is a semi-enclosed coastal body of water, which has free connection with the open sea, and within which sea water is measurably diluted with fresh water derived from land drainage.

Although widely accepted this definition significantly excludes coastal lagoons which do not usually have a free connection with the open sea, and may only be inundated with sea water at irregular intervals. It also excludes brackish seas, such as the Caspian Sea, which have salinities comparable to some parts of estuaries, but do not experience the regular fluctuations of salinity due to tidal effects. Additionally, McLusky (1989) notes that the Pritchard definition of 'semi-enclosed' excludes coastal marine bays, and the definition of 'fresh water derived from land drainage' excludes saline lakes with fresh water from rainfall only. McLusky (1989) therefore considered an estuary as being:

a dynamic ecosystem having a free connection with the open sea through which sea water enters normally according to the twice-daily rhythm of the tides. The sea water that enters the estuary is then measurably diluted with fresh water flowing into the estuary from rivers.

Davidson *et al.*, (1991) emphasises that estuaries are distinct from other kinds of wetlands in that they link three environments marine, freshwater and dryland ecosystems. For the purposes of their Nature Conservancy Council's Estuaries Review describe an estuary, or perhaps more accurately an estuarine area, as:

a partially enclosed area of water and soft tidal shore and its surroundings, open to saline water from the sea and receiving fresh water from rivers, land run-off or seepage.

A further definition by Fairbridge (1980) takes into account factors other than the dilution of sea water by freshwater (*i.e.* salinity) and provides one of the simplest and best definitions:

an inlet of the sea reaching into a river valley as far as the upper limits of tidal rise.

Entec (1995) considered that the tidal limit constituted a clearly definable boundary of the upper estuary but recognised that the lower estuary's boundaries were more difficult to locate. They recommended that the lower estuary limit should be defined as the zone where the water circulation changes and noted that in certain areas such as the Wash locally-based judgements were required to determine these limits.

In the Urban Waste Water Treatment Directive (91/271/EEC) an estuary is defined rather vaguely as:

the transitional area at the mouth of a river between freshwater and coastal waters.

while the Comprehensive Studies Task Team (CSTT 1993) presents two definitions which have been adopted by the UK for the purposes of operating this Directive. These are:

an area receiving freshwater inputs where the waters on a depth averaged basis have a salinity of less than 95% of the adjacent local offshore seawater for 95% of the time.

an inlet of the sea bounded by a line between such topographical features as define the seaward boundary of the estuary.

The first of these definitions is overly complicated and although many definitions describe the transition of salinity from freshwater to sea water, no previous reference has been found to the 95% salinity for 95% of the time described. The second definition is too vague and actually employs the term 'estuary' within the definition.

One of the key difficulties with obtaining a clear definition of estuaries is that estuarine systems differ widely in their geomorphological characteristics. As a result there are a number of different estuary classifications have been developed. For instance the balance of freshwater flow against tidal inundation within estuaries has been used for a classification of estuaries, with three main types being recognised: positive, negative and neutral estuaries.

Positive Estuary: the evaporation from the surface of the estuary is less than the volume of freshwater entering it from rivers and land drainage. The estuary is characterised by the outgoing freshwater floating on top of the saline water which has entered from the sea, with the water gradually mixing from the bottom to the top. This type of estuary is commonly found in temperate regions.

Negative Estuary: the evaporation from the surface of the estuary is greater than the volume of freshwater entering it from rivers and land drainage. This evaporation causes the surface salinity to increase which in turn increases the density of the saline surface water. As this surface water is then denser than that of the underlying water, it sinks. Thus the circulation pattern is the opposite to that of the Positive Estuary. This type of estuary is commonly found in the tropics.

Neutral Estuary: an estuary is termed neutral if the freshwater input equals the evaporation leading to a static salinity regime. This type of estuary however is rare as evaporation and freshwater inflow are almost never equal.

Another distinction between estuaries can be drawn by the strength of the tidal regime with three estuary types are usually identified based on the mean tidal range Davis (1973) these are:

- Macrotidal (tidal range >4m)
- Mesotidal 2-4m
- Microtidal <2m.

The Nature Conservancy Council's estuary review identifies the following nine categories of estuary on the basis of the geomorphological differences (Davidson *et al.*, 1991):

- **Fjords:** Essentially drowned glacial troughs, often associated with major lines of geological weakness. They are characteristic of areas once covered by Pleistocene ice sheets. Erosion by the ice deepened existing river valleys, and the movement and scouring action of the ice left characteristic shallow rock bars, particularly at the fjord mouths. Fjords generally have a close width-depth ratio, steep sides, and an almost rectangular cross section. They generally have rocky floors, and the 6 fjords identified by the Estuaries Review were found in north and west Scotland.
- **Fjards:** Structurally, fjards are more complex than fjords, with a more open and irregular coastline and often no main channel. Although ice-scoured rock basins and bars are characteristic features, fjards are relatively shallow, and although they are more exposed to wave action than fjords, they are sheltered in their upper reaches. The 20 fjards identified in Britain have a wider distribution than estuarine fjords, but are still confined to western and northern British coasts between Anglesey and Orkney.
- **Rias:** Rias have not been formed or modified by glacial processes, and are distinguished as drowned river valleys formed by tectonic subsidence of the land, a rise in sea level, or a combination of both. Rias are relatively deep, narrow, well-defined channels which are almost completely marine influenced. They have no entrance sill or ice-scoured rock bars and rock basins; and are shallower than fjords. The predominant substrata of the channel floors of most rias is bedrock, but the sheltered parts of bays and inlets adjacent to the main channels contain soft sediments. The 15 rias identified by the Estuaries Review occurred predominantly in the Carboniferous and Devonian rocks of Devon and Cornwall, although there are also two rias in south Wales, at Milford Haven and the Neath Estuary.
- **Coastal plain estuaries:** Coastal plain estuaries were formed during the Holocene transgression through the flooding of pre-existing valleys in both glaciated and unglaciated areas. Maximum depths in these inlets are generally less than 30 m, and they have the cross-section of normal valleys which deepen and widen towards their mouths. The width-depth ratio is usually large, and unlike rias, extensive mudflats and saltings often occur which tend to become increasingly sandy towards the mouth. Along with bar-built estuaries, coastal plain estuaries are one of the commonest estuary types in Britain with 35 Estuary Review sites being identified as coastal plain estuaries. In Britain, these estuaries have a wide distribution throughout England and Wales, particularly in Suffolk, West Sussex, Hampshire, south Wales, and along the north Wales-Lancashire stretch of the west coast. The Severn Estuary, The Dee Estuary and North Wirral, the Humber Estuary and The Thames Estuary complex are coastal plain type. Only two examples, the Dee and the Don (both in Aberdeen) occur in Scotland.
- **Bar-built estuaries:** Bar-built estuaries are also drowned river valleys which have been incised during the ice ages and subsequently inundated. However, they are distinguished by recent sedimentation, and have a characteristic bar across their mouths. These estuaries are only a few metres deep and often have extensive lagoons and shallow waterways just inside the mouth. A dominant feature of bar-built estuaries is the availability of abundant sediments in the coastal system, allowing tidal dynamics to rework and deposit this material along the coast and across river valley mouths. These

sand and shingle bars at estuary mouths can develop from a variety of sources. Because of their restricted cross-sectional area, current velocities can be high at the mouth, but in the wider parts further inland they rapidly diminish. During floods the bar may be swept completely away, but will quickly re-establish itself when the river flow diminishes. A total of 47 bar-built estuaries have been identified, with their main areas of distribution being in west Wales, the south coast of England, East Anglia and eastern Scotland.

- **Complex estuaries:** Estuaries in this group are of complex origin and therefore do not easily fit into the other types in the classification. In Britain, complex estuaries fall into two distinct categories. One is a group of the six major Scottish Estuaries (including the Solway Firth, the Moray, Cromarty and Dornoch Firths, and the Firths of Tay and Forth). The hydrodynamics of these estuaries are similar to those of coastal plain estuaries, but their overall size and geomorphological complexity sets them apart. The second group, of four estuaries in north-east England, are sites which reflect the influence of geological controls, glaciation and incision into relatively hard rock types during periods of relative sea-level change during the Quaternary.
- **Barrier Beaches:** Barrier beaches are open coast systems which characteristically develop as soft shores in shallow water, where the dissipation of wave and current energy offshore leads to the development of bars and barriers. Within these coastal systems sediment is abundant. Only two British Estuary Review sites are classified as barrier beach systems: the North Norfolk Coast, and Lindisfarne on the Northumberland coast. However, only that of North Norfolk is a classic beach barrier system in which extensive saltmarshes, creeks and tidal flats have developed behind a shingle and sand-dune barrier.
- **Linear shores:** The linear shore sites are those where there is a little indentation of the coast and the coastal outline is convex, linear or only slightly concave. Seven Estuary Review sites have been identified, including a cluster in south-east England at Dengie, Maplin, Southend and North Kent Marshes. Throughout such broad areas, in conditions of shallow seas and abundant sediments, where wave and tidal energy are dissipated out from the shore, soft sediments are deposited as in the development of barrier islands.
- **Embayments:** Embayments are formed where the line of the coast follows a concave sweep between rocky headlands. Soft sediments generally infill such embayments. 14 embayment coasts have been recognised in the Estuaries Review classification. These are widely distributed around the coast of Britain and include, Carmarthen Bay, Morecambe Bay and The Wash.

APPENDIX 4: BIOLOGICAL FEATURES OF ESTUARIES

To provide a brief overview of the biological characteristics of estuaries, the following aspects of estuarine ecology are reviewed here:

- the characteristic composition and distribution of species,
- the nature of food webs,
- the habitat types
- community types.

Composition and Distribution of Species

In estuaries, as with any environment, the communities present are the product of environmental forcing variables superimposed over which are the influences of biotic interactions from processes such as competition and recruitment.

The dynamic nature of estuaries has a profound effect on the composition and distribution of species by creating highly unstable conditions that in turn, restrict the diversity of the communities. The biggest single influence on species is the salinity regime. Salinity has a significant influence on the physiological responses of aquatic species and the large change in salinity along an estuary gradient, and its tide-driven fluctuations, affect their survival and therefore their distribution within the estuary.

Other influential factors such as the current strength, tidal amplitude, wave strength, and deposition of sediments as well as oxygen, temperature and the supply of nutrients also vary along the length of an estuary and are closely linked to the salinity distribution patterns. As a consequence, communities can be identified according to their position within the estuary in relation to salinity. McLusky, (1989) identifies several categories of estuarine flora and fauna based on the region of the estuary in which they are found and the associated salinity levels.

- **Oligohaline organisms:** include the majority of animals living in rivers and other freshwaters which generally do not tolerate salinities greater than 0.5, but some, can persist at salinities up to 5ppt.
- **True estuarine organisms:** these are mostly marine animals with marine affinities which live in central parts of the estuary. The majority of these organisms could live in the sea, but are apparently absent due to competition from other organisms. Most common at salinities of 5-18ppt.
- **Euryhaline marine organisms:** these constitute the majority of organisms living in estuaries with their distribution ranging from the sea into the central parts of estuaries. Many disappear by 18ppt, but a few survive at salinities down to 5ppt.
- **Stenohaline marine organisms:** these are marine organisms which live in the mouths of estuaries, at salinities down to 25ppt.
- **Migrants:** these animals, mainly fish and crabs, spend only part of their lives in estuaries, with some such as the flounder (*Platichthys flesus*) or the shrimp (*Pandalus montagui*) feeding in estuaries, whilst others such as the salmon (*Salmo salar*) or the eel (*Anguilla anguilla*) use estuaries as routes to and from rivers to the open sea.

As the majority of estuarine species are of marine origin there is typically there is a decrease in diversity of species that accompanies the decrease in the mean salinity levels up an estuary. As the salinity within the interstices varies far less than that of the overlying water, because of the slow rate of interchange across the sediment surface, it is usually possible for marine animals living buried within the sediment to penetrate further into estuaries, than for marine animals which live planktonically (McLusky, 1989). The greatest penetration of marine species often occurs in the intertidal areas where the interstitial salinity matches that of the high-tide salinity when it covers the mudflats, which may be considerably higher than the salinity of the estuarine waters at low tide at the same position in the estuary.

Although diversity is low in estuaries there is a often high input of allochthonous and a high production of autochthonous organic materials which provides a rich supply of food to those species which can adapt to the estuarine environment. Estuaries are therefore characterised by having a lower species diversity, but a greater number of individuals than adjacent aquatic environments. For the same reason, while the diversity declines up an estuary, the abundance of individual species often increases markedly because these tolerant species can avoid competition from marine or freshwater species and exploit the abundant food resource (Remane and Schleiper 1958).

The Nature of Estuarine Food Webs

The high production, input, trapping and recycling of organics and nutrients makes estuaries some of the most fertile and productive ecosystems in the world (Davidson et al., 1991; Nixon 1992). However because this productivity is driven by a range of nutrient sources and because of the complexity of the physical environment, the trophic pathways and food webs within estuaries are often very complicated.

In simple terms these food webs can be described by the passage of energy through progressively higher trophic levels from the primary producers to secondary or tertiary consumers. Plants, in the form of phytoplankton, microphytobenthos, macroalgae, and higher plants seagrasses and saltmarsh species convert the incoming nutrients through photosynthesis into living material. The producers are then consumed by primary consumers which can either graze directly on the macrophytes or filter plankton and suspended organics from the water column (e.g. the mussel, *Mytilus edulis*) or like *Macoma balthica*, *Corophium volutator* and *Hydrobia ulvae* consume deposited on organic material (Davidson et al., 1991).

Primary consumers are themselves eaten by secondary consumers. Davidson et al., (1991) notes that on estuaries there are three main categories of secondary consumer: invertebrates, fish and birds. Invertebrate predators include the shore crab (*Carcinus maenas*) and the shrimps (e.g. the common shrimp *Crangon vulgaris*), together with carnivorous polychaetes which live buried within the sediment such as the cat-worm, *Nephtys hombergi*. Large populations of fish, especially the flounder (*Platichthys flesus*) and gobies, (e.g. the sand goby *Pomatoschistus minutus*) feed on intertidal benthos at high tide when the water covers the mudflats, and may consume large quantities of invertebrates.

Numerous waterfowl species also feed estuaries. Some species such as redshank (*Tringa totanus*) and the shelduck (*Tadorna tadorna*) feed on the intertidal zone at low water on species such as *Macoma balthica*, *Hydrobia ulvae* and other infaunal invertebrates. Eider

ducks (*Somateria m. mollissima*) feed in the shallow water at low tide, particular on *Mytilus edulis*. Others, such as the diving ducks, (goldeneye, *Bucephala clangula* or scaup, *Aythya marila*), along with cormorants (*Phalacrocorax carbo*) and mergansers (*Mergus serrator*) feed at high water by diving for their prey (McLusky, 1989). Some waterfowl such as widgeon (*Anas penelope*) and several species of geese feed directly on plant material (primary producers) thereby bypassing the primary consumer level of the food web. Finally, secondary consumers are in turn eaten by tertiary consumers which in estuaries include birds of prey, piscivorous birds such as cormorants, herons and some diving ducks, and mammals such as otters and seals (two descriptions of estuarine food webs are presented in Figures 13 and 14).

The Habitat Types

Based on the JNCC review of British estuaries (Davidson *et al.*, 1991) this section provides a brief overview of the major estuarine habitat types, tidal flats, saltmarsh, sand dunes, shingle and lagoons as well as an overview of estuary community types based on JNCC's classification.

Tidal Flats: Tidal flats are a major part of the estuarine ecosystem and vary from soft muds to firm sandflats. In general, the finest suspended sediments are deposited in the most sheltered places, so mudbanks generally form in the inner reaches of estuaries. In the main river outflow channels closer to the mouth, and on more exposed shores, higher wave energy and/or faster tidal currents usually do not allow fine sediments to deposit. In these areas only the coarser sediments are deposited, resulting in sandflats in the outer parts. Tidal flats support large numbers and biomass of macrobenthos especially polychaete worms, and gastropod and bivalve molluscs and provide rich feeding grounds for fish and the many species of migrant and wintering waterfowl. The characteristic macrobenthic fauna varies according to different substrata types. Soft mudflats typically support high densities of the gastropod mollusc *Hydrobia ulvae*, bivalve molluscs such as the clam *Mya arenaria*, the polychaete ragworm *Nereis diversicolor* and the amphipod crustacean *Corophium volutator*. Sandier mudflats are often dominated by bivalve molluscs such as the cockle *Cerastoderma edule* and Baltic tellin *Macoma balthica*. The polychaete lugworm *Arenicola marina* is also characteristic of these stable muddy sandflats, and substantial beds of mussels *Mytilus edulis* can develop especially where hard substrata is present for attachment. In the UK there are over 265,000ha of tidal flats, comprising over 83% of the intertidal area of estuaries. Major British tidal flat areas include Morecambe Bay, The Wash, the Dee Estuary and North Wirral, the Humber Estuary, around the Thames Estuary, and in north-east Scotland.

Saltmarsh: Saltmarshes develop wherever tidal waters allow the deposition of fine sediment. At the lowest levels where immersion occurs on nearly every tide, plants tolerant of high salinities and physical movement by the incoming tide become established. These plants are represented by very few species such as eel-grass (*Zostera* spp.) and tasselweed (*Ruppia* spp.) which are essentially aquatic in type and lie flat when the tide is out. Above this zone, at the mean water level of the lower, neap tides, plants with a more terrestrial growth form predominate. These include a few salt-tolerant species such as the annuals glass-wort (*Salicornia* spp.) and seablite (*Suaeda maritima*) and the perennial cord-grass (*Spartina anglica*). Saltmarshes larger than 0.5ha are found on 135 estuaries. They total over 42,250ha comprising over 95% of the British saltmarsh resource and approximately

14% of the total intertidal area on estuaries. The Wash, Morecambe Bay and the Solway Firth each have over 3,000ha of saltmarsh.

Sand-dunes: Sand-dune systems form a major component of the physical environment within 30 British estuaries, and a further 25 sites have sand-dunes associated with them. Many form spits across estuary mouths, and sand-dunes on barrier islands shelter the shore from waves thereby allowing estuarine intertidal habitats to develop in their shelter. On their landward side, dune vegetation, especially marram grass binds the sand together resulting in grassy swards, wetter slacks and sometimes heath, scrub and woodland.

Shingle: Within Britain, there are 7 nationally important shingle structures of which 5 are associated with estuaries. Shingle is an important estuarine feature for its geomorphology, characteristic vegetation and invertebrate animals. Invertebrates associated with coastal and estuarine shingle tend either to be associated with the specialised flora of the shingle (e.g. sea kale, *Crambe maritima*) or to be dependent on the shelter provided by the shingle itself and the litter that accumulates on it.

Lagoons: Saline lagoons are present on 37 Estuaries Review sites throughout England and Wales with major concentrations around the Humber, the Thames Estuary and the Solent system. They support a highly specialised flora and fauna, often of very local distribution.

Coastal Grazing Marshes: Coastal grazing marshes are areas of flat low-lying grassland drained by complex networks of freshwater or brackish drainage ditches. Together with other lowland wet grasslands they are associated with at least 53 estuaries. Coastal grazing marshes have a wide range of salinities, which encourage the growth of a large variety of plants often including rare or scarce species, especially in the ditches.

Community Types

For the purposes of describing all marine and brackish-water communities the JNCC Marine Nature Conservation Review has developed a biotope classification for the UK coast. A biotope is a distinguishable community based on a series of physical and biological parameters, including sediment type, position down the shore, wave exposure, dominant species. The classification aims to encompass biotopes in both the littoral and sublittoral zones (Hiscock unpubl.) but at the present time classification is still at a working stage and, although it is reasonably well developed for the littoral zone, the sublittoral still requires substantial further development.

Within estuaries, sediment is a predominant feature and the following range of sedimentary habitats are available for colonisation by aquatic organisms. Sands and muds are the most common type of estuarine soft substrata and this is reflected in the high abundance of fauna which colonise these habitats. According to Davidson *et al.*, (1991) soft shore community types are widespread, with five communities being found on more than 20% of estuaries. These include an exposed sand community which is dominated by small crustaceans (i.e. *Haustorius arenarius* and *Pontocrates norvegicus*) and polychaete worms (i.e. *Nephtys cirrosa*, *Malacoceros fuliginosus* and *Paraonis fulgens*) in the outer parts of estuaries; mussel beds, widespread around Britain in both intertidal and shallow subtidal parts of estuaries; and beds of marine grasses associated with the lower intertidal mud and muddy sand areas within estuaries.

Davidson *et al.*, (1991) also comment that two other soft shore communities are particularly widespread, occurring on over 80% of the estuaries in Britain which were assessed by the Estuaries Review. Firstly a muddy sand community in more sheltered areas of variable or reduced salinity and secondly a mud community found in intertidal and shallow subtidal parts of estuaries.

The muddy sand communities are dominated by lugworms (*Arenicola marina*) although in intertidal regions, cockles (*Cerastoderma edule*), Baltic tellins (*Macoma balthica*) and several species of polychaete worms (*i.e.* *Pygospio elegans*, *Nephtys hombergii*, *Scoloplos armiger* and *Spiophanes bombyx*) are also abundant. The gastropod snail *Hydrobia ulvae* often occurs on the upper surface of the sediment and on the eelgrass *Zostera noltii* and *Zostera angustifolia*, which may occur in this community in upper parts of estuaries. The mud communities are dominated by clams and polychaetes, typically ragworms, *Nereis diversicolor*, with small gastropod snails (*i.e.* *H. ulvae*) and crustaceans also being abundant.

Hard substrata in the form of natural rocky shores are not a particularly common feature of estuaries in Great Britain. Rocky substrata is generally found in both the intertidal and subtidal zones within rias, fjords and fjards where sediment deposition is restricted. Rocky shore communities are often dominated by the 'wracks' such as *Ascophyllum nodosum* or *Fucus* spp. and appear to be more diverse on estuaries in south-west England and northern Scotland.

Estuaries of major marine conservation interest are currently those with many different aquatic communities, although those containing individual scarce communities are also of interest. Examples of these include the maerl beds that are found in the Fal Estuary, Helford Estuary and Milford Haven or communities that are dominated by razor shells.

**APPENDIX 5: EXAMPLES OF SPECIES WITH VARYING SENSITIVITIES
TO HYPOXIA (DIAZ AND ROSENBERG 1995).**

Species resistant to severe hypoxia	Species resistant to moderate hypoxia	Species sensitive to hypoxia
<i>Artica islandica</i>	<i>Capitella capitata</i>	<i>Diastylis rathkei</i>
<i>Astarte borealis</i>	<i>Abra alba</i>	<i>Nephrops norvegicus</i>
<i>Corbula gibba</i>	<i>Abra nitida</i>	<i>Echinarachnius parm</i>
<i>Ophiura albida</i>	<i>Amphiura filiformis</i>	<i>Brissopsis lyrifera</i>
<i>Halicryptus spinulosus</i>	<i>A. chiajei</i>	<i>Ampelisca agassizi</i>
<i>Malacoceros fuliginosus</i>	<i>Streblospio benedicti</i>	<i>Ampharete grubei</i>
<i>Metridium senile</i>	<i>Mediomastus ambiseta</i>	<i>Macoma calcarea</i>
<i>Phoronis mülleri</i>	<i>Mercenaria mercenaria</i>	<i>Gammarus tigrinus</i>
<i>Ophiodromus flexuosus</i>	<i>Spisula solidissima</i>	<i>Spisula solida</i>
<i>Pseudopolydora pulchra</i>	<i>Lumbrinereis verrilli</i>	<i>Asterias forbesii</i>
<i>Paraprionospio pinnata</i>	<i>Scoloplos armiger</i>	<i>Crangon crangon</i>
<i>Loimia medusa</i>	<i>Asychis elongata</i>	<i>Carcinus maenas</i>
<i>Modiola phaesolina</i>	<i>Nereis diversicolor</i>	<i>Magelona phyllisae</i>
<i>Nephtys hombergi</i>	<i>Pectinaria koreni</i>	<i>Neries pelagica</i>
<i>Ragacitis pulchra</i>	<i>Nereis micromma</i>	
<i>Calliactis parasitica</i>		
<i>Streblospio benedicti</i>		
<i>Goniadella gracilis</i>		
<i>Astarte castanea</i>		
<i>Mytilus edulis</i>		
<i>Munida quadrispina</i>		
<i>Heteromastus filiformis</i>		
<i>Arenicola marina</i>		
<i>Saduria entomon</i>		
<i>Magelona sp. H</i>		

APPENDIX 6: REVIEW OF STUDIES WHICH HAVE COMPARED NUTRIENT IMPACTS IN DIFFERENT ESTUARY SYSTEMS

Several authors have undertaken comparative studies of different estuarine systems on the basis of nutrient levels or trophic status. The results these provide an indication of the differences that have been observed between estuaries and therefore, of the problems that may arise when comparing different systems. Additionally, because these studies have generally required a large amount of data collation, they are often valuable sources of information. Summary details of these projects are shown in Table 10 and brief reviews of the approaches and results are described below.

Balls (1992) assessed inputs of nutrients to the Forth and Tay estuaries and found discernible differences in the nutrient cycling regimes of these two estuaries. Specifically the Forth had the larger turbidity maxima of the two systems and the rates of phosphate removal, bacterial mineralisation and nitrification of organic nitrogen were greater in this estuary.

Balls (1994) compared the riverine nutrient inputs and estuarine processes in nine estuaries including the Forth and Tay. To account for temporal changes, these estuaries were simultaneously sampled (within 3-5 days) on HWS tides. The major differences between estuaries were associated with catchment land uses, which influenced riverine nutrient concentrations, and flushing times which affected the extent to which internal biological processes could modify the nutrient input.

de Jonge et al., (1995) compared the developments in eutrophication in four large North Atlantic systems: the Dutch Wadden Sea, and the Baltic in NW Europe and Chesapeake Bay and Long Island Sound in the USA. It was found that nutrient loads had increased in all systems to varying degrees. With the exception of the Wadden Sea Primary, which has very short flushing times, production had also increased and a good correlation was obtained between annual primary production and the total nitrogen load. Apart from the Wadden Sea, a reasonable proportion of the locally produced organic matter was retained within the system and oxygen depletion occurred due primarily, to pycnocline formation and secondarily, to the mineralisation of the increased organics.

Elliott et al., (1994) assessed the trophic status of the Ore/Alde, Deben, Stour and Colne in Suffolk. All four systems had high freshwater inputs and the potential for nutrients to remain for a long period but all estuaries also had a high tidal exchange volume. Differences between estuaries were observed in the scale of the nutrient inputs and the length of flushing times but no estuary showed any 'well-developed' signs of eutrophication. This was attributed to the high turbidity levels in these areas.

Green et al., (1992) compared 109 estuaries in terms of the level of enrichment and the nature of the overwintering bird populations. They showed that there were four estuary types. One had species which favoured muddy conditions, another had those which favoured sandy shores and there were two further intermediate groups. They also used these procedures to describe changes in communities over time and compare species composition against environmental characteristics, in particular water quality related factors such as nutrient concentrations and DO concentrations. They observed a link between certain estuary characteristics and the species present. Wader community compositions were affected by salinity, ammonium, DO and BOD. The Grey Plover was

more abundant in estuaries with high ammonium concentrations, Sanderling and Purple Sandpiper were lower on estuaries with high DO concentrations and Avocet densities were lower on estuaries with minimum DO concentrations.

Fisher *et al.*, (1988) studied three large estuaries on the eastern seaboard of the USA (Hudson, Delaware and Chesapeake). Using mixing diagrams (nutrient/salinity plots) they described the transformations of nutrients in these estuaries. In the Hudson it was not possible to identify the transformations because of the lateral wastewater inputs from New York. In the other two estuaries there was a removal of nutrients (largely by phytoplankton blooms) but there were between-estuary differences in the specific nutrient fractions that were removed. There were also differences in the degree of nutrient limitation in the high salinity regions of these estuaries. In the Hudson there was no nutrient limitation, in the Delaware there was either slight limitation or no limitation and Chesapeake bay had widespread limitation.

Franklin *et al.*, (1991) compared the behaviour of organic and inorganic phosphorus (DOP and DIP) in four English estuaries (Itchen, Beaulieu, Humber and Thames). In all estuaries had similar range of DOP concentrations from below the limits of detection to a few μm . The relative proportion of DOP, as indicated by the DOP/DTP (Dissolved Total Phosphorus), varied greatly between the estuaries. DIP was dominant in the Itchen and Thames because of anthropogenic inputs while in the Beaulieu and Humber other forms of phosphorus were more important than DIP. In the latter two estuaries, particulate phosphorus was found to be most important because of interactions between dissolved and particulate phosphorus.

Lyons *et al.*, (1993) described the cumulative inputs of nutrients from 39 estuaries on the East Coast of Scotland to the Scottish North Sea Coastal Zone (SNSCZ). They noted that concentrations of nitrogen and phosphorus were greater in lowland areas than the highlands to the north of Scotland due to the differences in the soil composition. Concentrations from lowland areas were highly variable due to differences in the intensity of agricultural activity in the different catchments.

Mathieson and Atkins (1995) reviewed the TON concentrations in 43 Scottish estuaries and, as with Lyons *et al.*, (1993) the riverine TON inputs were related to catchment with low inputs from upland catchments with agriculture-free catchments and higher values in the lowland estuaries on the east coast and south-west coasts. They did however observe considerable variation in TON inputs even for geographically close sites.

McArthur and Johnson (1992) reviewed data from 12 east coast estuaries for the purposes of designating them under the UWWT and Nitrates directives. They analysed chlorophyll-*a*, conductivity, pH, DO, turbidity and a suite of nutrients in each of these estuaries. Although they proposed the designation of the Colne and Deben estuaries, Breydon water and Blyth, Blakney and Wells Harbours under the nitrates directive, they noted that these sensitive areas showed no evidence of any consistent causal characteristics. For instance the Ore and Alde had high residence times yet it was the Deben and Colne that exhibited 'more acute eutrophication'. There are similar STW inputs to the Orwell, Stour, Colne and Deben yet the former two showed no symptoms of eutrophication.

Table 10: Summary details of studies which involved the comparison of estuaries systems.

Reference	No. Estuaries	Site/Country	Research Subject/ Data collected
Balls (1992)	2	Scotland	nutrient behaviour in Forth and Tay
Balls (1994)	9	Scotland	nutrient inputs to estuaries, nutrients in surface waters and water quality data.
de Jonge <i>et al.</i> , (1995)	4	North Atlantic	nutrients loads and symptoms of eutrophication
Elliott <i>et al.</i> , (1994)	9	Suffolk	Trophic Status
Green <i>et al.</i> , 1992	109	UK	Organic pollution and waterfowl populations
Fisher (1988)	3	USA East Coast	Phytoplankton, nutrients and turbidity
Franklin <i>et al.</i> 1991	4	England	Behaviour of phosphorus
Lyons <i>et al.</i> (1993)	39 (riverine inputs)	Scotland East Coast	Fluxes to estuaries and coastal zone, Nitrate, phosphate, silicate and Flow.
Mathieson and Atkins (1995)	43 (riverine inputs)	Scotland	TON
McArthur and Johnson (1992)	12	East Anglia	Biological and Chemical Data to designate estuaries under UWWT and Nitrates directives.
Parr and Wheeler 1995	12 SMAs	England	Biological and Chemical Data
Villate <i>et al.</i> 1991	5	Spain	Hydrography and Seston
Wulff and Ulanowicz 1989	2	Chesapeake and Baltic	Ecosystem Structure

Parr and Wheeler (1994) reviewed the literature and nutrient data for 12 Sensitive Marine Areas identified by English Nature with the aim of demonstrating links between nutrient loads and ecological impacts. In none of these SMAs was it possible to clearly identify trends in nutrient loading or cause/effect relationships between nutrient status and ecology due to the variability of the environments and of the sampling and analysis protocols.

Villate *et al.*, (1991) conducted a comparative study of hydrography and seston in five estuarine systems of the Basque Country (Bay of Biscay). In Autumn SPM related to allochthonous inputs from rivers and bottom sediments. SPM was smaller in three estuaries with high freshwater flows. In the other two resuspended materials seem to have been a major contributor. In summer SPM was mostly derived from plankton blooms (autochthonous) which were greater in the less-stratified estuaries.

Wulff and Ulanowicz (1989) compared the ecosystem structures of Baltic and Chesapeake Bay and showed that the former system was trophically more efficient. As a result it was concluded that nutrient inputs were more likely to contribute to an increase in productivity in the Baltic. In Chesapeake Bay however, where the productivity of the upper trophic levels have already begun to decline, nutrient inputs were less likely to result in greater productivity and instead may have a detrimental effect on the outputs of the system.

APPENDIX 7: SUMMARY DETAILS OF THE ENVIRONMENT AGENCY'S PROPOSED GQA SCHEME FOR BENTHIC COMMUNITIES.

GQA: A measure of ecological quality based on benthic invertebrates

Codling *et al.*, (1995) evaluated the usefulness of macrobenthic communities as indicators of the 'general quality' of estuaries and coastal waters within the EA's GQA scheme. For this study they assessed the value of various statistical descriptors of community structure that could be employed in such a scheme. They observe that there are numerous indicators of community-type but that for the purposes of a GQA scheme they noted that the selected indicators requires the following characteristics:

- 1) Predictable response to pollution/disturbance.
- 2) Either a predictable response, or an insensitivity, to natural environmental changes.
- 3) Cost effective to derive.

There was some evidence that the primary community variables of: species number (S), organism abundance (A) and biomass (B) could be used as descriptors of macrobenthic community structure in estuaries. The generalised response of these SAB statistics to organic enrichment was described by Pearson and Rosenberg (1978). However, according to Codling (1995) this relationship would only be applicable on a limited spatial scale because on an estuary-wide basis salinity variations would have a significant additional impact.

SAB statistics have been used in Chesapeake Bay by Dauer (1993) who developed biological criteria along with more specific measures of the biomass of animals below 5cm and the biomass of 'equilibrium' species typical of estuarine conditions. These parameters were then plotted against salinity to graphically show the changes down the estuary. This approach however, had the benefit of a large dataset of benthic community information for this estuary and therefore, accounted for the influence of salinity.

Codling *et al.*, (1995) concluded that for estuaries there is insufficient accompanying data on environmental conditions to demonstrate the necessary insensitivity of SAB statistics to environmental change and that the amount of work necessary to adapt such a scheme for most estuaries would be prohibitive.

Another approach is the use of predictive models multiple regression models to predict univariate measurements from environmental variable (Elliott and O'Reilly 1991). This approach has an added benefit as a potential means of determining background conditions because if accurate correlations can be made between environmental variables and biological conditions, historical changes in community structure can be predicted from historical changes in physical conditions.

Codling *et al.*, (1995) also referred to two newly available univariate measures (often at phylum level) which can be employed as indicators of disturbance.

Biological Quality Index BQI: Designed for use specifically in estuaries by Wilson and Jeffrey (1987). This is based on SAB statistics with the ultimate designation of the communities being either abiotic (A), opportunistic (B) or stable (C). A single value index

is obtained by initially separating the intertidal areas on sediment types and the percentage cover of communities A, B and C (such that $A+B+C=1.0$). Then the single value BQI is obtained from the equation $BQI = \text{antilog}_{10}(C-A)$ which has a range 1-10. A BQI of <1 represents poor conditions and a BQI >1 represents good conditions. Elkaim *et al.*, (1994) studied estuaries on the English Channel and northern France and considered that BQI and the Pollution load Index (PLI) were the most appropriate indices.

Benthic Index: Developed by the USEPA this requires that suitable reference and degraded sites are selected based on the responses to sediment toxicity tests (Effects Range Median ER-M values) and sediment water quality criteria (Dissolved Oxygen Concentration). From these reference and degraded stations 28 ecological descriptors were identified which were tested to ensure that they showed no significant correlation with environmental variable. The next task was to obtain a linear combination of these 28 descriptors that best discriminates between degraded and reference sites and that explains the greatest proportion of the 'explainable variation' and also has some ecological relevance. With this aim two equations have been developed:

Discriminant Score = -0.68 (mean abundance (A) of opportunists) + 0.36 (B:A for all species) + 1.14 (mean S/grab). (Schimmel *et al.*, 1994).

Discriminant Score = 2.38 (proportion of expected diversity) + 1.67 (proportion of total A of Tubificids) + 0.668 (proportion of total A of bivalves).

The authors also reviewed the use of ABC plots but they could not recommend this approach for use as a GQA because r-strategist polychaetes are naturally present in physically dynamic estuaries, and because ABC is designed to reflect changing species (with accompanying changing biomass) rather than changing sizes of individuals of a species.

Multivariate statistics were also considered and specifically the role of 'Meta analysis', a new technique which takes account of the two problems with multivariate techniques which are their: sensitivity to differences in species compositions and the lack of a value judgement on the observed results. Warwick and Clarke (1993) tested this approach on available data and found that there was no relationship between the meta-analysis and environmental variables and that the differences between samples along a pollution gradient were driven by the distribution of the 5 phyla. They also tested phylum-level production for the same samples and observed a very good correlation and probably a better indicator than abundance.

The conclusion of this review was that available data were insufficient to determine the most suitable community descriptors. It was recommended that further data should be collected from a selection of estuaries to assess the usefulness of existing and new univariate measures.

APPENDIX 8: DETAILS OF THE COMPREHENSIVE STUDIES DEVELOPED BY THE COMPREHENSIVE STUDIES TASK TEAM (CSTT 1993).

The UK methodologies for the application of the Urban Waste Water Treatment Directive (UWWTD) are detailed in the Comprehensive Studies Task Team Report (CSTT 1993). As the directive does not define adverse effects, the CSTT report aims to offer guidance to dischargers and regulators on the objectives, standards and methodology that must be applied in undertaking comprehensive studies in order to ensure that no adverse effects will occur from discharged waste water.

The CSTT report includes guideline standards that can be used to define the critical levels for a number of environmental parameters as well as the levels of acceptable change for these variables and models to predict future effects due to increased nutrient inputs. In this report estuaries and coastal waters are considered separately. In estuaries, where only small discharges qualify, dissolved oxygen (DO) standards were derived, in coastal water however, standards were derived for DO, nutrients, chlorophyll-*a*, and organic enrichment on the benthos. A range of criteria and acceptable levels of change in response to nutrient inputs are identified.

For estuaries the median DO which at 7mg l⁻¹ was considered to be at a safe level and a change of <1mg l⁻¹ from this level is considered acceptable. However, where the median is less than 7mg l⁻¹ then further study is required to determine critical levels of change and each case needs to be considered on an individual basis. In coastal waters a median DO of 7mg l⁻¹ was again considered safe and a change of <0.5mg l⁻¹ was considered to be acceptable. For nutrient levels, hypereutrophication was considered to exist where summer nutrient concentrations exceed 12mmol Dissolved Available Inorganic Nitrogen (DAIN) in the presence of >0.2mmol Dissolved Available Inorganic Phosphorus (DAIP). Although hypereutrophication should not be seen as a problem in itself it may become so when a substantial proportion of the nutrients are converted to algae. Eutrophic conditions were considered to exist where chlorophyll-*a* concentrations regularly exceed 10mg m⁻³ however, only treatment-related increases in the predicted maximum of >1mg m⁻³ were considered to be significant impacts.

The impacts of discharged suspended solids are measured by the change that occurs to the benthos due to organic enrichment. The acceptable levels of change within the sphere of influence of the discharge are: the total abundance is +200% of a reference station, the total taxa is +50% of a reference station and/or the biomass is +50% of a reference station. There is evidence also that the Infaunal Trophic Index (ITI) could be useful for describing an adverse effect.

As well as these standards levels the CSTT report also includes guidelines for predicting the trophic state of the waters surrounding the waste discharge based on nutrient concentrations and phytoplankton bloom dynamics. The first step in the analyses is an assessment of the potential steady state nutrient concentrations using the following 'mixed-box' model:

$$S = S_0 + ((S_i + S_d))/E.V$$

Where:

- S_o nutrient concentration in adjacent seawater
- S_i total local inputs from sources other than the discharge under investigation
- S_d nutrient input from the discharge under consideration
- E is the relative water exchange measured in days defined by: $E = (\Delta V/\Delta t) \cdot (1/V)$ where V is the volume of the dispersion zone and $\Delta V/\Delta t$ is the mean rate of volume exchange. E can be estimated using alt budgets or numerical models for worst case conditions and, in the absence of hydrographic data $E=0.1 \text{ d}^{-1}$ can be used.

The calculations should be carried out in late winter with S_o measured in February before the spring phytoplankton bloom depletes the nutrients. The calculations for present and future nutrient discharges with and without secondary treatment (S_d) should relate to mid-summer conditions.

Hypernutrification may be considered to exist when the predicted winter values of S exceed $12 \text{ mmol DAIN m}^{-3}$ in the presence of $0.2 \text{ mmol DAIP m}^{-3}$. Anthropogenic hypernutrification exists where at least $5 \text{ mmol DAIN m}^{-3}$ of the nutrient concentration can be related to human activities.

The value of S can be incorporated into another model to estimate the maximum biomass of phytoplankton. The maximum biomass, which is considered here to represent the 'worst-case' for potential eutrophication, is predicted by assuming that all available nutrients are assimilated by phytoplankton. The following equation is used:

$$X_{\max} = X_o + q \cdot S \text{ mg m}^{-3}$$

where:

- X_o is the concentration of phytoplankton chlorophyll- a in the surrounding water
- q is the yield of phytoplankton from nutrients 1.1 to $2.8 \text{ mg chlorophyll (mmol DAIN)}^{-1}$, 50 - $100 \text{ mg chlorophyll (mmol DAIP)}^{-1}$.

This calculation should be done for both early spring and mid-summer.

Further analyses were also described for assessing whether conditions are suitable for phytoplankton growth by computing the relative rate of light-controlled growth (μ). This measure is based on the relationship between light conditions and the photosynthetic efficiency of phytoplankton. The potential maximum phytoplankton can be realised where $\mu > E+L$ where L is the relative loss rate of phytoplankton due to grazing.

Therefore a region is potentially eutrophic if $\mu > E+L$ and summer $X_{\max} > 10 \text{ mg chl m}^{-3}$

Further recommendations for monitoring, testing and evaluating these conclusions are also included in the CSTT report.

APPENDIX 9: DOE CRITERIA FOR DEFINING EUTROPHICATION IN ESTUARINE AND COASTAL WATERS

Eutrophication is defined in the Urban Waste Water Treatment Directive (91/271/EEC) as 'the enrichment of waters by nutrients, especially compounds of nitrogen and/or phosphorus, causing accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the waters and to the quality of the waters concerned'. The UK Government (DoE 1993) expanded the definition to include the following criteria for estuarine and coastal waters:

1. **Nitrate Concentrations:** Winter (February) nitrate-nitrogen concentrations significantly enhanced relative to a background concentration for a defined geographical area based on salinity.
2. **Occurrence of exceptional algal blooms:** Attention should be given to the occurrence of unusual blooms of phytoplanktonic species or blooms of unusual scale or with unusual toxicity characteristics. It should be noted that blooms of algae in coastal waters normally reach densities of 5×10^5 cell per litre, and chlorophyll-*a* concentrations of around 10 mg m³.
3. **Duration of Algal Blooms:** It could be considered exceptional if the normal spring bloom algal densities persisted through the summer until the autumn bloom without the typical nutrient-limited decline in the summer.
4. **Oxygen Deficiency:** Attention should be given to decreased oxygen concentration at the surface, as well as in deeper water layers, including areas where sedimentation and/or stratification may occur.
5. **Changes in Fauna:** Substantial increases or decreases in benthic biomass, shifts in species composition and mortality of benthos and fish.
6. **Changes in Macrophyte Growth:** These can be relatively minor, such as the disappearance of red algal species, or a reduction in depth of the photic zone, or more significant, *e.g.* dense and widespread growth of *Enteromorpha* spp.
7. **Occurrence and Magnitude of Paralytic Shellfish Poisoning (PSP):** The occurrence of PSP-causing species (*e.g.* *Alexandrium* sp.) is endemic in areas around the UK coast even where there is no nutrient enrichment and blooms of varying significance occur each year. However, their scale may be enhanced by nutrient enrichment, extending the duration and geographic area of effect. Such an extension could indicate eutrophication but could also be due to a variety of natural causes.
8. **Formation of Algal Scums on Beaches and Offshore:** Dense blooms of colonial or chain-forming species (*e.g.* *Phaeocystis*, *Chaetoceros*) can result in drifts of cells on the strand-line. The significance of these phenomena should be placed in a historical perspective as such phenomena have been regularly recorded in some UK coastal waters for over 100 years.

APPENDIX 10: DRAFT MONITORING GUIDELINES PRESENTED BY THE UK TO THE OSPAR CONVENTIONS FOR THE PREVENTION OF MARINE POLLUTION AD HOC WORKING GROUP ON MONITORING (1995).

The UK's draft guidelines for monitoring eutrophication are presented in Agenda item 8 of the 'Oslo and Paris Conventions for the Prevention of Marine Pollution: ad hoc working group on monitoring'. In this article the following five eutrophication parameters monitoring were identified: nutrients, chlorophyll-*a*, phytoplankton species composition, benthos and oxygen in water. The literature which detail the accepted practices for monitoring these five parameters were noted and a few potential inconsistencies in recording techniques were highlighted.

Nutrients: Sampling practices are described in the ICES documents (CM 1991/Poll:4 Ref. C+E Sect 1-6) and the principles for the determination of nutrients are outlined in 'Nutrients: Practical notes on their determination in Seawater' by Kirkwood (ICES 1996). The analytical procedures in these documents are for: nitrogen (nitrate, nitrite, ammonium, total dissolved nitrogen, total nitrogen and particulate nitrogen); phosphorus (orthophosphate, total dissolved phosphorus and particulate phosphorus); silicon (dissolves reactive silicate); and supporting information on temperature, salinity and total suspended load. It was recommended that data submitted should be accompanied by QA information based on participation in ICES and QUASIMEME intercomparison exercises.

Chlorophyll-*a*: This is an indirect indicator of phytoplankton biomass. The recommended procedures are detailed in Tett (1987) and the HELCOM guidelines. There is some debate about the best method but chlorophyll-*a* can only be distinguished by liquid-chromatographic techniques. The results of whole pigment extracts should be considered as indicators of chlorophyll plus chlorophyllide. The procedures for the intercomparison of chlorophyll-*a* determinations are not well developed.

Phytoplankton species composition: There is a need to consider the hydrography of the survey area to account for potential vertical, horizontal (spatial) and temporal variation to establish the frequency and location of sampling. As for the chlorophyll-*a* measurements principles and practices are outlined in Tett (1987) and the HELCOM guidelines.

Benthos: There are two relevant ICES reports, one deals with the methodology for the treatment and collection of samples of soft-bottom macrofauna (Rumohr 1990) and the second is concerned with approaches to monitoring communities at point source discharges (Rees et al., 1991). Also valuable is Holme and McIntyre (1984), Gray *et al.*, (1992) and Kramer *et al.*, (1994) for sampling tidal estuaries.

In addition, Rees et al., (1990) considers schemes for judging the 'acceptability' of changes in benthic communities and MAFF (1993) assessed a range of community measures for deriving acceptable standards for change in response to sewage-sludge disposal based on the Pearson and Rosenberg (1978) model of enrichment.

Oxygen: Principles and practice are outlined in Grasshoff (1983).

APPENDIX 11: PRELIMINARY DESCRIPTION OF THE FRAMEWORK FOR THE ASSESSMENT OF BIOLOGICAL INFORMATION TO BE CARRIED OUT DURING STAGE 2 OF THE MONITORING PROTOCOL.

Symptom	Parameter	L	I	M	H	Technique					Accuracy				
	blooms present					skilled eye									
	species comp'					microscopy									
	amount					counts									
	productivity					C14 analyses etc.									
Benthic	Presence					Skilled eye					Qualitative				
Micro/macrophankton	Cover					Aerial photography					Semi-Quantitative				
	Thickness					Groundtruthing					Quantitative				
	Dry Weight/m2														
Macrophytes	Presence					Skilled eye					Qualitative				
	Cover					Aerial photography					Semi-Quantitative				
	Thickness					Groundtruthing					Quantitative				
	Dry Weight/m2					CASI									
Organic enrichment						Occurence					Qualitative				
						Total Load					Semi-Quantitative				
						Carbon Budget					Quantitative				
Macrofauna	Presence										Qualitative				
	Abundance										Semi-Quantitative				
	Community										Quantitative				
	Biomass														
Fish/Birds	Presence										Qualitative				
	Abundance										Semi-Quantitative				
	Community										Quantitative				
Level of percieved risk	L = Low					M = Medium									
	I = Intermediate					H = High									

APPENDIX 12: PRELIMINARY DESCRIPTION OF THE FRAMEWORK FOR THE ASSESSMENT OF PHYSICAL INFORMATION TO BE CARRIED OUT DURING STAGE 3 OF THE MONITORING PROTOCOL.

Symptom	Parameter	L	I	M	H	Technique					Accuracy				
Size	length					Literature Review					Qualitative				
	width					Mapping/GIS					Semi-Quantitative				
	Surface Area					Aerial image					Quantitative				
	Volume					Survey									
Stratification	Mixed layer depth					Literature Review					Qualitative				
						Mapping/GIS					Semi-Quantitative				
						Single Survey					Quantitative				
						Multiple Survey									
Tide	Range					Literature Review					Qualitative				
	Prism					Mapping/GIS					Semi-Quantitative				
	HW volume					Single Survey					Quantitative				
	Midtide volume					Multiple Survey									
Freshwater Flow	River Discharge					Literature Review					Qualitative				
						Mapping/GIS					Semi-Quantitative				
						Single Survey					Quantitative				
						Multiple Survey									
											Qualitative				
											Semi-Quantitative				
											Quantitative				
											Qualitative				
											Semi-Quantitative				
											Quantitative				
Level of perceived risk	L = Low					M = Medium									
	I = Intermediate					H = High									

APPENDIX 13: PRELIMINARY DESCRIPTION OF THE FRAMEWORK FOR THE ASSESSMENT OF NUTRIENT INPUT INFORMATION TO BE CARRIED OUT DURING STAGE 3 OF THE MONITORING PROTOCOL

Symptom	Parameter	L	I	M	H	Technique					Accuracy				
Diffuse loads (N & P)	FSI conc					Single survey					Qualitative				
	Seasonal Variation					Multiple Survey					Semi-Quantitative				
	Diurnal/tidal variation					Continuous Monitor					Quantitative				
						Catchment survey									
Point sources (N & P)	DIN, DIP					Single Survey					Qualitative				
						Multiple Survey					Semi-Quantitative				
						Continuous Monitor					Quantitative				
Total Input (N & P)	DIN, DIP					Single Survey					Qualitative				
						Multiple Survey					Semi-Quantitative				
						Continuous Monitor					Quantitative				
Ambient Water	DIN					Single Survey					Qualitative				
Nutrient Levels	DIN, DIP, DIS					Multiple Survey					Semi-Quantitative				
						Continuous Monitor					Quantitative				
						Diatom-Trophic Index									
											Qualitative				
											Semi-Quantitative				
											Quantitative				
											Qualitative				
											Semi-Quantitative				
											Quantitative				
Level of perceived risk	L = Low					M = Medium									
	I = Intermediate					H = High									

APPENDIX 14: FIGURES

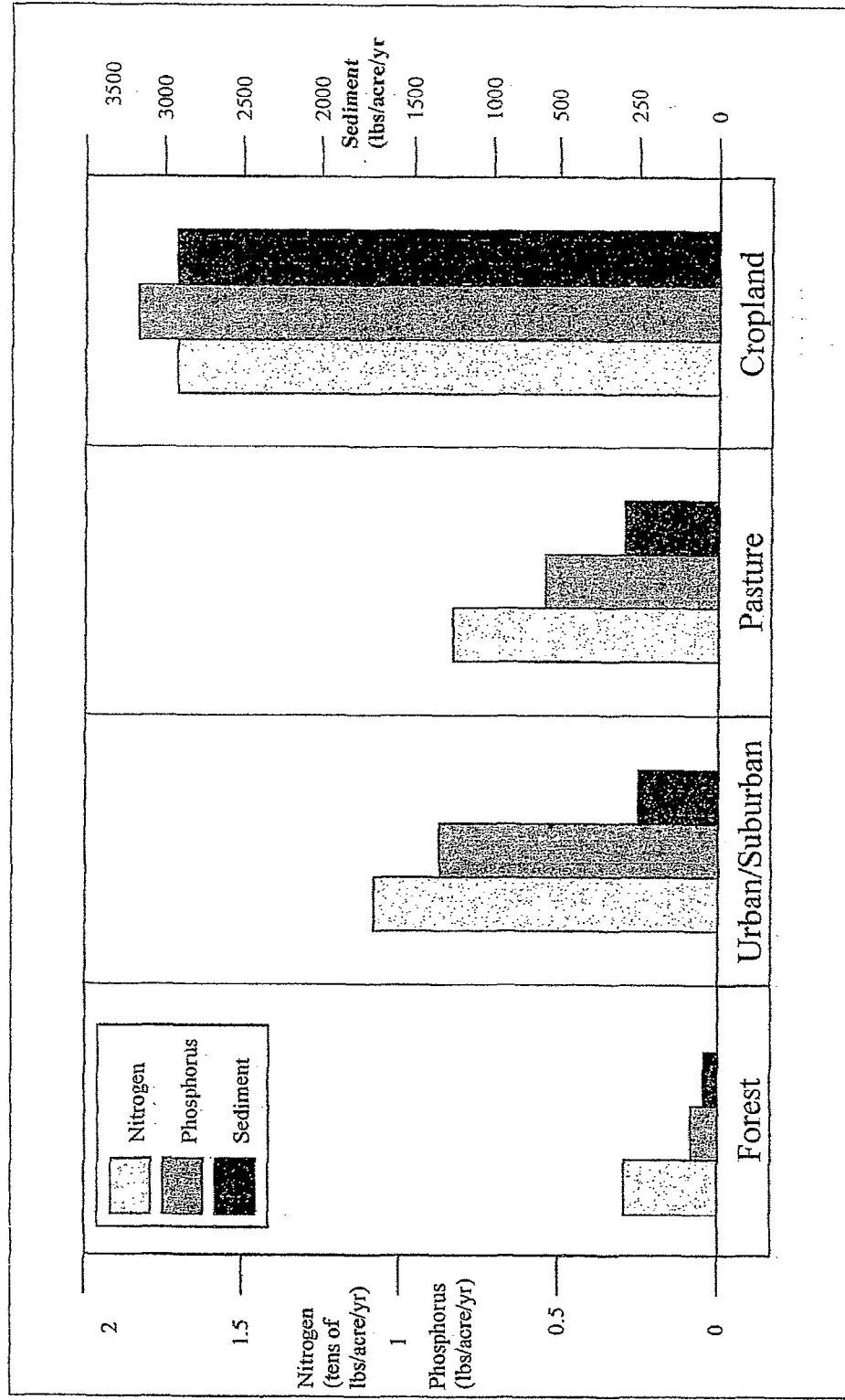


Figure 1: Nutrient and Sediment Pollution entering Chesapeake Bay from areas of the catchment with different land-uses (Magnien et al., 1995)

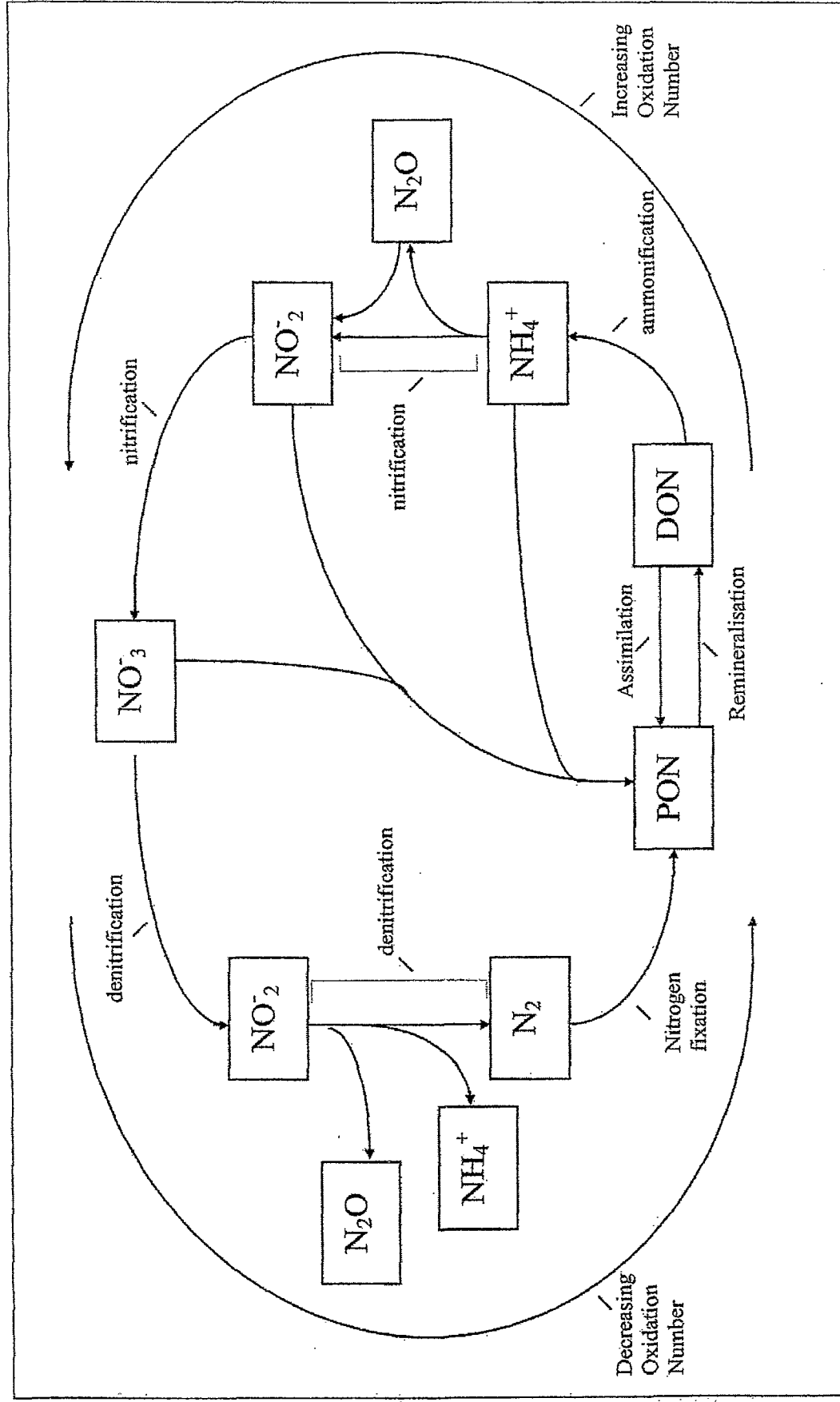


Figure 2: The Nitrogen Cycle (Libes 1992)

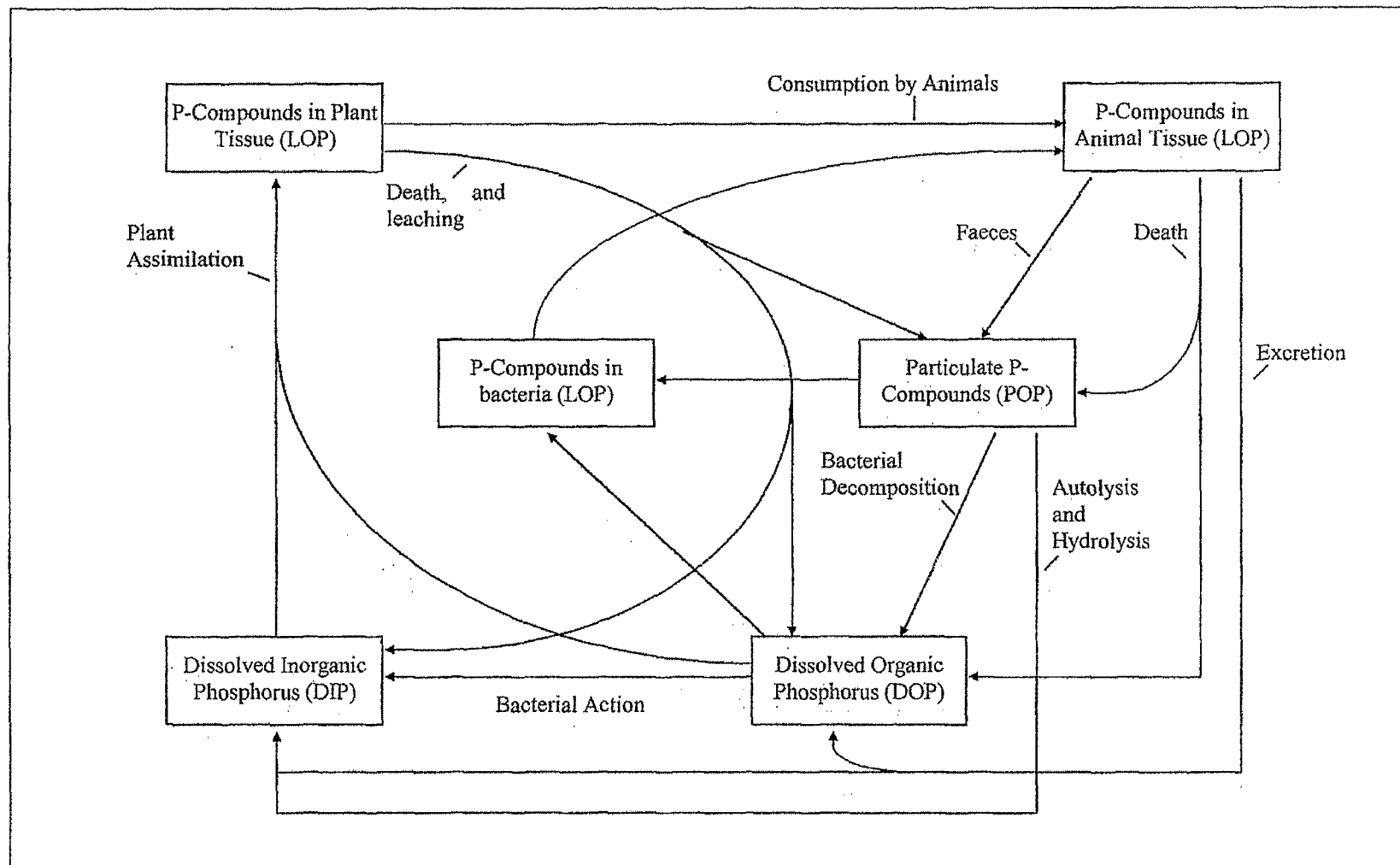


Figure 3: The Marine Phosphorus Cycle (Tait 1981)

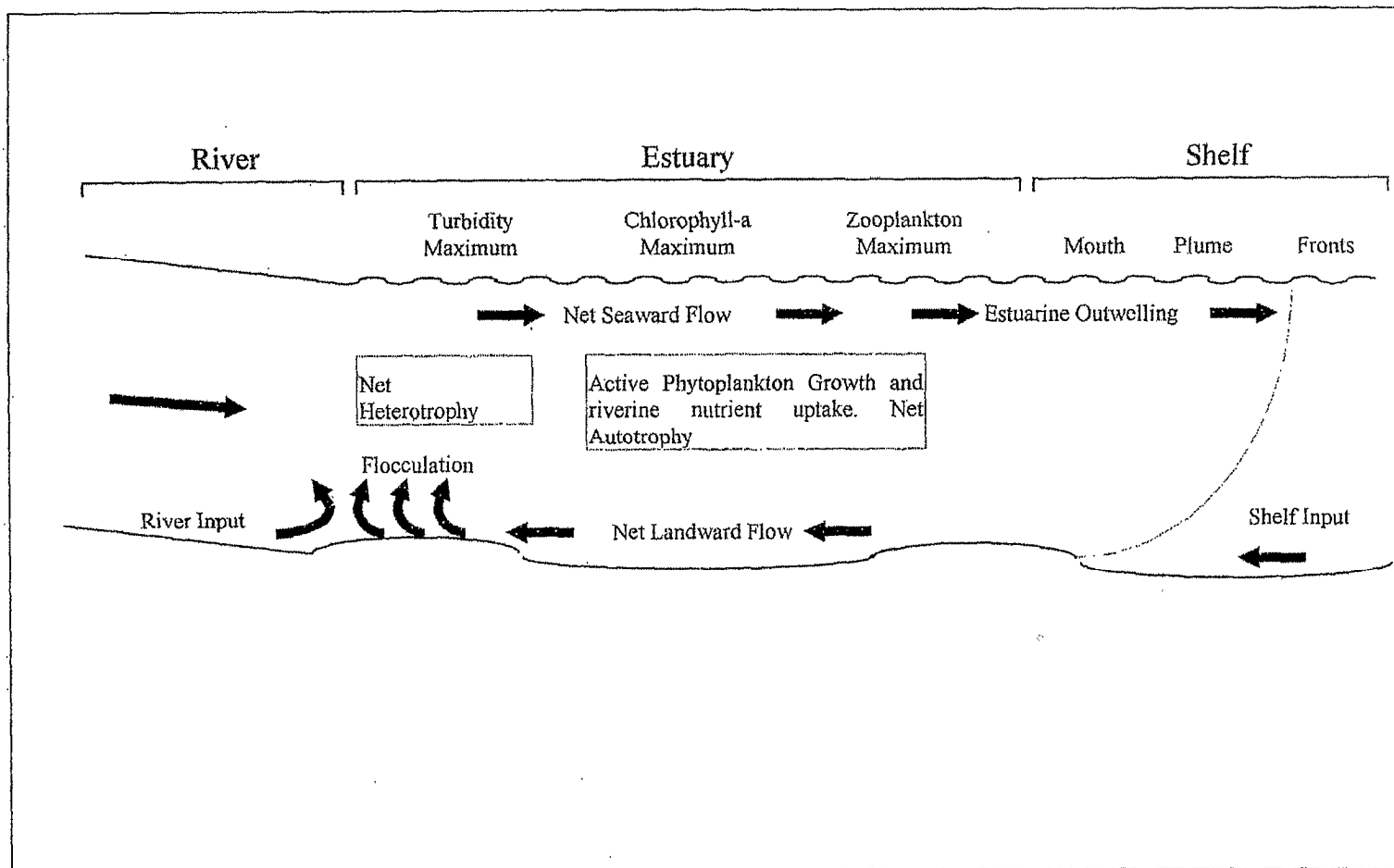


Figure 4: Conceptual model of the spatial changes along the salinity gradient of deep slow flushing estuaries (Fisher et al., 1988).

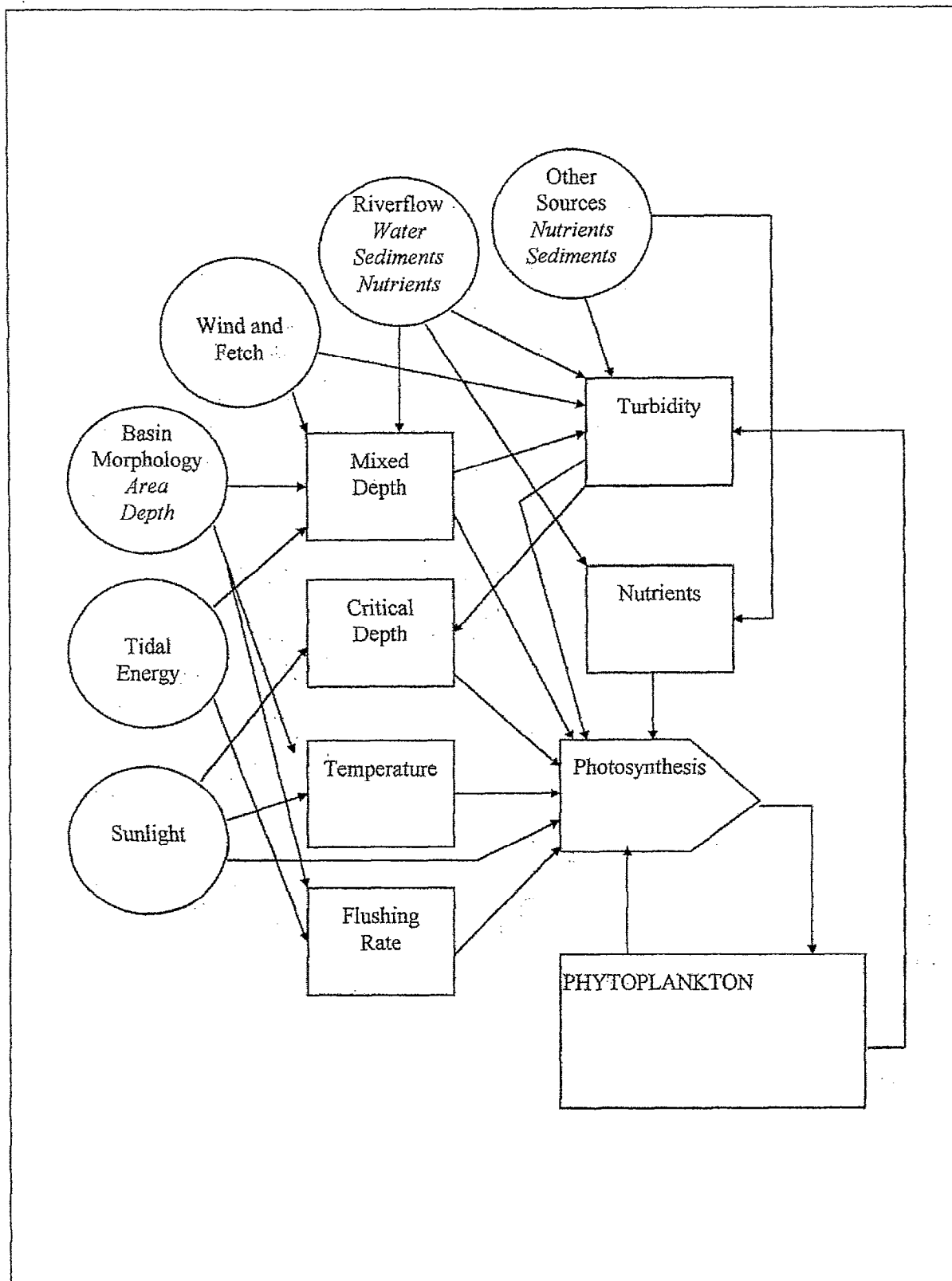


Figure 5 A conceptual model describing the factors influencing phytoplankton production in estuaries (adapted from Boynton et al., 1982)

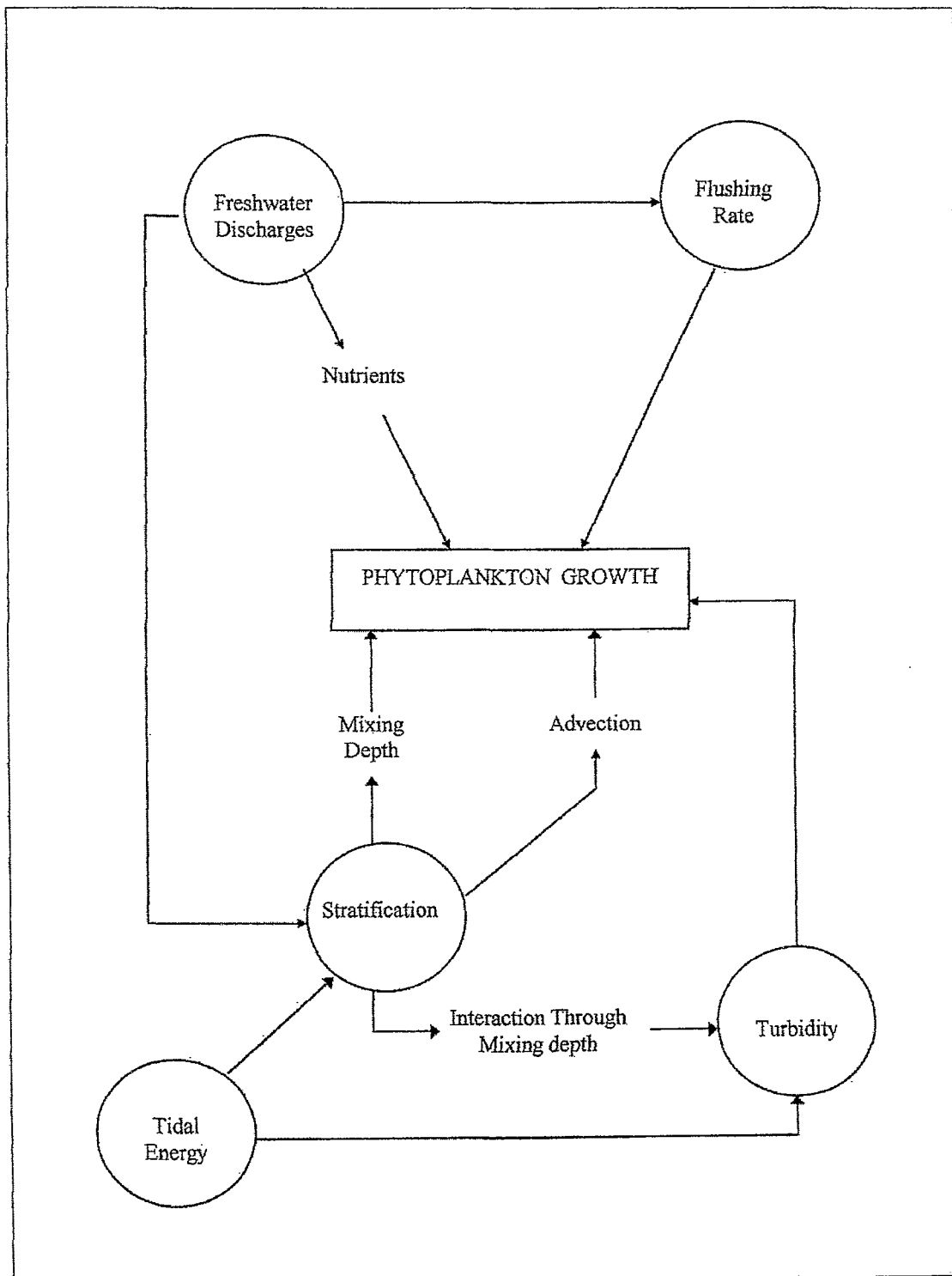


Figure 6 A conceptual model describing the factors influencing phytoplankton production in estuaries (Wilkinson unpubl.)

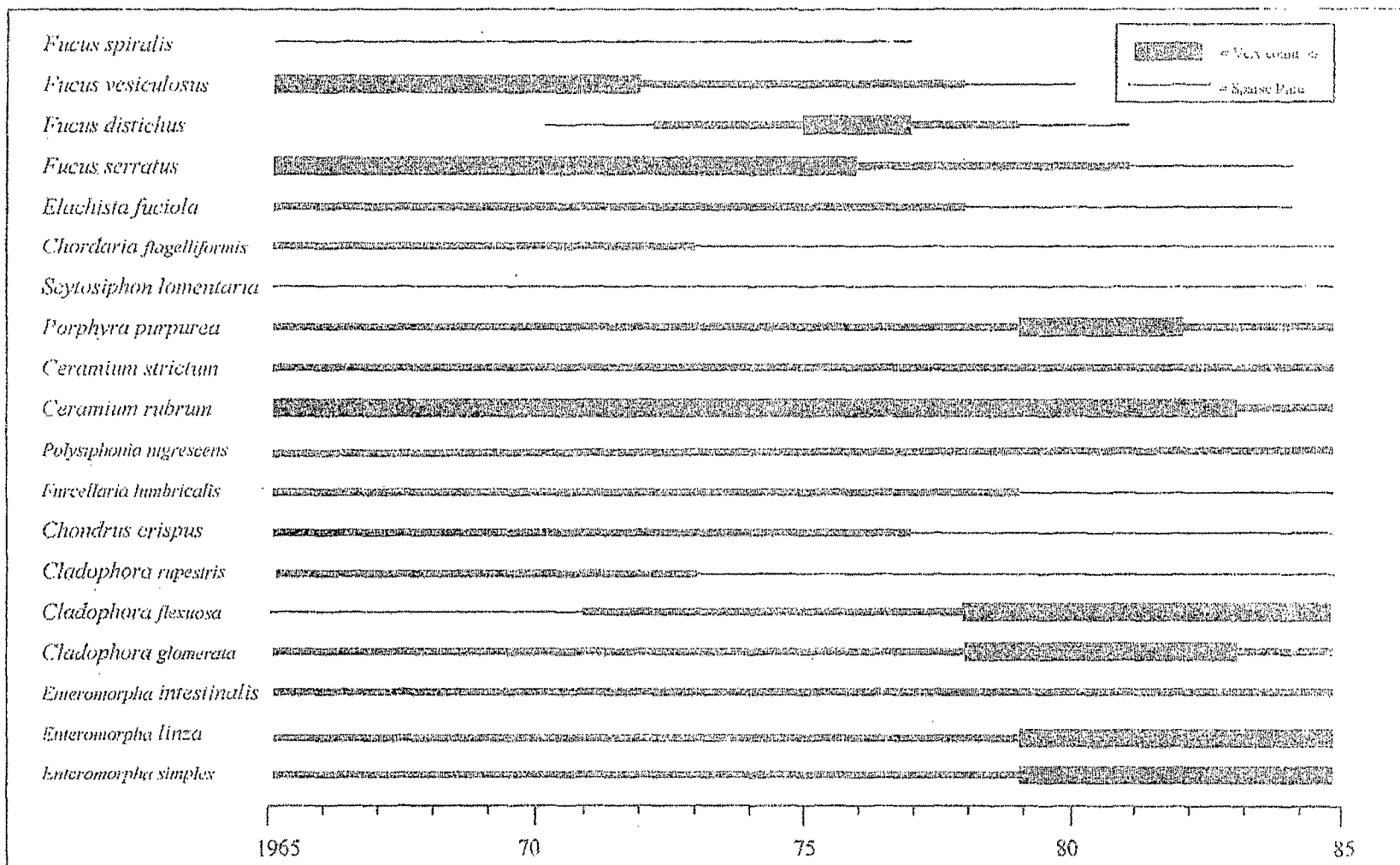


Figure 7: Changes in abundance of macroalgae in Laholm Bay (Kattegat) attributed to nutrient enrichment (Baden et al., 1990).

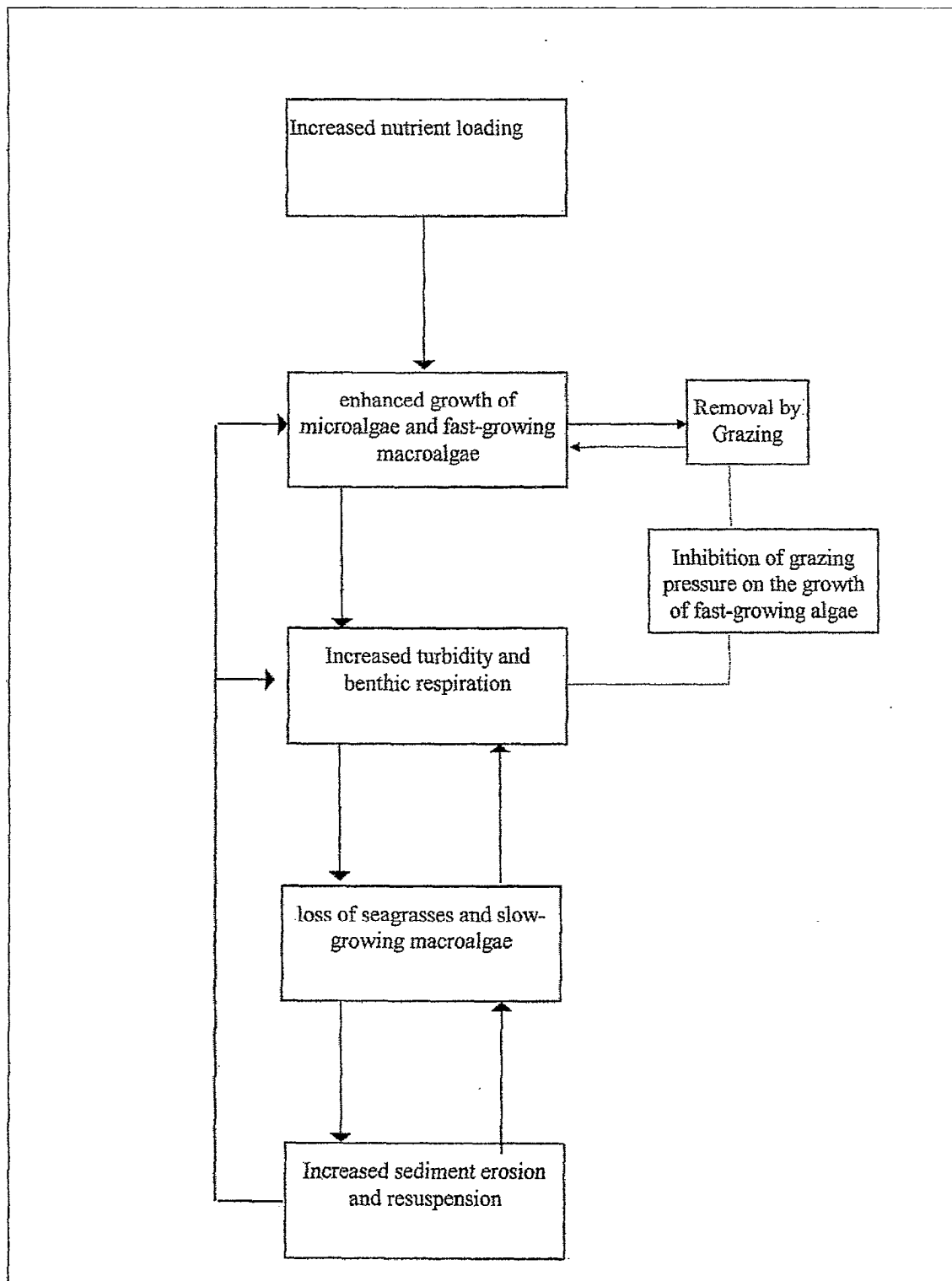


Figure 8: Schematic description of the self-perpetuating process by which nutrient enrichment results in enhanced algal growth (adapted from Duarte 1995)

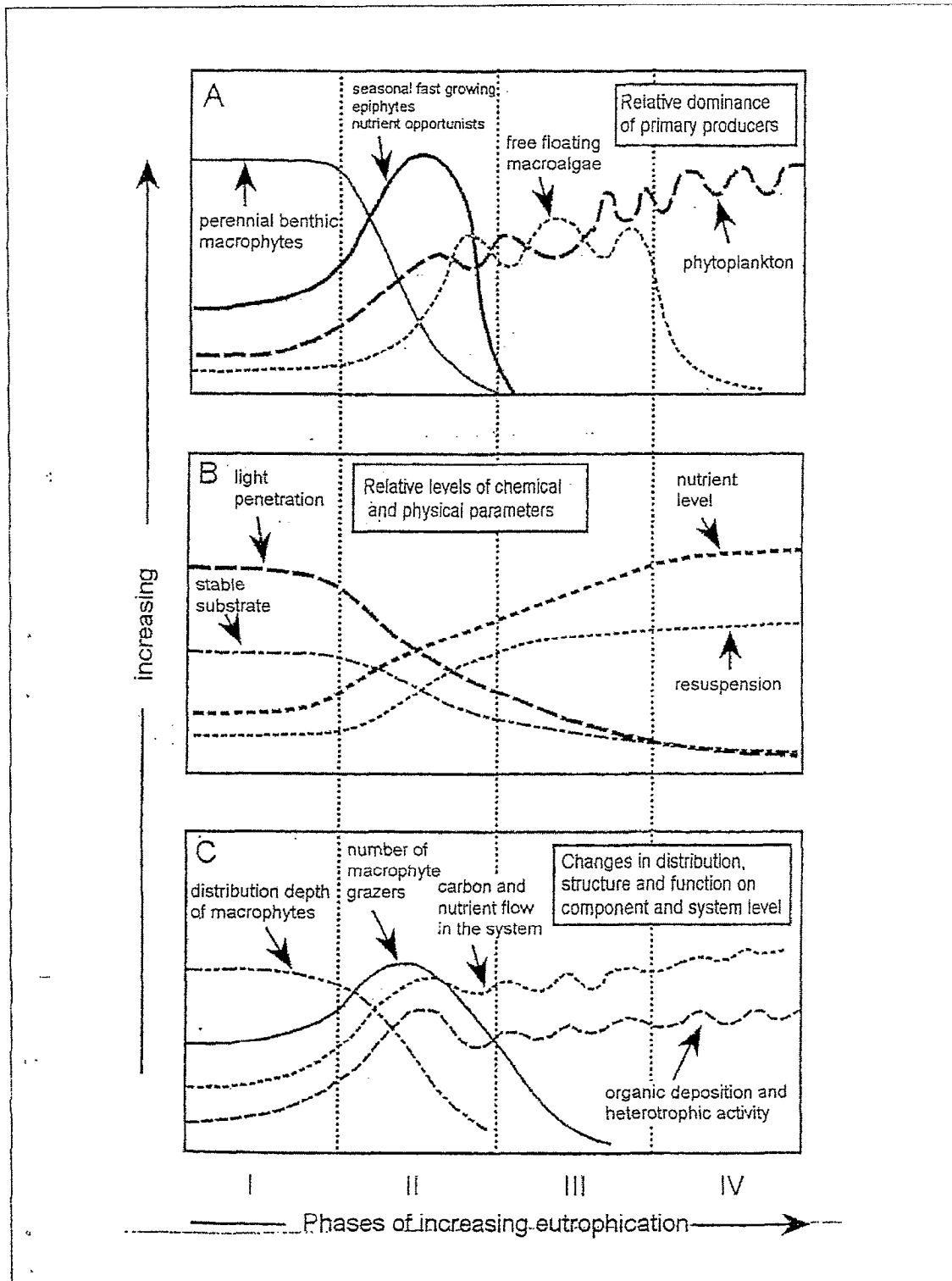


Figure 9: Schematic description of the changes to primary producers and related parameters during successive phases of eutrophication (Schramm 1996).

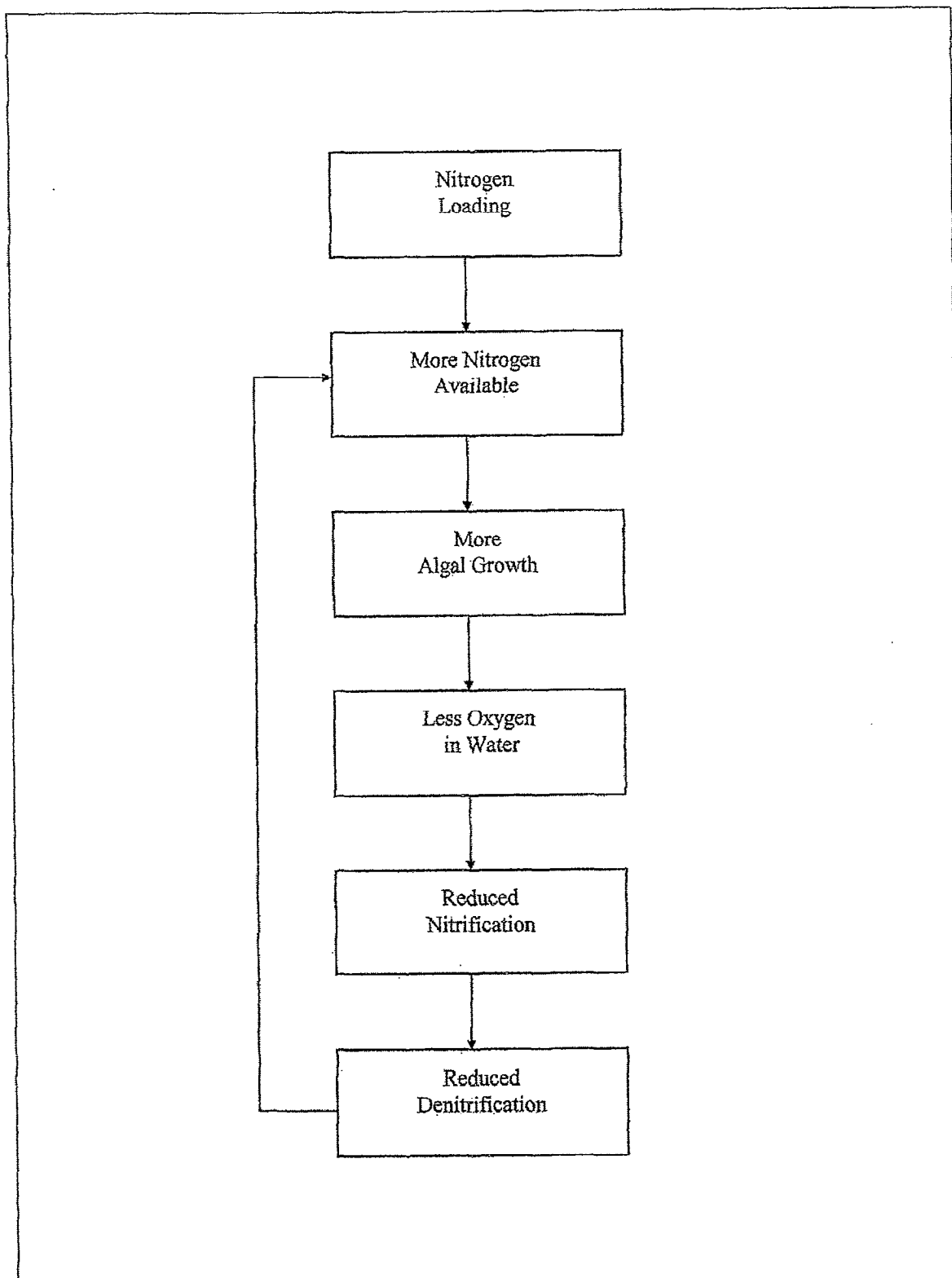


Figure 10: Schematic description of the self-perpetuating processes by which an uncoupling of nitrification and denitrification drives further nutrient enrichment (Harding et al., 1992).

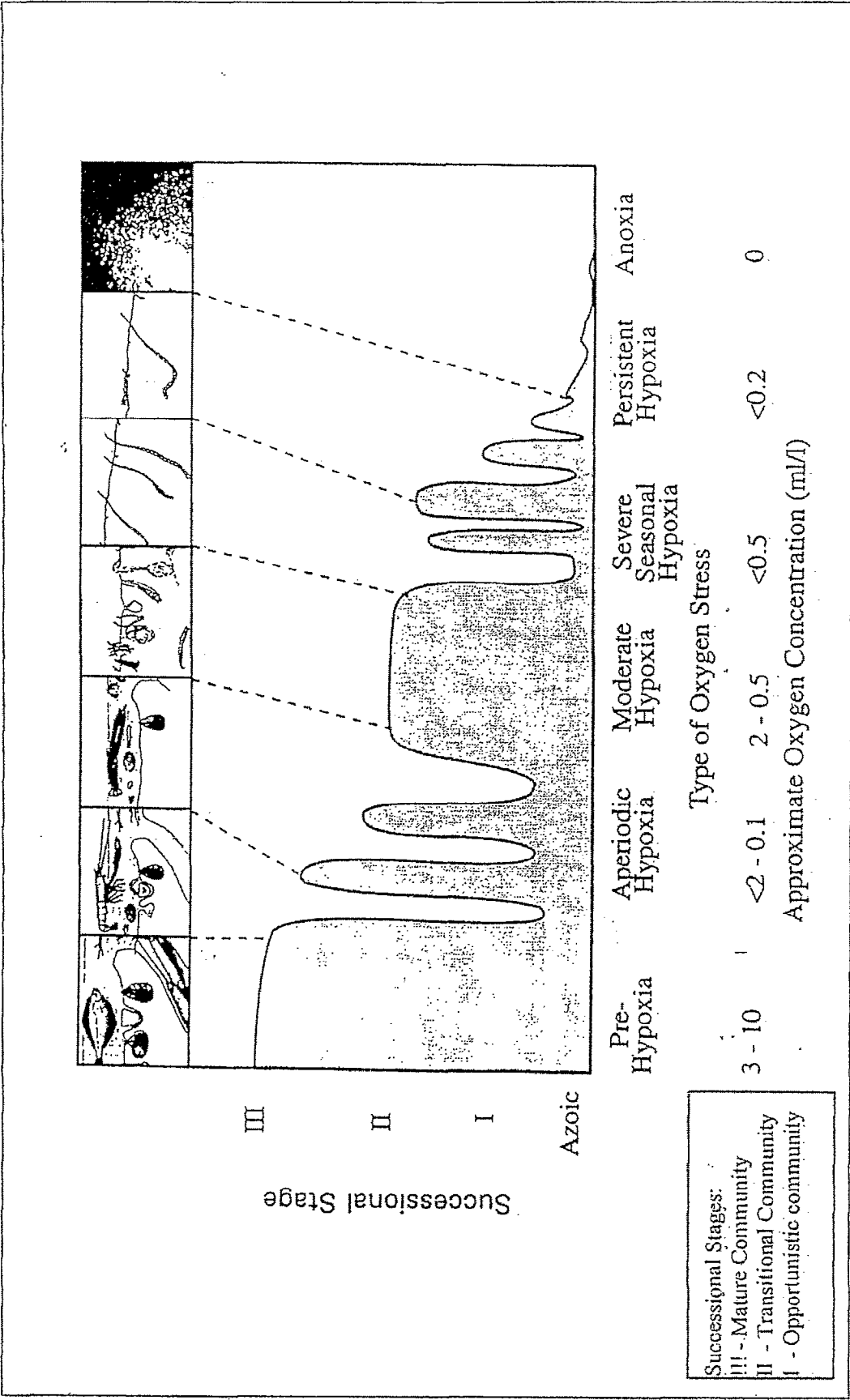


Figure 11 Influence of changes in the severity and periodicity of hypoxic events on benthic fauna (Diaz and Rosenberg 1995).

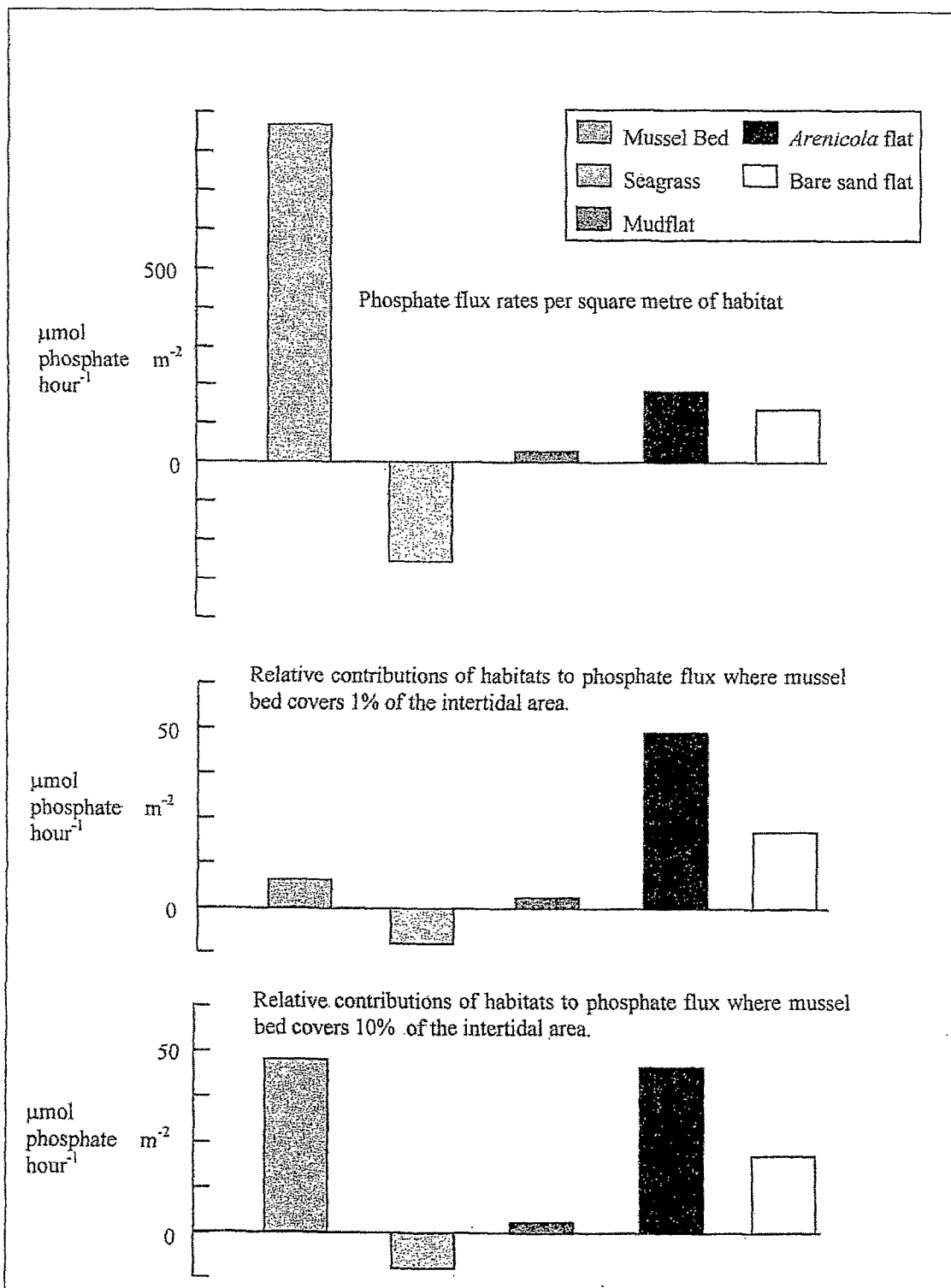


Figure 12 Comparative rates of phosphate flux from five different intertidal benthic communities (Asmus et al., 1995).

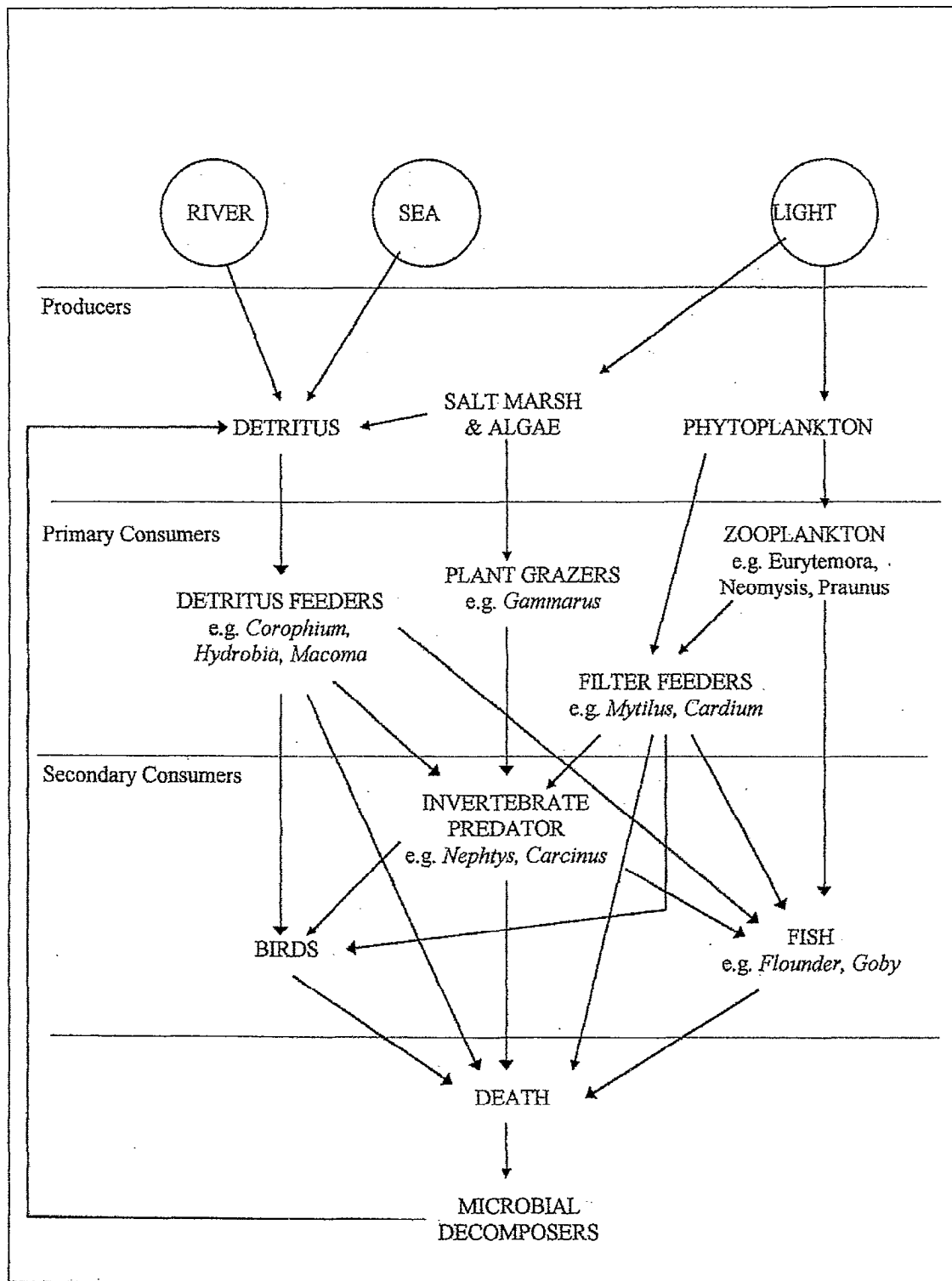


Figure 13: An estuarine food web for a typical northern hemisphere estuary (McLusky, 1989).

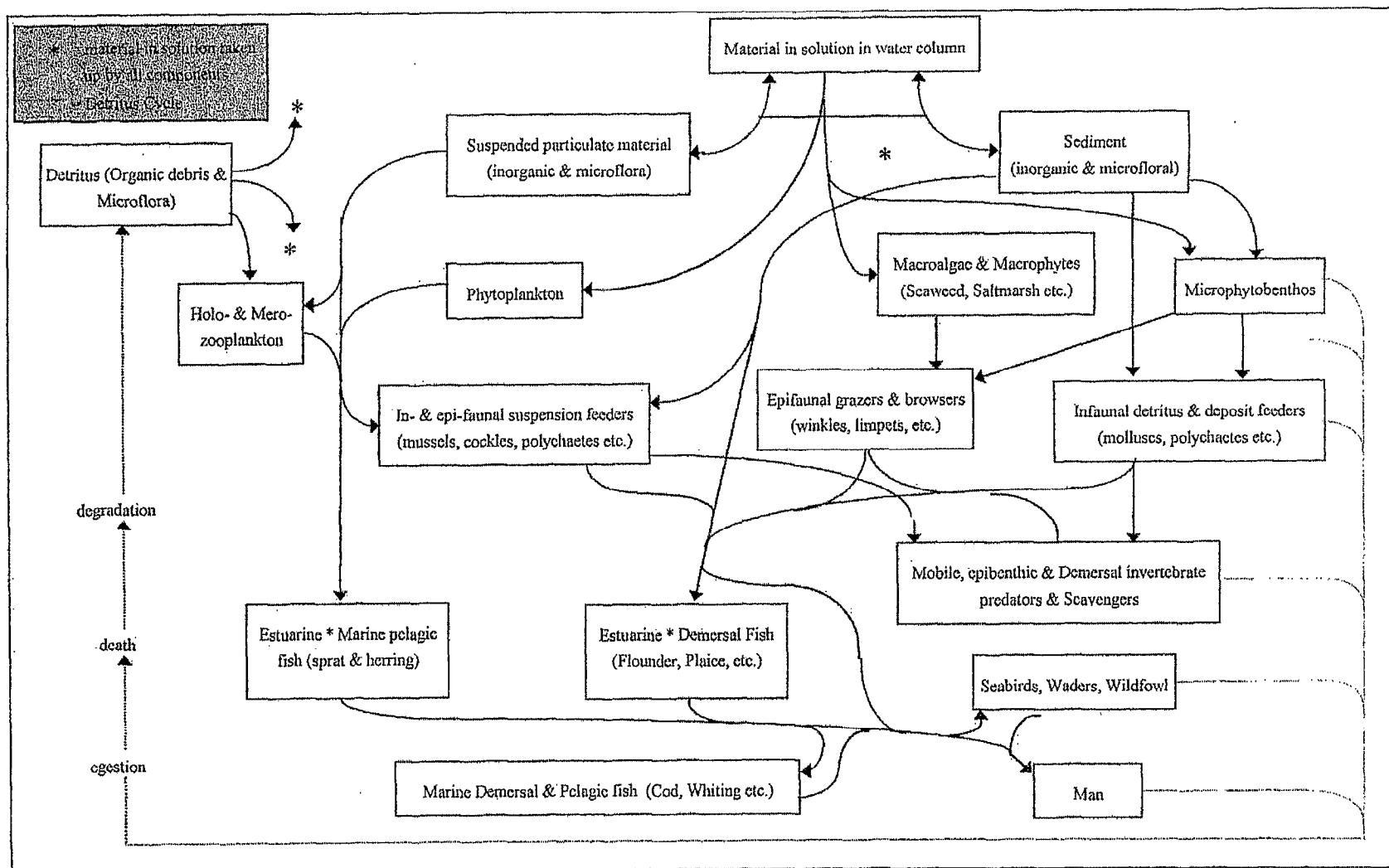


Figure 14: Energy and Pollutant Transport in Estuarine and Coastal Systems (Elliott unpubl.)

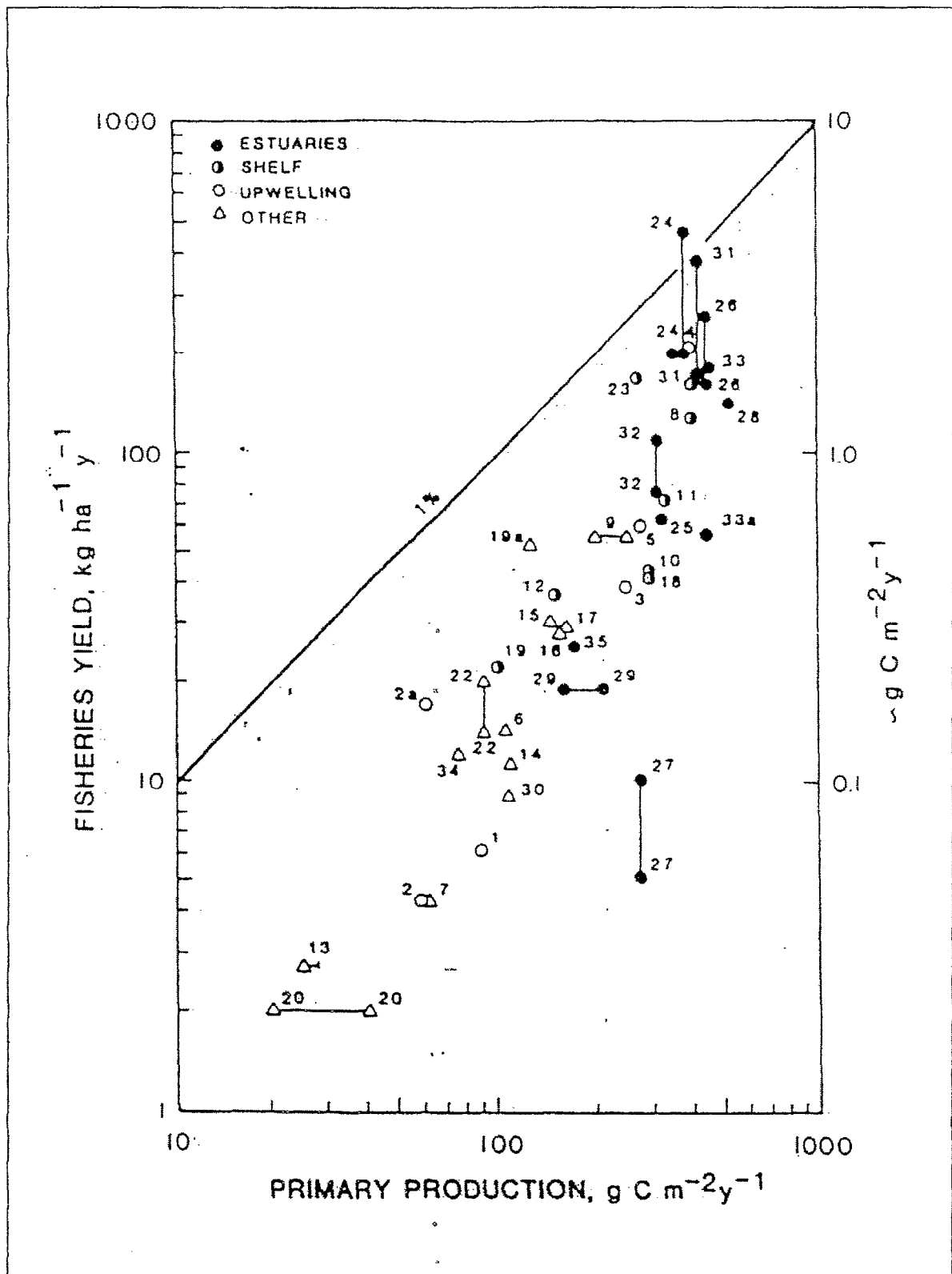


Figure 15: Description of the relationship between nutrient input and primary production for a range of aquatic systems Nixon (1992).

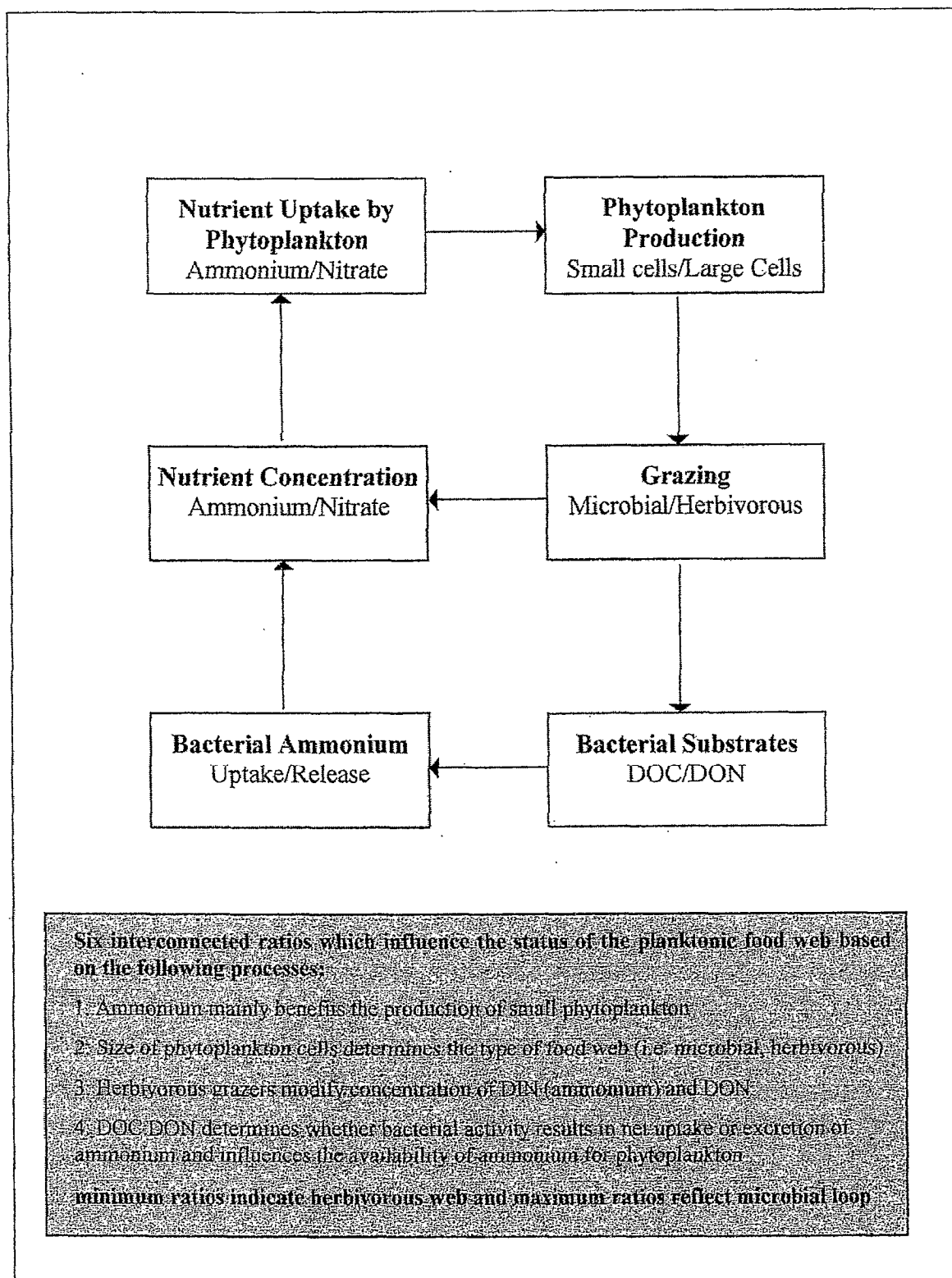


Figure 16: Schematic description of the relationship between nutrient availability and planktonic food webs (from Legendre and Rassoulzadegan 1995)

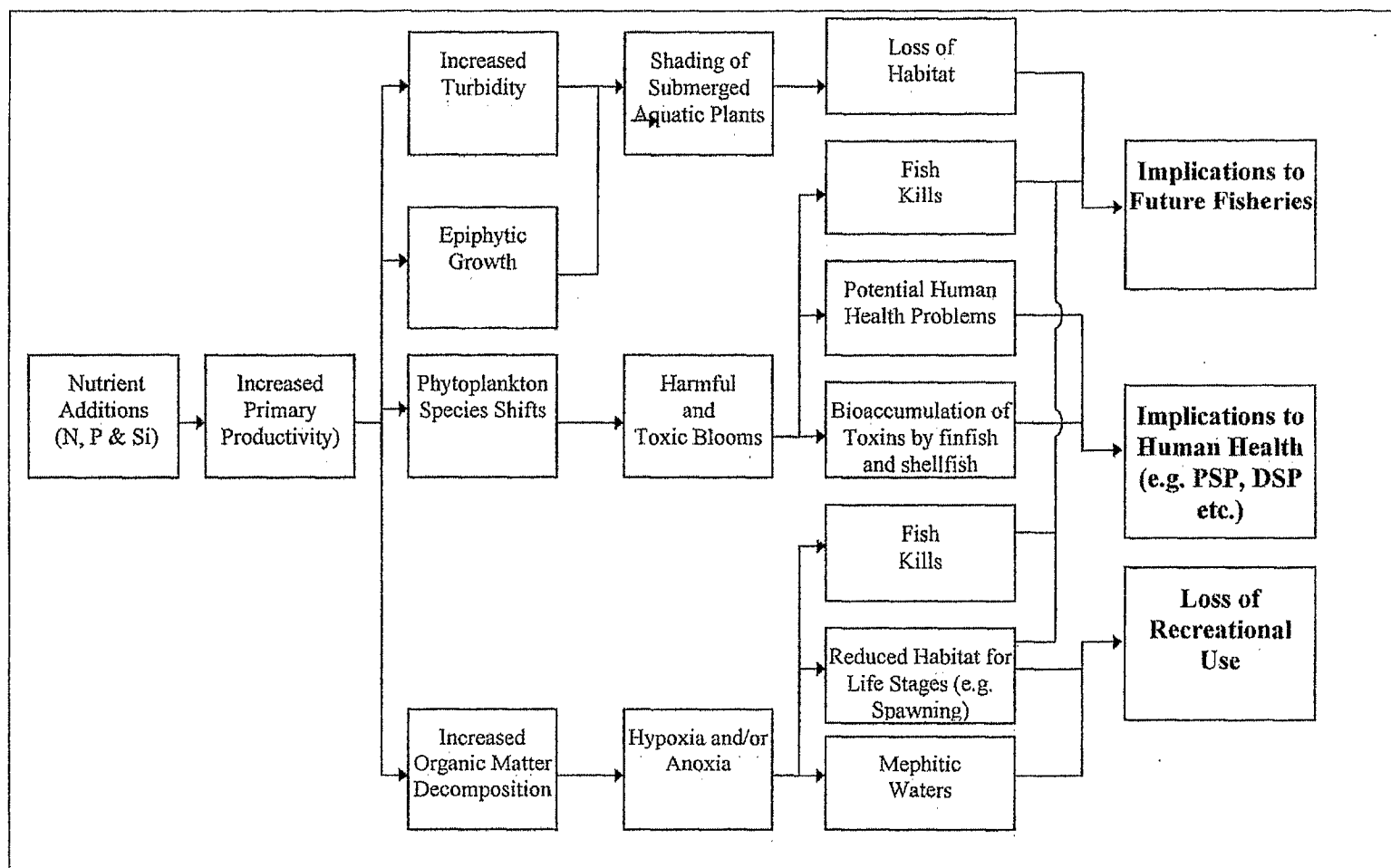


Figure 17: An overview of the potential effects of eutrophication (NOAA 1993).

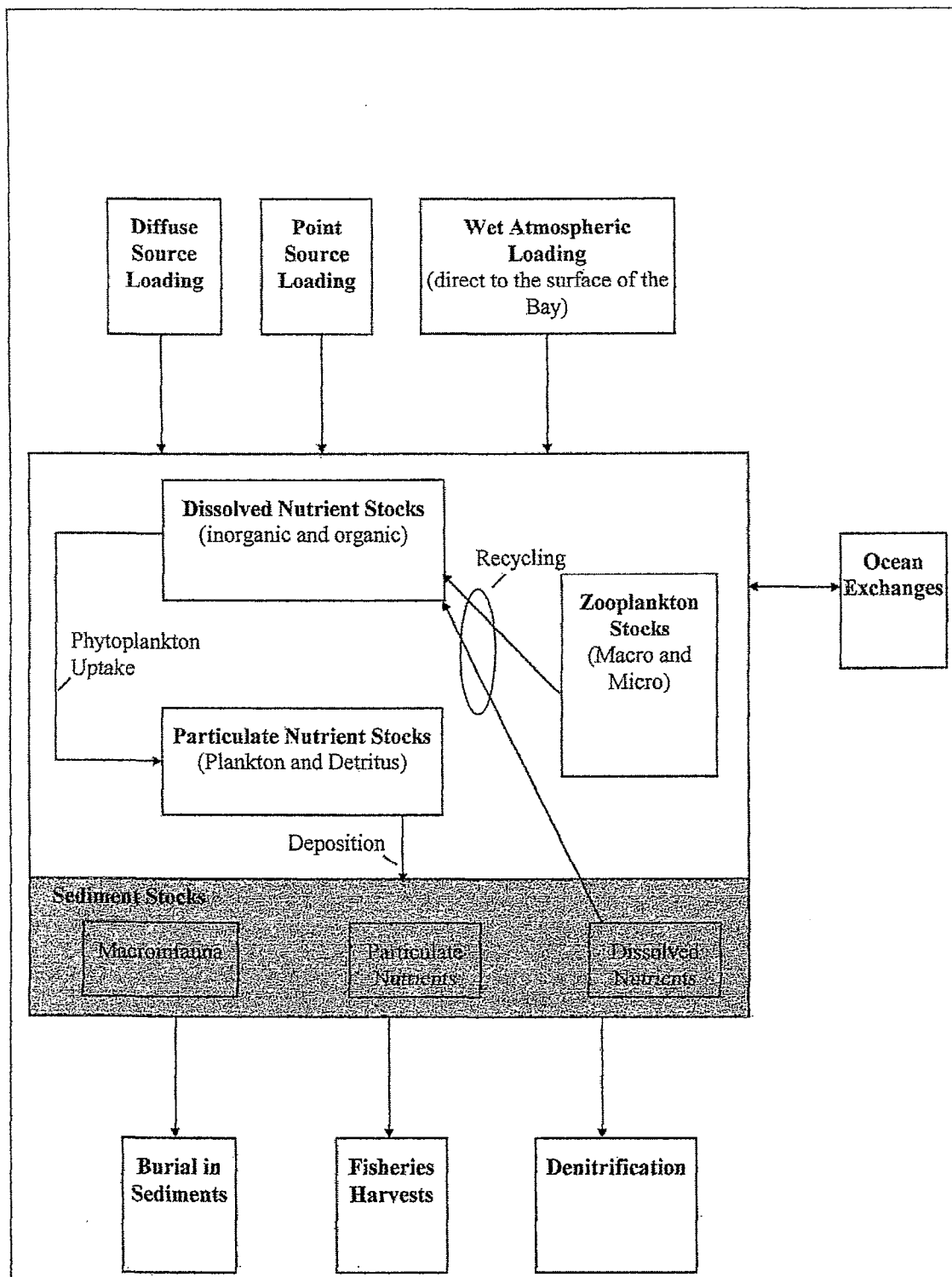


Figure 18: Conceptual model used to develop nutrient budgets for Chesapeake Bay (Boynton et al., 1995).

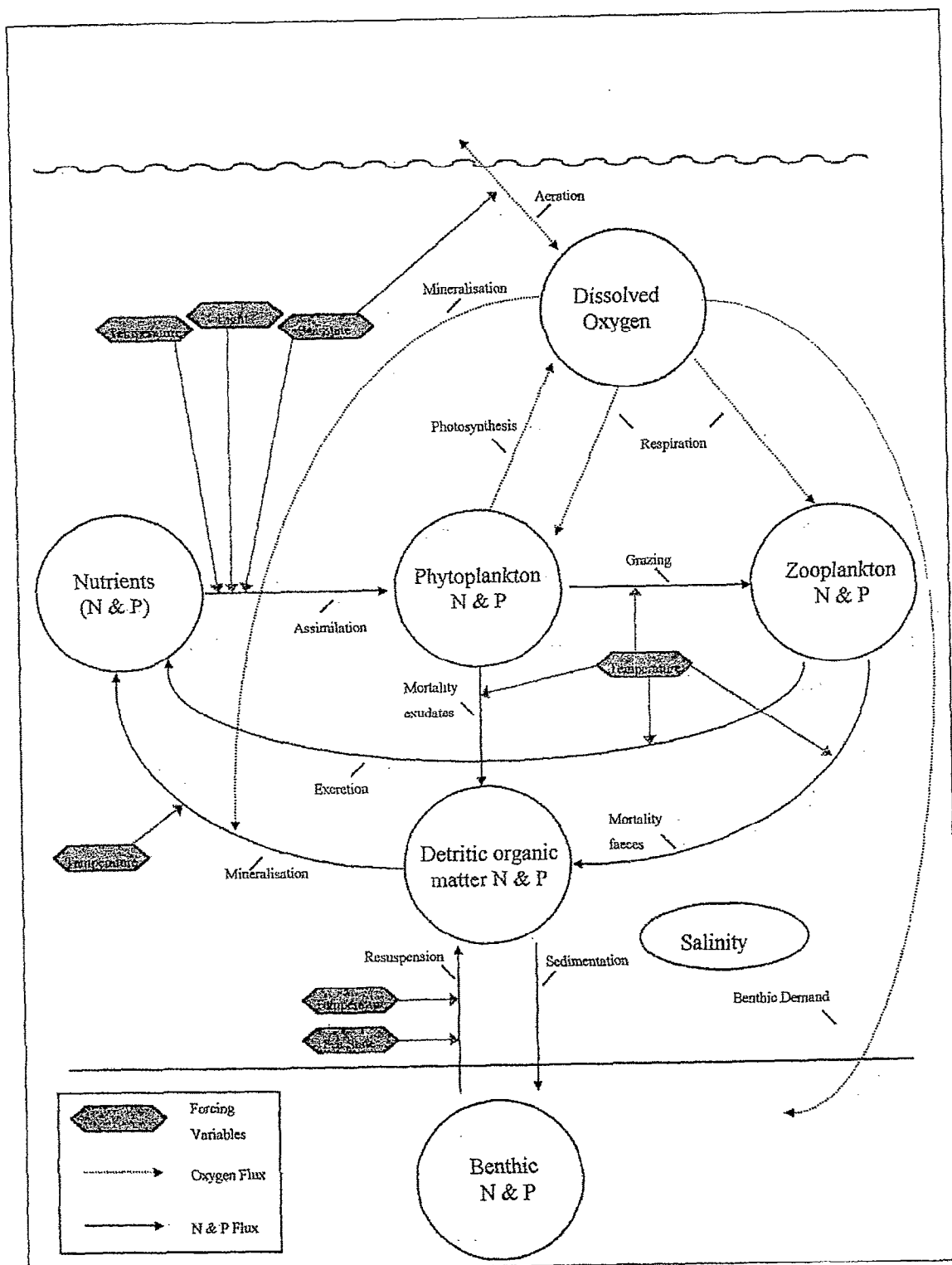
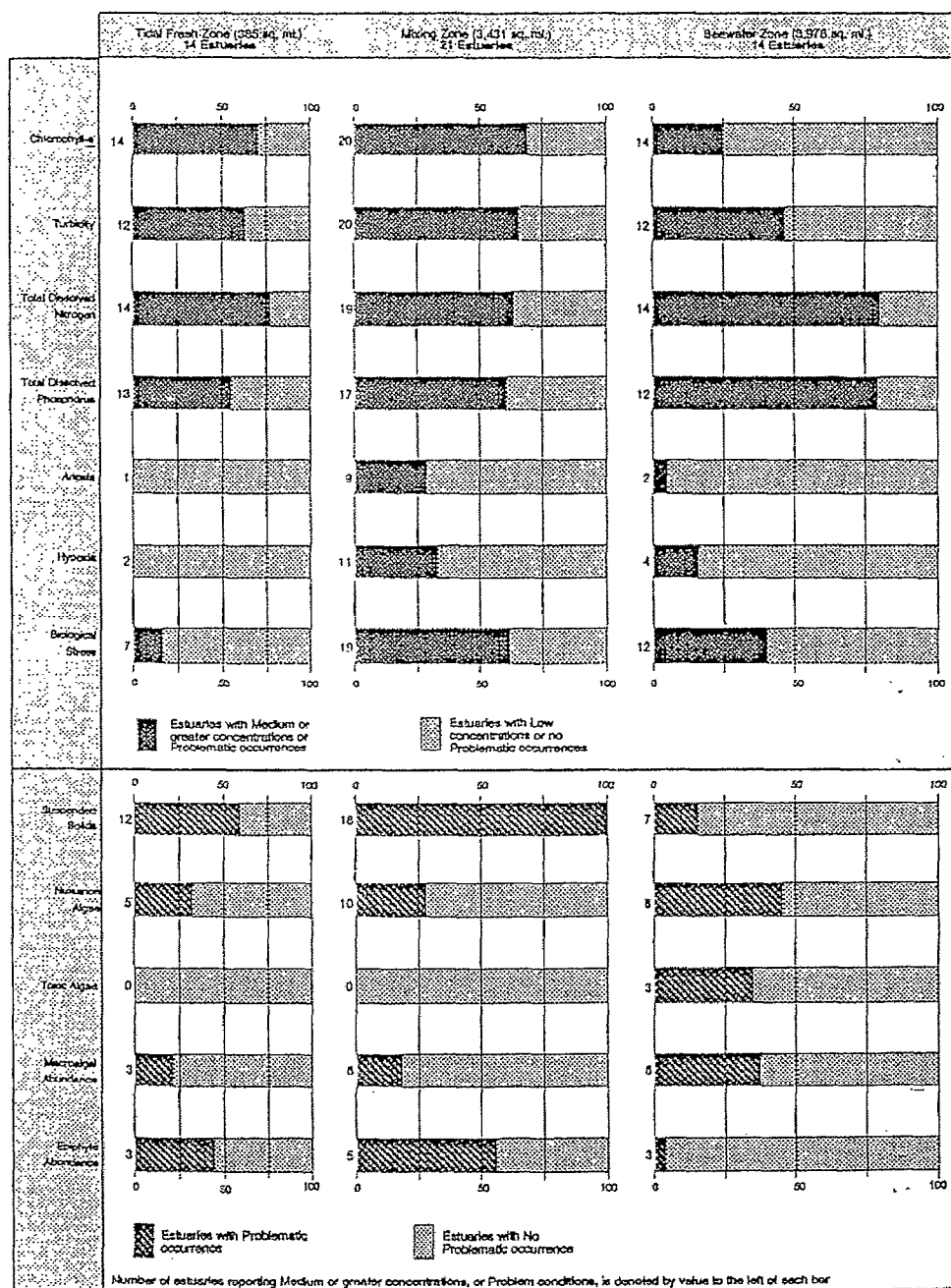


Figure 19 A biological model of eutrophication in Vilaine bay, France (Chapelle et al., 1994).



Conditions are based on values of those eutrophication indicators which suggest problem conditions expressed as cumulative percentage surface area for each salinity band for all estuaries combined

Figure 20: Summary of existing conditions in estuaries on the USA mid-Atlantic coast (NOAA 1995).

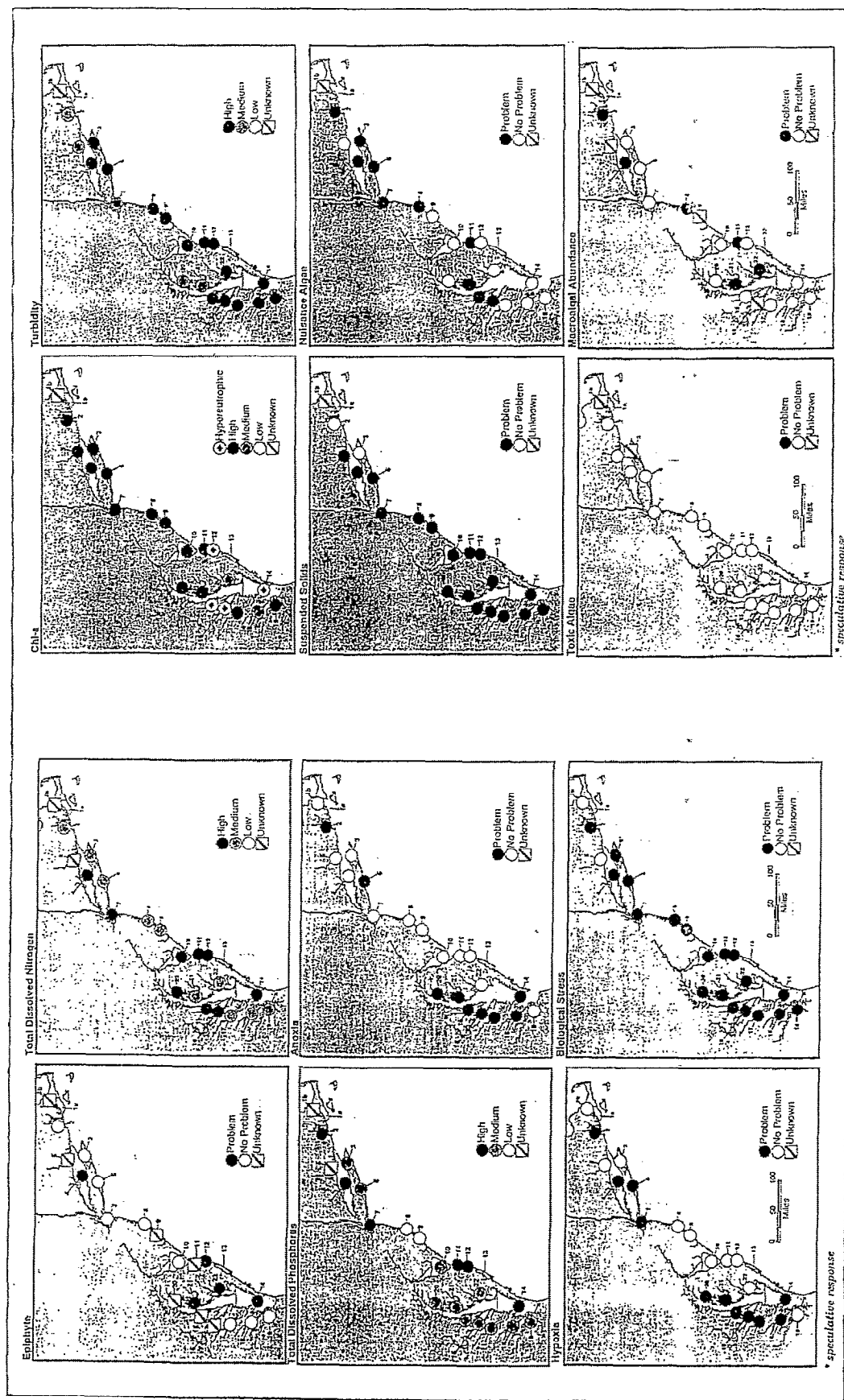


Figure 21: Summary of existing conditions in estuaries on the USA mid-Atlantic coast based on values of eutrophication indicators in the mixing zone (identified as the most heavily impacted region) of estuaries (NOAA 1995).

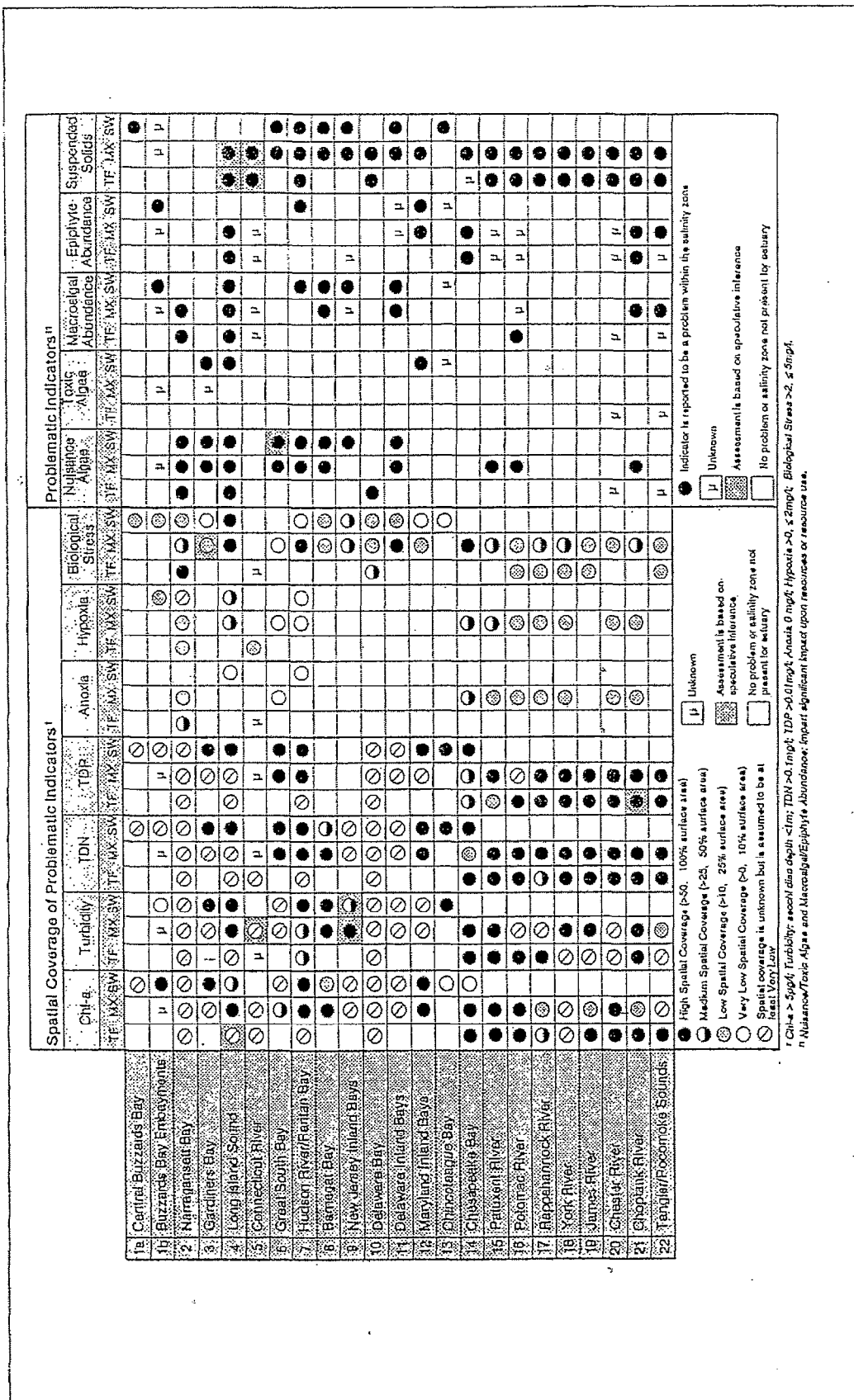


Figure 22: Summary of existing conditions in estuaries on the USA mid-Atlantic coast based on the areal coverage of eutrophication indicators showing problematic conditions in each estuary and in each salinity band (NOAA 1995).

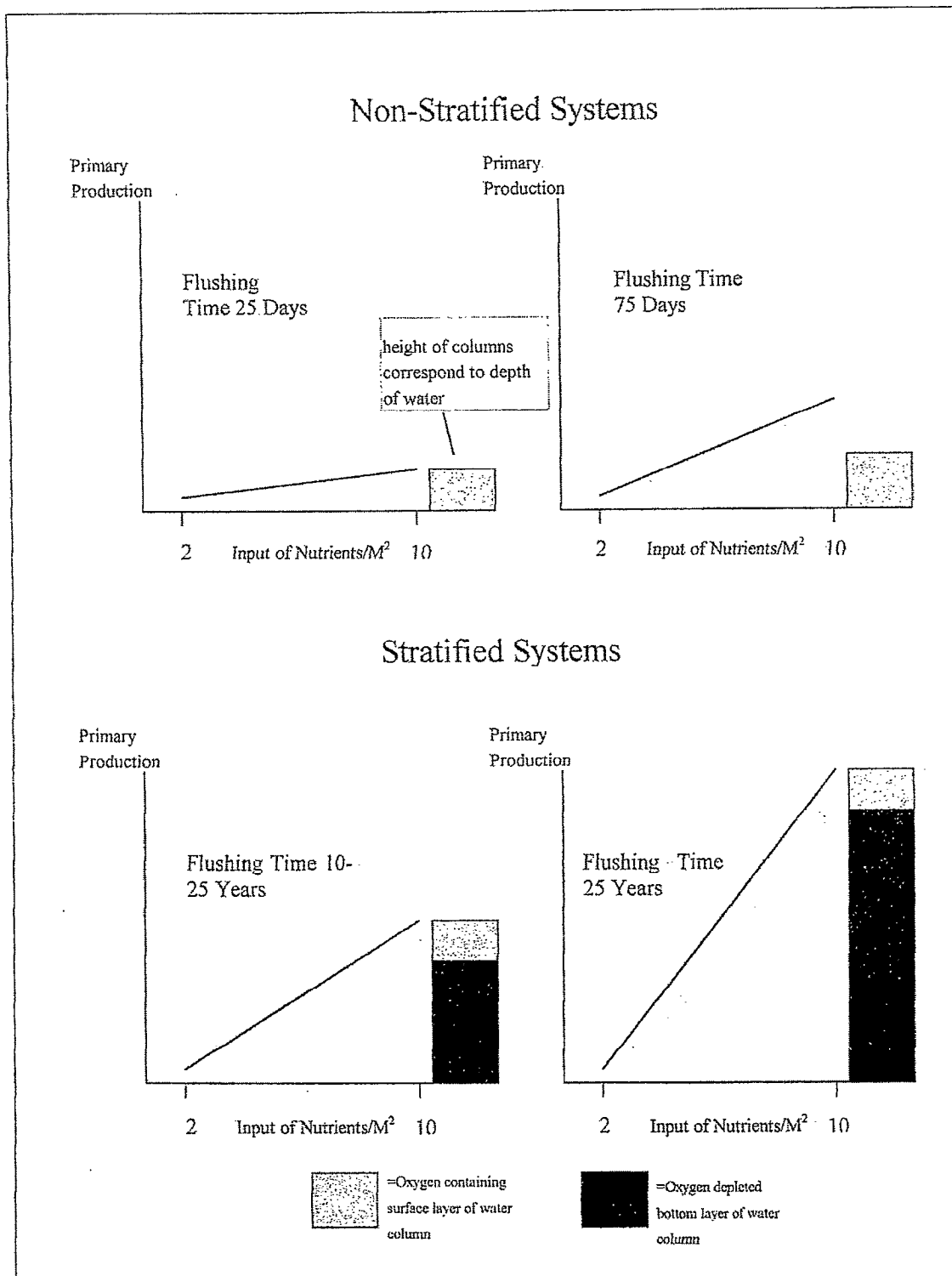


Figure 23: Characteristics of the four estuary types identified by Perdon (1995) showing the: water depth, presence of stratified layer and primary productivity as a function of nutrient input.

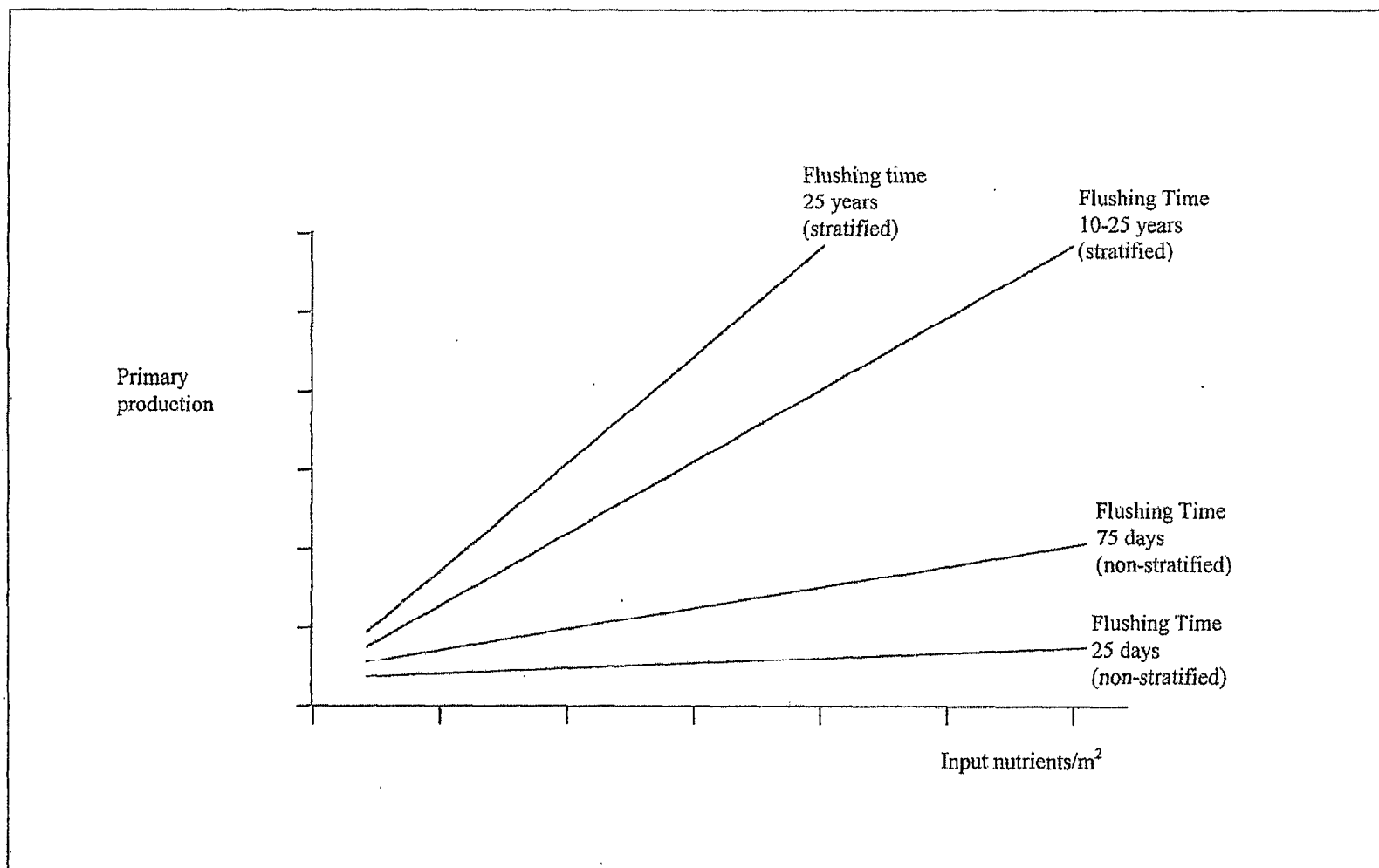


Figure 24: The four estuary classes identified by Perdon (1995) described in a single plot which separates stratified, non-stratified, slow flushing and fast flushing estuaries on the basis of primary productivity levels for given nutrient inputs.

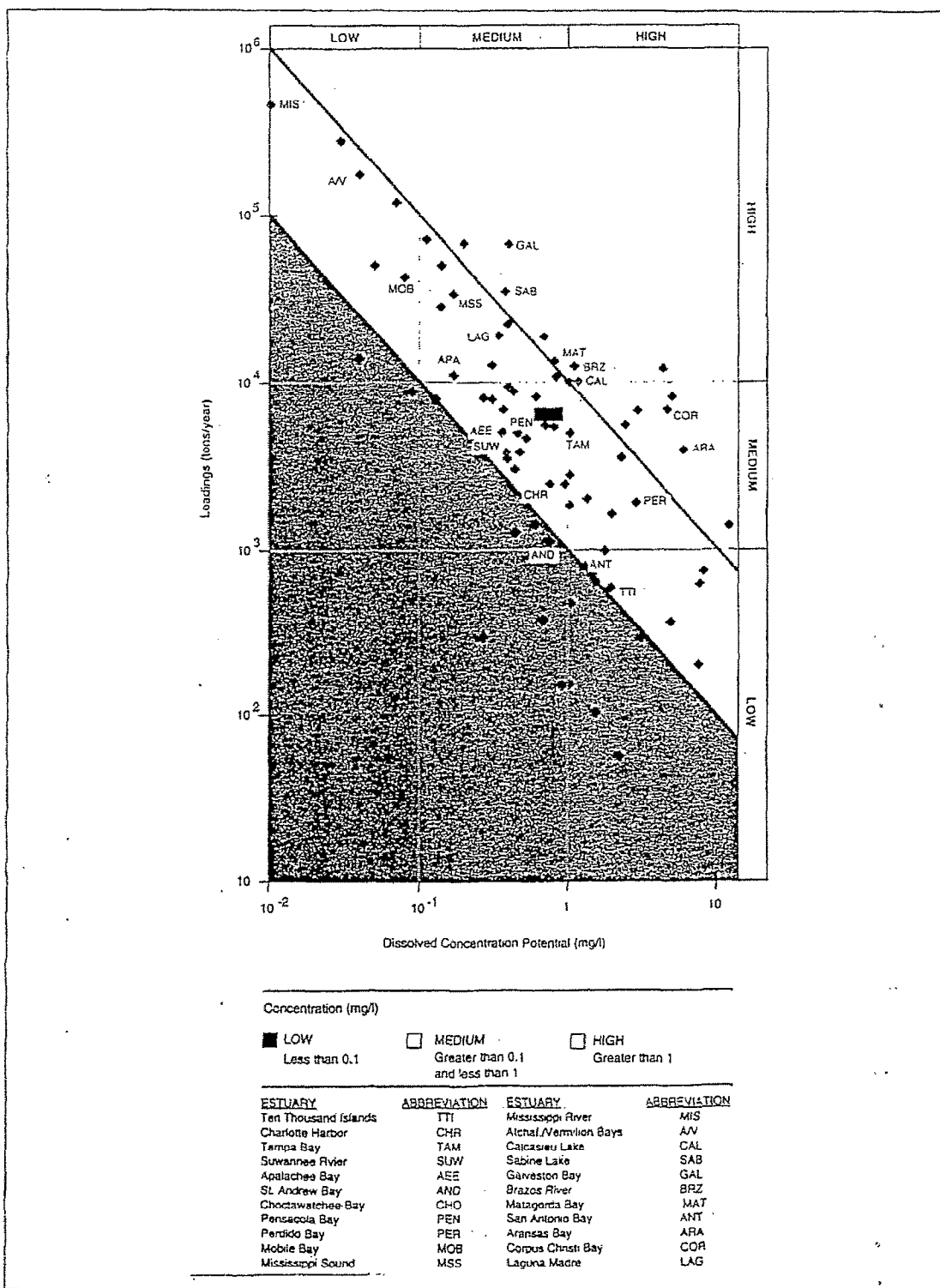


Figure 25: NOAA (1993) classification approach using measure of the capacity of estuaries for retaining nutrients (*Dissolved Concentration Potential*) plotted against nutrient loading.

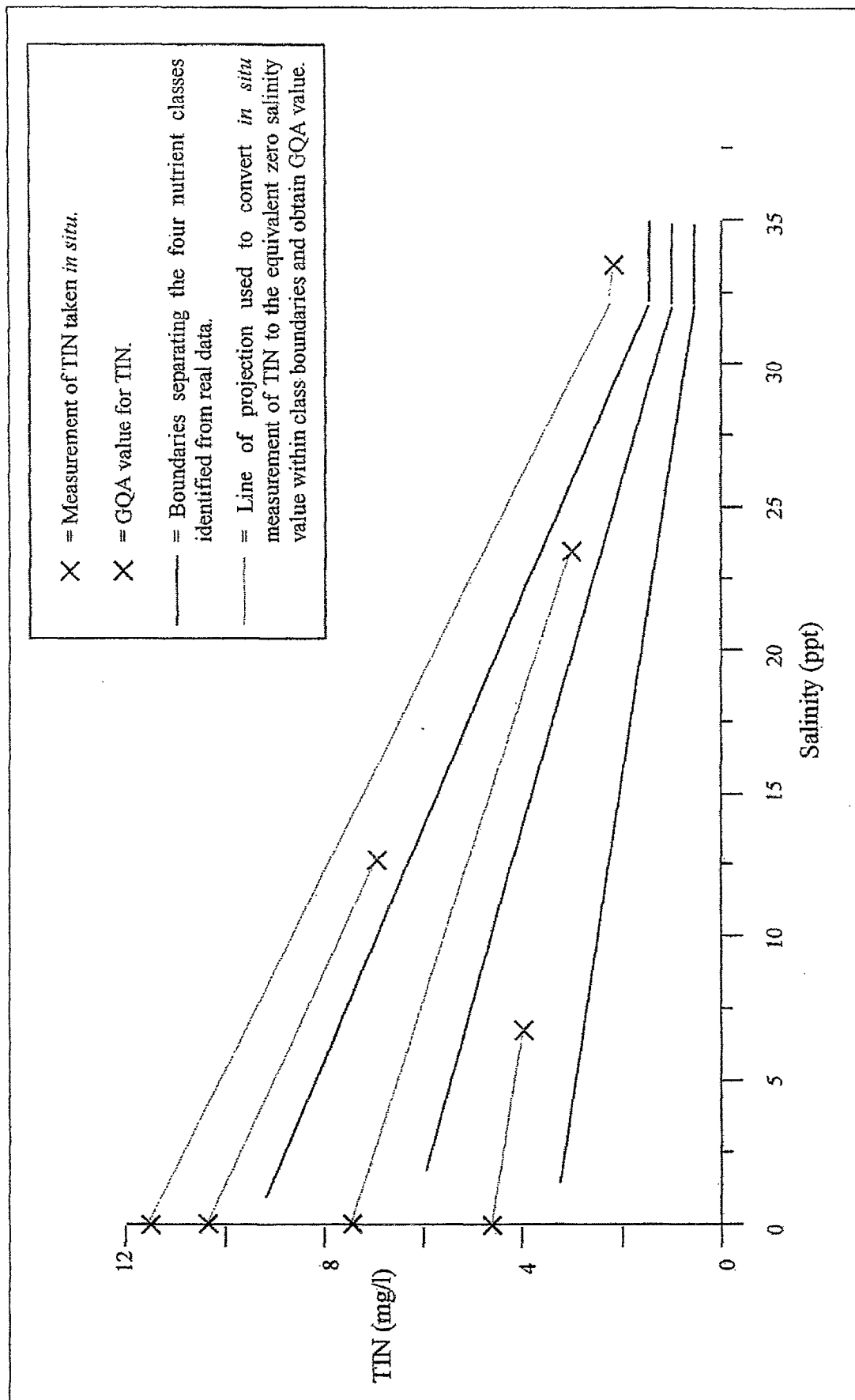


Figure 26: Graphical presentation of the estuarine nutrient classification scheme developed by Gunby et al., (1995).

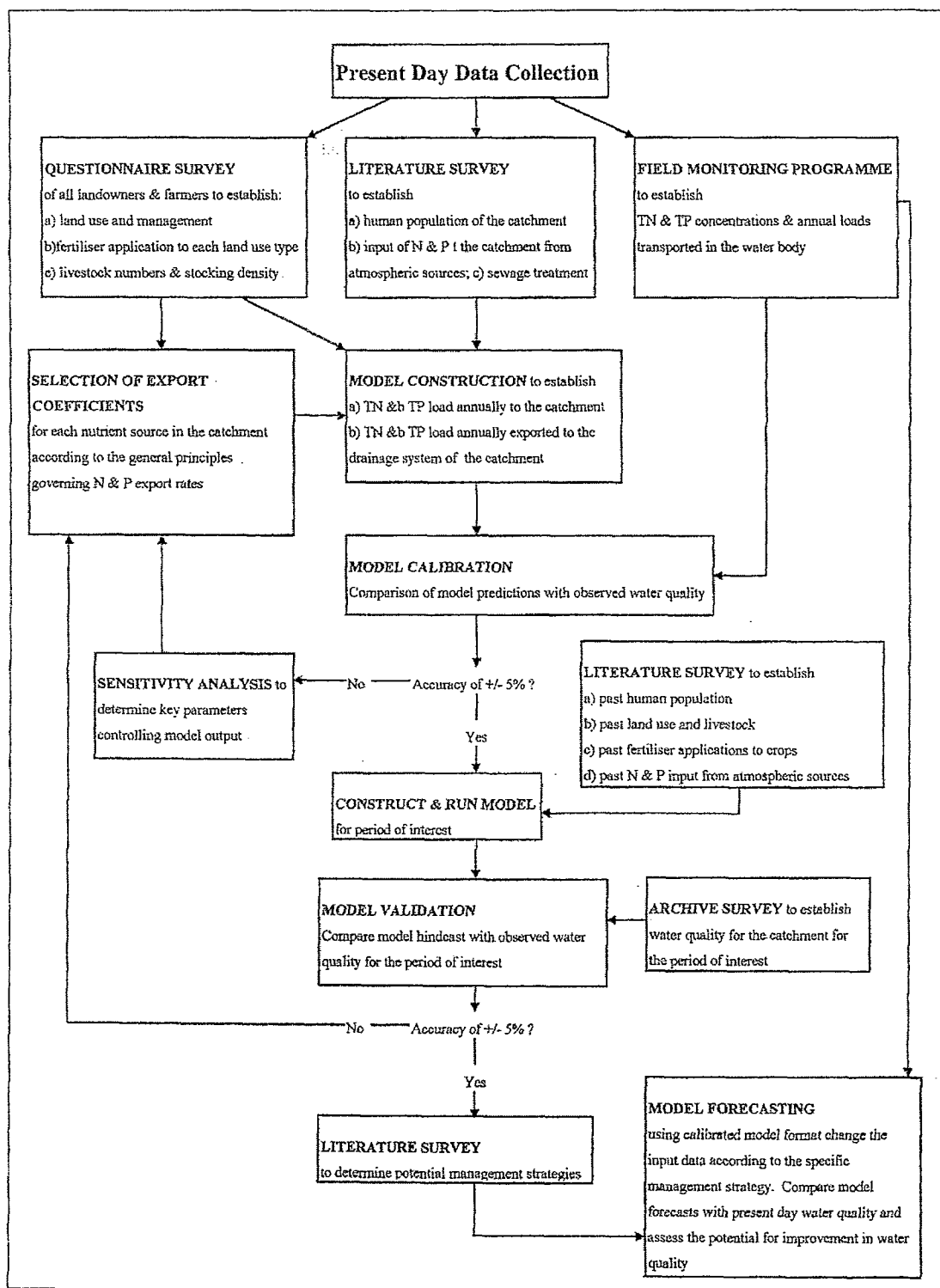


Figure 27: Procedure for modelling nutrient export from catchments (from Johnes, 1994)

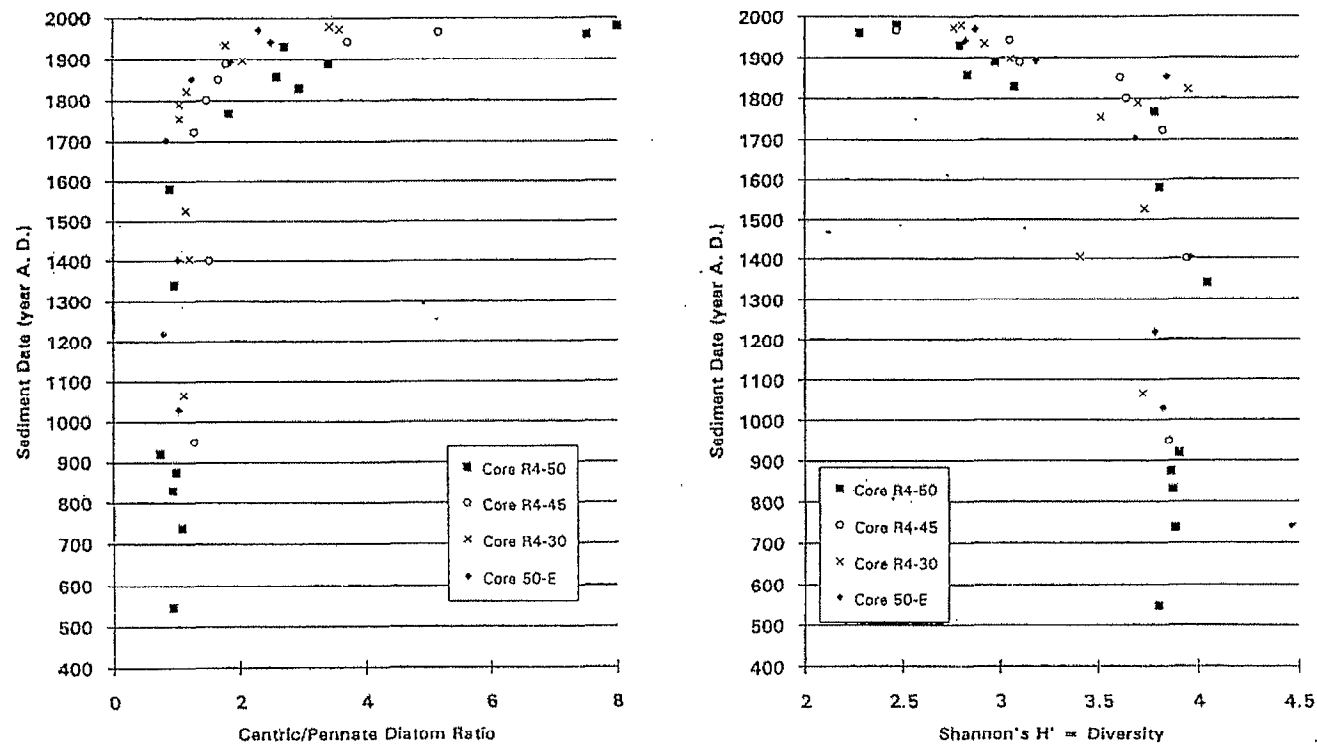


Figure 28: Sedimentary profiles from Chesapeake Bay showing characteristics of diatom populations as stratigraphic indicators of environmental change over time (Cooper and Brush 1993).

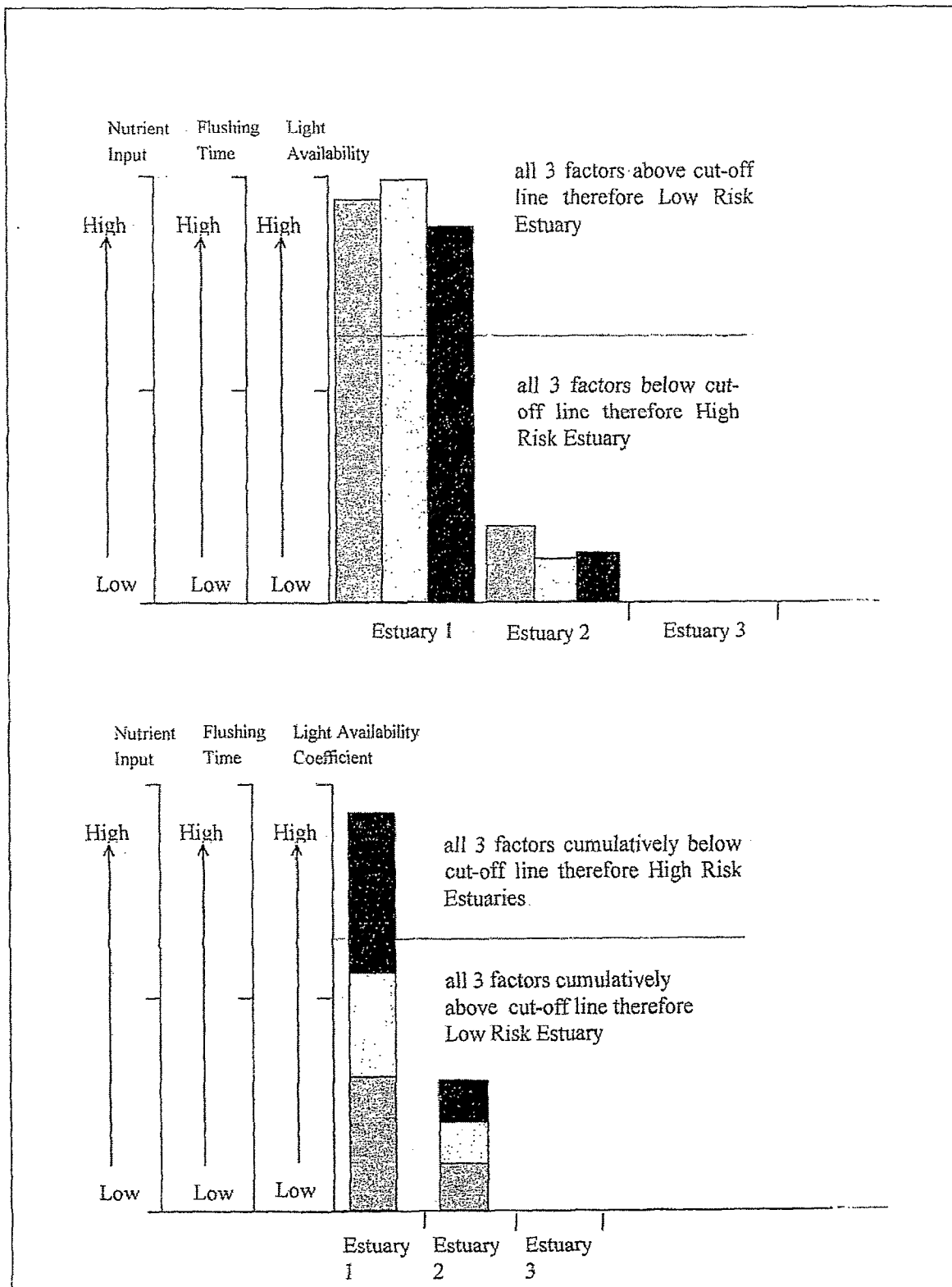


Figure 29: Proposed approach for classifying estuaries based on: flushing time, light extinction and nutrient input using histogram plots (a = individual factors plotted, b = factors plotted cumulatively).

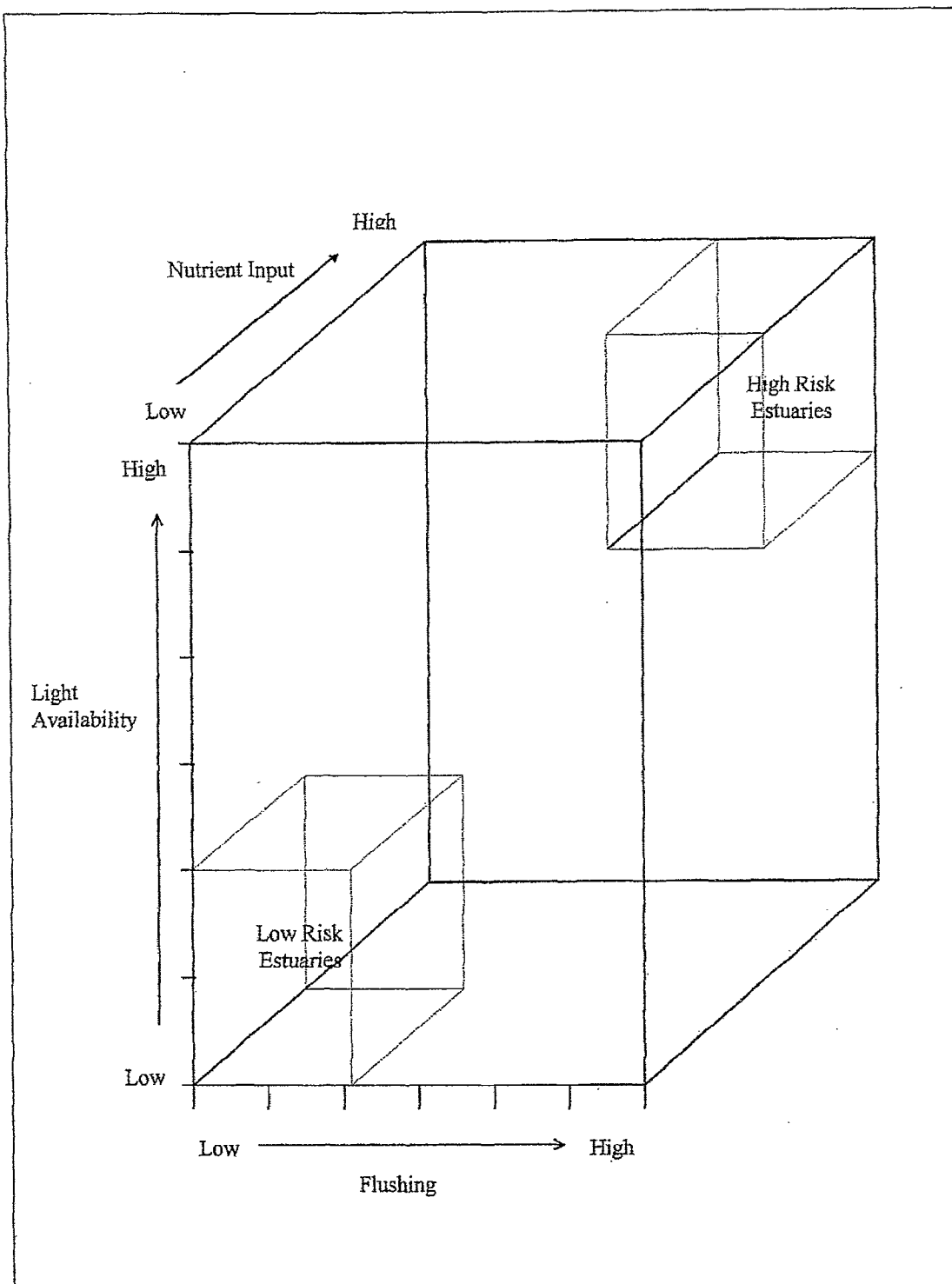


Figure 30: Proposed approach for classifying estuaries based on: flushing time, light extinction and nutrient input using an XYZ scatter plot.

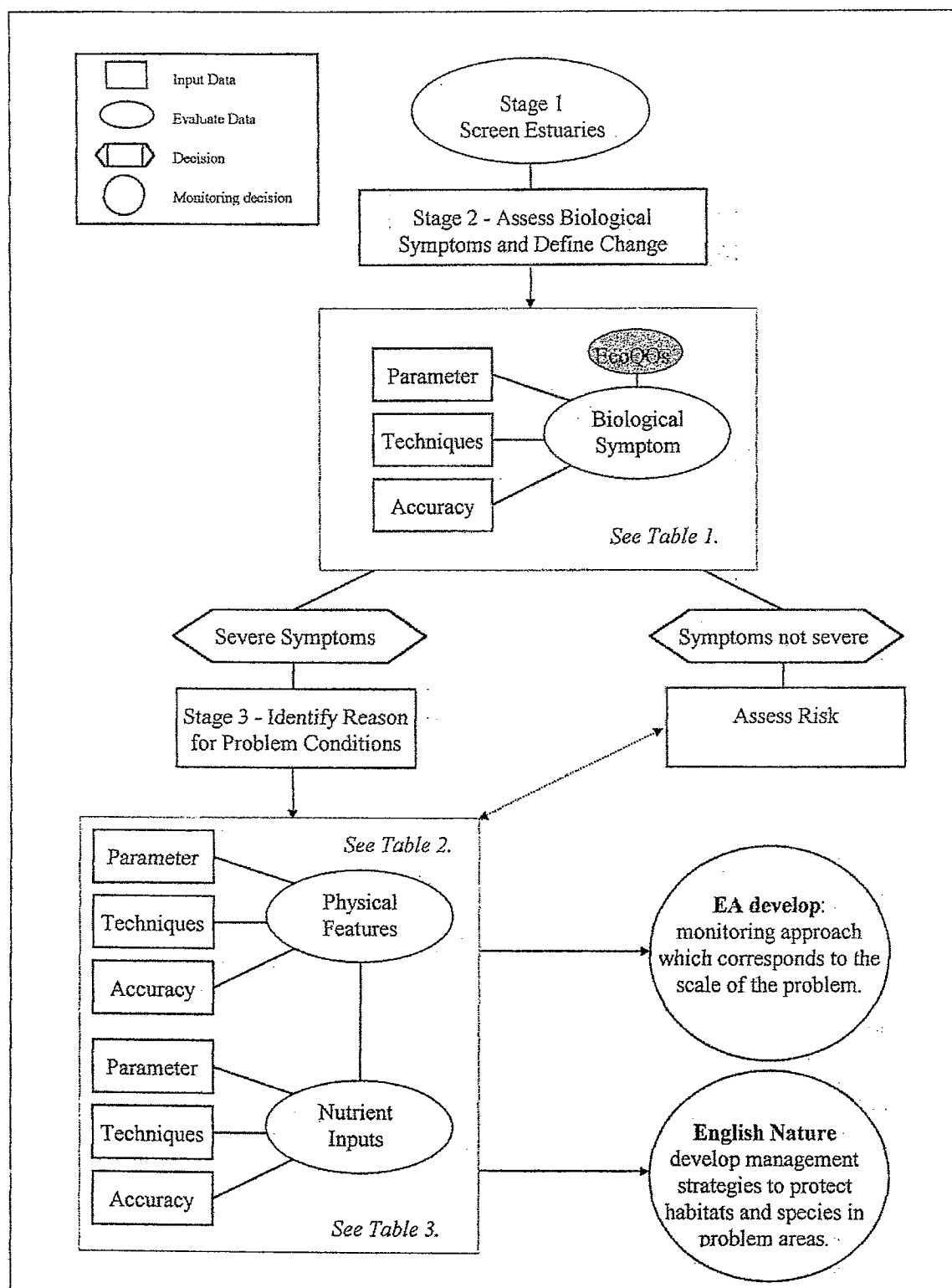


Figure 31: Proposed protocol for driving the monitoring and management of nutrient enrichment in estuaries in the UK.

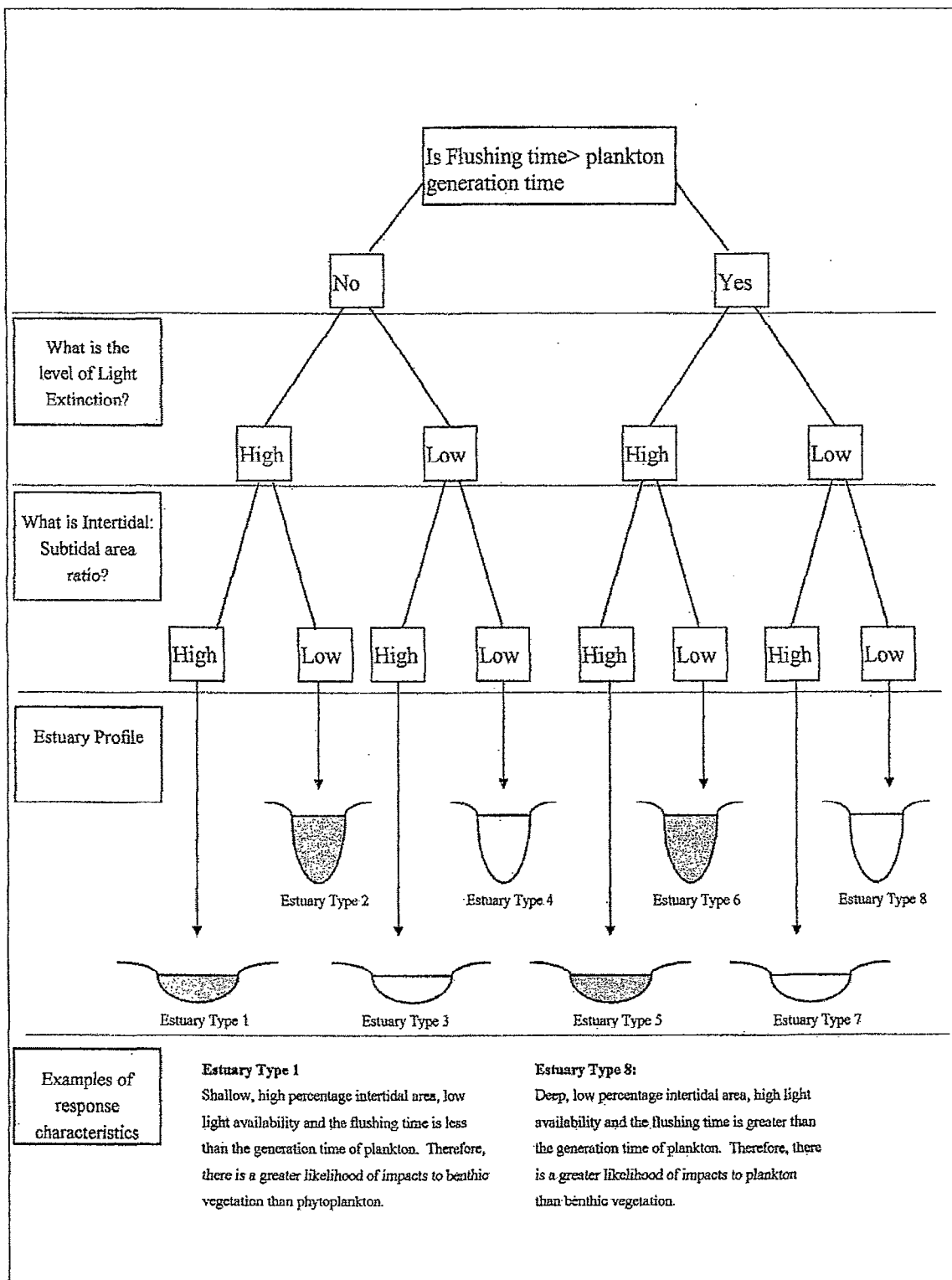


Figure 32 A possible approach for determining the response characteristics in estuaries and tailoring monitoring programmes to take account of site-specific responses to nutrient input.

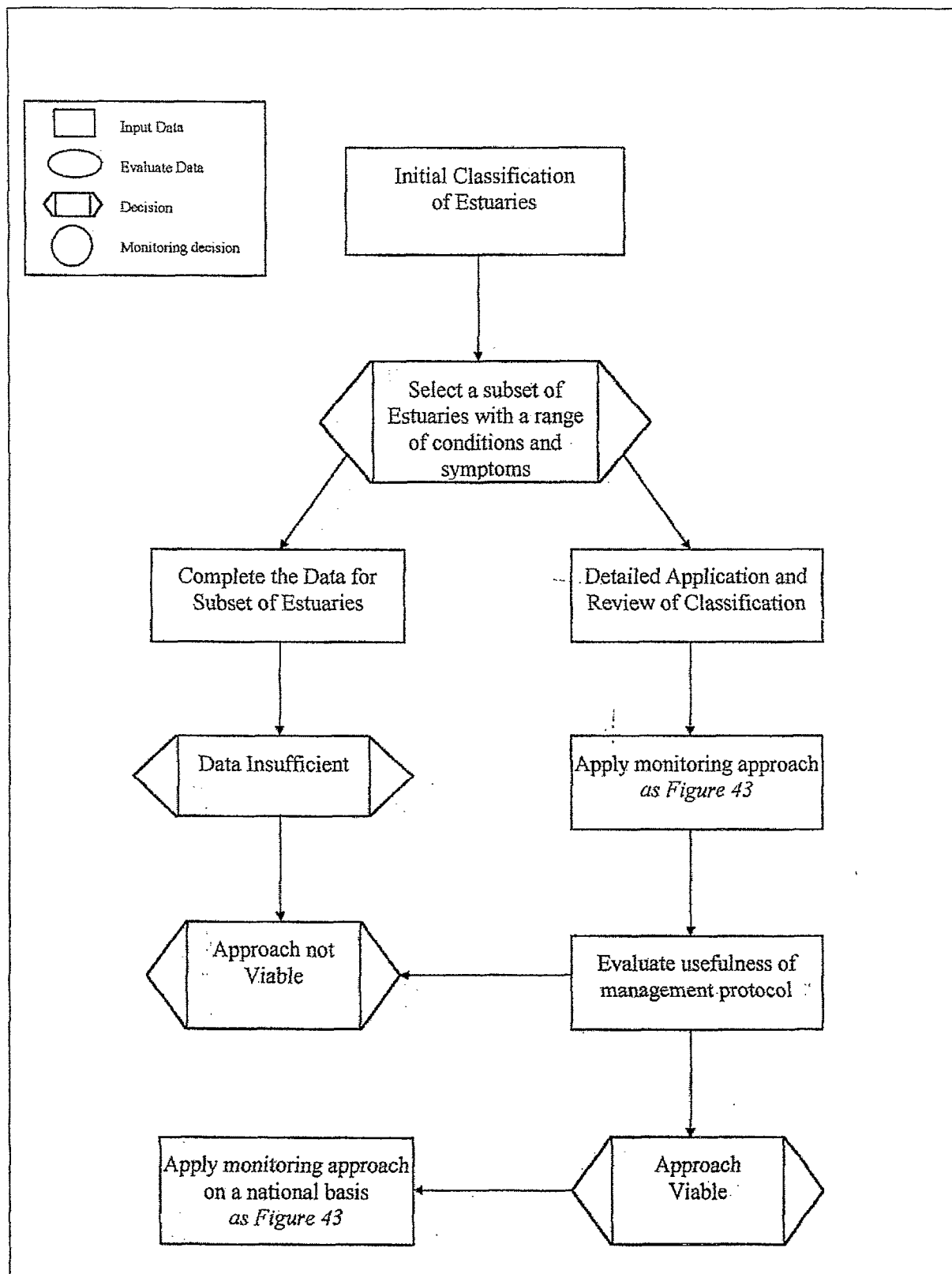


Figure 33: Proposed protocol for carrying out Phase 3 of the 'Impact of Nutrients in Estuaries' Project.

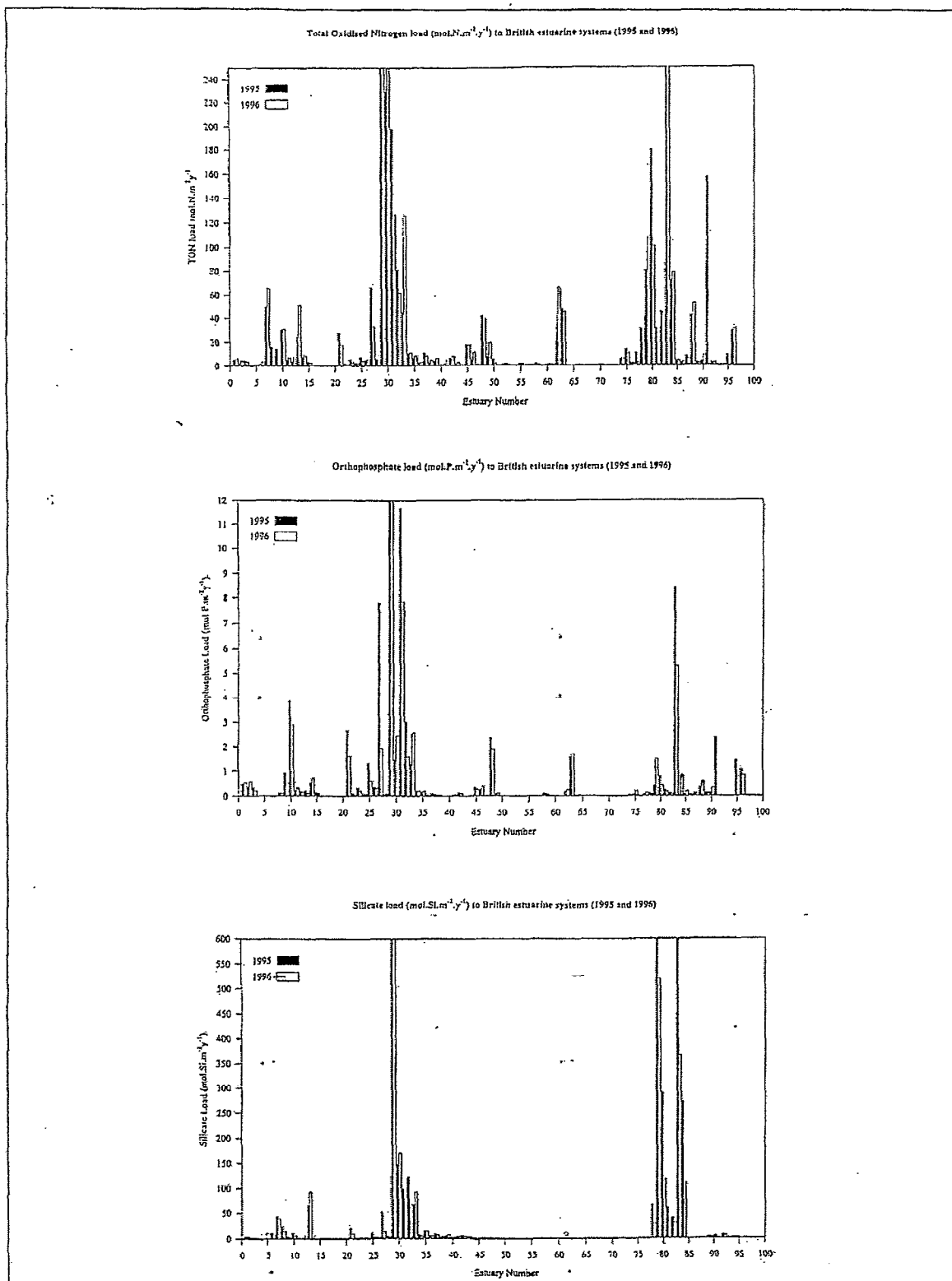


Figure 34: Analysis of Harmonised Monitoring Data (1995 and 1996) for 96 UK estuaries (Nedwell pers. com.).