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Classification Options for Trophic Status

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CLASSIFICATION OPTIONS FOR TROPHIC STATUS

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EXECUTIVE SUMMARY

The Water Act 1989, now consolidated into the Water Resources Act 1991, requires that all controlled waters may be subject to a system of classification and that Statutory Water Quality Objectives (SWQOs) may be set in relation to such waters by the Secretary of State. After lengthy consultations the NRA has proposed that there would be different Use Classifications (UCs) for setting targets relating to the actual or potential use of the water, on a statutory basis i.e. Statutory Water Quality Objectives, plus a General Quality Assessment (GQA) scheme, for assessing general overall progress on a periodic basis. The GQA scheme would periodically examine the general state of the waters using different criteria - basic chemical, biological, aesthetic and nutrient status. It was also recognised by the NRA that such a GQA scheme for estuaries and coastal waters might be improved if a direct assessment of trophic status could be included with an assessment of nutrient levels.

The many factors and inherent variability affecting algal production in tidal waters would have to be accounted for in any classification (and the associated monitoring programme) based on trophic status. A number of options for assessing trophic status are considered in detail. These are: discrete sampling for chlorophyll and phytoplankton biomass; *in situ* fluorimetry; monitoring for nutrient levels; macroalgal biomass; the use of algal growth bioassays; the use of statistical models; and remote sensing from aircraft. It was concluded that there was at present no clearly-defined best option for assessing trophic status in all tidal waters. The most favoured options at present appear to be sampling for chlorophyll and phytoplankton biomass, remote sensing from aircraft and the potentially most robust but most expensive option, the use of statistically based models. However, there are still technical and operational difficulties associated with these options. For example, chlorophyll can be an unreliable indicator of phytoplankton biomass, remote sensing might not be applicable to estuaries, and statistical modelling has not been widely tested in UK waters. All options would require an increased level of sampling (and hence increased cost) in tidal waters by the NRA. The amount of extra sampling would depend, amongst other factors, on the desired level of precision of the classification and what is affordable. Once the appropriate database has been established the classification of trophic status could be based upon the range of observed primary production in tidal waters of England and Wales, with class boundaries calculated using simple statistical rules to divide the frequency distribution into suitable groups.

KEYWORDS

Trophic status, general quality assessment, tidal waters.

1. INTRODUCTION

The Water Act 1989, now consolidated into the Water Resources Act 1991, requires that all controlled waters may be subject to a system of classification and that Statutory Water Quality Objectives (SWQOs) may be set in relation to such waters by the Secretary of State. The NRA has consulted publicly on its initial proposal that SWQOs should consist of the following three elements:

- achievement of relevant use-related EQOs (by compliance with relevant EQSs);
- achievement of target class of relevant classification scheme;
- compliance with relevant EC Directives.

Following the results of the consultation, the NRA has reviewed and refined its proposals for submission to the Secretary of State for his consideration (NRA 1992). It is now proposed that there would be:

- different Use Classifications (UCs) for setting targets relating to the actual or potential use of the water, on a statutory basis i.e. Statutory Water Quality Objectives; plus
- a General Quality Assessment (GQA) scheme, for assessing general overall progress on a periodic basis.

The GQA scheme should periodically examine the general state of the waters using different criteria - basic chemical, biological, aesthetic and nutrient status. It was also recognised by the NRA that such a GQA scheme for estuaries and coastal waters might be improved if a direct assessment of trophic status could be included with an assessment of nutrient levels. In addition, given public concern over eutrophication, the classification scheme is likely to be better received if it includes a direct measure of trophic status. This project is, therefore, aimed at reviewing the options for assessing trophic status with a view to its possible inclusion in a general classification scheme.

2. OBJECTIVES

The overall objective as described in the project investment appraisal is:

"To review the options for including a reliable and practical assessment of trophic status in terms of algal productivity in a general classification scheme for estuaries and coastal waters in England and Wales".

The specific objectives are:

1. To establish the options for assessing trophic status in terms of algal productivity of estuaries and coastal waters in England and Wales.

2. To review the advantages and disadvantages of these options from the viewpoint of general class-limiting thresholds, inclusion in a routine monitoring programme, practicality, reliability, cost-effectiveness, ease of application and monitoring and universal applicability in estuaries and coastal waters in England and Wales.

3. To establish the most appropriate timing and frequency of sampling for these options for assessing trophic status.

4. To establish the likely availability of data for the identified options which could be used for defining threshold values.

5. To establish the extent to which measures of trophic status could or should be linked to other criteria proposed (e.g. nutrients) for inclusion in the general classification scheme for both estuaries and coastal waters.

6. To identify whether or not an appropriate measure of trophic status suitable for inclusion in a general classification scheme for either estuaries or coastal waters, or both, does exist which can meet the criteria identified in 2.

3. THE REQUIREMENTS OF A CLASSIFICATION SCHEME BASED ON TROPHIC STATUS

Any option for assessing the trophic status of tidal waters would ideally have to take account of the following aspects:

1. The natural variability of water quality and primary productivity in tidal waters. This variability would include temporal variability over periods as short as a tidal cycle in estuaries, and as long as seasonal and year-to-year differences. Year-to-year differences might reflect differences in growing conditions, such as light availability and temperatures between years rather than differences in nutrient supply. Spatial variability in primary productivity might relate to location of sources of nutrients, retention times in estuaries, clarity of water (turbid estuaries compared to less turbid coastal waters) and tidal currents effects (e.g. the effects of residual currents in coastal waters).

2. The constraints associated with the objectives of the scheme must be met. For example, if the objective was solely to detect the development of nuisance algal blooms or maximum algal productivities, sampling frequency and sampling period constraints would have to be adhered to. In addition, as primary productivity is mainly limited to certain periods of the year a sampling programme over the whole year might not be required. Alternatively if peak production was missed by too infrequent sampling, a true reflection of trophic status may not be obtained.

3. The potentially different indicators of trophic status of tidal waters would have to be considered. The measurements of winter nutrient levels might enable the potential trophic status of the waters to be assessed but not give an indication of what might actually occur considering the many other abiotic and biotic factors affecting trophic status e.g. light availability. In addition, trophic status or changes in trophic status might not be solely reflected in production of phytoplankton. In some bodies of water such as estuaries and harbours the major response to nutrification might be changes in macroalgal or benthic algal communities. The selected options must, therefore, use an appropriate indicator of trophic status and, if possible, be universally applicable throughout England and Wales.

4. It is desirable that the most cost-effective and affordable option is selected. For example, if chlorophyll measurements were required weekly over the growing season, would this be best achieved by discrete sampling by boat compared to remote sensing. An option that encompasses other uses or meets other needs of the NRA might be favoured even though it might, in the short term, be the most expensive. For example, an appropriate model might be able to define the sensitivity of bodies of water or be used as a water quality management tool. The possibility/practability of including sampling for trophic status within current NRA routine monitoring programmes would also be of benefit.

5. The options must also be suitable for defining the thresholds between the classes of tidal waters and ideally class widths would reflect changes and differences in trophic status. For example, the most affected (in terms of algal production) waters might be termed hypertrophic, with less affected waters described as eutrophic and

mesotrophic, and least affected, oligotrophic. However, the setting of thresholds between classes is always going to be somewhat arbitrary as effectively it is a continuum of degrees of contamination and primary productivity that is being divided up. Any method (analytical or operational) must be capable of defining precisely the differences between classes. For example, if there were a 10 µg l^{-1} chlorophyll class width then the analytical technique would need an accuracy of +/- 0.1 to +/- 0.2 µg l^{-1}. Given the large amount of variability in tidal waters the classification would have to be made on some form of statistical assessment of quality at any one site for classification. Structured sampling programmes might offer the opportunity of reducing variability to more manageable levels. It is also probable that the concentration of the determinand used to define class would change throughout the growing season depending not only on nutrient availability but also on other factors such as light. Consideration would, therefore, have to be given as to whether maximum productivities achieved or averages were used in the classification, particularly if maxima were associated with nuisance blooms.

6. Coastal waters may also respond proportionately more than estuaries in terms of primary production to the same nutrient load or nutrient concentration because of factors such as greater water clarity in coastal waters, and different flushing rates/residence times. This would be a shortcoming of classifying on nutrient levels alone without due allowance for factors such as retention times, turbidity and light availability.

7. The selected option would also have to be practicable and appropriate. For example, the required technology (e.g. *in situ* samplers or remote sensors) should be proven to give reliable and relatively easily obtainable data with the appropriate degree of accuracy and precision. Some options would not be applicable to all tidal areas of England and Wales, for example the habitat required for macroalgal growth may not be widespread enough for use in a general classification scheme. As with all classifications all data would need to be directly comparable and as such would have to be collected using standard procedures and methods. Any option that could be wholly or partly incorporated in routine monitoring programmes using well-tested techniques would be of obvious benefit to the NRA.

An illustative example of how a classification based on trophic status might be structured is given in Table 3.1.

Table 3.1 Illustrative example of how a classification based on trophic status might be structured.

Class	A	B	C	D	E
1	Oligotrophic	Minimum production		Minimum loads	Low production
----------1/4(max-min)----(1 µg l^{-1})----------					
2	Mesotrophic			Non-critical load	No algal blooms or problems
----------1/2 (max-min)----(10 µg l^{-1})----------					
3	Eutrophic			Critical loads	Subject to occasional algal blooms or problems
----------3/4 (max-min)----(100 µg l^{-1})----------					
4	Hypertrophic	Maximum production		Excessive loads	Subject to frequent algal blooms or problems

Descriptor or defining criteria ------> A, B, C, D, E

Key:

A. Descriptor of trophic status.

B. Classification based on levels of production with thresholds based on proportions of maximum or average production levels found in tidal waters of England and Wales.

C. Speculative class threshold values based on the chlorophyll concentration (10 µg l^{-1}) given to define 'sensitive' estuaries and coastal waters in terms of the Urban Wastewater Treatment Directive (DoE/Welsh Office 1992).

D. Classes based on assessment of (critical) loads of nutrients discharged into tidal waters.

E. Class descriptors based on occurrence of algal blooms or nuisance algal growths (e.g. macroalgae).

The rest of this report describes in more detail the main elements of algal productivity in tidal waters, the factors affecting trophic status and the considered main options for assessing trophic status. Also considered is the availability of appropriate data for developing the scheme, and finally how the scheme might be linked with other classification criteria is discussed.

4. ALGAL PRODUCTIVITY IN TIDAL WATERS

Because the additional nutrients responsible for marine eutrophication are derived primarily from land, the coastal and estuarine waters of intensively populated and managed land have higher nutrient levels than in the open sea. It is, therefore, not surprising that the most dramatic effects of marine eutrophication occur in coastal and estuarial regions. The biological effects of marine eutrophication are most apparent with regard to primary producers, but the effects carry on up the food chain so that ultimately all biota are affected, either because of changes in food availability or because of other changes, e.g. oxygen deficiency, especially in stagnant waters such as below a pycnocline. The options for classification considered in this report are, however, restricted to primary producers. Algae are responsible for the great bulk of primary production in the waters around the British Isles, with both macroalgae and microalgae being involved (Wood 1987).

Macroalgae are limited to shallow areas where the substratum is suitably firm and light levels high. There are qualitative as well as quantitative changes in algal distribution in response to changes in illumination because different types of algae have specific requirements and utilise light of different wavelengths (Wood 1987). Green algae are generally restricted to shallow waters. The production of macroalgae can be high on marine rocky shores, although in estuaries populations of seaweeds tend to cover a very small proportion of the total area, being confined to rocky outcrops, quays and piers, etc. Nevertheless, the mudflats of estuaries can support profuse growth of some green algae such as *Enteromorpha* if nutrient levels are high and physical parameters are suitable.

Microalgae form films on all types of substrata and also occur in the interstitial spaces in sediments but a much greater range and number occur as phytoplankton. Diatoms and other microalgae occur in the upper 1 cm of estuarine mudflats with the largest populations occurring in the lowest part of intertidal areas. Some workers have also reported that there is no marked seasonal fluctuation in numbers and biomass (McLusky 1989). These microalgae can have a significant input into the production of estuarine ecosystems; within the the Lynher estuary it has been estimated that the primary production of epibenthic algae is comparable with phytoplankton in the overlying water (McLusky 1989).

Seagrasses (*Zostera* spp.) contribute to primary production in some areas. The relative importance of each of these components of primary production fluctuates from one area to another but there is little doubt that in nearshore areas kelp (*Laminaria hyperborea*) forests are of major importance.

4.1 Phytoplankton

Phytoplankton are often divided into one of two or three categories depending on their size: netplankton (which are all >20 µm): nannoplankton (some authors consider all phytoplankters below 20 µm in size to belong to this group, but others include only those falling into the 2-20 µm size category. This group consists predominantly of solitary flagellates, chlorophytes and diatoms); and picoplankton (<2 µm).

Whilst it is netplankton which are the most obvious algae in a marine water sample, it is more likely to be the nanoplankton which dominate the phytoplankton community in terms of biomass and cell numbers, with reported values of >80% for the contribution made towards the total phytoplankton productivity and total chlorophyll-a of coastal water samples being quite frequent (e.g. Malone 1971a,b, El-Sayed and Turner 1977, Bruno *et al.,* 1983).

Smaller phytoplankters tend to have faster nutrient uptake rates (per unit biomass) and growth (doubling) rates than larger phytoplankters, e.g. Friebele *et al.* (1978) found that nannoplankton accounted for 75% of the total uptake rate by a natural estuarine phytoplankton population, despite only making up 21% of the total algal biomass.

Eutrophication affects marine and estuarine phytoplankton in two ways: firstly, there is an increase in biomass; and secondly, a change in species composition. It is more difficult to say with any certainty when the latter occurs because of natural fluctuations in phytoplankton species composition, although areas of marine/brackish water of increasing nutrient status, are known to have undergone changes to their phytoplankton community, e.g. an increase in dinoflagellates and decrease in diatoms in the North Sea (Reid 1978), although this trend now appears to have stopped, and possibly reversed (Reid *et al.*, 1990).

Perhaps the greatest ecological damage that extensive phytoplankton blooms can cause occurs as a result of massive bloom formation as a result of ideal weather conditions, the growth of which is then limited by nutrient depletion. The bloom enters a stationary phase and as a result of nutrient stress, diverts carbon metabolism to the synthesis of extracellular polysaccharide (Jensen 1984). This may sink to the sea/estuary bed where the resulting microbial degradation subsequently results in massive deoxygenation of hypolimnetic waters, causing wide-scale benthic mortality. For other algae, e.g. *Phaeocystis* (Lancelot *et al.*, 1987), the factors controlling polysaccharide production/excretion are much less well understood. This alga forms dense colonial blooms which on dying can blanket intertidal and shallow inshore sediments with a slowly decomposing mucilage (Owens *et al.*, 1989). Fortunately, however, the resulting damage in UK coastal and estuarine regions has been very much less than that experienced in the Adriatic and off the Scandinavian coast, but blooms of other algae such as non-toxic dinoflagellates which excrete only a fraction of the organic carbon that *Phaeocystis* does, have also been implicated. in the wide-scale death of benthic invertebrates (Hepper *et al* 1974).

Phaeocystis pouchetti has been recorded in scum-forming densities at a number of sites around the UK, as well as along much of the west European shore, stretching from France to Denmark. Also, the south UK coast has been affected by blooms of *Gyrodinium aureolum* (responsible for toxic fish kills) and blooms of *Gonyaulax* sp. have caused toxic PSP events (Zabel and Miller, 1992).

4.2 Macroalgae

Sewage effluent is reported to enhance the growth of green macroalgae to a greater extent than brown seaweeds (e.g. Ford *et al* 1983, Sawyer 1985). The effects of such artificial fertilisation are most noticeable in sheltered coastal waters and estuaries, such as parts of the Firth of Clyde (Perkins and Abbot 1972). Also, a three-fold increase in the biomass of intertidal macroalgae in Dublin Bay over the period 1977-1987 has been attributed to more general eutrophication (ISSG 1990).

The major effects of eutrophication on macrophytic algae are twofold: decreased depth of penetration, e.g. *Fucus vesiculosus* in the Baltic Sea (Cederwall and Elmgren 1990); and an increase in the area and standing crop of opportunistic green algae where light is not limiting, e.g. *Enteromorpha* in the Ythan Estuary, Scotland (Raffaelli *et al.*, 1989).

A presumed 2-3 fold increase in nitrogen levels in the river feeding the Ythan Estuary between 1960 and 1984 has led to a large area of previously exposed intertidal sediment becoming colonised by mats of the green alga *Enteromorpha*. In areas where standing crop has exceeded 1 kg wet weight m^{-2}, there is a significant decline in the amphipod *Corophium volutator*, an extremely important food item of wading birds, and at standing crops above 3 kg wet wt m^{-2} *Corophium* disappears (Hull 1987).

In areas within the photic zone which have a high input of nutrient-rich ground water, this may stimulate growth of benthic macroalgae (and epiphytic microalgae), with less of an effect than would have been predicted had the land-derived nutrient load been emptied into the coastal waters as a fluvial discharge (e.g. Sewell 1982).

The increased deposition on the bottom of organic matter derived from algal production can, if vertical mixing is not sufficient, increase the sediment oxygen demand to a point where deoxygenation of the bottom waters takes place, so causing mortality of large areas of invertebrates (Cederwall and Elmgren 1990).

Occurrences of filamentous algae (mainly *Cladophora* species) and sea lettuce (*Ulva lactuca*) have increased in some coastal waters around Denmark, particularly in enclosed waters with high nutrient loads. In many areas perennial macrophytes have disappeared completely (FCC 1991). The composition of plant species found on the sea bottom and the areas they occupy have changed significantly since the turn of the century. The eelgrass (*Zostera* spp.) which covered areas in coastal waters have been reduced by half because the depth to which sunlight penetrates has been reduced. The depth limit for eelgrass in open waters has been reduced from 9-10 metres in 1890 to 5-6 metres today and in fjords from 6-7 to 2-3 metres. For other forms of seaweed the depth limit has been reduced from 30-35 metres to 10-12 metres during the same period.

5. FACTORS AFFECTING TROPHIC STATUS

5.1 Nutrient enrichment

Increasing the nutrient status of tidal waters (nutrification) does not imply a *pro rata* effect on algal growth, since nutrients have to be the growth-limiting resource for this to occur. Whilst a single classification scheme for both coastal and estuarine waters may be desirable, it should be realised that other factors make the two types of environment very different. Water clarity tends to be very much lower in estuaries than coastal waters due to the higher levels of suspended sediment, the pattern of which follows the diurnal tidal cycle. The higher nutrient levels and reduced light penetration in estuaries mean that light, rather than nutrients, is more likely to be the major factor limiting algal growth, especially when half-saturation constants for nitrogen uptake are so low (typically 0.1-14 μg N l^{-1} for freshwater and marine phytoplankton; Barnes and Hughes 1988, Reynolds 1984). Phytoplankton blooms have been shown to occur naturally in saline water of low nutrient status. However, a large body of evidence shows that in saline waters, phytoplankton standing crop is increased following anthropogenic inputs of nutrients to semi-enclosed regions of water, even in some estuaries with high levels of suspended sediment. In addition, large standing crops of maroalgae such as kelp can develop in relatively nutrient poor waters. Factors such as a suitable substrate might be much more important for the growth of macroalgae than nutrient supply.

However as nutrients are only a precursor of algal growth, it is not surprising that in other examples chlorophyll and nutrient levels show little correlation. Warren *et al* (1992) present an example of this using data from the Adriatic Sea (effectively a huge stratifying estuary).

5.2 Nutrient dynamics

Nutrient dynamics in coastal and estuarine waters are controlled by a number of factors (Aston 1980, Kennish 1986), including:

1. The diurnal tidal cycle produces short-term temporal changes in the contributions of nutrients from land and sea.

2. Nutrient levels vary sinusoidally on an annual basis (e.g. Radach 1984), with nitrogen and phosphorus reaching minima in May/June, and maxima in November/December in temperate coastal waters.

3. Circulation and stratification may cause vertical and horizontal differences in nutrient concentrations.

4. Some topographic features (e.g. fjords) may reduce mixing between freshwater inputs and coastal waters.

5. The pattern of sediment deposition and resuspension influences nutrient budgets. Nutrients may dissolve or be deposited as a result of chemical reactions when

(desorb)

be realesed into solution

5. Adsorption / desorption at freshwater - salt water interface.
6. Microbial decomposition of organic matter releases orthophosphate into pore water. May adsorb onto iron oxides at aerobic sed - water interface.

freshwater mixes with seawater. The behaviour of phosphorus in sediment-water systems is dependent to a large extent on redox potential (E_h), an indicator of which is taken by many workers to be dissolved oxygen levels. At low E_h/DO levels, the equilibrium between ~~solid~~ adsorbed and dissolved phosphorus swings to the dissolved side of the equation, with the effect that formerly non-bioavailable particulate phosphorus adsorbed becomes bioavailable dissolved phosphorus.

6. The uptake and release of nutrients by growing and senescent algae. Because nutrients are sequestered by phytoplankton, high levels of dissolved inorganic nitrogen and phosphorus tend to coincide with low chlorophyll levels and *vice versa*.

7. Sediment-water bacterial ecology (nitrification, denitrification and factors affecting these processes). Ammonia is derived from the bacterial breakdown of organic nitrogen, and is itself converted to nitrite and nitrate by nitrifying bacteria. These are the three major nitrogen sources for algae, though some low molecular weight organic compounds such as urea and some amino acids may also be taken up and metabolised. In coastal areas and sediments, ammonia may be denitrified (via NO and N_2O) to form nitrogen gas (N_2). Denitrification is a major cause of nitrogen loss in some estuaries, e.g. Seitzinger (1987) calculated that in Ocholocknee Bay an equivalent 54% of the river input of dissolved inorganic nitrogen (DIN) was removed in this manner.

Despite the fact that the factors controlling nutrient dynamics in estuaries and coastal waters are very similar, the relative importance of these factors is different enough to ensure that the behaviour and distribution of macronutrients is far more variable in estuarial waters than in the coastal environment. In estuaries, nutrient concentrations are regulated by complex biological processes as well as a dynamic hydraulic regime which is dependent on weather conditions (Pritchard and Schubel 1981). For example, data presented by FCC (1991; see Figure 5.1) shows no correlation between wind velocity and inorganic phosphorus levels during winter in Danish coastal waters. Although strong winds are often associated with heavy rainfall, which in turn is associated with increased nutrient run-off from land, the increased nutrient levels in coastal waters are not due to greater run-off of nutrient-rich water from land, since salinity actually increases as wind velocity rises (Figure 5.2). Rather, the pattern appears to be related to the increased depth of mixing in coastal and estuarial waters so that phosphorus-rich interstitial water is mixed into the main body of water above the sediment. Thus, a plot of inorganic phosphorus concentration versus salinity shows a positive correlation (Figure 5.3), not the negative correlation which a conservatively mixed estuary would show.

5.3 Sources of nutrients

Data for quantifying the nutrient input to the marine environment around the UK will necessarily have a high degree of uncertainty associated with it, but a clear picture emerges that the major source of nutrients in UK coastal waters is the Atlantic Ocean, which annually contributes over 100 million tonnes of nitrogen and 28 million tonnes of phosphorus to the Irish Sea. This compares with a yearly input of just 93 580 tonnes of

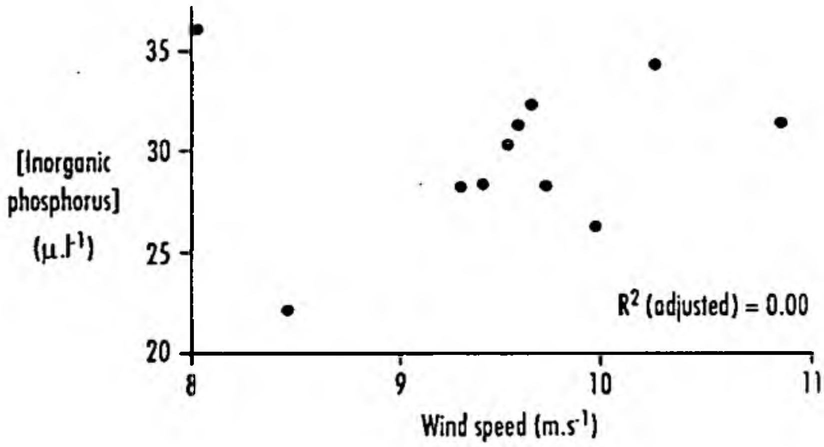

Figure 5.1 Relationship between winter wind velocity and inorganic phosphorus concentration in Danish coastal water (data from FCC 1991)

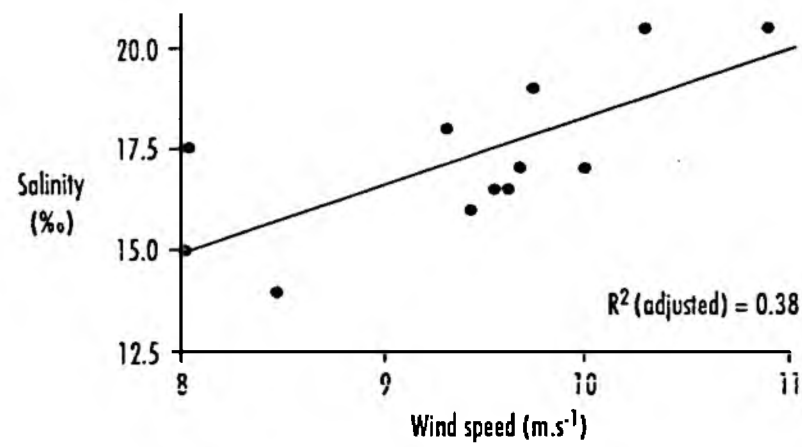

Figure 5.2 Relationship between winter wind velocity and salinity in Danish coastal water (data from FCC 1991)

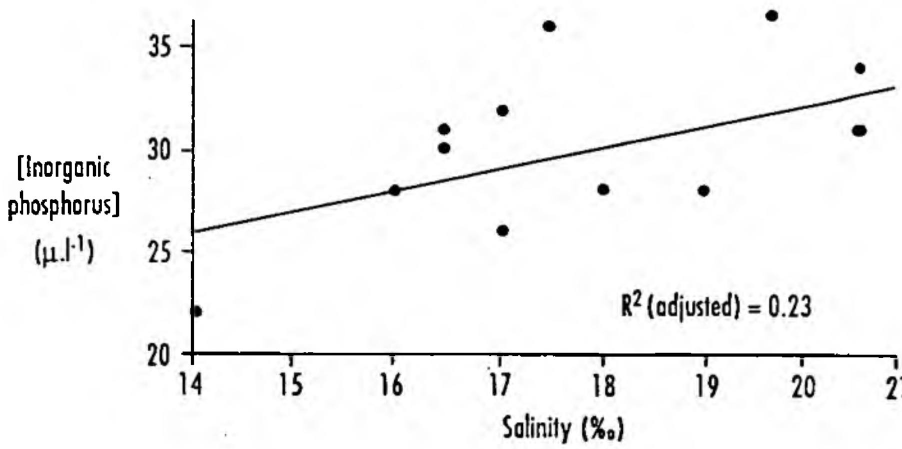

Figure 5.3 Relationship between winter inorganic phosphorus concentration and salinity in Danish coastal water (data from FCC 1991)

nitrogen and 22 010 tonnes of phosphorus from the UK and Ireland (i.e. less than 0.1% of the total nutrient input; Table 5.1).

Table 5.1 Nutrient inputs to the Irish Sea (tonnes/year; ISSG 1990)

Source	Nitrogen	Phosphorus
Atmosphere	43 000	2 000
Rivers	76 400	6 120
Domestic	10 700	1 910
Industry	2 640	16 260
Sewage Sludge	3 840	720
Atlantic	100 000 000	28 000 000

5.4 Ratio of nitrogen to phosphorus

The general value most often given for the ratio of nitrogen to phosphorus in algae is 16:1 (on an atomic basis) or 7.2:1 (on a weight basis). However, this ratio can vary a great deal both between species, and within a single species, depending on the state of growth of the bloom and the levels of other nutrients. This ratio is often employed as an indicator of which of these macronutrients is limiting - in freshwaters where the ratio is usually greater than 7.2:1 (w/w), phosphorus is usually considered to be limiting; but in the marine environment generally (e.g., Goldman *et al.*, 1973, Ford *et al.*, 1983, FCC 1991) and in coastal waters more particularly, nitrogen is more likely to limit algal growth. This emphasis on nitrogen being the limiting nutrient in coastal waters is due partly at least to the loss of nitrogen by denitrification, primarily by sediment bacteria (some 15-75% of the sediment-water nitrogen flux is accounted for by N_2; Seitzinger 1987). The one major exception to nitrogen limiting algal production in the marine environment appears to be the Mediterranean Sea (Berland *et al* 1980). In estuaries a point is reached where the excess nitrogen from the freshwater input is countered by the excess phosphorus from the marine inflow and an ideal nutrient ratio is reached for algal growth.

However, too much emphasis on this ratio may lead to misunderstanding since nutrients are not the only factors which can limit growth, and in the majority of cases, it is one of these other factors which limits primary productivity (light and temperature) and/or biomass (grazing pressure). In estuaries in particular, light is often the single factor most limiting to algal growth, but the fact remains that in estuaries where the nutrient status increases, intertidal/subtidal macrophyte biomass often increases as a result, as does the frequency and/or severity of phytoplankton blooms.

5.5 Other factors affecting algal production

5.5.1 River velocity and estuarine flushing/residence time

High nutrient levels in the upper reaches of estuaries are usually associated with catchment runoff during high river flows while high concentrations in the lower reaches tend to be associated with sediment remineralization and oceanic exchange. In some estuaries phytoplankton levels follow nutrient concentrations so that chlorophyll levels follow conservative nutrient dynamics (Gibbs *et al.*, 1992), but in others, high flushing rates do not allow sufficient time for phytoplankton to grow before being swept into the lower reaches or out of the estuary altogether. In yet another group of estuaries, high flushing rates from both the river and from tidal intrusions, mean that chlorophyll maxima occur in the middle reaches of such estuaries (e.g. Campbell *et al.*, 1991). Even taking the freshwater input into account, tidal amplitude can have an overwhelming effect on phytoplankton standing crop (Monbet 1992).

In temperate estuaries, large seasonal fluctuations occur in phytoplankton standing crop, but this becomes less pronounced as ambient temperature and solar radiation increases, so that in subtropical countries such as South Africa, spatial variance in phytoplankton standing crop is greater than temporal variance (Campbell *et al.*, 1991).

In the River Thames, the critical mean discharge for algal bloom formation appears to be 0.48 ms^{-1}. (Reynolds 1988), a value very similar to the maximum velocity at which Gessner (1955) concluded that true phytoplankton could be supported. However, Bowles and Quennell (1971) found that the spring diatom bloom in the lower Thames occurred only when the mean velocity was <0.3 ms^{-1}, with peak algal levels occurring at velocities of <0.1 ms^{-1}.

The potential for a major bloom occurring increases as nutrient levels increase, and also as the retention or flushing time of an estuary increases. As the doubling times for estuarine/marine phytoplankton are typically in the order of 3-4 days, estuaries with retention times of less than 1 to 2 weeks are likely to have a much decreased risk of major algal blooms, though outside the estuary mixing zone the risk of blooms may still be high. The retention time of an estuary is dependent on freshwater flow. For example, the retention time of the Forth estuary under mean river flows is 12 days, but in summer with reduced river flow it may be as much as 10 weeks (McLusky 1989). This, along with the higher temperatures and higher incident light levels would increase the potential for an algal bloom. Other estuaries would have much shorter retention times (e.g. 4 days for the Mersey estuary; Hughes 1958) which means that such effects are unlikely to occur therein.

The flushing time of many British estuaries is of the order of a few days, e.g. the Mersey Estuary 4 days (Hughes 1958), whereas sea lochs tend to be a little longer, e.g. Loch Striven, Loch Etive, Loch Ewe have residence times of at least a week (Edwards and Edelsten 1976, Landless and Edwards 1976, Gowen *et al.*, 1983). However, as with such generalisations, there are always exceptions to test the rule (and find it lacking), e.g. Loch Ardbhair (1-2 days, Gowen *et al.*, 1983). The shape of the loch also makes a great deal of difference; for example, the Inner Clyde lochs tend to have narrow mouths and long basins so exchange with the sea is restricted which makes the lochs more susceptible to algal bloom formation. In addition, many sea lochs have a pronounced shallow bar across the mouth which makes them more prone to stratification, e.g. Loch Etive is 120 m deep but has an entrance sill which rises to 5 m below sea level.

5.5.2 Turbidity

Most estuaries (e.g. the Shannon and Ems-Dollard estuaries) contain a gradient of suspended solids of autochthonous and allochthonous origin, increasing upstream from the estuary mouth (Colijn 1982, McMahon *et al.*, 1992). In estuaries of short retention time in particular, turbidity is strongly related to tidal current velocity and follows a diurnal pattern, whilst this relationship is weaker in estuaries where the retention time is long enough to allow blooms to form. Light is often regarded as the single most important factor limiting phytoplankton biomass, (and therefore chlorophyll levels) in estuaries (e.g. McMahon *et al.*, 1992). Indeed, the nutrient levels observed in UK estuarine waters are usually much higher than the half-saturation constants for nitrogen uptake, typically 0.1-14 $\mu g\ N\ l^{-1}$ for freshwater and marine phytoplankton (Reynolds 1984, Barnes and Hughes 1988).

In coastal waters, the photic zone extends to a greater depth, so a closer relationship between nutrient levels and chlorophyll-a levels would be expected, though the data to prove such relationships is not available. The infrequent sampling programmes for tidal waters are rather 'hit and miss' with regard to whether a bloom is recorded or not, even when bimonthly sampling is practised. To be certain of recording close approximations of maximum and minimum chlorophyll levels, weekly sampling as practised in many freshwater reservoirs is required; though, clearly, this is prohibitively expensive. Moreover, chlorophyll-a is only a crude analogy of phytoplankton biomass (see Sections 6.2.1 and 6.2.4).

5.5.3 Stratification

Another physical process which makes the biological response within estuaries and coastal waters to nutrification different is that of stratification. Some estuaries and, more notably, sea lochs stratify during warmer months, initially because of a temperature differential which results in less dense surface waters; but the even lower density freshwater input may then mix only with the water above the thermocline, so reducing its density further and stabilising the pycnocline. In coastal and oceanic waters a thermocline may develop during periods of calm weather, but this is not stabilised by the epilimnion being diluted. However, the greater depths of these waters mean that greater wind speeds are required to fully mix the water column than for estuaries.

Indeed the depth of mixing has to be considered carefully when considering the effects of turbidity: for example, although turbidity levels were highest in the upper reaches of the Shannon Estuary, because the depth of mixing was low, less light limitation of phytoplankton occurred there than in the lower reaches (McMahon *et al.*, 1992). Because nutrients are sequested within algae in the photic zone, which then sink, levels of dissolved nutrients are much lower in surface waters when stratification (thermal or density) occurs. However, along coasts, on banks and at the frontal zones separating shallow stratified and unstratified areas, areas of upwelling can bring nutrient-rich bottom water to the surface. If the water then restratifies, this makes ideal conditions for algal bloom formation in surface waters.

5.5.4 Salinity

All algae grow more efficiently at some salinities than others. This limits the distribution of all algae so that some are confined to freshwaters, others to brackish or marine waters and yet others to hypersaline environments such as stromatolites or saline lakes. For this reason an algal bloom occurring in a large lowland river is unlikely to act as an inoculum for a lower estuarine or coastal water bloom downstream, since the cells will die or have their growth rate inhibited to such an extent that they are succeeded by other algae.

In the same way that algal metabolism is affected by salinity, so are bacteria. The bacteria responsible for nitrification and denitrification therefore tend to be distributed unequally down estuary salinity gradients. Thus, Furumai *et al.*, (1988) measured peak concentrations of ammonium, nitrite and nitrate at salinities of 5, 17 and 30 ppt, respectively.

6. OPTIONS FOR ASSESSING TROPHIC STATUS

6.1 General sampling requirements for options

Each of the options discussed in this section involves a level of sampling for the determinand selected to be indicative of the trophic status of the body of water. There are, therefore, general statistical and sampling considerations common to each. This is discussed in this subsection before more detailed descriptions of the main options.

Sampling and monitoring for trophic status would have to cope with the inherent variability associated with tidal waters particularly in estuaries and to some extent inshore coastal water (Section 3). Past experience has shown that, even when estuarine water quality is expressed as a simple overall mean or percentile, it is difficult to do sufficient sampling to provide an adequately precise quality assessment (for example against a defined standard) except in the most clear-cut of cases. There are two related reasons for this:

1. Given the complexity and size of spatial and temporal variations typically to be expected in estuarine quality, any approach relying solely on random sampling theory is very likely - at present or foreseeable future sampling levels - to produce unacceptably large errors.

2. The problem is not simply the likely number of samples required. The major spatial and temporal quality variations will manifest themselves over time periods as short as a tidal cycle. Thus, to meet any worthwhile objective over so compressed a timescale will call for an intensity of sampling that will be very difficult to provide without the use of some form of automatic monitoring.

So, to repeat the point, these practical problems apply even in assessing compliance with the present, simply-defined estuarine quality standards. Given the scale of this variability any monitoring approach relying solely on random sampling theory is very likely to produce unacceptably large errors. There are potentially two approaches to addressing this difficulty: the use of a structured sampling programme, and the use of statistical/mathematical models. To devise the latter, the former would have to be undertaken.

If properly designed, a structured sampling strategy may be able to reduce much of the variability observed in tidal waters and especially in estuaries. In this way comparable standardised data would be obtainable for classification purposes. Ideally, the strategy would be designed from an assessment of the relative variability associated with the factors affecting the selected determinand such as chlorophyll levels. This might entail a detailed examination of existing datasets (if they are comprehensive enough) or from an intensive monitoring programme over a number of years and sites: lengthy datasets can be statistically analysed to remove seasonality. The programme would need to be designed to study all the factors likely to affect the selected determinand and would attempt to assess the relative size of sources of variation such as seasonality, tidal stage, depth, position in the estuary, sampling depth etc. Such an assessment of variability could be used to determine the sampling frequency (number of samples, scans etc.) required to achieve a desired precision in classifying tidal waters. As an example the source of the variability associated with the measurement of chlorophyll in a Scottish Sea Loch has been examined (Freeland *et al.*, 1980). It was concluded that method error was responsible for only a very small part of the total variation. Small-scale patchiness contributed more but most of the variability about the generalised annual cycle of phytoplankton biomass came equally from year-to-year variation and spatial variability on a scale of a few kilometres.

As indicated earlier in Section 3 consideration would have to be given to whether the maximum determinand values were required, annual average values or averages over a rolling number of years. If the ultimate aim of the classification (in conjunction with SWQOs) is to identify the most eutrophic waters and those that might be subject to events (such as blooms) that seriously compromise the uses of bodies of controlled waters, then the detection of maximum productivity levels may be of more relevance. However such catastrophic effects may only occur periodically and at unpredictable times when other factors (such as light) are optimum, and when sampling or measurement may not be possible (especially for remote sensing techniques).

IMPORTANT

Sampling frequency and how this relates to the development of algal biomass and the duration of peak production within tidal waters must, therefore, be considered. For example, Weisse *et al.* (1986) reported a spring diatom bloom (April to May) in the German Bight lasting 29 days. This was followed in May and June by a bimodal *Phaeocystis* bloom which lasted for approximately 36 days. Between 1978 and 1988 significant blooms of *Phaeocystis* occurred in the Eastern Irish Sea during 8 of the 11 years (Irish Sea Study Group 1990). The duration of the blooms ranged from just 2 days to 49 days, with an average of 22 days. Over the same period there were 3 years when diatom blooms occurred and 2 years when dinoflagellate blooms were present. Measurements of *Chrysochromulina polylepsis* (a unicellular flagellate) undertaken off the east coast of Denmark indicated that cell numbers could change dramatically within and over a number of days (CEC 1989). For example, 40 million cells per litre could be found on one day, zero cells a few days later but increasing again up to 5 million cells per litre. In addition, maximum cell numbers were always found in a very narrow layer of water of less than 1 m, often close to the pycnocline at around 9 m. This later case also highlights the possible need for sampling throughout the water column rather than just at the surface.

In contrast the development of green macroalgae such as *Ulva* is relatively slower. For example, Menesguen (Barth and Fegan 1990) reports that along the Brittany coast there is an initial rapid growth in April and May with peak biomass occurring at the end of June.

The above would suggest that sampling would ideally be required at least weekly during the main growing season (approximately beginning of April to end of October), with some additional assessment of resting or ambient-levels (March and November), in order to obtain a reasonable estimate of algal productivity. Measurements would also be required throughout the water column in waters that might be stratified in any way. This level of sampling is likely to be prohibitively expensive for universal application: a compromise between accuracy and affordability might be monthly samples between March and November (9 surveys).

Spatial coverage of a sampling programme would also be important. As indicated above algal blooms can develop in one area of coast and then progressively move to other areas. This might reflect the same body of water moving with the residual sea currents and/or it might reflect the seeding of other geographic areas. The spatial unit of classification, particularly in coastal waters, required would, therefore, have to be considered.

It is likely that there will be more tidal variability of determinands such as chlorophyll within estuaries than in coastal waters. As for the proposed nutrient classification scheme (Rees *et al.*, 1992) specific times of sampling might have to be defined. Surveys undertaken on east coast estuaries indicate, for example that chlorophyll-a may vary up to 6-fold at any one location over a tidal cycle (Wheeler 1991). Depending upon where the main primary production is occurring within an estuary (e.g. in the more turbid, nutrient-rich headwaters or in the less turbid relatively nutrient poor downstream waters) allowance may have to be given to salinity to account for dilution effects.

If macroalgal biomass was to be used for assessment of trophic status then it is likely that sampling would have to be undertaken at low water when there will be maximum exposure of algae for sampling. Benthic green macroalgae, which appear to respond more vigourously to nutrient enrichment than brown macroalgae, are more likely to be found in the intertidal and shallow water areas, than in deep subtidal waters where remote sampling would probably be required.

The following options for assessing trophic status are discussed in more detail in the following subsections, including an assessment of the advantages and disadvantages of each, and a description of a possible basis of a classification on that option, and on the availability of data for developing the classification:

- discrete sampling for chlorophyll and phytoplankton biomass;
- *in situ* fluorimetry;
- monitoring for nutrient levels;
- macroalgal biomass;
- the use of statistical models;
- remote sensing from aircraft;
- algal growth bioassays.

6.2 Discrete sampling for chlorophyll and phytoplankton biomass

6.2.1 Methodology

Coulter counter

Coulter counters are useful because they measure both intracellular volume and number of particles (i.e. plankters). However, with the exception of multisizers, they are probably best suited to measuring particles of a similar size (which in effect means phytoplankton populations dominated by a single species) because the size of the aperture in the electrode determines the size range of solids which can be measured/counted.

Providing the size range of individual species does not overlap to a large extent, multisizers can be used to determine the volume and number of different species. Coulter counters are not suitable for measuring filamentous or colonial algae, only unicellular species, and while they are suitable for measuring larger phytoplankton, the 'noise' from bacteria precludes their use with picoplankton. Coulter counters measure the volume inside the plasma membrane only, so take no account of elaborate frustules or cell walls which may comprise a large proportion of the phytoplankton biomass. Problems may also be encountered with the electrode aperture becoming blocked.

Flow cytometry

This techniques allows the measurement of phytoplankton pigments through its fluorescence and other optical properties on an individual cell basis. Populations of phytoplankton can be recognised by characteristic flow cytometric 'signatures' made up of groups of correlated parameters such as light scatter and fluorescence at specified wavelengths (Sosik *et al.*, 1989).

Flow cytometry is a widely used analytical technique in biology, clinical medicine and oceanography but it has not been widely applied or tested for water quality and environmental monitoring (Harman 1992).

It is capable of rapidly counting and sizing waterborne particles whilst simultaneously discriminating between phytoplankton, zooplankton and non-living material. Some research laboratories have flow cells that are capable of handling colonial algal species but they are not normally found on commercially available equipment. Flow cytometry is used in a variety of ways with sizing, autofluorescence, cytochemical stains and immunofluorescent probes (Yentsch 1990). The instrument can be used shipboard as well as onshore.

The large size and high cost (£16 000 to £70 000 depending upon specification) of present commercial flow cytometers, lack of information about the optical properties of freshwater phytoplankton and problems associated with handling the wide range of algal cell and colony sizes present in natural waters are contributing factors affecting the application of flow cytometry in this area (Harman 1992).

Utermöhl settling technique

This technique must be undertaken on land and cannot be done on board ship since it requires the sample to be kept still. It is widely used for larger microplankters, but controversy remains over its use to quantitatively estimate numbers of nanoplankton and smaller algae. It is time-consuming and quantitative results are inaccurate with regard to nanoflagellates, but more especially with respect to picoplankton. These groups tend to be underestimated (Reid 1983).

Nucleic acid content

This is a rather complicated and non-automated (i.e. expensive) method of analysis for routine plankton biomass determination. The method is only known to have been used in selected studies, but nucleic acid content does appear to correlate with biomass, since the protein content of cells is directly proportional to cell volume and the RNA content of cells is directly related to protein synthesis. However, larger phytoplankton cells would only be expected to contain a similar amount of DNA as smaller cells. The method has been used primarily in academic studies rather than as a routine biomass monitoring technique, and this is where it is most useful.

ATP content

In phytoplankton, as in higher plants and animals, ATP is the energy currency of cells and as such the size of the intracellular ATP pool is indicative of the general 'well-being' of cells, as well as the biomass. Nevertheless Bondarenko *et al.* (1991) found a good relationship between ATP content and phytoplankton biomass. Other workers have found similar trends for both natural phytoplankton samples and laboratory-grown algal cultures, though perhaps with relationships that are not quite as strong when other biomass parameters are used. For example, Brezonik *et al.* (1975) obtained correlation coefficients of 0.244 - 0.860 between dry weight and ATP content of four laboratory grown algal cultures. For the same algae, ATP/chlorophyll-a correlation coefficients were in the range 0.539 - 0.725. The correlation between ATP content and biomass as measured by cell counts, chlorophyll, etc. would be expected to be weaker in natural samples than in axenic laboratory grown cultures, since the ATP content of bacteria is also included in the analysis. As with nucleic acid content, ATP analysis is a lengthy, non-automated and expensive process.

In vitro chlorophyll-a analysis

Here chlorophyll is used as a surrogate for phytoplankton biomass. The standard method for chlorophyll analysis entails the initial extraction of chlorophyll from particulate material quantitatively and without degradation. The chlorophyll in the extract is then determined without interference from related pigments, in particular the phaeophytins. The phaeophytins are often termed 'dead chlorophyll' and are the degradation products of chlorophyll which do not undertake photosynthesis. Standard methods involve a filtration, extraction and analysis step.

Chlorophyll-a is the only pigment found in almost all algae. The exception to this rule are a very small number of non-pigmented heterotrophic algae (e.g. *Probothecu*, a colourless relative of the green algae). As such, chlorophyll-a content should be expected to provide an adequate surrogate for total algal biomass. Several factors can greatly influence the amount of chlorophyll per cell (Richardson *et al.*, 1983), a criticism which can be levelled at all methods relying on chlorophyll determination, whether *in situ* or in discrete samples. Integrated samplers (effectively a long tube or pipe, *ca* 5 m long, which is lowered into the water, the top end sealed, and then withdrawn), although more often used in freshwater, offer the same major advantage over conventional discrete sample collection in the marine environment, i.e. they collect a more representative sample than conventional discrete samplers.

Fluorescence as a method of measuring chlorophyll first gained favour in the 1960s after Lorenzen (1966) demonstrated that it could be used use for continuous measurement, so negating the requirement for discrete sample collection. However, numerous factors have since been found to affect the consistency of *in vitro* fluorescense (IVF) results, including species, environment and photoinhibition at high light intensities (Vandevelde *et al.*, 1988).

6.2.2 Basis of classification

The classification could be based upon the range of observed chlorophyll concentrations in tidal waters of England and Wales, with class boundaries calculated using simple statistical rules to divide the frequency distribution into suitable groups: this approach formed the basis of the nutrient classification proposed by Rees *et al.*, 1992.

The classification could have an extra component that attempted to divide sites/areas/estuaries into broad categories reflecting the production status prior to anthropogenic inputs. This classification may take the form of two parallel classifications one describing the current situation and the other describing the 'natural' situation. Alternatively the natural condition could be integrated with the current condition to present one classification. A potential problem is that if the natural variation in chlorophyll production was not sufficiently accounted for, the potential for comparing the level of anthropogenic contamination of different sites would be reduced.

6.2.3 Availability of data

The review of factors affecting trophic status (Section 5, this report), shows that while there is a large investment in research, data suitable for forming the basis of a classification scheme is not to be found in the published literature. In general, data are patchy and restricted to small geographical areas, or not directly relevant to the British coastal waters under the statutory control of the NRA.

Possible sources of unpublished data on levels nutrients have recently been approached by WRc (Rees *et al.*, 1992), in relation to nutrient classification. This information has been updated and augmented to take account of the range of possible approaches for the assessment of trophic status. The time scale of the project did not allow a comprehensive list to be compiled and information from some areas may be incomplete. Potentially one of the major sources of data are the NRA tidal waters surveys, though here again it is unlikely that data on which to base a trophic classification scheme is comprehensive enough. The reason for this is that the vast majority of NRA sampling is driven by legislation and is thus collected on a need basis.

The available data are summarised in Appendix A. Routine monitoring by the NRA and RPB's has included chlorophyll analysis in some regions. Regions which appear to have the most extensive programmes are Anglian, Northumbria, South West and additionally Clyde RPB in Scotland. A summary of this data, mainly relating to Scottish waters, is presented in Table 6.1.

Table 6.1 Summary of chlorophyll ($\mu g\ l^{-1}$) data from Scottish waters and from the Colne estuary, Anglian Region

Survey Area	Month	Year	CHLOROPHYLL Salinity zone (ppt) <20	20-30	>30
Colne	Sep	90	79.2	14.95	4.11
	Oct	90	3.65	6.1	0.48
Firth of Clyde	May	89	*	*	1.3
	Jul	89	*	*	0.37
	Aug	89	*	*	0.36
	Sep	89	*	*	7.52
	Oct	89	*	1.06	0.53
	Oct	89	*	*	0.33
	Nov	89	*	0.62	0.51
	Mar	90	0.26	*	*
	Mar	90	0.33	0.1	0.07
	Apr	90	*	3.61	1.57
	Apr	90	*	7.55	8.2
	Apr	90	*	*	5.64
	May	90	*	*	0.5
	Jul	90	*	*	3.16
	Jul	90	*	*	2.26
	Oct	90	*	0.51	0.18
	Oct	90	*	0.49	*
	Oct	90	*	0.3	*
Gare Loch	Apr	89	*	3.4	*
	Jul	89	*	*	0.78
	Oct	89	*	0.41	0.27
	Mar	90	0.23	*	*
	Apr	90	*	1.43	*
	Jul	90	*	0.41	*
	Oct	90	*	0.6	*

Table 6.1 Continued...../2

Survey Area	Month	Year	CHLOROPHYLL Salinity zone (ppt)		
			<20	20-30	>30
Holy Loch	Apr	89	*	0.73	*
	Jul	89	*	*	0.12
	Oct	89	*	0.36	*
	Mar	90	0.2	*	*
	Apr	90	*	*	5.82
	Jul	90	*	*	0.46
	Oct	90	*	0.33	*
Irvine Bay	Apr	89	*	5.33	0.76
	May	90	*	*	4.52
	Aug	90	*	*	1.39
Loch Fyne	Mar	90	*	*	0.27
Loch Goil	Apr	89	*	1.47	*
	Jul	89	*	*	0.68
	Oct	89	*	0.66	*
	Mar	90	0.15	*	*
	Apr	90	*	14.1	*
	Jul	90	*	*	5.12
	Oct	90	*	0.54	*
Loch Long	Apr	89	*	1.9	*
	Jul	89	*	*	1.98
	Oct	89	*	*	0.65
	Mar	90	0.39	*	*
	Apr	90	*	9.89	*
	Jul	90	*	*	4.94
	Oct	90	*	0.85	*
Striven	May	89	*	*	1.09
	Sep	89	*	*	1.12
	Oct	89	*	*	0.43
	May	90	*	*	3.55
	Oct	90	*	0.55	0.36
Number			8	27	36
Mean			10.6	2.90	1.98
Median			0.30	0.73	0.77
Minimum			0.15	0.10	0.07
Maximum			79.2	14.95	8.20
First quartile			0.21	0.49	0.39
Third quartile			2.84	3.61	3.45

The data in Table 6.1 represents the average chlorophyll found in a particular salinity band for individual surveys undertaken in the 'growing' season i.e. March to November.

Also presented in the table are summary statistics for each salinity band. It can be seen that there is a wide range of chlorophyll concentrations found with relatively high levels being found in the Colne estuary. Levels in the Firth of Clyde varied between 0.07 $\mu g\ l^{-1}$ to 8.2 $\mu g\ l^{-1}$ in the >30 ppt salinity band during 1990. This gives an indication of how difficult it might be to set threshold values and to determine class in a classification scheme based on chlorophyll.

6.2.4 Advantages and disadvantages

All of the techniques described above rely on the taking of discrete water samples and the subsequent determination directly or indirectly of chlorophyll concentrations or phytoplankton biomass. The determinations can be undertaken in a laboratory or possibly on board the sampling vessel. The main advantage of these methods is that reasonably precise and accurate results can be obtained under laboratory controlled conditions for a given sample. In addition, samples could be taken during sampling cruises for other determinands and might not entail extra cost. However, like all discrete sampling methods the question is how many samples and at what frequency would be required to obtain the desired level of precision in classifying tidal waters based on these parameters.

Because phytoplankton can grow or die, chlorophyll is not necessarily a conservative property in tidal waters (Tett 1987). A plot of chlorophyll against salinity may indicate whether a distribution of phytoplankton results mainly from physical properties, such as mixing between two water masses of differing salinities and chlorophyll concentrations, in which case the plot will be a straight line, or whether local growth or grazing are important in which case there will be deviations from a straight-line plot.

An example of such a plot is given in Figure 6.1 where salinity is plotted against chlorophyll concentration at different tidal stages in the Colne estuary (Wheeler 1991). It can be seen that generally higher chlorophyll concentrations were found at lower salinities. At the low salinity sites the highest levels of chlorophyll were found around mid-flood tide and lowest at around high water. This again indicates how variable chlorophyll may be in relation to salinity and temporally at fixed geographic locations.

Even though the most practically effective method for the estimation of phytoplankton biomass is still that of measurement of photosynthetic pigment there are still shortcomings. For example, the chlorophyll:biomass ratio is variable, ranging from 20 to 100 mg carbon per mg chlorophyll, and there is, because of variation in cell sizes and samples, often only a weak correlation between chlorophyll and phytoplankton numbers (Tett 1987). For freshwater species the range appears to be even wider than this, with differences of an order of magnitude between extreme species (Reynolds 1984). Very large differences can also be found within a single species, notably the change that occurs when algal productivity is highest prior to peak chlorophyll concentrations being measured and the situation which may occur perhaps a fortnight later during bloom senescence.

As indicated elsewhere in this report, (Section 8) it is suggested that sampling would have to be undertaken at a minimum of monthly intervals during the growing season to ensure

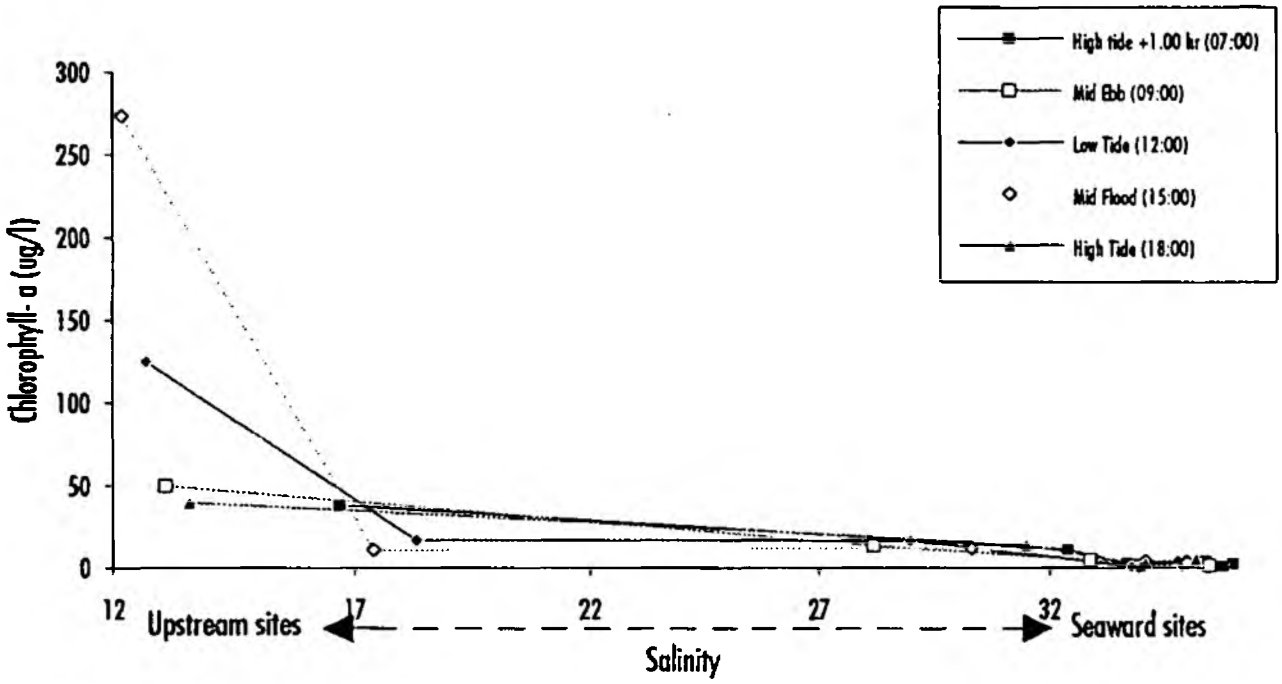

Figure 6.1 Relationship between chlorophyll and salinity Colne estuary, September 1990.

that an adequate representation of algal productivity was obtained. Some of the variability associated with chlorophyll concentrations could perhaps be removed by stipulating specific sampling times or windows in relation to the tide: this could be more fully assessed through examination of a suitable database of chlorophyll concentrations. The required surveys could also be incorporated to some extent in current NRA monitoring programmes though a more extensive temporal and spatial coverage would be required than is currently undertaken.

Implementation of this option would potentially entail a large amount of sampling effort. The cost of implementing such a sampling programme in the tidal waters of England and Wales has been estimated using cost figures used in previous NRA reports (Nixon 1991 and Pentreath 1992), and is given in Table 6.2. The assumptions used at arriving at the costs in the Table are also given.

Table 6.2 Estimated costs of implementing discrete sampling for chlorophyll and phytoplankton biomass option

Number of locations	Samples per site	Survey frequency (£'000)	Sample cost (£'000)	Boat costs (£'000)	Staff costs (£'000)	Total
Estuaries						
178	15	9	108	954	135	1197
Coastal waters						
Inshore						
430	5	9	87	243	32	362
Offshore						
43	5	9	8.7	-	-	8.7

Assumptions:

- Cost of analysis of chlorophyll in samples, £4.50 a sample (Pentreath 1992);
- Monthly surveys are required over the growing season, March-November, i.e. 9 surveys;
- 15 sample sites per estuary;
- 5 sampling sites per coastal location;
- Number of estuaries, 178 (Nixon 1991);
- Number of coastal locations: inshore 430 (1 every 10 km); offshore 43 (1 every 100 km) (Nixon 1991);
- Boat costs £600 per day (Pentreath 1992);
- NRA personnel cost £84.39 a day (grade 3) (Nixon 1991);
- Sampling by boat in coastal waters costs £27 K (boat) and £3.6 K (personnel) per survey;
- Sampling by boat in estuaries costs £106 K (boats) and £15 K (personnel) per survey.

From Table 6.2 it can be seen that the estimated cost for sampling each estuary as specified would be in the order of £1197 K per year and coastal waters (inshore and offshore) £371 K per year.

What if sampling trips already planned?

6.3 *In-situ* chlorophyll analysis (fluorimetry)

6.3.1 Methodology

In situ spectrophotometry has been used on far fewer occasions than *in vitro* fluorimetry. The method is more usually used to determine suspended solids but can be used to measure chlorophyll-a if an appropriate wavelength (660 nm) is used. However, even discounting the disadvantages of all methods which rely on chlorophyll as a surrogate of biomass, the accuracy of results obtained will be less than conventional sample collection/chlorophyll extraction/spectrophotometry.

Instruments are now available which incorporate a fluorimetric capability (e.g. Aquatracka) which could be used to measure chlorophyll in tidal waters. These instruments would be capable of obtaining a considerable amount of spatial data *in situ* which could be used for classification purposes. At present a number of NRA Regions (e.g. Anglian) has at least one of these instruments in use. They are, however, quite expensive to purchase costing around (£100 000). Accepting the fact that adequate data could be obtained from any one trip the question would then become how many trips would be needed to obtain the desired precision in classification. Indeed, these instruments have the potential for remote deployment and continuous recording of chlorophyll concentrations. If for example nine sampling trips were required this would entail considerable manpower effort on part of the NRA and may not be feasible within current financial and resource constraints.

6.3.2 Basis of classification

The basis of the classification could be as described in Section 6.2.2 that is on a comparison/ranking of the observed levels in tidal waters of England and Wales. An advantage of this method that much more data would be obtained on the spatial distribution of chlorophyll on a particular sampling trip, giving an indication of patchiness in chlorophyll and perhaps any relationships with salinity. This option would not be so easily (as discrete sampling) incorporated or included into current NRA monitoring programmes.

6.3.3 Availability of data

As already mentioned the NRA has purchased a number of Aquatracka devices with capability of chlorophyll measurement. These are currently deployed on the NRA's Vigilance class boats. The extent of their use is rapidly increasing and are routinely used on cruises to and from specific survey site areas, this adds considerably to the amount of data available. In addition, Aquatrackas are used in the sea-truth studies undertaken for the current remote sensing programme.

6.3.4 Advantages and disadvantages

The main advantage would appear to be the large amount of *in situ* data that could be obtained both horizontally and vertically throughout the water column. Due allowance would have to be given for suspended sediments and dissolved organic matter that might interfere with the chlorophyll signal and this might potentially raise the limit of detection. The main drawback appears to be the cost, both in terms of purchase of the instrument and in resources (e.g. manpower and boats) required to sample at a sufficient frequency. The estimated costs, with the underpinning assumptions, of implementing this option throughout NRA tidal waters are given in Table 6.3.

Table 6.3 Estimated costs of implementing *in situ* fluorimetry option

Number of locations	Survey frequency	Boat costs (£'000)	Staff costs (£'000)	Total costs (£'000)
Estuaries				
178	9	954	135	1089
Coastal waters				
Inshore				
430	9	243	32	275
Offshore				
43	9	-	-	
Total				1364

Assumptions:

- Sample analysis not required
- Continuous *in-situ* measurement of chlorophyll in estuaries and coastal waters;
- Monthly surveys are required over growing season, March-November, i.e. 9 surveys
- Number of estuaries, 178
- Number of coastal sites: inshore 430 (1 every 10 km); offshore 43 (1 every 100 km);
- Boat costs £600 per day;
- NRA personnel cost £84.39 a day (grade 3).
- Sampling by boat in coastal waters costs £27 K (boat) and £3.6 K (personnel) per survey;
- Sampling by boat in estuaries costs £106 K (boats) and £15 K (personnel) per survey.

6.4 Monitoring for nutrient levels

6.4.1 Methodology

This option was considered in detail in a previous report for the NRA (Rees *et al* 1992). The report proposed a classification for tidal waters based on existing levels of inorganic nitrogen and phosphorus in UK estuarine and coastal waters. The scheme was designed to reflect the extent of contamination from natural and anthropogenic sources and was not intended to relate directly to algal production or eutrophication.

A single scheme was proposed for estuarine and coastal waters based on the measurement of salinity related nutrient (total inorganic nitrogen and soluble reactive phosphorus) in the winter months. Sampling in winter would remove a lot of the inherent variability in water quality already discussed. It was proposed that a minimum three sampling runs were carried out over the winter. Furthermore to reduce the bias caused by unusually 'wet' or 'dry' years the estuary class would be based on a three year rolling average. It was further suggested that sampling would be undertaken at around high water of neap tides. A number of sampling sites would also be required in each estuary to reflect the differences in salinity and locations of sources of nutrients.

The position of the class boundaries in any classification scheme which is not based on effects will be somewhat arbitrary and can only be decided by subjective judgements. In the case of the nutrient classification scheme three approaches were considered:

- Establish background levels and to base subsequent class boundaries at multiples of this level;

- Set class boundaries so that given percentages of UK estuaries would initially fall into each class;

- Determine the levels of nutrients at the most contaminated sites available and base the class boundaries (as percentages) of these.

The same basic scheme was proposed for coastal waters divided into an inshore and offshore zones, which would be sampled separately. The inshore zone would be sampled at high water, preferably on a neap tide. No such restriction was suggested for the offshore zone. The number of coastal sites would also have to be carefully considered so that the desired precision was obtained.

An alternative method of obtaining inorganic nitrogen concentrations might be through the deployment of *in situ* nitrate analysers which have been developed by SOAFD. They use a colorimetric method for the analysis of nitrates in water samples periodically pumped into the analyser's housing suspended beneath a moored buoy. Data could be telemetred to a land-based station. However, the cost of these instruments is in the order of £10 000 each and the wide scale deployment of these by the NRA is likely to be prohibitively expensive.

6.4.2 Availability of data

Nutrient monitoring forms part of every region's tidal monitoring programme, although currently there is no unified approach and different determinands are employed in different regions. Table A1 (Appendix A) summarises the extent of NRA and RPB monitoring of tidal waters, this table was drawn from Rees *et al.* (1992) and has been updated to accommodate changes in sampling regimes. Table A2 drawn from MPMMG (1991) shows long term nutrient data sets that predate the organisational change in the water industry, the accessibility of these data sets is not known and would require further investigation. Nutrient monitoring data are the most extensive of all options considered and therefore, has the most extensive database on which to base a trophic classification scheme.

In addition, to NRA and RPB routine sampling under MPMMG monitoring guidelines, a number of other potentially useful datasets exist. MAFF have extensive data on the Wash and Humber collated under the auspices of the JONUS project. This project was designed to assess the fate of nutrients as they pass through estuaries, data include anchor stations in both estuaries from a number of cruises.

6.4.3 Advantages and disadvantages

A potential advantage of monitoring nutrient levels is that a lot of the variability associated with water quality and primary productivity of tidal waters might be reduced giving a more reliable comparison between different water bodies. Again there would be major sampling effort required but this might be achieved most cost effectively through the use of a helicopter. However, this option would only reflect levels of contamination, and the possible potential for eutrophication in terms of algal growth problems, rather than a true assessment of trophic status or response of the system to nutrients. This disadvantage could potentially mean that any initiatives to improve water quality, as a result of the classification, might be ill-directed. In addition, some sites might have 'naturally' elevated levels of nutrients which would be difficult to separate from anthropogenic sources.

The estimated costs, with the underpinning assumptions, of implementing this option throughout NRA tidal waters are given in Table 6.4.

Table 6.4 shows that the total cost of implementing this option would be in the order of £548 K a year, somewhat less than that for the discrete sampling option (£1568 K) and the *in situ* fluorimetery option (£1364 K).

Table 6.4 Estimated costs of implementing the monitoring of nutrient in estuaries and coastal waters option

	Number of sites or estuaries	Number of samples/ location	Number of sampling trips	Total samples	Estimate of costs of sample analysis (£'000)	helicopter hire (£'000)	sampler (£'000)
Estuaries							
	178	15	3	8010	80	180	5
Coastal							
-inshore	430	5	3	6450	65	203	8
-offshore	43	5	3	645	6.5	-	-
Total costs				151.5	383	13	

Assumptions:

- Length of coast within NRA's responsibility 4296 km (Nixon 1991)
- Inshore coastal zone spilt into 10 km blocks
- Offshore coastal zone split into 100 km blocks
- Number of samples: estuaries, 5 in each of three salinity zones (Rees *et al.*, 1992); 5 per block of coastal water
- Sampling undertaken three times between December and February
- Sample cost, £10 a sample for saline nutrients (Pentreath 1992)
- Helicopter hire: £380 per flying hour; 157.6 hours required for each estuary survey (Nixon 1991); 178.1 hours for each coastal waters survey (Nixon 1991)
- Sampler costs: based on grade 3 NRA person at £84.39 per day (Nixon 1991);
- 20 man days required for each estuary survey; 31 man days required for each coastal waters survey

6.5 Macroalgal biomass

6.5.1 Methodology

The biomass of macrophytic algae is simply measured by clearing an area of known size and directly weighing the material collected, either while fresh, or more usually and usefully after drying in an oven. Populations of such fast growing ephemeral green species can fluctuate very rapidly and whole populations of even attached plants can be removed by a single storm. Frequent sampling would, therefore, be needed to obtain a complete picture of biomass production. It is likely that sampling would be restricted to low water during the growing season and this may impose logistic problems. There is no data available on which to base the sampling frequency that would be required but it is likely that at least nine surveys would be required during the growing season. This survey frequency would possibly ensure that the sudden loss of macroalgal growths as the result of storms etc. would be correctly allowed for.

6.5.2 Basis of classification

It is likely that the classification would be based on a comparison of observed macroalgal biomasses with class boundaries set according to one of the criteria described in Section 6.4.1. Again due consideration would have to be given to natural growth of macroalgae compared to that stimulated by anthropogenic inputs.

6.5.3 Availability of data

Reported data on the use of macroalgal biomass appears to be restricted to a single intense study on the Ythan estuary (Raffaelli *et al.*, 1989) which has an excellent 25 year data set of macroalgal biomass changes, together with nutrient load data taken from the harmonised monitoring programme. There may also be other data sets relating to local areas or problems such as work undertaken on some south coast harbours such as Langstone Harbour (Soulsby *et al.*, 1985). Such measurements are not however generally undertaken in routine monitoring programmes, the sampling of macroalgae being largely concerned with bioaccumulation monitoring.

6.5.4 Advantages and disadvantages

The sampling methodology would be quite simple but results would potentially be very variable over a growing season and probably between years. Changes might become more apparent over long time spans perhaps reflected also by changes in community structure, such as dominance of greens over browns in the intertidal zone.

A major disadvantage would be the lack of applicability around all coasts and estuaries of the UK as the conditions and habitat required for their growth would not be universally present. However, as indicated earlier, the inclusion of macroalgae as one of the criteria for trophic status assessment might improve the robustness of the classification as it would cover the situation where the primary response to nutrification is seen in the macroalgae rather than solely in the phytoplankton. The growth of macroalgae is very dependent on substrate suitability and (at the same levels of nutrients), light availability. Moreover, under the same conditions, different algal species can achieve very different biomasses. (Need figures for comparison, laminaria cf entermorpha.) A classification based on macroalgal biomass would therefore, need to be species based and may well be overly complex.

Even though macroalgal biomass determinations are strictly speaking discrete samples, because the growth period is so much longer than for phytoplankton, they effectively represent a cumulative measure of trophic status.

The estimated costs, with the underpinning assumptions, of implementing this option throughout NRA tidal waters are given in Table 6.5.

Table 6.5 Estimated annual cost of implementing macroalgal biomass option

	Number of locations	Number of sites per location	Survey frequency	Number of samples (£'000)	Sample costs (£'000)	Staff costs (£'000)	Total costs
Estuaries	178	3	9	48060	144	270	414
Coastal waters	430	3	9	116100	348	653	1001
Totals							1415

Assumptions:

- Monthly surveys are required over the growing season, March-November, i.e. 9 surveys;
- Number of estuaries, 178;
- Number of coastal locations, 430 (1 every 10 km); 100 km);
- Number of sites per estuary or coastal location, 3;
- Number of replicate samples per site, 10;
- NRA personnel cost £84.39 a day (grade 3);
- A day required per each estuary and coastal site per survey;
- Two people are required to sample;
- Sample analysis estimated to be £3 a sample (30 samples a day).

The cost of implementing this option would be in the order of £1415 K per year.

6.6 Algal bioassays

6.6.1 Theory and methodology

Algal bioassays are more often used to assess toxicity, but can be used to measure trophic potential, i.e. to give an indication of the algal biomass which will be produced under ideal conditions (*ex situ* bioassays), or to measure algal productivity under more natural coastal water physical conditions (*in situ* bioassays).

Algal bioassays rely on measuring the biomass achieved over a given period of time, from days to weeks. However, the nutrient concentrations in many tidal waters are well in excess of typical phytoplankton K_S values, so differences in short-term productivity are likely to be minimal between, say eutrophic and hypertrophic waters. Because of this, the maximum biomass achieved, regardless of time, could be a better approach to adopt in terms of measuring trophic status, but as some algae tend to produce large amounts of polysaccharide when nutrients are limiting, test species must be carefully selected. It may be necessary, for example, to measure chlorophyll on a daily basis to ensure that peak levels are measured prior to culture senescence.

In situ (dialysis bag) algal bioassays

A dialysis bag containing either a filtered sample of water or an artificial medium is inocculated with an algal starter culture and is then attached to a moored buoy at a given depth (usually close to the surface) for a fixed period of time. The dialysis bag allows exchange of water within the bag with that outside, so the algae are effectively maintained in a fixed position, but experience the dynamic range of water quality that occurs outside the bag.

Single or multi-species tests may be used, in which the algae are chosen for their ecological relevance and suitability to the water being assessed (i.e. the inoculum used grows naturally in waters which are chemically similar to the body of water being assessed).

Macroalgal bioassays have also be used in estuaries primarily for pollution assessment studies but could be adapted to measure trophic potential: in this case the growth of transplanted discs of *Ulva, Enteromorpha* or *Fucus* was measured after a period of deployment in the field (Wilkinson *et al.*, 1992).

The potential advantages of this option include the provision of a direct measure of trophic status with reference to phytoplankton (or potentially macrolagae), the species of which could be ecologically relevant. In addition, dynamic changes in water quality and some physical parameters (i.e. temperature and turbidity) are taken into account. Once techniques are established the method should be relatively inexpensive to undertake. However, the method does not take account of depth of mixing and reservations have been raised over result reproducibility. In addition, dialysis bags may be washed away or ripped during inclement weather.

Ex situ (discrete sample) algal bioassays

In *ex situ* bioassays a discrete water sample is collected and filtered to remove any algae that are present in the water sample. The sample is then inocculated with either a single algal species or several species and the culture is grown under fixed, 'ideal' laboratory conditions for at least 5 days, during which time the maximum rate of growth is determined.

Again a direct measure of trophic status with reference to phytoplankton is provided and if the bioassay is devised for a specific country or climate, the species selected should be ecologically relevant. However, if a standard experimental protocol which has been developed in another country is used, the algal species selected may not be ecologically relevant. The method also relies on analysis of a discrete sample which may not be representative of the body of water being assessed and no account is taken of natural physical conditions as growth conditions are optimised in the laboratory.

Ex situ macroalgal bioassays

A number of macroalgal growth bioassays have been used in toxicity tests, such as the growth of rhizoids by sections of *Enteromorpha* thallus or the rate of germination of *Enteromorpha* spores. Some may be suitable to adapt as bioassays for determining trophic potential, but only if genetic homogeneity could be guaranteed by culturing genetically uniform algae on a large scale and using this as the reference material for all bioassays. With *Enteromorpha* this may be possible, since the alga reproduces both sexually and asexually, naturally releasing 'swarmers' (gametes and spores) maximally 3 to 5 days before the highest tide of each lunar period (Christie and Evans 1962). This pattern can be manipulated and the release of swarmers stimulated in the laboratory by changing the lighting pattern. This option is not known to have been favoured by any organisation charged with monitoring water quality as part of a wide-scale tidal water trophic status assessment.

In this case a direct measure of trophic status with regard macroalgae is obtained. This type of bioassay has only been used in small local studies and hence there is little information on result reproducibility. Again a disadvantage is that it relies on analysis of a discrete sample which may not be representative of the body of water being assessed. In addition, no account is taken of natural physical conditions and the maintenance of 'stock' algae may be more troublesome than for phytoplankton.

6.6.2 Basis of classification

The classification could be based on a comparison of biomass or other related determinand such as chlorophyll from the standardised bioassay. If a bioassay universal to all tidal waters (e.g. salinity limitations) was possible then direct comparisons would be achievable. It is likely though, that *in situ* bioassays would give a more robust and relevant measure of the trophic status of a body of water.

The estimated costs of implementing the algal bioassay option with the underpinning assumptions is given in Table 6.6. It has been assumed that the *in situ* algal bioassays would be used with monthly deployments of the bioassays in all tidal waters. These costs must be treated as very approximate as there is no operational experience to base costs upon.

There does not appear to be any existing data which could be used for developing the classification, and these techniques are not used in routine NRA monitoring programmes.

Table 6.6 Estimated costs of implementing algal bioassay option

	Number of locations	Samples per site	Survey frequency	Bioassay cost (£'000)	Boat costs (£'000)	Staff costs (£'000)	Total (£'000)
Estuaries	178	15	9	240	954	135	1329
Coastal waters							
Inshore	430	5	9	193	243	32	468
Offshore	43	5	9	19.3	-	-	19.3

Assumptions:

- Cost of *in situ* algal bioassay, £10 a sample (chlorophyll and biomass analysis);
- Monthly surveys are required over the growing season, March-November, i.e. 9 surveys;
- 15 sample sites per estuary;
- 5 sampling sites per coastal location;
- Number of estuaries, 178 (Nixon 1991);
- Number of coastal locations: inshore 430 (1 every 10 km); offshore 43 (1 every 100 km) (Nixon 1991);
- Boat costs £600 per day (Pentreath 1992);
- NRA personnel cost £84.39 a day (grade 3) (Nixon 1991);
- Sampling by boat in coastal waters costs £27 K (boat) and £3.6 K (personnel) per survey;
- Sampling by boat in estuaries costs £106 K (boats) and £15 K (personnel) per survey

6.7 The use of statistical models

6.7.1 Theory and methodology

The fundamental principle adopted by Vollenweider more than 20 years ago is the concept that nutrient loading, rather than concentration, is the key to understanding the trophic responses of water bodies. The statistically derived relationship between nutrient loading and algal response, usually in terms of chlorophyll-a concentrations, provides a means of controlling and managing water quality. The concept was developed initially for freshwater lakes and reservoirs where, the controlling nutrient is phosphorus. In more recent years the concept has been applied to marine waters (coastal embayments) where the controlling nutrient is usually nitrogen.

The OECD (1982) approach to predicting chlorophyll levels in freshwater lakes has been tried using data from the north Adriatic coastline (Giovanardi and Tromellini 1992). Here, 'typical' peak chlorophyll levels are approximately 5-11 times mean chlorophyll levels at a given phosphorus concentration and 'exceptional' peak levels are approximately 6-23 times mean chlorophyll levels. In freshwater lakes, peak chlorophyll levels are approximately 3 times annual mean concentration for the same waterbody (OECD 1982).

Temporal fluctuations in chlorophyll levels in coastal waters can therefore be expected to be much larger than the fluctuations observed in lentic freshwaters.

In some coastal waters a statistical relationship can be established between nutrient concentrations and chlorophyll. For example, a relationship between annual, depth-averaged, concentrations of total inorganic nitrogen and the annual, average, surface concentrations of chlorophyll in some Hong Kong waters has been found (Lack *et al.*, 1990). In this latter work it was possible to identify 'responsive' sites where there was a clear statistical relationship between the two determinands (Figure 6.2). Vollenweider also recognised the importance of the retention time of the body of water. Allowance for this was found to improve the statistical relationship. Therefore, the relationship was improved by applying a factor to the total inorganic nitrogen concentrations which took into account the retention or flushing time of the bays and semi-enclosed bodies of water (Figure 6.3).

In the Hong Kong case it was found that with flushing corrected values of TIN less than 1 mg l^{-1} there was little response in chlorophyll concentrations, varying around 4 μg l^{-1}. Above a flushing corrected TIN of 1 mg l^{-1}, the algal response was clear and marked. The 'trigger' of response appeared to be in the range of 0.4 to 1 mg l^{-1} and the authors considered this to be equivalent to Vollenweider's freshwater mesotrophic zone where there is a balance between nutrient concentration and algal response.

The next stage in the Hong Kong study was to relate TIN levels to the total nitrogen loadings to the bays and coastal waters. Again, a statistically significant relationship was established between the two determinands. In the Hong Kong case this relationship provided a predictive tool that could be used to predict changes in TIN and chlorophyll (and excessive algal growth) that would occur with changes in nitrogen load to particular bodies of water. The 'trigger' concentration was found to correspond to a particular range of total nitrogen load. Below this trigger load range the waters were considered to be unpolluted by nitrogen and those above as eutrophic, and those in between in transitionary or mesotrophic stage.

If this type of relationship could be established for tidal waters in England and Wales then there would potentially be a means of relating a classification based on nutrient concentrations to that relating to trophic status in terms of algal productivity. This approach would probably be more applicable to estuaries and enclosed marine bays rather than open coastal waters. The statistical relationship is also likely to be site-specific as different bodies of water would have different flushing times and also other physical and environmental factors affecting algal production (e.g. water turbidity and stratification). In estuaries the relationship between salinity and productivity would also have to be considered. For example, for the same nutrient concentration low salinity sites might show less response (in terms of primary productivity) than a high salinity site due (as an example) to the former possibly having higher turbidity and growth of marine phytoplankton may be inhibited in low salinity waters. However, it might be found that geographically similar estuaries might fit the same general model.

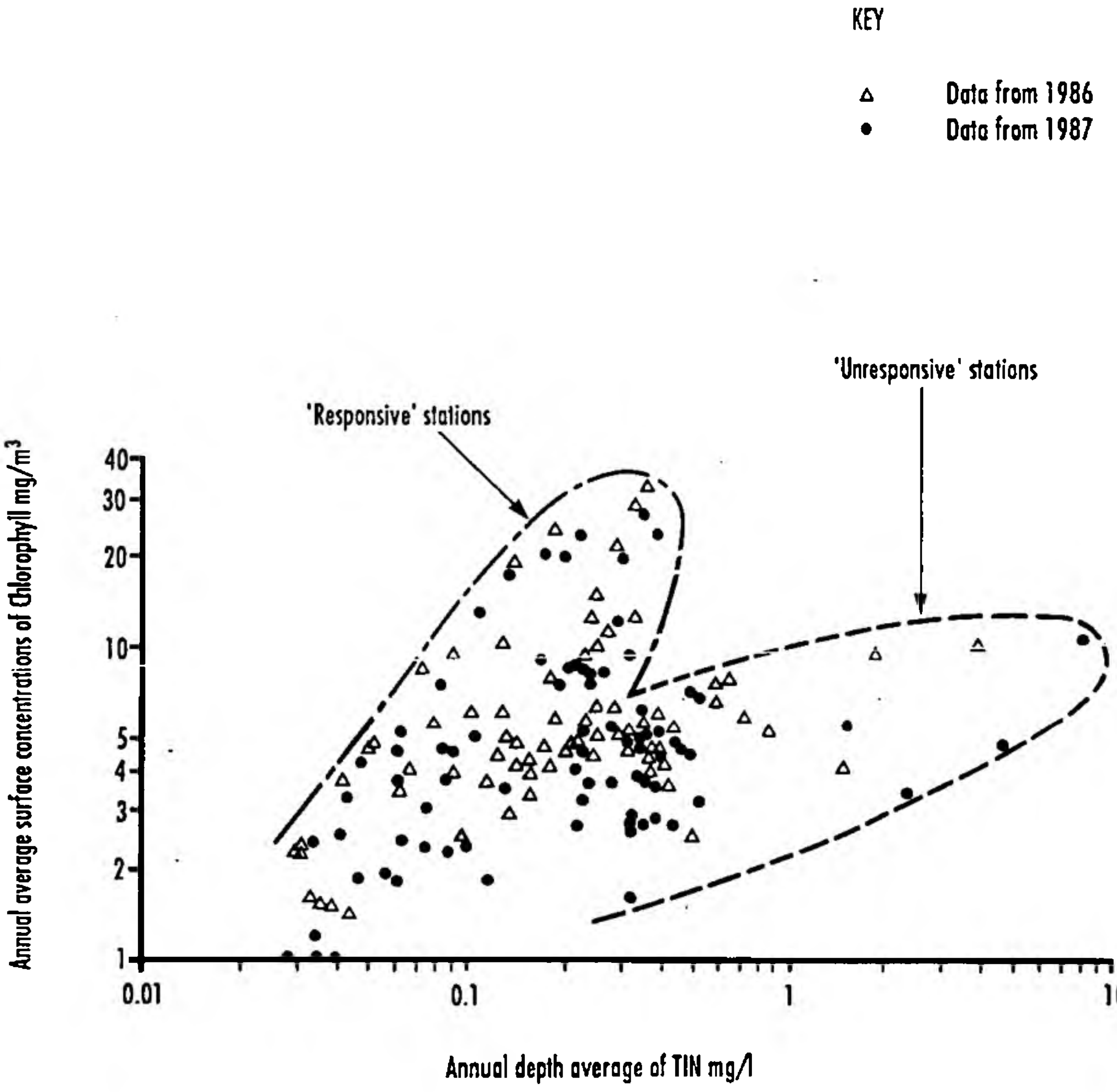

Figure 6.2 Annual depth averaged concentration of total inorganic nitrogen and average surface concentrations of Chlorophyll in Hong Kong waters

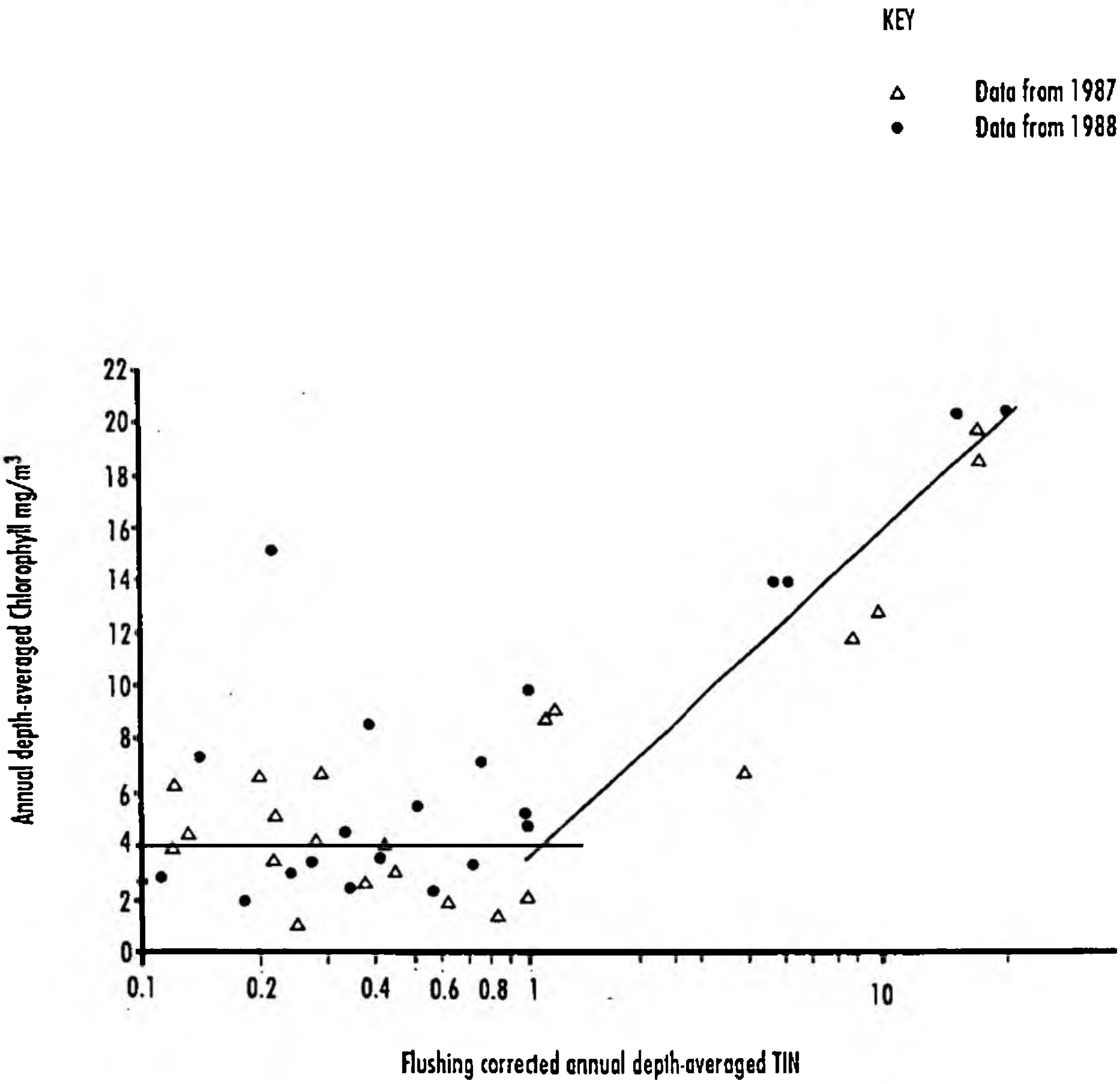

Figure 6.3 The relationship between flushing corrected total inorganic nitrogen (TIN) and chlorophyll

A similar approach is being adopted by American workers who are undertaking a national estuarine eutrophication project (Nixon 1992). The aim is to use various types of loading versus effects plots to investigate the relationship between nutrient loadings to estuaries, the physical characteristics of estuaries and the expression of effects of nutrient loadings in estuaries. The work is seeking information on nutrient concentrations and loading data for as many estuaries as possible. Other data required will be on incident light, sediment type, phytoplankton (e.g. abundance, primary production, growth limitation studies, species composition, unusual blooms/species, surface scums), macrophyte surveys (e.g. vascular plants, macroalgae), benthic organisms surveys (e.g. species composition), oxygen survey (e.g. bottom water dissolved oxygen, rate of oxygen decrease in deep water). They are also suggesting that an estuary would be spilt up into 4 salinity zones: 0 - 0.5, 0.5 - 5, 5 - 20, >20 ppt.

6.7.2 Basis of classification

Once statistical relationships for individual estuaries, bodies of water or, more ideally geographic areas have been established, the classification could be based on where a particular site or location lies on the nutrient/load/production curve. Those sites that were above the 'trigger' load or concentration could be classed as sensitive or eutrophic, those well above as severely nutrient polluted or 'hypertrophic'. Sites around the trigger level could form a further class band (mesotrophic) and those showing no response as the best quality.

This form of model may also serve to highlight those bodies of water where productivity is naturally high: on the response curve they would tend to respond more than would be anticipated from the loadings or nutrient concentrations.

6.7.3 Availability of data

Although the Vollenweider approach is as yet untested in UK tidal waters a number datasets exist which go some way to fulfilling the data requirements. For example, Anglian NRA has commissioned a number of water quality models for their major estuaries and to date these include the Nene, Great Ouse, Orwell Stour, Colne, Blackwater and the Crouch and Roach.

These water quality models quantify nutrient loads and have some data on resulting environmental contaminant levels and chlorophyll. In addition to these models the Humber estuary has also been modelled in a generic sense and has the capability to be tuned to any environmental variable including chlorophyll.

A number of other studies are potentially useful for this modelling approach:

Harmonised Monitoring Points

This sampling programme has in excess of 100 points located at the tidal limits of rivers, thus they should provide information for quantifying loads entering systems. The sampling programme has the advantage of being carried out over a long time scale and at least in earlier days used standardised methods. The sampling programme began in 1974 and was co-ordinated and funded by the DoE and although funding stopped in the 80's the programme continued, though in not in a harmonised way. Throughout its history sampling regimes have varied from region-to-region and even site-to-site with exact sampling frequency and determinands being predicated by local needs. The sampling programme did, however, contain a core of determinands which contain nutrients.

North Sea Declaration and Paris Commission data

The Convention for the Prevention of Marine Pollution from land-based sources ('Paris Convention') requires Contracting Parties to adopt measures to combat marine pollution. In order to prioritise the measures to be taken it is necessary to have information on the relative share of inputs from the various sources and on the relevant pathways (Oslo and Paris Commissions, 1992). The NRA are responsible for implementing sampling programmes to obtain and since 1990 have annually reported the findings. The sites sampled for Paris Convention are in the main those used in the harmonised monitoring programme.

LOIS

The Land-Ocean Interaction Study (LOIS) a NERC community research project, which is currently under way may well provide some data of use in assessing trophic status. A part of the study, the River Atmosphere Coast study (RACS), programme involves measuring and characterising the nutrients in the coastal zone together with rates of primary productivity. The area chosen for this study are 'representative portions of the coast, between Berwick-on-Tweed and Great Yarmouth, together with the associated fluvial fluxes'.

6.7.4 Advantages and disadvantages

This option has a clear advantage in that it is based on a statistical relationship between nutrients and algal production (in terms of phytoplankton chlorophyll). It would also provide a valuable management tool to identify those estuaries that are 'responsive' to increased nutrient loads and which are potentially at risk from eutrophication and possibly nuisance algal growths.

Disadvantages are the potential costs for establishing the models. This would require detailed quantification of loads into estuaries and coastal waters. However, a lot of this information should be readily available in some form or another. Loads could simply be based on population levels, consent conditions and land-use maps with crude estimates of run-off. Such an estuarine nutrient load model is currently being developed by WRc on behalf of SNIFFER. Whether this information would be sufficient for the model would have to established at an early stage of the development phase. In addition, if models were too site-specific, a large amount of work might be required establishing models for every estuary and body of coastal water. However, once established the method potentially would require less sampling for validation purposes. Loads would have to be monitored (this would be routinely done anyway) and a certain amount of monitoring of chlorophyll and nutrient analysis would be required.

Another major shortcoming is that the method has not been tested in UK waters, and may not be applicable in open coastal waters.

Data for establishing and validating this type of model in some estuaries might already be available from the development of water quality mathematical models. Data availability is addressed in the next subsection.

The estimated costs, with the underpinning assumptions, of implementing this option throughout NRA tidal waters are given in Table 6.7.

As can seen from Table 6.7 the estimated costs are very high being in the order of £1731 K a year for estuaries and £479 K for coastal waters. Against this, the potential robustness of the option must be borne in mind, together with the models potential use as a tool for designing control strategies , e.g. as part of consent setting procedures.

6.8 Remote sensing from aircraft

6.8.1 Methodology

The colour of the sea contains information on suspended solids (types and concentrations), productivity (chlorophyll concentrations), plankton types, dissolved humic substances, and many dissolved and suspended organic and inorganic pollutants. Remote sensing of marine algae is based either on the absorbing or the fluorescence characteristics of the chlorophyll-a pigment.

The NRA has recently assessed the feasibility and practicability of undertaking coastal water monitoring and surveillance using ships and aircraft (Pentreath 1992, and Boxall *et al.*, 1992). The objective is to put into place an annual programme to operate from early spring into summer to monitor the three mile coastal strip. This would include the measurement of chlorophyll and suspended sediment load (ship/aircraft), basic water determinands (ship), nutrients (ship) and dangerous substances (ship). An initial assessment study, undertaken in 1991, involved four regional passes by aircraft using remote sensors between early spring and summer; these were linked to regional shipboard cruises. Two basic types of sensor were considered, those relying on colour or thermal imagery and those using some sort of spectral analyser.

Table 6.7 **Estimated annual costs for implementing the use of statistical models option**

No. of locations \estuaries	Sample analysis		Boat costs		Personnel costs		Load calculations	
	per survey (£'000)	per year (£'000)	per survey (£'000)	per year (£'000)	per survey (£'000)	per year (£'000)	per site (£'000)	total (£'000)
Estuaries								
178	23	279	106	1272	15	180	0.4	75
Coastal waters								
43	9.4	112	27	324	3.6	43	0.4	17

Assumptions:

- 178 estuaries;
- 43 blocks of coastal waters each 100 km (parallel to shore) x 5.6 km (3 nautical miles offshore) = 560 km^2 in area;
- 5 sampling points and 3 depth samples in each coastal block;
- 3 sampling points and 3 depth samples in each estuary;
- 12 surveys each year;
- Sample analysis cost of £14.50 a sample for chlorophyll and saline nutrients;
- Boats taking samples at a cost of £27 K (boat) and £3.6 K (personnel) a survey for coastal waters and £106 K (boat) and £15K (personnel) for estuaries;
- Preliminary calculation of loads into each estuary and coastal zone block based on 5 man days at grade 3 per unit.

Colour and thermal image sensors

With visible or infra red radiation sensors such as the Airborne Thematic Mapper (ATM) the most tractable data sets are those obtained over water whose optical properties are dominated by a single component e.g. phytoplankton and suspended sediment. To relate the image to chlorophyll levels it is necessary to obtain 'sea-truth' data on actual chlorophyll and other determinand concentrations, and from these produce algorithms from which chlorophyll concentrations can be derived. When a mixture of different components are present in the target water it is much more difficult to devise reliable algorithms. A variety of multiband algorithms have been designed to estimate algal biomass in coastal waters but they tend to be somewhat insensitive and very site specific. Even though the ocean emits both infra red and visible light, colour is the only parameter that includes information from the integrated surface layers (1 to 80 m depending upon water clarity), the others are only surface values (surface mm).

Therefore, variations in water quality parameters other than chlorophyll-a (e.g. suspended particulate matter, salinity and the composition of different phytoplankton species) affects the accuracy of measurements which use the green/blue reflectance ratios or near infra-red/red band ratios (vegetation indices). Because of this, these techniques are rarely applicable to the majority of British estuaries, but can be useful in relatively still waters such as inland lakes.

As well as determining phytoplankton chlorophyll concentration remote sensing could potentially determine the cover of intertidal macroalgae and macrophytes. For example, using the Landstat TM false colour image it was possible to classify the intertidal areas of Langstone and Chichester harbours into areas of sand and gravel, *Enteromorpha* and *Ulva* species, *Spartina* species and upper salt marsh species (Goldsmith 1991). Better images are produced when there are relatively homogeneous, monospecific vegetation types. Satellite images have also been used to differentiate between and to map sediment types (e.g. the Wash, Goss-Custard 1993). In this latter case this information was used to predict the bird carrying capacity of the intertidal mudflats.

To be useful for classification, cover measurements would have to be related to biomass: this would be species-specific and potentially, therefore, site-specific. Ground truth data would, therefore, be required.

Multispectral analysers

On the other hand, multispectral sensors such as the Fluorescence Line Height Imager (FLHI) measure an unique index of phytoplankton biomass which is not readily contaminated by other optical components. The Compact Airborne Spectrographic Imager (CASI) is also particularly effective in quantifying chlorophyll in mixed waters though knowledge of what depth the signal is coming from might be needed. Measured shifts in the recorded spectrum may solve this problem.

The FLHI method is more applicable to estuarine/coastal waters due largely to the relative independence of the fluorescence of chlorophyll-a to salinity.

Absolute accuracy is largely untested in British waters as yet, due to lack of results which correlate imagery with *in situ* measurements. The NRA's initial assessment study (Boxall *et al.*, 1992) was unable to investigate this aspect as there was a need to look at a considerable length of coastline and a number of different techniques and pollutants.

The following should be noted with respect to FLHI.

1. The sensitivity of measurement of chlorophyll-a (limit of detection imposed by instrument/platform/atmospheric induced noise) for CASI (Compact Airborne Imaging Spectrometer) was found to be about 0.1 to 0.2 µg l^{-1} (Gower, 1990).

2. Initial results suggest the accuracy of measurement of chlorophyll-a to be of the order of 0.5 µg l^{-1}. In a study in the Skagerrak off the coast of Norway, of 12 controlled sites (in-situ measurements obtained within 8 hours of the imagery), with salinity and light-beam attenuation variation of up to 50%, 8 of the FLH-calculated values were within 0.5 µg l^{-1} of observed valves over a range 0.6-2.6 µg l^{-1}. All were within 1.2 µg l^{-1} of observed values (Pettersson, 1990). However, a significant mismatch was obtained where the maximal chlorophyll concentration was found at 10 m depth.

3. Accuracy may be affected by optical properties of the water (turbidity) and also by phytoplankton physiology under certain conditions, e.g. where fresh water mixing with sea water causes cell damage to marine algae, thus altering the degree of fluorescence (Pettersson, 1990).

Platforms

Sensors can be mounted on aircraft or satellites. This has also been investigated by the NRA (Briggs *et al.*, 1992). Aircraft mounted scanners have a resolution of 1 to 10 m with cloud cover affecting results to varying degrees. In comparison satellite scanners have spatial resolutions ranging from 30 m to 1 km, the latter may be of little benefit in a 3 nautical mile coastal zone. The greater the resolution from satellites the smaller the area covered in a single overpass. Satellite systems such as Landsat can achieve good resolution, 30 m, but with less than two repeat overpasses at any one location per month and with typical cloud conditions, only three to four clear visible or infra red images for a particular site would be available per year.

There may be quite limited opportunity for adequate flying conditions for optical airborne sensors like CASI during the year. Low cloud (<1000 m) generally precludes image acquisition, although it is possible to obtain images under conditions with cloud cover at greater altitude. Low-altitude surveys reduce the image coverage and the costs of aircraft and data processing increase dramatically. Cloud heights of <500 m persist for lengthy periods in Northern and Western areas during Winter months. An aircraft on standby is ideally required, adding considerably to the cost of a monitoring facility.

Airborne imagery tends to be more geometrically distorted than satellite imagery, although newer instruments like CASI (a linear-array scannner) are better than the scanners which have been in use to date in this country - e.g. ATM (a single detector, 'sweep' scanner). Generally, techniques developed using airborne sensors can be applied to satellite data obtained from a similar instrument with little modification, although there are additional atmospheric corrections to be made.

A satellite-borne sensor with similar spectral capability to CASI, MERIS (Medium Resolution Imaging Spectrometer) is proposed for the European Polar Platform, scheduled for launch in 1997. This will have 250 m/1 km spatial resolution over a 1500 km scan-width and a 3-5 day image repeat cycle.

Satellite data carry the advantages of regular imaging (although highly dependent on cloudcover), better mapping accuracy and much lower cost (partly because of the size of the area which may be imaged in a single scan and also because it generally benefits from central funding).

Airborne sensing will remain an important technique for supplementing satellite data and where greater control is required over spectral and spatial resolution. Airborne systems may be deployed in areas of persistent cloudcover where satellite imagery is scarce, providing the cloud base is at least 500 m. Airborne systems have the flexibility to obtain imagery at short notice only if aircraft and sensor are retained on standby. Limited opportunity exists for obtaining optical data over northern and western Britain during Winter months from any platform.

With respect to the usefulness of sensors like CASI and MERIS for the measurement of chlorophyll-a, the limiting factor is likely to remain that of control data. The uncertainty in estuarine environments, in particular, dictates that temporally, and preferably also spatially (surface), distributed control data in the form of simultaneous *in situ* measurements are necessary for complete confidence. This is likely to remain the case even if the methods can be proven under a wide range of conditions. In this respect, such techniques cannot themselves replace traditional surveys but offer a valuable source of spatial (and, to a lesser extent, temporal) sea-surface information as a cost-effective supplement.

6.8.2 Basis of classification

Providing enough images can be obtained (and afforded) this option offers the greatest potential for obtaining a very large amount of spatial data (albeit a snapshot in time) covering all tidal waters in England and Wales. Once obtained, suitably calibrated and corrected for chlorophyll then direct comparisons could be made and a classification established as for the options previously described. There may also be a need to specify specific times for imaging in terms of high and low water in estuaries, this may reduce the likely tidal cycle variability of chlorophyll in estuaries.

6.8.3 Availability of data

Apart from the remote sensing undertaken by the NRA there are seemingly very few other studies which may prove to be useful sources of data for a future classification system. Briggs *et al.* (1992) recently reviewed potential remote sensing applications for the NRA and included information on earlier studies. It is apparent from this study in particular that while remote sensing of chlorophyll in lakes is well documented and there are a large number of studies involving satellites for coastal and ocean areas; there are very few studies involving aircraft remote sensing and any measurement of chlorophyll status. One notable and geographically relevant example carried out in 1989 in the Skagerrak (Pettersson 1990) revealed a high correlation between sea fluorescence and the concentration of chlorophyll-a.

There should also be a large amount of spatial data available from the planned remote sensing surveys around the whole coastline of England and Wales during 1993.

6.8.4 Advantages and disadvantages

The three major advantages offered by satellite imagery for estimating phytoplankton abundance are: the very large areas which can be monitored; the ease with which such monitoring can be undertaken (providing meteorological conditions are suitable); and the relatively low cost compared to intensive manual sampling/analysis. A major disadvantage is the frequency of data that would be required from any one geographic location. In addition, the method is not yet fully validated in terms of suitability for estuarine use and further work is required to assess the need for sea truth data and information on the vertical profile of chlorophyll in the water column.

Ideally weekly remote sensing data would be required to monitor and measure the growth of phytoplankton in tidal waters. This is likely to be prohibitively expensive and so as for the other options involving the measurement of chlorophyll, a minimum of nine monthly surveys is suggested. Once obtained there would be the same disadvantages and difficulties in handling and interpreting the data as in common with other chlorophyll methods e.g. discrete sampling and *in situ* fluorimetry. Such a difficulty would be interpreting data in relation to the different cell contents of chlorophyll in phytoplankton species.

In terms of the instrumentation and techniques available for remote sensing, the advantages of airborne sensing data compared to that obtained by satellite were considered by Briggs *et al.* (1992) to be:

- resolution or scale of imagery;
- control in respect of timing of the overflight;
- better capability to mitigate against adverse weather conditions;
- repeat cycle frequency;
- flexibility of sensor systems;
- ability to combine more than one sensor type per platform giving overall synergy in the dataset;

* easier co-ordination with ground data collection.

The main disadvantages were seen to be:

* greater difficulties of geometric registration of airborne data;

* one or two orders of magnitude or so increase in acquisition costs per area;

* greater inconvenience/involvement in data acquisition; it is not possible to simply buy the data from a single agency, such as a national point of contact for satellite data;

* restricted areal coverage;

* the need for a greater degree of understanding of the characteristics of the sensor system in terms of its calibration aspects etc;

* the problems of wide view angle encountered when covering a reasonable physical swathe at relatively low altitude, leading to radically different viewing geometries for different parts of the overall scene.

The limitations of both satellite and aircraft remote sensors to weather conditions (e.g. cloud cover) would mean that a truly random monitoring programme could not be achieved. For example during the randomly designated day for sampling there may be too much cloud cover for successful measurement, also measurements during dark hours would not be possible. In addition a systematic monitoring programme would not be readily achievable. In a situation when it was specified that measurements were required at specified times of day or tide then again conditions may prevent measurement. (This may also be true of ship sampling as well, although this is likely to be the case less frequently.) To obtain a complete picture and assessment of algal production in tidal waters in England and Wales it is suggested that a minimum of nine surveys would have to be undertaken from the beginning of March to end of November, a total of nine surveys.

The number of algorithms required for ATM type sensors (and possibly spectral analysers) for the coastal strip would have to be considered, and whether the algorithms change over seasons and from year-to-year. Ship cruises might, therefore, have to be undertaken at the same time. It is likely that the limit of detection and sensitivity of the sensors would be adequate for classification purposes.

It would appear, therefore, that sensors based on fluorescence spectral analysis and mounted on aircraft offer the best opportunity for obtaining reliable chlorophyll data from tidal waters. It is likely that in the short term at least sea truth data would be required to calibrate and validate the method. In particular, no data would be obtainable from subsurface water layers and this might be very important in situations where some form of stratification is occurring.

The situation with regard to estuaries is also complicated by the presence of greater concentrations of interfering substances such as suspended sediment and organic matter compared to that generally found in coastal waters. There will also be greater differences in salinity which may also confound the results. It is suggested, therefore, that the method be tested in estuaries before its general applicability and suitability for all tidal waters is accepted. In estuaries, there may also be the issue of degree of spatial resolution required in relation to the width and length of the estuary channel. The limit of detection and accuracy of imaging would also have to be assessed against the need to differentiate images between different trophic classes.

The estimated annual costs of implementing the remote sensing option have been taken from Pentreath 1992 and are based on only four surveys a year. As there is no information on how the costs are made up, no attempt has been made to scale up the costs to reflect the increased number of surveys suggested. They must, therefore, be treated as the minimum estimate.

Table 6.8 **Estimated annual costs of implementing remote sensing option (modified from Pentreath 1992)**

	£K
Salaries and hired labour costs	150
Analysis cost for samples	10
Costs of operating survey vessels	75
Aircraft operating	17
Total revenue cost$	252
Capital costs (one off)	355.5

NB:

These costs are based on 4 surveys per year only and include the analysis of sample for saline nutrients and chlorophyll only, at 174 sites per survey in the coastal zone. In reality, at least month, and preferably weekly sampling would be required.

7. LINKAGE WITH OTHER CLASSIFICATION CRITERIA

7.1 NRA proposed classification scheme

It was initially proposed (NRA 1991) that the classification scheme for estuaries and coastal waters would be based on a number of components: aesthetic quality; water quality (ammonia and dissolved oxygen), biological quality, nutrient contaminants in water and sediment contaminants for estuaries; and for coastal waters all the above components except for water quality which would be based on microbiological quality.

As has been described in previous sections of this report the most obvious linkage with other classification criteria would be the nutrient component. It has been suggested that the latter would be based on winter nutrient contamination levels giving direct comparisons rather than an assessment of what these could represent in terms of productivity or potential productivity of the waters. This method could therefore classify waters as 'poor' when other environmental factors means that the system may not be very productive.

A measure of trophic status could only be indirectly linked with the other proposed classification criteria. For example, an indirect effect of eutrophication could be the production of large masses of organic matter which on degradation could decrease oxygen levels. Similarly algal scums and rotting seaweeds could decrease the aesthetic value of coastal waters and estuaries. There could also be secondary effects on the benthic ecology which could potentially be reflected in changes in community structure and hence in biological classification.

7.2 Criteria for sensitive areas

It is also desirable that the classification of trophic status could be relevant to other needs of the NRA such as identifying sensitive estuaries and coastal waters in terms of the Urban Waste Water Treatment Directive (CEC 1991a). This Directive lays down minimum standards for the provision of sewerage systems and sewage treatment. The sewage treatment standards vary according to the nature and sensitivity of the area receiving the sewage discharge and the size of the discharge. The Directive specifies secondary treatment as the norm but provides for higher standards for discharges to sensitive areas, and at least primary treatment for discharges to less sensitive areas. Member states are required to identify sensitive and less sensitive areas for the purpose of this Directive by the end of 1993.

One of the criteria for identifying sensitive areas is:

" freshwaters, estuaries and coastal waters which are eutrophic or which in the near future may become eutrophic if protective action is not taken."

The Department of the Environment has produced a consultation paper on criteria and procedures for identifying sensitive and less sensitive areas (DoE 1992). The criteria for identifying sensitive estuaries and coastal waters are given below.

a) Nitrate concentrations

Winter (February) nitrate nitrogen concentrations significantly enhanced relative to a background concentration for a defined geographical area based on salinity.

b) Occurrence of exceptional algal blooms

Occurrence of unusual blooms of phytoplanktonic species or of blooms of unusual scale or blooms with unusual toxicity characteristics. Blooms of algae in coastal waters normally reach densities of at least 5×10^5 cells per litre and chlorophyll-a concentrations of around 10 mg m^{-3}.

c) Duration of algal blooms

Considered exceptional if the normal spring algal densities persisted through the summer until the autumn bloom without the typical nutrient-limited decline in the summer.

d) Oxygen deficiency

Decreased oxygen concentrations at the surface, as well as in deeper water layers, including in areas where sedimentation and/or stratification may occur.

e) Reductions in fauna

Substantial increases or decreases in benthic biomass, shifts in species composition and mortality of benthos and fish.

f) 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, for example dense and widespread growth of *Enteromorpha* spp.

g) 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 of the present chronic phenomenon. Such an extension could indicate eutrophication but could also be due to a variety of natural causes.

h) 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 sea surface or on the strand-line, or slimy deposits on fish nets or drogues). 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.

It can be seen from the above that the criteria are in most cases rather descriptive rather than entirely objective. However, they include most of the options discussed in this report. Hence changes in macroalgae growth, winter nutrient concentrations and chlorophyll concentrations are included. The other option for assessing trophic status, modelling, also offers the opportunity of identifying sensitive areas particularly of estuaries and semi-enclosed coastal waters.

7.3 The Nitrate Directive

In June 1991, the EC adopted a directive concerning the protection of fresh, coastal and marine waters against pollution caused by nitrates from diffuse sources (CEC 1991b). It requires Member States to identify by mid-December 1993 waters affected by pollution from nitrates and waters which could be affected by pollution from nitrates if protective action is not taken. One of the main criteria which are to be used in identifying polluted waters includes:

" freshwater, estuaries, coastal and marine waters which are eutrophic or which in the near future may become eutrophic if protective action is not taken."

Annex 4 of the Directives lays down the frequency with which the member states must monitor their waters. Section 1c stipulates that members must review the eutrophic state of their surface, estuarine and coastal waters every three years.

The options for assessing the trophic status of tidal water would also, therefore, be of direct relevance to the requirements of this directive.

8. COMPARISON OF CLASSIFICATION OPTIONS FOR TROPHIC STATUS

The different options described in the previous sections are compared against a number of criteria in Table 8.1. Also given is an estimate of the cost of undertaking the required sampling for each option in all coastal waters and estuaries in England and Wales: the basis of these costs was given in Section 6. To try to assess the cost-effectiveness of each option, scores have been assigned to each option. The scores were derived by assessing each option against the described criteria, with the 'best' option scoring 6 and the 'worst' 1. Even though this is rather arbitrary as no weighting of relative importance of criteria (and therefore scores) has been given, it was felt that it was worth doing to give an indication of which option might be the most cost-effective.

Class-limiting thresholds

If the modelling option was successfully developed it could potentially define the class limiting thresholds most precisely as the definition would be based on statistical relationships. As indicated earlier in the report there is, however, uncertainty as to whether the method would be applicable, particularly in offshore coastal waters. The use of *in situ* fluorimetry would also potentially give a good definition of thresholds, particularly since much more data would be available to form the basis of the classification. Remote sensing would also be potentially as good as a very large spatial coverage of tidal waters would be obtained, though only again as a snap shot in time. The discrete sampling, nutrient monitoring, algal bioassay and macroalgal biomass options would suffer in comparison because of the lack of spatial and also temporal coverage of data which would not allow the thresholds to be set with as much certainty.

Applicability

The most applicable options would be those sampling the water column for chlorophyll-a as a surrogate for phytoplankton biomass (discrete sampling, algal bioassay and *in situ* fluorimetry) and nutrient monitoring. On the other hand macroalgae may not grow in certain estuary and coastal areas (e.g. lack of suitable substrata), and may not, therefore, be universally applicable. Remote sensing has not yet been fully validated particularly for estuary use and the need for sea truth data to validate outputs has to be assessed. Similarly the modelling approach has not been tried and tested in UK waters and there is uncertainty whether it would be successful, particularly in coastal waters.

Ease of application

The options based on established techniques would be the easiest to apply with some limitations and constraints such as weather being well understood. Most of the analytical techniques are also well developed and tested. On the other hand a lot of additional data and effort (e.g. definition of loads) would be needed to set up the modelling option. There are also some potential logistic problems with remote sensing, such as the need for synoptic sea truth data, the limitations of cloud cover and the precision of chlorophyll determinations. The sampling of macroalgae may also cause logistic problems with regard to the need to sample intertidally. Extra effort would also be required for the application of the *in situ* algal bioassay option where potentially buoys would have to be deployed and maintained at many sites.

Table 8.1 Comparison of different options for classifying coastal waters on trophic status

OPTION Criteria	A	B	C	D	E	F	G
Timing of sampling	Mar-Nov	Mar-Nov	Dec-Feb	Mar-Nov	Mar-Nov	All year	Mar-Nov
Frequency of sampling	Monthly (9 per season)	Monthly (9 per season)	3 per winter	Monthly (9 per season)	Monthly (9 per season)	Monthly (12 per year)	Monthly (9 per season)
Class-limiting thresholds	2.5	5.5	2.5	2.5	2.5	7	5.5
Applicability	5.5	5.5	5.5	1	5.5	2.5	2.5
Ease of application	6	6	6	2.5	2.5	2.5	2.5
Reliability	5.5	5.5	3.5	1.5	1.5	7	3.5
Practicality	5.5	5.5	5.5	3	5.5	1.5	1.5
Routine monitoring	4	4	4	4	4	4	4
Data availability	4	6	7	2	1	4	4
Total score	33	38	34	16.5	22.5	28.5	23.5
Cost (£'000)	1569	1364	548	1416	1817	2306	504*
Cost-effectiveness	48	36	16	86	81	81	21

Key:

A = Discrete sampling
B = *In situ* fluorimetry
C = Nutrient monitoring
D = Macroalgal biomass
E = Algal Bioassay
F = Statistical model
G = Remote sensing
* = Costs based on four surveys and an estimate for estuaries
7 = Best by comparison
1 = Worst by comparison

Reliability

Reliability here is taken to mean the level of certainty that would be associated with the classification scheme, particularly when classifying a very variable parameter. Again the most reliable option would appear to be that associated with a statistical assessment of loads and effects i.e. the modelling option. In the case of discrete sampling and *in situ* fluorimetry there would be uncertainty associated with the level of data collection. The number of samples taken or the amount of data collected, (and whether it is affordable), will directly relate to the precision of the classification. Similarly remote sensing has not been fully tested in estuaries and the need to measure subsurface chlorophyll may be a major problem in some areas. The nutrient monitoring option may also turn out to be less reliable as it is based on contaminant levels rather than on measured effects. The growth of macroalgae would appear to be subject to even more variability than sampling water (e.g. lack of suitable habitat, changes in substrata from year to year). The reproducibility of results arising from the algal bioassays is also unknown which could potentially add extra uncertainty. These latter two options are, therefore, scored as the least favoured options under this criterion.

Practicality

Remote sensing and the modelling option are the least favoured options under this criterion, again largely due to the remaining uncertainty about the techniques and how practical it would be: for example, limitations to flying in the former case; the ease of assessing loads in the latter. The options involving water sampling are considered to be equally practical as they are relatively straight forward (weather permitting) whereas there may be practical difficulties in measuring macroalgal biomass (e.g. large samples to be handled).

Routine monitoring

All options as suggested involve an increased level of sampling from that already routinely undertaken by the NRA. For example, macroalgae are sampled in some estuaries for contaminant monitoring, samples for nutrients are taken as part of the harmonised monitoring programme, and discharges are sampled for consent purposes. The options have, therefore, been scored equally.

Data availability

Though there is at present not enough data to develop a classification based on any of the options, the most favoured option under this criterion would appear to be nutrient monitoring, followed by *in situ* fluorimetry which has produced good spatial data in some areas. Potentially there would be extensive spatial data from a limited number of remote sensing surveys on which a preliminary classification could be attempted. Little data were identified on macroalgal biomass and none on the algal bioassay option.

Costs

A comparison of the estimated costs of implementing the different options in the tidal waters in England and Wales is presented in Table 8.2. The basis of individual costs has been given in Section 6. It should be noted that the cost of the remote sensing option is based on four surveys only and should, therefore, be considered as an estimate of minimum costs only.

Table 8.2 Comparison of estimated annual costs of implementing different options for assessing trophic status

	A (£'000)	B (£'000)	C (£'000)	D (£'000)	E (£'000)	F (£'000)	G$ (£'000)
Estuaries	1197	1089	265	414	1329	1806	252*
Coastal waters	372	275	283	1002	488	496	252
Total	1569	1364	548	1416	1817	2306	504

Key:

A = Discrete sampling
B = *In situ* fluorimetry
C = Nutrient monitoring
D = Macroalgal biomass
E = Algal Bioassay
F = Statistical model
G = Remote sensing
$ = Based on four surveys, no estimate available for more frequent surveys
* = Assumed to be equivalent to coastal survey

On this basis the most expensive option would be the modelling option and the cheapest nutrient modelling. Other possible monitoring scenarios for options may, of course, be cheaper (e.g. use of helicopters for discrete sampling) and some monitoring may also be included in existing monitoring programmes.

Cost effectiveness

An estimate of cost-effectiveness of each option has been arrived at by dividing the estimated costs (in Table 8.2) by the total score under the defined criteria (Table 8.1). No weighting was given to any criterion which might be used to reflect its relative importance. The most cost-effective option in this assessment would appear to be nutrient monitoring and the least macroalgal biomass monitoring. Though remote sensing appears to be very cost-effective, there is a large level of uncertainty associated with the cost estimates, since the costs for this option are known to be artificially low.

9. CONCLUSIONS

1. It would be desirable and scientifically justifiable to include a measure of trophic status in classification or general quality assessment schemes for tidal water. This would enable a comparison of quality to be made on a defined biological effect, and counter measures to reduce this effect (if necessary) would be more cost-effectively targeted than in the case where comparisons were made on contaminant levels alone. The inclusion of such a well-publicised effect might also be more publicly or politically acceptable than one that did not.

2. Increasing the nutrient status of tidal waters (nutrification) does not imply a *pro rata* effect on algal growth, since nutrients have to be the growth-limiting resource for this to occur. Whilst a single trophic status classification scheme for both coastal and estuarine waters may be desirable, it should be realized that other factors make the two types of environment very different. Water clarity tends to be very much lower in estuaries than coastal waters due to the higher levels of suspended sediment. The higher nutrient levels and reduced light penetration in estuaries means that light, rather than nutrients, is more likely to be the major factor limiting algal growth.

3. Even though a large body of evidence shows that in saline waters phytoplankton standing crop is increased following anthropogenic inputs of nutrients to semi-enclosed regions of water, phytoplankton blooms have been shown to occur naturally in saline water of low nutrient status. In addition, large standing crops of macroalgae such as kelp can develop in relatively nutrient poor waters.

4. Nutrient dynamics, and hence the potential for primary production, in coastal and estuarine waters are controlled by many factors. Despite the fact that these controlling factors in estuaries and coastal waters are very similar, their relative importance is different enough to ensure that the behaviour and distribution of macronutrients is far more variable in estuarial waters than in the coastal environment.

5. There are many factors effecting algal production intidal waters, these include : residence time in estuaries and semi-enclosed coastal bays, water turbidity, salinity, stratification and biotic factors such as grazing by primary consumers. These factors will have to be accounted for in any classification of trophic status.

6. Sampling and monitoring for trophic status (for classification) would have to cope with the inherent variability associated with nutrient dynamics and algal production in tidal waters. Past experience has shown that, even when estuarine water quality is expressed as a simple overall mean or percentile, it is difficult to do sufficient sampling to provide an adequately precise quality assessment. This will also apply to an assessment of the class of body of tidal water. There are potentially two approaches of addressing this difficulty: the use of a structured sampling programme, and the use of statistical/mathematical models.

7. If properly designed, a structured sampling strategy may be able to reduce much of the variability observed in tidal waters and especially in estuaries. In this way

comparable standardized data would be obtainable for classification purposes. Ideally, the strategy would be designed from an assessment of the relative variability associated with the factors affecting the selected determinand such as chlorophyll levels. This would entail a detailed examination of existing datasets (if they are comprehensive enough) or from an intensive monitoring programme over a number of years and sites.

8. The sampling programme would need to be designed to study all the factors likely to affect the selected determinand and would attempt to assess the relative size of sources of variation such as seasonality, tidal stage, depth, position in the estuary, sampling depth etc. Such an assessment of variability could be used to determine the sampling frequency (number of samples, scans etc.) required to achieve a desired precision in classifying tidal waters.

9. A number of options for assessing trophic status have been considered in detail. These were: discrete sampling for chlorophyll and phytoplankton biomass; *in situ* fluorimetry; monitoring for nutrient levels; macroalgal biomass; the use of algal bioassays; the use of statistical models; and remote sensing from aircraft.

10. It was concluded that there was at present no clearly-defined best option for assessing trophic status in all tidal waters. The most favoured options at present appear to be sampling for chlorophyll and phytoplankton biomass, remote sensing from aircraft and the potentially most robust but most expensive option, the use of statistically based models.

11. There are still technical and operational difficulties associated with the favoured options. For example, chlorophyll can be an unreliable indicator of phytoplankton biomass, remote sensing might not be applicable to estuaries, and statistical modeling has not been widely tested in UK waters.

12. All options would require an increased level of sampling (and hence increased cost) in tidal water by the NRA. The amount of extra sampling would depend, amongst other factors, on the desired level of precision of the classification and what is affordable.

13. Once the appropriate database has been established the classification of trophic status could be based upon the range of observed primary production in tidal waters of England and Wales, with class boundaries calculated using simple statistical rules to divide the frequency distribution into suitable groups.

14. In terms of remote sensing from aircraft the usefulness of sensors like CASI for the measurement of chlorophyll-a, is likely to be limited by the need of control (sea truth) data. The uncertainty in estuarine environments, in particular, dictates that temporally, and preferably also spatially (surface) distributed control data in the form of simultaneous in-situ measurements are necessary for complete confidence. This is likely to remain the case even if the methods can be proven under a wide range of conditions. In this respect, such techniques cannot themselves replace traditional

surveys but offer a valuable source of spatial (and, to a lesser extent, temporal) sea-surface information as a cost-effective supplement.

15. Providing enough images can be obtained (and afforded), the remote sensing option offers the greatest potential for obtaining a very large amount of spatial data (be it a snapshot in time) covering all tidal waters in England and Wales. Once obtained, suitably calibrated and corrected for chlorophyll, direct comparisons could be made and a classification established as for the other options involving the assessment of chlorophyll levels. There may also be a need to specify specific times for imaging in terms of high and low water in estuaries, this may reduce the likely tidal cycle variability of chlorophyll in estuaries. The method is not yet fully validated in terms of suitability for estuarine use and in terms of the need for sea truth data and for information on the vertical profile of chlorophyll in the water column.

16. The three major advantages offered by satellite imagery (compared to aircraft) for estimating phytoplankton abundance are: the very large areas which can be monitored; the ease with which such monitoring can be undertaken (providing meteorological conditions are suitable); and the relatively low cost compared to intensive manual sampling/analysis. A major disadvantage is the frequency of data that would be required from any one geographic location.

17. The number of algorithms required for colour and thermal image type sensors (and possibly spectral analysers) for the coastal strip would have to be considered, and whether the algorithms change over seasons and from year-to-year. Ship cruises might, therefore, have to be undertaken at the same time. It is likely that the limit of detection and sensitivity of the sensors would be adequate for classification purposes.

18. It would appear, therefore, that sensors based on fluorescence spectral analysis and mounted on aircraft offer the best opportunity for obtaining reliable chlorophyll data from tidal waters. It is likely that in the short term at least sea truth data would be required to calibrate and validate the method. In particular, no data would be obtainable from subsurface water layers and this might be very important in situations where some form of stratification is occurring.

19. The assessment and classification of the trophic status of tidal waters would be directly relevant to the requirements of the NRA (and DoE) to comply with the requirements of the EC Urban Wastewater Treatment and Nitrates Directive.

20. Trophic status could most readily and relevantly linked with the proposed classification based on nutrient contamination levels, particularly if the statistical modeling option was pursued, where potentially both primary productivity and nutrient levels could be linked with nutrient loads and hence tidal water quality management.

21. Trophic status classification would only be indirectly related to other components of the proposed GQA schemes through secondary effects such as deoxygenation arising from algal biomass production and related potential changes in marine benthic

invertebrate communities. The presence of algal scums would also affect the aesthetic quality of tidal waters.

10. RECOMMENDATIONS

1. It is recommended that a comparison of trophic status is included in a GQA scheme for tidal waters. – why?

2. The more favoured options should be further investigated with the view of assessing their universal applicability, technical feasibility and cost effectiveness. The aim would then be to establish a database of the determinand associated with the preferred option from which the classification could be established.

3. It is recommended that the statistical model approach is tested in a pilot study on a number of estuaries and enclosed coastal bays covering a range of different geographic and water quality types, and for which there may already be a database of information that could be used.

4. It is recommended that a sampling programme is instigated to determine levels of chlorophyll, phytoplankton numbers and biomass in a number of estuaries and coastal waters, and to assess the relative importance of the different sources of variability. This could form the basis of a suitable database of information and could be used to devise a more cost effective sampling/monitoring strategy. This study could also form part of the study associated with recommendation 3 above.

5. The situation with regard to the remote sensing of estuaries is complicated by the presence of greater concentrations of interfering substances such as suspended sediment and organic matter compared to that generally found in coastal waters. It is recommended, therefore, that the method be tested in estuaries before its general applicability and suitability for all tidal waters is accepted. In estuaries, there may also be the issue of degree of spatial resolution required in relation to the width and length of the estuary channel. The limit of detection and accuracy of imaging would also have to be assessed against the need to differentiate images between different trophic classes.

6. It is recommended that a preliminary classification of waters in the coastal zone around England and Wales is made from the remote sensing surveys planned for the spring/summer of 1993. From this a preliminary assessment can be made as to whether the precision and accuracy of image analysis for chlorophyll is adequate for the differentiation between areas and classes. It might also be possible to define the optimum unit (in terms of area) for classifying coastal waters.

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APPENDIX A SUMMARY OF AVAILABLE DATA

Table A1 Summary of tidal monitoring programmes

Regulatory Authority	Location	Sampling Sites	Frequency pa	Spring/Neap Tidal Stage	Daily Tidal Stage	Depth	Determinands NO_3	NO_3	NO_2	TON	TN	SRP	TP	Si	chl-a
Anglian[1]	Welland	3	12	-	L+H	-	-	X	-	X	-	X	-	X	X
	Nene	6	12	-	L+H	-	-	X	-	X	-	X	-	X	X
Northumbrian	Blythe	16	4	-	HTC	S,MD,B	X								
	Tyne	6	8	N	-	S,MD,B	X	X	X	-	X	X	X	X	X
	Wear	10	4	N	L	S,MD,B	X	X	X	-	X	X	X	X	X
	Tees	13	4	N	M	S,MD,B	X	X	X	-	X	X	X	X	X
Severn Trent	Humber	No information received													
Southern[2]	Medway	14	12	-	L+H	-	X	X	X	X	-	X	-	X	-
	Stour	8	12	-	L+H	-	X	X	X	X	-	X	-	X	-
	Ouse	2	12	-	L+H	-	X	X	X	X	-	X	-	X	-
	SE coast[3]	21	4	-	-	0.5 m	X		X	X	-	X	-	X	X
South West	Axe	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Otter	1	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Teign	3	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Avon	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Erme	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Yealm	6	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Lynher	6	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Looe	4	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Restronguet Ck	3	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Truro	5	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Carrick Roads	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		x	
	Helford	5	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Hayle	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Gannel	2	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Camel	7	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Torridge	4	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Tawe	4	4	-	FTC	S,MD,B	X	-	-	X	-	X		X	
	Dart	6	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	

Table A1 Continued...../2

Regulatory Authority	Location	Sampling Sites	Frequency pa	Spring/Neap Tidal Stage	Daily Tidal Stage	Depth	Determinands NO_3	NO_3	NO_2	TON	TN	SRP	TP	Si	chl-a
	Kingsbridge	5	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Plym	3	3-4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Plymouth Sound	3	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Tamar	11	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Tavy	1	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Fowey	9	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Penryn	2	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Tresillian	2	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Percuil	2	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
	Fal	4	4	-	FTC	S,MD,B	X	-	-	X	-	X	-	X	
Thames	30	40	-		M	S	X	-	-	X	-	X	-	X	X
Welsh[4]	Severn	23	4	N	H	S	X	X	X	X	-	X	X	-	-
	Teifi	- (5)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	Dovey	- (7)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	Dee	- (24)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	Cardigan Bay[5]	- (32)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	N-Wales coast[5]	- (29)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
Wessex	Poole H	4	V	-	-	S	X	X	X	X	-	X			
	Portland H	2	V	-	-	S	X	X	X	X	-	X			
	Severn	23	4	N	H	S	X	X	X	X	-	X		X	
	SW coast[3]	48	4	-	-	0.5 m	X	-	X	X	-	X		X	X
Yorkshire	Humber	4	6	-	-	S	X	-	-	-	-	-	-	-	-
	Conwy														
North West	Dee	- (24)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	Mersey	22(19)[6]	12(2)	S(-)	H(-)	S	X	X	X	X	-	X	X	-	-
	Ribble	12 (19)[6]	5(2)	S(-)	(-)	S	X	X	X	X	-	X	X	-	-
	Solway	- (6)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
	NW coast[5]	- (80)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-

Table A1 Continued...../3

Regulatory Authority	Location	Sampling Sites	Frequency pa	Spring/Neap Tidal Stage	Daily Tidal Stage	Depth	Determinands NO_3	NO_3	NO_2	TON	TN	SRP	TP	Si	chl-a
	Irish Sea[7] (Mersey to IOM)	- (27)[6]	(2)	(-)	(-)	S	X	X	X	X	-	X	X	-	-
Solway	Solway	No information received													
Clyde	Clyde Estuary	9	6	-	-	DP	X	X	X	X	-	X	-	-	X
	Clyde Firth	27	6	-	-	1	X	X	X	X	-	X	-	X	-
	Gare Loch	2	6	-	-	DP	X	X	X	X	-	X	-	X	X
	Loch Long	2	6	-	-	DP	X	X	X	X	-	X	-	X	X
	Holy Loch	1	6	-	-	DP	X	X	X	X	-	X	-	X	X
	Loch Goil	2	6	-	-	DP	X	X	X	X	-	X	-	X	X
	Loch Striven	2	6	-	-	DP	X	X	X	X	-	X	-	X	X
	Loch Fyne	3	1	-	-	DP	X	X	X	X	-	X	-	X	X
	Loch Creran	v	v	-	-	DP	X	X	X	-	-	X	-	-	X
	Irvine/Ayr Bays	47	4	-	-	1	X	X	X	X	-	X	-	X	X
	Oban Bay	7	-	-	-	DP	X	X	X	-	-	X	-	X	X
Highland	Inverness/ Beauly Firth	17+10	1/2	-	-	1	X	X	X	-	-	X	-	-	-
	Cromarty Firth	10	2	-	-	1	X	X	X	-	-	X	-	-	X
North East	Ythan	3	3-6	-	-	S	X	-	X	X	-	X	X	-	-
	Dee/Aberdeen H	9	<1	-	H+M+L	S	X	-	X	X	-	X	X	-	-
Tay	Tay	No information received													
Forth	Forth estuary	17	24	S + N	-	1+B	X	X	X	-	-	X	-	-	X
	Forth firth	33	1	-	-	DP	X	X	X	-	-	X	-	X	-
Tweed	Tweed	2	3/4	-	-	S	X	-	-	X	X	X	-	X	-
	Eyewater	1	6	-	-	S	X	-	-	X	X	X	-	X	-
Northern Ireland	Larne Lough	5	4-6	-	-	2	X	-	-	X	-	X	-	X	-
	Carlingford Lough	4	4-6	-	-	2	X	-	-	X	-	X	-	X	-
	Strangford Lough	4	4-6	-	-	2	X	-	-	X	-	X	-	X	-
	Belfast Lough	6	4-6	-	-	2	X	-	-	X	-	X	-	X	-

Table A1 Continued...../4

Regulatory Authority	Location	Sampling Sites	Frequency pa	Spring/Neap Tidal Stage	Daily Tidal Stage	Depth	Determinands NO_3	NO_3	NO_2	TON	TN	SRP	TP	Si	chl-a
	Lough Foyle	5	4-6	-	-	2	X	-	-	X	-	X	-	X	-
	Coastal sites	4	4-6	-	-	2	X	-	-	X	-	X	-	X	-

Notes:

1 Other estuaries which are monitored in this region are : Humber, Witham, Wash, Great Ouse, Bure, Yare, Waveney, Blythe, Alde/Ore, Deben, Orwell, Stour, Colne, Blackwater, Crouch, Roach.

2 Other estuaries monitored in this region are: Rotter, Swale, Milton Creek, Faversham Creek, Beualieu, Hamble, Itchen, Test, Lymington, Solent, Langstone, Portsmouth, S'hampton Water, Arun, Adur, Cuckmere, Chichester.

3 Monitoring was conducted as part of the coastal baseline survey at points 1-2 miles from the shore.

4 Other estuaries which are monitored in this region are: Afan, Ogmore, Loughor, Neath, Tawe, Swansea bay, Menai Straits.

5 Monitoring was conducted 1-2 miles from the shoreline.

6 Sampled for the NORSAP programme summer 90, winter 90/91 and summer 91.

7 Offshore run from the Mersey to the Isle of Man.

Key:

Determinand

TON	Total oxidised nitrogen
TN	Total nitrogen
SRP	Soluble Reactive Phosphorus
TP	Total phosphorus
chl-a	Chlorophyll-a
Si	Silicate

Tidal stage

S	Spring tide
N	Neap tide
H	High tide
F	Full tide
L	Low tide
M	Mid tide
FTC	Full tidal cycle
HTC	Half tidal cycle

Depth

S	Surface
B	Bottom
MD	Mid-depth
DP	Depth profile

Table A2 Relevant surveys identified by the MPMMG (1991)

Source	Location	Objectives / Description of Programme	Frequency	Determinands
Yorkshire WA Humber Est Committee	Humber	Assess compliance with EQS set by HEC. Water quality programme is part of a co-ordinated to look at inputs , water quality and sediment quality. Store based sampling at HW and LW at 18 stations throughout estuary as far downstream as Spurn.	8/Yr	NH_4 NO_3
Thames WA	Thames	Water Quality sampled at 1m depth at 15 sites (Richmond - Barrow No 7 Buoy)	Weekly	P04, NO_3 chl-a
Southern WA	20 Estuaries	Estuarine water quality monitoring programme. Spot surface samples at 82 sites Medway (14), Stour (8), Rotter (1), Swale (18), Milton Creek (5), Faversham Creek (4), Bealieu (1), Hamble (2), Itchen (2), Test (2), Lymington (1), Solent (1), Langstone Harbour (1), Portsmouth Harbour (2), Southampton Water (5), Arun (5), Adur (2), Ouse (2), Cuckmere (1), Chichester (5)	12/yr	NH_4 NO_3 NO_2 PO_4
Southern WA	Medway & Swale	Water Quality monitoring. Medway (17), Swale (18).	12/yr	NH_4 NO_3 NO_2 PO_4
Wessex WA Severn Estuary Joint Committee	Severn	Helicopter sampling at 28 stations	4/Yr from 1975 chl-a	NH_4, NO_3 N02, PO_4
Severn-Trent	Severn	Full estuary survey 30 sites, on spring tides	2/Yr	NO_3, NH_4 PO_4
		Lydney Tidal Cycle, 12 samples over one tidal cycle	2/Yr	NO_3, NH_4
		Upper estuary neap survey. 14-17 sites sampled once.	6/Yr from	NO_3,NH_4

Table A2 Continued...../2

Source	Location	Objectives / Description of Programme	Frequency	Determinands
			1978	chl-a
		Upper estuary spring survey 20-23 sites sampled once.	3/Yr	NO_3, NH_4 chl-a
Welsh WA	Usk	HW and LW sites at 2 sites on spring tides	6/Yr	NO_3,NO_2 PO_4,NH_4
	5 estuaries	Ebbw (1), Rhymney (2), Taff (2), Ely (2), Thow (1), all at LW	6/Yr	NO_3,NO_2 PO_4,NH_4
	4 estuaries	Ogmore (1), Kenfig (1), Afan (1), Tawe (3)	12/Yr	NO_3,NO_2 PO_4,NH_4
	Lougher	LW at 2 sites, HW ±2 at 1 sites	12/Yr	NO_3,NO_2 PO_4,NH_4
Welsh WA	Gwendrath, Taff & Twyi	Under summer low flow spring tide conditions, sample LW at 2 sites	*ad hoc*	NH_4,NO_3 NO_2,PO_4 chl-a
	Dee	9 stations sampled mid-flood to high water in main channel.	12/Yr	NH_4,NO_3 NO_2,PO_4
	Menai Straits	24 stations throughout Straits at HW and LW	4/Yr	chl-a
	North Wales Coastal Waters	Inshore surface water quality. A grid of 20 stations between Dee and LLandudno	6/Yr NO_2,PO_4	NH_4,NO_3

Table A2 Continued...../3

Source	Location	Objectives / Description of Programme	Frequency	Determinands
	Conwy	Conwy tunnel water quality monitoring programme. Fixed site at 1m at 1 site throughout a tidal cycle, samples taken at 1/2 hour intervals.	3*/week	HN4,NO_3 N02,PO_4 chl-a
North West WA	Mersey	Mersey estuary water quality monitoring programme based on an assessment of quality data collected 1962-1982		
		8 bankside stations sampled on neap tides at high and low water.	12/Yr	NH_4,NH_3 NO_2,PO_4
		21 stations in main channel on spring tides at high water	Monthly May-Oct	NH_4,NH_3 NO_2,PO_4