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Defining and detecting undesirable disturbance in the context of marine eutrophication

Paul Tett [a,*], Richard Gowen [b], Dave Mills [c], Teresa Fernandes [a], Linda Gilpin [a], Mark Huxham [a], Kevin Kennington [d], Paul Read [a], Matthew Service [b], Martin Wilkinson [e], Stephen Malcolm [c]

[a] School of Life Sciences, Napier University, 10 Colinton Road, Edinburgh EH10 5DT, United Kingdom
[b] Aquatic Systems Group, AFESD, Department of Agriculture and Rural Development, Newforge Lane Belfast BT9 5PX, United Kingdom
[c] CEFAS, Pakefield Road, Lowestoft, Suffolk NR33 OHT, United Kingdom
[d] University of Liverpool, School of Biological Sciences, Port Erin Marine Laboratory, Port Erin, Isle of Man IM9 6JA, United Kingdom
[e] School of Life Sciences, Heriot-Watt University, Edinburgh EH14 4AS, United Kingdom

Abstract

An understanding of undesirable disturbance to the balance of organisms is needed to diagnose marine eutrophication as defined by EU Directives and OSPAR. This review summarizes the findings of the UK Defra-funded Undesirable Disturbance Study Team, which concluded that 'an undesirable disturbance is a perturbation of a marine ecosystem that appreciably degrades the health or threatens the sustainable human use of that ecosystem'. A methodology is proposed for detecting disturbance of temperate salt-water communities dominated by phytoplanktonic or phytobenthic primary producers. It relies on monitoring indicators of ecosystem *structure* and *vigour*, which are components of health. Undesirable disturbance can be diagnosed by accumulating evidence of ecohydrodynamic type-specific changes in: (i) *bulk indicators*; (ii) *frequency statistics*; (iii) *flux measurements*; (iv) *structural indicators*; and (v) *indicator species*. These are exemplified by (i) chlorophyll, transparency, dissolved oxygen, and opportunistic seaweed cover; (ii) HABs frequency; (iii) primary production; (iv) benthic and planktonic 'trophic indices'; (v) seagrasses and *Nephrops norvegicus*. Ecological Quality Objectives are proposed for some of these. Linking the diagnosis to eutrophication requires correlation of changes with nutrient enrichment. The methodology, which requires the development of a *plankton community index* and emphasizes the importance of primary production as an indicator of *vigour*, can be harmonized with the EU Water Framework Directive and OSPAR's *Strategy to Combat Eutrophication*.

1. Introduction

The EU Urban Waste Water Treatment Directive (UWWTD) and Nitrates Directive, and OSPAR's 'Strategy to Combat Eutrophication', provide similar definitions of eutrophication. The first part of the OSPAR (2003) definition is representative:

> "Eutrophication" means the enrichment of water by nutrients causing an accelerated growth of algae and higher forms of plant life to produce *an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned ...*

A water body identified as suffering from eutrophication is labelled as *sensitive* under the UWWTD, *nitrate-polluted* under the Nitrates Directive, and a *problem area* under OSPAR's strategy. The consequences of such identification are more stringent treatment of urban waste water, reduction in the use of nitrate fertilizers on land, and measures to reduce or to eliminate the anthropogenic causes of eutrophication. The last is an explicit requirement of OSPAR's strategy and might well be required under the Water Framework Directive (WFD). The practical implications

* Corresponding author. Tel.: +44 131 455 2633; fax: +44 131 455 2291.
E-mail address: p.tett@ichrachan.u-net.com (P. Tett).

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of these measures extend beyond the issues of sewage treatment and nitrate fertilizer use, to include the need to control nutrient release by agriculture, aquaculture, transport and urban development in general.

UK waters considered to be at risk from eutrophication have until recently been identified mainly by measurements of winter concentrations of nitrate and phosphate and summer concentrations of phytoplankton chlorophyll, which were compared with thresholds such as the 10 mg chl m^{-3} in summer or the 12 μM winter Dissolved Available Inorganic Nitrogen (DAIN) proposed by the CSTT (1994). However, nutrient enrichment and accelerated algal growth are not in themselves harmful, and because these bulk measurements provide little information on the extent of change in the *balance of organisms*, they cannot adequately identify harmful consequences of nutrient enrichment. The UK Department of Environment, Food and Rural Affairs (Defra) therefore commissioned a study aimed at providing (i) a scientifically based definition of *undesirable disturbance* in the context of marine eutrophication and (ii) a monitoring strategy for detecting disturbance and unambiguously diagnosing eutrophication. This paper summarizes and updates the study's findings, which are reported in detail by Anon (2004).

2. The scientific basis: a theory of undesirable disturbance

The *Undesirable Disturbance Study Team (UDST)* dealt with UK marine ecosystems from an estuarine inner limit where the flora and fauna cease to have a substantial marine component, to the edge of the continental shelf. These ecosystems include those in which the characteristic primary producers are seaweeds, seagrasses or microphytobenthos as well as those dominated by phytoplankton. Undesirable Disturbance was defined as

> 'a perturbation of a marine ecosystem that appreciably degrades the health or threatens the sustainable human use of that ecosystem'.

'Ecosystem' is used in the sense of Odum (1959), meaning

> 'any area of nature that includes living organisms and nonliving substances interacting to produce an exchange of materials between the living and nonliving parts ...',

although this factual definition lacks the normative implications associated with the idea of *ecosystem health*. According to Costanza (1992), a healthy ecosystem, like a healthy human body, is a system that functions well and is able to resist or recover from disturbance. This is more than a metaphor, because *ecosystem health* has quantifiable components of *vigour*, *organization*, *resistance* to disturbance, and *resilience* (Mageau et al., 1995).

The *vigour* of an ecosystem lies in its biologically mediated fluxes of energy and materials as well as its ability to recover from disturbance by means of recolonization and population growth. Although these processes, and the food

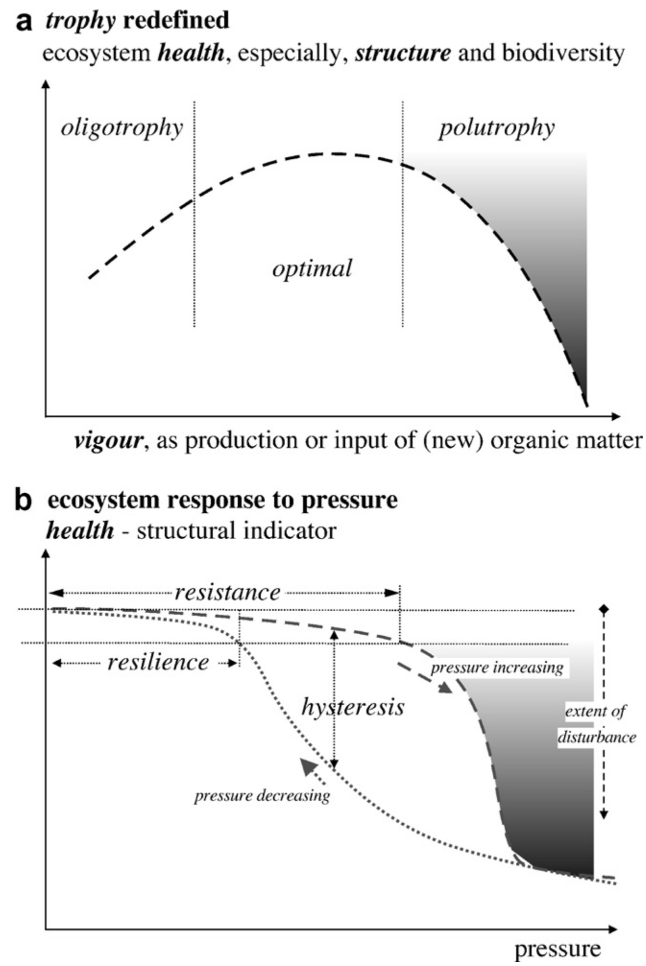

Fig. 1. **Ecosystem health and undesirable disturbance**. The primary components of *health* are good *structure* and optimum *vigour*. These lie behind the ecosystem's resistance to pressure and its *resilience* in recovering from disturbance. Part (a) of this conceptual diagram relates health to vigour as the latter increases with nutrient enrichment; part (b) shows the response of structure to pressure. The two parts of the diagram should be read together to understand why the process of (anthropogenic) *eutrophication* is now understood to imply a change for the worse: increasing pressure from nutrient enrichment might overcome ecosystem resistance and so result in a *polutrophic* state.

supply available to higher levels in marine food chains, depend on primary production, the relationship between production and ecosystem health is not linear (Fig. 1(a)). Exceeding a moderate supply of organic matter can result in a state in which eukaryotic consumers fail to deal effectively with organic input. It is, however, not so much the size of the input as the *uncoupling* between production and use that can lead to the problems associated with eutrophication: Harmful Algal Blooms (HABs); the spread of opportunistic macroalgae; and the deoxygenation of deep water or sediment resulting from the death and decay of excess biomass. The problem state is that which is now commonly called *eutrophic*, but such labelling goes against the Greek etymology (ευτραφεσ, *well-fed, thriving*; ευτϖ ροφοσ, *nourishing, healthy* (Liddell and Scott, 1940)) and original meaning of the German scientific term *eutraphent*

(Hutchinson, 1969). Better naming options include *hypereutrophic* (Hutchinson, 1969), *polluted eutrophic* (Rodhe, 1969), and *hypertrophic* (Nixon, 1995). Here, we use *polutrophic* from *πολυτροφϊα* meaning 'excess of nourishment' in classical Greek (Liddell and Scott, 1940).

The *organization* (or *structure*) of an ecosystem comprises its biodiversity, its food web, and its biophysical structure. A coral reef (high diversity, complex physical structure) and a subpolar pelagic system (low diversity, little physical structure) exemplify the structural variety of marine ecosystems. So far as biodiversity is concerned, a proper balance amongst guilds or life-forms is thought to be more important for ecosystem health than the presence of many species (Hooper et al., 2005; Loreau et al., 2002). A *guild* is a group of species, not necessarily closely related, that have similar ecosystem functions. An example is provided by the large burrowing animals that keep pore waters well flushed and thus help maintain the geochemical state of the sediment and provide suitable environmental conditions for other macrobenthos. The term *lifeform* is more commonly used for functional categories of primary producers, with fucoid seaweeds, seagrasses, diatoms and autotrophic dinoflagellates providing relevant examples. Recent studies (Biles et al., 2003; Bolam et al., 2002) support the hypothesis that the marine shallow-water benthos only functions well when all expected guilds are present, although each guild needs flourishing populations of only a few species.

Fig. 1(b) illustrates how the structural component of ecosystem health could respond non-linearly to increasing ecological pressure, such as nutrient enrichment or toxic pollution. An ecosystem shows *resistance* by initially reacting little to such increases. However, pushed beyond a certain threshold, structural changes can occur rapidly, culminating in a radically altered state from which recovery is slow. A key operational need is therefore to detect a trend towards a widespread undesirable disturbance before the ecosystem has reached the limit of its resistance to nutrient and organic enrichment. Resistance also depends on *ecohydrodynamics*, the risk of polutrophy (for example) occurring at a given level of vigour being dependent on physical conditions and consumer populations. An example of overloading and structural deterioration is to be found in the Baltic Sea, where the occurrence of extensive deep water anoxia and widespread elimination of macrobenthos is ascribed to the nutrient enrichment of a system in which the deep water is only replaced at long intervals (Karlson et al., 2002; Laine et al., 1997).

Resilience is the ability of the ecosystem to recover from disturbance, and ecosystem theory holds that a structurally damaged system has little resilience. This may mean that recovery lags behind reduction in pressure. Studies of the plankton in the nutrient-enriched freshwater plankton of Lake Washington (Edmondson, 1991), and of oil- and detergent-damaged rocky shore communities (Southward and Southward, 1978), have provided classic demonstrations of such hysteresis. In a worse case, the ecosystem could switch to a new stable state (Krebs, 1988; Scheffer and Carpenter, 2003; Scheffer and van Nes, 2004; Tett and Mills, 1991). Although we earlier discounted species richness in relation to community organization and function, species diversity within guilds or lifeforms may be important in aiding resistance and resilience, contributing a variety of detailed strategies and genotypes and so increasing the probability that some species will survive increased pressure. For example, monospecific stands of mangroves appear more likely to be killed by local changes in hydrodynamics (Blasco et al., 1996).

Ecologists distinguish episodic *pulse* from sustained *press* disturbances (Bender et al., 1984). Local pulse disturbances are not considered to be a threat to ecosystem health; indeed, they can increase biodiversity according to intermediate disturbance theory (Connell and Sousa, 1983). A widespread pulse disturbance would be of concern if it brought a weakly resistant ecosystem to the point in Fig. 1(b) at which the graph of structure against pressure begins to descend steeply. Extensive press disturbances, evidenced by widespread and marked deterioration in ecosystem structure, are undesirable. Movement of ecosystem state *towards* a crisis should also be a cause for concern. Such a shift could be difficult to identify from subtle changes in structure, but easier to detect from changes in vigour.

Small-scale anthropogenic pressures, and changes affecting only a small part of an ecosystem, are generally not a cause for concern. They match natural disturbances (e.g., the local anoxia beneath a dead whale) and, in most cases, are comparatively simple to regulate—as in the example of the 'Allowable Zone(s) of Effect' consented beneath salmonid farms in Scotland (Read and Fernandes, 2003). The UK CSTT (1994) sought to distinguish such local perturbations (referred to as a waste discharge's 'zoneA') from impacts on water bodies as a whole. It is these latter that should be, with one set of exceptions, the main subject of concern in relation to undesirable disturbance. The exceptions are where a conservation feature could be disturbed, and these are governed by legislation (e.g., national implementations of the EU Habitats Directive) that could apply irrespective of any undesirable consequences for ecosystem health.

3. Indicators of disturbance

Table 1 lists indicators of change in ecosystem health, based on the theory given above. They fall into five groups: *bulk indicators, frequency statistics, flux measurements*, *structural indicators* and *indicator species*. Roughly speaking, the first three groups relate to *vigour* and *coupling* and the last two groups to the changes in community *structure* that are required to confirm a diagnosis of *undesirable disturbance*. Most of these indicators allow definition of

Table 1
Water types and indicators for disturbance

Ecohydrodynamic water type	Subcategory and notes	*Indicators* that can be used to show disturbance (see Table 2 for EcoQOs); see main text for further guidance	*Correlation* of the following change with nutrient increase requires further study and contributes to a diagnosis of eutrophication when there is evidence of undesirable disturbance
1. *Shallow clear waters*, phytobenthos dominant under reference conditions. Although this category includes the littoral zone, salt marshes are not considered here	(a) General	1.a.1. Water transparency (Secchi depth or diffuse attenuation coefficient)	Decreasing Secchi depth, increasing attenuation coefficient
		1.a.2. Depth of lower limit of macrophytobenthos (if present)	Decreasing depth limit
		1.a.3. Water column chlorophyll concentration (mg/m^3)	Increasing mean concentration
	(b) Seagrass meadows: natural condition (before wasting disease) in soft, moderate-energy substrates in shallow water, typically with reduced tidal range	1.b.1. Extent (area, m^2) of seagrass bed	Decreasing extent
		1.b.2. Mean seagrass biomass ($/m^2$)	Decreasing biomass
		1.b.4. Opportunistic macroalgal or epiphytic microalgal incidence	Increasing incidence
	(c) Perennial macroalgal communities: natural condition on hard or mixed soft/hard littoral and shallow sublittoral substrates; the indicators are proposed only for soft or mixed intertidal substrates; 'seasonal' refers to the growth season	1.c.1. Maximal seasonal % cover of opportunistic seaweeds	Increasing cover
		1.c.2. Maximum seasonal biomass of opportunistic seaweeds	Increasing biomass
		1.c.3. Occurrence of widespread macrobenthic death or of anoxic sediment	Increasing frequency of occurrence
	(d) Microphytobenthos dominant; natural condition in shallow energetic or depositional waters	1.d.1. Benthic chlorophyll (mg/m^2)	Increasing abundance
2. Optically deep *mixed waters*	May be physically deep, or shallow and turbid. Insufficient light for plant or algal growth	2.1. Pelagic chlorophyll concentration (mg/m^3)	Light limitation likely to prevent change in state caused by nutrient enrichment
3. *Offshore stratified waters* with phytoplankton dominant and marked seasonal cycle	Includes: regions of seasonal thermal stratification and Spring–Autumn blooms; those with additional haline stratification and extended growth season; and (tidal mixing) frontal regions which may exhibit natural Red Tides;	3.1. Mean or maximum pelagic chlorophyll concentration (mg/m^3) during growth season	Increasing concentration
		3.2. Frequency of HABs	Increasing frequency
		3.3. Net annual microplankton primary production (NMP)	Increasing annual NMP. *Gross (phytoplankton) primary production (GPP) is expected to correlate with nutrients in this water type and so is more a cause than an indicator of disturbance; see text*
		3.4. Plankton community index	Increasing deviation from reference condition
		3.5. Mean or minimum oxygen concentration in deep water when there is a pycnocline	Decreasing concentration
		3.6. Thickness of sediment oxic layer/depth of RPD	Decreasing thickness or depth
		3.7. Macrobenthic community index, e.g., ITI or AMBI	Change in value of index from reference condition
		3.8. Population density (numbers, or burrows, $/m^2$) of *Nephrops novegicus*	Decreasing population density

(*continued on next page*)

Table 1 (*continued*)

Ecohydrodynamic water type	Subcategory and notes	*Indicators* that can be used to show disturbance (see Table 2 for EcoQOs); see main text for further guidance	*Correlation* of the following change with nutrient increase requires further study and contributes to a diagnosis of eutrophication when there is evidence of undesirable disturbance
4. *Regions of Freshwater Influence (ROFIs)* with variable blooms of phytoplankton	Sediment and benthos highly physically disturbed by tidal and wind-wave stirring, and so benthic indicators not proposed	Indicators 4.1–4.4. same as 3.1–3.4	*Correlations same as those for 3.1–3.4*
		4.5 Occurrence or magnitude of *Phaeocystis* blooms or beach-foam incidents	Increasing frequency or magnitude
		4.6. Frequency and extent of anoxic sediment or death of macrobenthos	Increasing frequency or extent
5. *Regions of Restricted Exchange (RREs)* where phytoplankton abundance depends on flushing rate	Semi-enclosed transitional and coastal waters fall into this category, for which it is necessary to take account of flushing rate as well as optical conditions. In some fjords, haline stratification may persist throughout year, and the flushing of *basin deep water* becomes an important issue	Treat as type 1–4 if appropriate, with following variation	
	(a) Large RREs of EHD type 3	3.5. may become 5.5: mean or pre-flushing minimum oxygen concentration in basin deep water	Decreasing concentration
	(b) Small RREs of EHD types 3 or 4: list of indicators simplified in interests of cost-effectiveness, but option of using more complete list remain	Could use reduced list, i.e., 3.1 and 3.2, plus 5.5 in case of Basin Deep Water	
		Indicator 3.1 could be replaced by: 5.1.b. Maximum summer chlorophyll calculated by CSTT model	(Unless the system is light- or flushing limited, maximum predicted chlorophyll will automatically increase with nutrient loading)

Ecological Quality Standards (EQSs) to provide thresholds to undesirable disturbance. Table 2 presents EQSs in the form of *Ecological Quality Objectives (EcoQOs)* which require indicator values to be within a defined range unless the ecosystem is to be considered disturbed. As will be considered later, the actual diagnosis of *undesirable disturbance* relies on the accumulation of evidence; a transgression of a single EcoQO will rarely be conclusive.

3.1. Bulk indicators

Chlorophyll concentration, a common measure of phytoplankton biomass and photosynthetic potential, is much used as an indicator of trophic status in freshwaters (OECD, 1982) and the sea (Painting et al., 2005). However, assessment of change should take seasonal variation into account, perhaps using the method of comparison with a reference envelope shown in Fig. 2. Increased chlorophyll concentration decreases transparency and thus impacts on the phytobenthos in shallow waters. Transparency can be roughly estimated from *Secchi depth*, and it has been claimed that decreasing *Secchi depth* tracks eutrophication in the Baltic (Kratzer et al., 2003; Sandén and Håkansson, 1996)). Opportunistic green and brown seaweeds, with an annual lifecycle, can be easily distinguished from perennial seaweeds and seagrasses, and their cover impacts directly on the natural fucoid, laminarian or seagrass flora. *Cover or biomass of opportunistic seaweeds* have thus been proposed by the UK Marine Plants Task Team (MPTT) as indicators of eutrophication in shallow waters. A century-long time-series showing decreasing *deep-water oxygen* has been used as evidence of eutrophication in the Baltic Sea (Fonselius and Valderrama, 2003; Jansson and Dahlberg, 1999), and regular measurements of dissolved oxygen should be made beneath the pycnocline of persistently stratified waters that might be at risk from nutrient enrichment.

3.2. Frequency statistics

Harmful Algal Blooms (HABs) are natural phenomena that can be rendered more frequent by nutrient enrichment, as exemplified in the Inland Sea of Japan (Nakanishi et al., 1992; Prakash, 1987). However, there is much confusion about what they are. Although the acronym HAB has become widely used, some HABs are not harmful, others are not algal, and some are not sea-discolouring 'blooms' (Anderson and Garrison, 1997). It is thus useful to distin-

Table 2
Indicators and EcoQOs for undesirable disturbance

Indicators	Possible EcoQOs	Apply in EHD types	Source, status, comments
Water column chlorophyll concentration (mg/m^3)	Chlorophyll concentration in summer should not exceed 10 mg m^{-3} Note that 'chlorophyll' is what is measured by standard survey methods; it should be free of 'pheopigments', but referring to it as 'chlorophyll *a*' implies more precision than is typically achieved without the use of HPLC	All	Based on the original UK standard for undesirable disturbance in the context of eutrophication, that of CSTT (1994). There is a need for EHDts EcoQSs. Painting et al. (2005), following OSPAR, proposed that *maximum and mean chlorophyll a concentrations during the growing season should remain below elevated levels, defined as concentrations >50% above the spatial (offshore) and/or historical background concentration*, with 10 mg/m^3 as the offshore EQS and 15 mg/m^3 as the nearshore EQS for maximum chlorophyll. However, this implies that inshore waters are less sensitive to enrichment, which may not always be the case
Oxygen concentration in deep water	(i) Oxygen concentration should not remain below 4 mg/L nor fall below 2 mg/L	3, some 5	(i) Pearson and Rosenberg (1978) indicate that oxygen concentrations between 4 and 2 mg/L can alter the species composition and abundance of benthic organisms. Gray et al. (2002) considered metabolism affected below 4 mg/L. In some basin deep waters the oxygen concentration can naturally fall below these levels
	(ii) Oxygen concentration, decreased as an indirect effect of nutrient enrichment, should remain above region-specific oxygen deficiency levels, ranging from 4 to 6 mg oxygen per litre		(ii) EcoQO quoted from Painting et al. (2005)
Pelagic GPP and NPP	None proposed	2, 3, 4, some 5	Nixon (1995) suggested that annual (gross?) production greater than 300 g C/m^2 indicated eutrophic conditions, and greater than 500 g C/m^2 indicated hypertrophic conditions; he did not consider typology. See text for detailed discussion
Phaeocystis blooms	Region/area-specific phytoplankton eutrophication indicator species should remain below respective nuisance and/or toxic elevated levels (and increased duration)	4, some 5	EcoQO quoted from Painting et al. (2005). Quantitative EQS/EcoQO desirable
Extent (area, m^2) of seagrass bed	Decrease in cover should be less than 10% in 3 years	1(b)	EQS is the boundary between WFD *moderate* and *poor* proposed by the UK MPTT (Marine Plants Task Team); other class boundaries also proposed
Epiphyte cover, as % of seagrass leaf area	Epiphyte cover should be less than 55%	1(b)	EQS is the boundary between WFD *moderate* and *poor* proposed by the UK MPTT; other class boundaries also proposed
Maximal seasonal cover of opportunistic seaweeds, as percent of available intertidal	Maximum cover should be less than 15%	1(c)	EQS is the boundary between WFD *moderate* and *poor* proposed by the UK MPTT; other class boundaries also proposed; soft intertidal sediments only
Maximum biomass of opportunistic seaweeds	Maximum biomass (as wet weight) should be less than 1 kg/m^2	1(c)	EQS is the boundary between WFD *moderate* and *poor* proposed by the UK MPTT; other class boundaries also proposed; soft intertidal sediments only
Frequency and extent of anoxic sediment or death of macrobenthos	None proposed	1(c), 3, 4, some 5	EcoQO needed. That proposed by Painting et al. (2005), following OSPAR, that *there should be no kills in benthic animal species as a result of oxygen deficiency and/or toxic phytoplankton species*, seems too stringent; such kills can occur under natural conditions
Thickness of sediment oxic layer/depth of RPD	Depth of RPD should exceed 2 cm	3, 4, some 5	Tentative proposal made during the UD study. Needs further study

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Table 2 (*continued*)

Indicators	Possible EcoQOs	Apply in EHD types	Source, status, comments
Macrobenthic community structure as measured by index such as ITI or AMBI	(i) The value of AMBI should not exceed 4.3 (ii) The value of the ITI should not fall below 30	3, some 5	(i) AMBI described by Borja et al. (2000), EQS taken from proposal by Borja et al. (2003) for boundary between WFD *moderate* and *poor* classes (ii) ITI described by Word (1990). The EQS is that used by the Scottish Environment Protection Agency to mark the edge of the (small) 'Allowable Zone of Effect' beneath fish farms

Only included here are indicators for which EcoQOs have been proposed.

Note about terminology. Usage of terms such as Environmental or Ecological Quality Standard (EQS) and Ecological Quality Objective (EcoQO) is complex and changing. We use 'EQS' to mean the value of an indicator at a threshold, and 'EcoQO' to refer to the desirability of not transgressing this EQS, which may be an upper or lower threshold. This usage corresponds to modern European norms (see Painting et al., 2005 for discussion), but differs from e.g., that of Elliott (1996) for whom EcoQOs were both more general and a form of testable scientific hypothesis. Because of the lack of full scientific evidence for type-specific EQSs, our suggested EcoQOs are indeed, also hypotheses: if the objective is breached, then: undesirable disturbance will result. Finally, EHD refers to EcoHydroDynamic and EHDts to EHD-type-specific.

Note about mapping to WFD. Some of the EcoQOs have been taken from proposed values of WFD biological quality element indicators at the *moderate/poor* quality boundary. As argued in the main text, transgression of this boundary would be an undesirable disturbance. The sources given for these tools also propose values at the *good/moderate* quality boundary. Transgressing this boundary might indicate a trend towards undesirable disturbance, especially if the trend correlates with a trend in ecological pressure. The indicators proposed here do not comprise a full set for WFD purposes, because they are intended for efficient diagnosis of *undesirable disturbance in the context of eutrophication*.

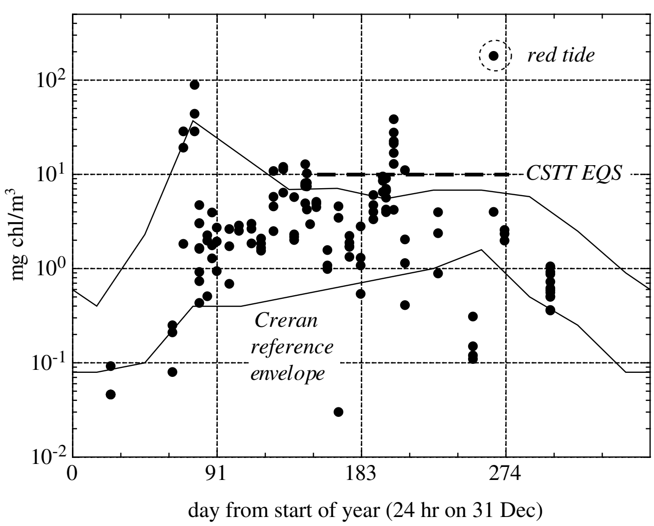

Fig. 2. **Use of a reference envelope to assess disturbance**, illustrated by chlorophyll concentrations in Loch Striven, 0–10 m, during 1980 (Tett et al., 1986), compared with the CSTT (1994) summer threshold of 10 mg chl m^{-3} and a smoothed envelope of seasonal variation in Loch Creran, 1972–1976 (Tett and Wallis, 1978) as an example of a reference condition. The comparison is intended only to be indicative; although both these small fjords on the west coast of Scotland belong to category 5 ('RREs') in Table 1, they differ in their detailed hydrodynamics. The 'Red Tide' in loch Striven was described by Jones et al. (1982).

guish two categories of HABs. The first is of large-biomass events that visibly colour the sea (hence the alternative and sometimes appropriate term, 'Red Tide'). Some of these blooms have killed benthic organisms through smothering (Helm et al., 1974) or weak toxicity (Jones et al., 1982; Roberts et al., 1983). Other blooms give rise to the nuisance of algal-generated foam on beaches (Lancelot et al., 1987). In some cases (Crawford et al., 1997), however, no harm is evident. Monitoring of the occurrence of this category of HABs seems desirable; although local nuisances due to blooms (e.g., foam on beaches, mortalities of fish or benthos) do not diagnose undesirable disturbance, a trend of increasing HAB frequency would be a cause for concern. In some UK waters, such as those in the north-western North Sea (Miller, 2001), satellite remote sensing can be used to monitor the occurrence and geographical extent of Red Tides, even if lack of sea-truth often prevents explanation of nature or cause.

The second category of HABs is that of occurrences of highly toxic micro-algae in comparatively low abundances (a few hundred or thousand cells per litre). These can pose a threat to the health of humans, sea-birds or marine mammals when their toxins are concentrated by shellfish (Coulson et al., 1968; Todd et al., 1993). For the present, incidents involving such Shellfish-Vectored Toxins (SVTs) should not be counted, because the link between such incidents and nutrient enrichment is controversial (Tett and Edwards, 2003). However, continued studies of the relationship between shellfish toxicity, the abundance of SVT-producing algae, and nutrient availability, are desirable.

3.3. Flux measurements

Quantification of vigour could involve measurements of larval settlement, benthic oxygen demand, or nutrient mineralization fluxes, but the best single indicator is undoubtedly *annual primary production*. It should be reported in grams of carbon per square metre to allow comparison between phytobenthos and phytoplankton, or amongst water bodies of different depth. Precise measurements of macrophytobenthic production are not proposed here, because the standing crop of seaweed or seagrasses at the end of the growth season serves as a rough measure of annual production, and the suggested bulk indicators involving cover of opportunistic algae seem sufficient. In contrast the biomass of planktonic algae typically turns over every few days, and repeated measurements are needed to estimate their annual production.

The state of the art in the measurement of pelagic production is discussed in a recent book (Williams et al., 2002). Techniques include free-water budgets of nutrient removal or oxygen production, and the use of remote sensing, numerical models and sophisticated opto-electronics. Despite the development of new instruments, core methods remain those involving the incubation of water samples containing phytoplankton, either in the sea, on the deck of a ship under natural light conditions, or in the laboratory under controlled illumination. The *radiocarbon* method involves measuring the incorporation of ^{14}C-labelled bicarbonate ($H^{14}CO_3^-$) into particulate organic matter. Short incubations (1–3) hours are thought to measure *gross primary production* (GPP)—i.e., the total organic matter made during photosynthesis, before any is lost to respiration. In the *light and dark bottle oxygen* method, GPP can be estimated from the difference between the changes in oxygen concentration in transparent and opaque bottles. *Net primary production* is GPP less respiration and can be estimated from the change in oxygen over time in a transparent bottle. Because water samples also include bacteria and protozoa that consume products of photosynthesis and use oxygen, what is measured in such incubations is best called *net microplankton* production (NMP, shortened from the *net microplankton community production* of Williams and Raine (1979). Longer term ^{14}C incubations (either from dawn to dusk or 24 h), give results that are less than GPP (because some of the ^{14}C label is returned to the water by way of algal and microheterotroph respiration) but more than NMP.

Results from short incubations can be graphed against irradiance to obtain a *p–I* (photosynthesis–irradiance) curve and values of the photosynthetic parameters that define the curve (Jassby and Platt, 1976; Lederman and Tett, 1981). The parameter values can be used with solar radiation, water transparency and chlorophyll data to estimate hourly and daily water column production in mg carbon m^{-2} (Herman and Platt, 1986). These estimates then can be *scaled up* to give an estimate of annual production.

There is a final complication. Much euphotic zone production is fuelled by *recycled* nitrogen excreted by zooplankton feeding on phytoplankton (Dugdale, 1967). Only *new production*, supported by nitrogen (mostly nitrate) introduced from outside the euphotic zone, can be exported (Eppley and Peterson, 1979). In the context of undesirable disturbance due to nutrient enrichment, only this fraction of production has the potential to cause disturbance. Methods exist for the estimation of new production from the uptake of isotopically labeled nitrate, but their reliability in shelf seas, where some nitrate may be recycled and some ammonia may be new, is unclear. The alternative method of estimating the consumption of winter nitrate also has difficulties because the link between N assimilation and organic carbon production is variable. It may be that only models soundly based in algal theory can adequately estimate new production in coastal waters. For the present, *vigour* is in our view best indicated by measurement-derived estimates of annual GPP and NMP. GPP comes closer to the idea of vigour as a potential for growth and activity, whereas NMP indicates potential for undesirable disturbance.

Bearing in mind all these issues, we suggest a two-step method drawn from several sources (Gowen and Bloomfield, 1996; Herman and Platt, 1986; Joint and Pomroy, 1993; Tett et al., 1988). In the first step, water samples are incubated (a) with $H^{14}CO_3^-$ under a light gradient, for 1–3 h, to measure carbon fixation and (b) in darkness for 24 h to estimate microplankton respiration by oxygen change. The results are used to estimate chlorophyll-related photosynthetic and respiratory parameters. In the second step, the parameter values are used with vertical profiles of chlorophyll and submarine light, taking account of diel changes in sea-surface irradiance, to estimate euphotic zone GPP and NMP. These daily column production values are regressed upon euphotic zone chlorophyll and the regression used with chlorophyll maps obtained during repeated surveys to estimate annual GPP and NMP. Such regressions explained up to 70% of the variance in production in the North Sea and Irish Sea (Gowen and Bloomfield, 1996; Joint and Pomroy, 1993). Improvements in accuracy could be made by using additional chlorophyll data (e.g., from remote sensing and moored or ship-mounted fluorometers) and by taking account of day-to-day variations in sea-surface and submarine light using accessory models. In the long run, the best estimates might be obtained by assimilating bio-physical models to observed chlorophyll and local productivities.

Finally, there is evidence (Pearson and Rosenberg, 1978) that organic enrichment results in a shallowing of the depth of the Redox Potential Discontinuity (RPD) in soft sediments, as benthic organisms consume oxygen faster in relation to its diffusion or its biological pumping into the sediment. This aspect of vigour (with its potential for overloading the capacity of a sediment to assimilate organic matter) can be estimated by Redox probes or by Sediment Profile Imaging (Nilsson and Rosenberg, 1997).

3.4. Indicators of ecosystem structural health

The impact of organic matter on the macrobenthos of temperate shelf seas is well understood (Pearson and Rosenberg, 1978; Rosenberg, 2001), and *tools* to assess the resulting change in community structure include the Infaunal Trophic Index (ITI) (Word, 1990), and the AZTI Marine Biotic Index (AMBI) (Borja et al., 2000, 2003a). Although AMBI has been assessed against several sources of disturbance (Borja et al., 2003b), it and ITI may prove insensitive to the low-level, wide area, organic enrichment that may be expected to occur during eutrophication. In any case, it is the community structure of the primary producers that holds the key to diagnosing eutrophication, because algae and cyanobacteria provide the initial response to nutrient enrichment. It is thus unfortunate that indicator tools for change in seaweed communities or the phytoplankton are less developed than those for macrobenthic change.

In the case of seaweeds, the main indicators proposed by the UK MPTT are those of bulk cover by opportunistic seaweeds. However, species richness and the balance amongst *functional form groups* or *ecological status groups* (Orfanidis et al., 2001) have also been considered. In the case of freshwater phytoplankton, shifts from desmids, chrysophytes or diatoms to cyanobacteria are known to be associated with nutrient enrichment (Hutchinson, 1969; Talling and Heaney, 1988). In contrast, and excepting the Baltic Sea, where blue–green bacteria have increased with nutrient enrichment (Finni et al., 2001), the marine situation is less clear. Increases in the ratio of N to Si may cause increases in the proportion of non-silicified algae (Gillbricht, 1988; Tett et al., 2003b), and this has led to proposals for indicators based on the ratio of diatoms to dinoflagellates. Care must be taken in the use of simple, growth-season-averaged, ratios of this sort, since they can underestimate the effect of nutrient pressure on well-stirred waters where diatoms, including resuspended benthic diatoms, are natural dominants. Setting EQSs from such ratios tends to reflect the view that 'diatoms' are 'good' and 'flagellates' or 'dinoflagellates' are 'bad', which misunderstands the multiple roles that each group plays in marine ecosystems. For example, dinoflagellate lipids can make important contributions to the diet of crustacean zooplankton. More generally, it is apparent that the phytoplankton encompasses a wide range of biochemical, taxonomic and functional diversity (Delwiche et al., 2004; Jeffrey and Vesk, 1997; Tett et al., 2003b), and it seems unwise to ignore this diversity in assessing the health of the plankton. It is also desirable to take into account the natural, especially seasonal, variability that is an essential part of phytoplankton ecology.

Tools know generically as (marine) Phytoplankton Community Indices (PCIs) are being developed to satisfy these requirements. All start with the idea of defining ecosystem state in terms of values of state variables, which can be plotted into a multidimensional 'state variable space'. Fig. 3(a) illustrates this in two dimensions, where the axes are the state variables, y_1 and y_2. The shaded 'doughnut' region includes all those states of the ecosystem that are normal for the type-specific conditions, taking account of seasonal and interannual, variation and spatial patchiness. The system is healthy while its state remains within, or is capable of returning rapidly to, the 'doughnut'. Sustained movement away from the 'doughnut', constitutes an undesirable disturbance.

The main difficulty is that of identifying state variables. Graphs in a 'phytoplankton species abundance' space containing dozens of dimensions would be too complex to be useful. One route to simplification involves empirical multivariate analysis to extract a few key dimensions (representing groups of regularly co-occurring species) that include most of the variation in community composition. A second route involves the use of 'lifeforms' based on function and taxonomy (Tett et al., 2003b). This route

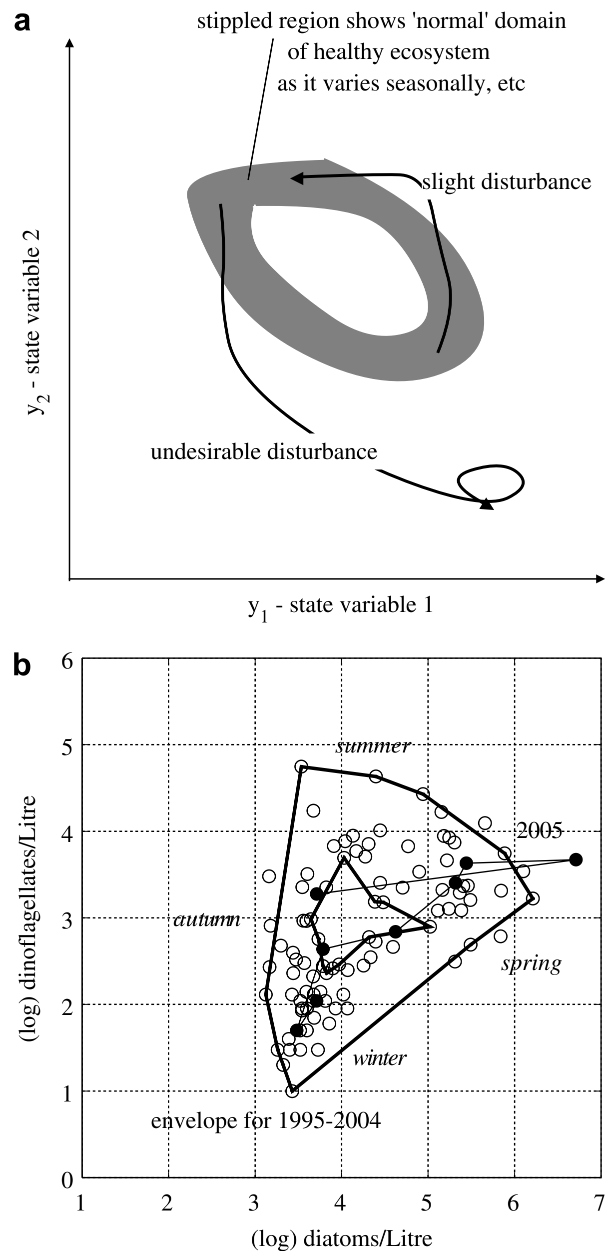

Fig. 3. **Ecosystem state**. Part (a) is a generalized diagram showing a state-space defined by two variables; a 'normal' or 'reference' domain is shown by the shared 'doughnut' region, and a disturbance is a movement outside this region. Part (b) provides a concrete instance of a state-space diagram by plotting Port Erin Marine Laboratory data from the Cypris station near the coast of the Isle of Man in the Irish Sea. The data are inverted microscope counts of cells in each category, averaged over a month. Results from 10 years of sampling (1995–2004) have been plotted, and an envelope drawn to include 95% of the points. Results from 2005 are also plotted, to show how new data can be assessed against a previous condition. The proposed PCI will be based on the proportion of new points outside the envelope, compared with the 5% expectation.

is exemplified in Fig. 3(b) using data for diatoms and dinoflagellates from the Cypris station near the Isle of Man, which has been regularly sampled by the Port Erin Marine Laboratory (Shammon et al., 2005). The PCI here is not the diatom:dinoflagellate balance, but the deviation of new observations from the reference envelope, which takes account of seasonal and interannual changes. Use of such indices will require regular monitoring of phytoplankton in water samples or by the Continuous Plankton Recorder survey (Brander et al., 2003; Warner and Hays, 1994).

3.5. Indicator species

Although there are no species that could serve as universal indicators of nutrient-induced disturbance, there are some species that may serve as indicators of disturbance in particular water types. These include seagrasses, *Nephrops norvegicus* in deep muddy sediments, and the colonial prymnesiophyte *Phaeocystis* spp. in regions of intermittent, freshwater-driven, stratification. In each case, the indicator is the abundance or health of the species' population and not its presence/absence.

In the case of seagrass meadows, ecosystem health largely depends on the real health of the seagrass primary producers. Decreased water transparency, or increased growth of opportunistic seaweeds or epiphytic microalgae, can degrade seagrass cover, biomass or health, and in some cases lead to the replacement of the community by one in which the dominant primary producers are macro-algae, benthic micro-algae, or phytoplankton (Den Hartog, 1994; Hauxwell et al., 2000; McGlathery, 2001). Any such change should be seen as an undesirable disturbance, because seagrass meadows are now uncommon in UK waters, following a decline during the XXth century (Cleator, 1993).

N. norvegicus is sensitive to hypoxia (Diaz and Rosenberg, 1995) and so observations during DARD benthic surveys of good populations of this decapod in the seabed of the western Irish Sea points to the maintenance of high oxygen levels in the deep water. Its presence both creates (through flushed burrows), and indicates, good benthic health, and its burrow density can be estimated by towed underwater TV (Marrs et al., 1998). Spring blooms of *Phaecystis* spp. are a feature of the southern North Sea (Lancelot et al., 1987) and Liverpool Bay in the Irish Sea (Gowen et al., 2000; Jones and Haq, 1963), and their magnitude might indicate an excess of available N (or P) in relation to dissolved silica.

4. (Eco)Hydrodynamics

Using the concept of ecosystem health to assess disturbance requires the spatial extent of a marine ecosystem to be defined. It should be large enough for structure and vigour to be controlled more by internal processes than by outside forcing, and should also comprise a hydrographic and hydrodynamic unit. Combining this requirement with the concept of *biomes*, which are defined by the lifeform of their typical primary producers, leads to the view that the methodology described above should be applied to EcoHydroDynamic (EHD) units characterized by: (i) their physical conditions; (ii) typical primary producers (in the absence of anthropogenic interference); and (iii) significant ecosystem features emerging from such primary producer dominance and from biogeography. For example, the seasonally stratified and frontal waters, and underlying seabed, of the western Irish Sea are sufficiently extensive to be a largely self-contained unit (Gowen et al., 1995). The unit is a phytoplankton-dominated ecosystem, too deep for phytobenthic growth and with a zooplankton dominated by small copepods (Gowen et al., 1998). There is insufficient ecological knowledge to set up such a typology for all UK waters, but the following types can be identified (Fig. 4) on the basis of the existence and duration of stratification and its bio-optical consequences:

1. Shallow clear waters, in which the euphotic zone includes the seabed, and where ***phytobenthos*** are expected to be important primary producers.
2. Optically deep ***mixed waters*** where phytoplankters are unlikely to be stimulated by nutrient enrichment because light is either absolutely or relatively limiting.
3. Offshore ***stratified waters***, which in a natural state have a nutrient-depleted surface layer during summer; extra nutrients can stimulate phytoplankton production here with a risk of oxygen depletion as organic matter sinks below the pycnocline; the category includes waters with: (a) seasonal, mainly thermal, stratification; (b) thermohaline or haline stratification that persists for most of the year.
4. ***Regions of Freshwater Influence (ROFIs***), which are highly variable nearshore waters characterized by tidal and wind-wave stirring and high turbidity, and with a significant freshwater content and intermittent stratification from river or estuarine discharges.
5. ***Regions of Restricted Exchange (RREs)***, which are inshore, semi-enclosed waters whose dynamics and eutrophication risk depends on the rate of water exchange with the sea; the category includes *fjords* (some of which have *basin deep water*), rias, other types of estuary, and coastal embayments and straits.

The typology was designed, in particular, to key into Table 1, and to enable the identification of type-specific *reference conditions*. In principle, such conditions exist in the absence of significant anthropogenic pressures, although there is debate about where to find them. As an example, neglecting the historic impacts of human activities on populations of large marine mammals and fish, reference conditions for the stratified western Irish Sea might be deduced from observations here during the 1950s and 1960s, before subsequent increases in Winter nutrient concentrations (Allen et al., 1998; Gowen et al., 2002). Fig. 2 presents an example for Scottish fjordic RREs, showing the use of

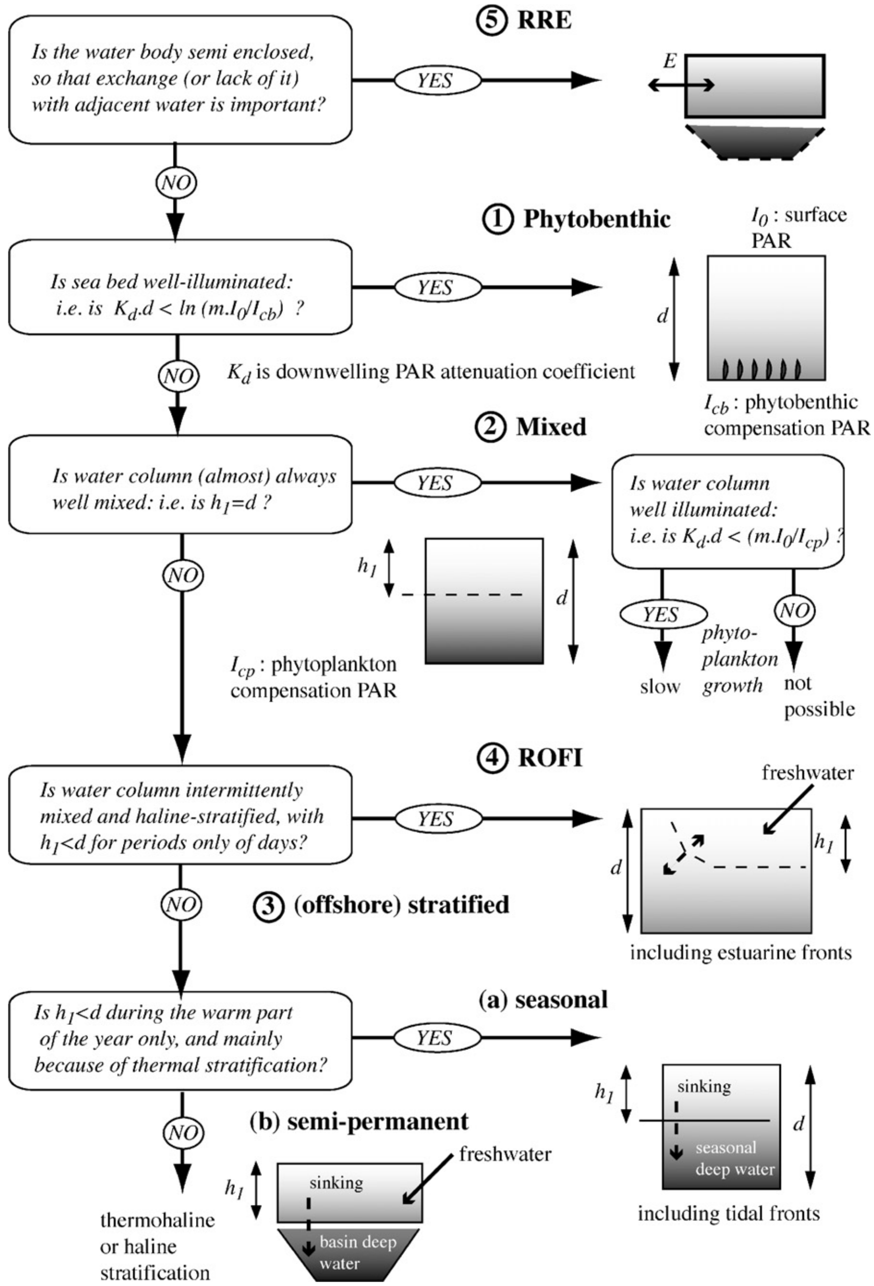

Fig. 4. **Ecohydrodynamic typology.** The diagram defines the types considered in this paper and provides a key for their identification from physical and bio-optical properties. The latter involve PAR, which is photosynthetically active (ir)radiance. The comparisons of 'optical depth' (the product of actual depth and the PAR attenuation coefficient K_d) and the ratio of 24 h mean surface PAR (I_0) to the compensation irradiance (I_c) of the relevant primary producers, use approximations (Tett, 1990) involving the correction factor m.

conditions during the 1970s in loch Creran (Tett and Wallis, 1978) to provide a reference envelope for chlorophyll in the nutrient-enriched loch Striven (Tett et al., 1986).

5. Undesirable disturbance and eutrophication

The indicators in Table 1 can be used to monitor against deterioration in ecosystem state and function, and, together with the proposed EcoQOs in Table 2, provide a methodology for recognizing undesirable disturbance. They do not, however, allow such disturbance to placed uniquely in the context of eutrophication. There are two remaining difficulties. First, some disturbances of marine ecosystems are natural, but the implication of the OSPAR and EU definitions of eutrophication is that disturbance is undesirable (and preventive or remedial action is required) only when it is human-generated. Second, undesirable disturbance might be caused by a mixture of pressures, of which nutrient enrichment is but one.

Thus, Jennings et al. (2001) could not determine whether increases in benthic biomass and production in the North Sea were caused by trawling disturbance, climate change or eutrophication. An increase in the ‘green colour’ detected by the Continuous Plankton Recorder survey, which was at first interpreted as evidence of eutrophication in the North Sea, was subsequently shown to be so widespread that climate change is a more likely explanation (Edwards et al., 2001). The diatom: flagellate balance in shelf seas is controlled by physical processes and selective grazing as much as by nutrient element ratios (Tett et al., 2003b). Although increases in the abundance of *Phaeocystis* spp. in the Wadden Sea have been associated with eutrophication (Cadeé and Hegeman, 1986), North Sea populations have shown long-term fluctuations which appear unrelated to nutrient enrichment (Gieskes and Kraay, 1977). In the case of the phytobenthos, the replacement of dominant wracks or seagrasses by opportunistic green or brown algae can occur for a variety of reasons in addition to eutrophication (Fletcher, 1996; Morand and Briand, 1996).

So far as vigour is concerned, it has been suggested that a lake is *polluted eutrophic* if its annual production exceeds 350 g C m^{-2} yr^{-1} (Rodhe, 1969) and that a marine water is *hypertrophic* above 500 g C m^{-2} yr^{-1} (Nixon, 1995). In relation to our own definitions of production, it can be argued that pelagic GPP provides the desired indicator of eutrophication because it responds directly to nutrient enrichment. It takes account of ecohydrodynamic conditions which diminish light for photosynthesis or keep biomass low by high flushing. However, it is likely to be a poor predictor of disturbance because of the sensitivity of organic impact to ecohydrodynamic type and the efficiency of coupling. NPP may be a better predictor if the argument that pelagic protozoans are the most important control on biomass formation (Tett et al., 2003a), is correct. But other pressures, such as toxic pollution, may harm the protozoan community and so increase NPP, and in any case we think it desirable that EcoQOs for NPP are set on the basis of EHD type. This is also the case for production’s proxy, chlorophyll.

Thus we conclude that there appear to be no unambiguous and universal indicators of disturbance due to marine eutrophication, either amongst species, lifeforms, fluxes or bulk or frequency indicators, and hence no single, precise EcoQO can be proposed. Instead, a multi-step method is needed to diagnose undesirable disturbance due to eutrophication. The steps are:

1. Identify ecohydrodynamic type, and thus, in principle, reference conditions.
2. Assess nutrient loading and identify water bodies/ecosystems where there is potential for undesirable disturbance in the context of eutrophication, because such water-bodies are sufficiently well-illuminated to allow nutrient to be converted into primary producer biomass, flushing is sufficiently low to allow blooms to develop, and, in some cases, stratification allows organic accumulation and oxygen consumption in deep water.
3. Use the simpler, bulk and frequency, indicators, in comparison with reference conditions, to detect a trend towards disturbance or, with reference to an EHD-type-specific EcoQO, to make a provisional diagnosis of undesirable disturbance.

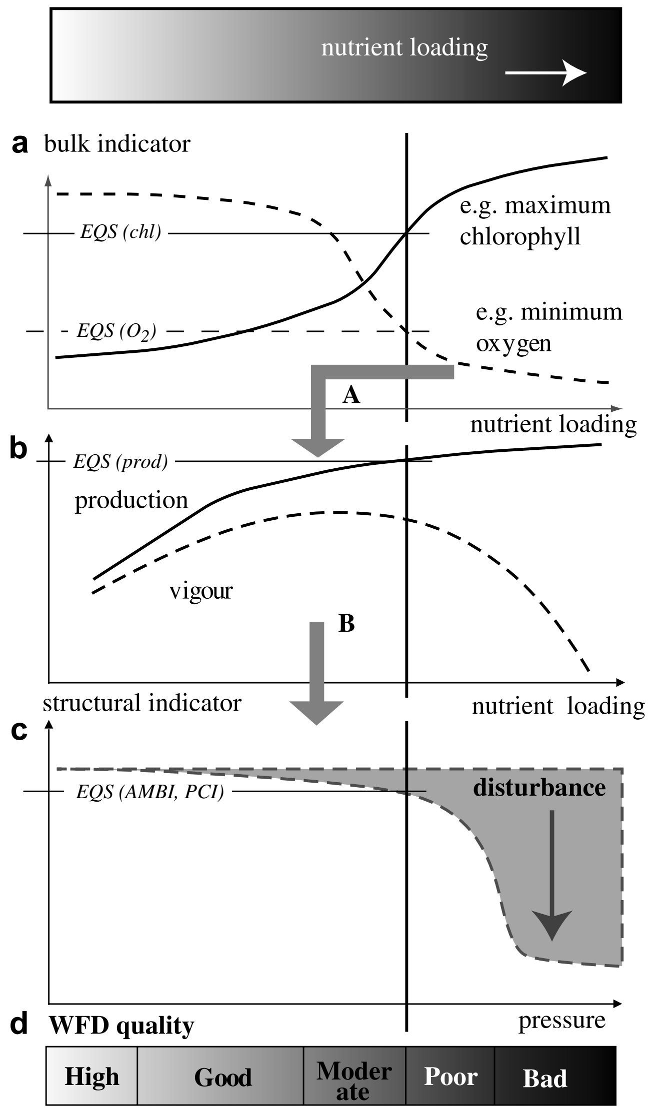

Fig. 5. **Identifying undesirable disturbance in the context of eutrophication.** The diagram assumes that the water body and ecosystem under investigation is able to suffer eutrophication, and that nutrient enrichment is the only significant anthropogenic pressure: the horizontal axis in (c) is labelled ‘pressure’ to remind the reader that there may be other causes of disturbance. Part (d) is a suggested mapping between the ecosystem health approach to undesirable disturbance and WFD Annex V quality categories. In order to use the methodology, specific monitoring variables and EcoQOs need to be identified for each ecohydrodynamic water type. Bulk indicators (exemplified by chlorophyll and dissolved oxygen in part (a)) and frequency indicators should be routinely monitored; production measurements (part b) and indicators of structure (part c) are more expensive, and the arrows A and B show the application of step 5 (measuring production and structure) of the proposed monitoring strategy when concern has been triggered by trends in the indicators of part (a). The vertical arrangement of the four parts should *not* be read as suggesting that breach of EcoQOs in part (a) necessarily implies breaches in parts (b) and (c) or that water body ecological quality in (d) has necessarily fallen below WFD *good*.

4. Use correlation between (adverse) trends in these indicators and in nutrients, aided by purpose-made models such as that of the UK CSTT (CSTT, 1994, 1997; Tett et al., 2003a; Painting et al., 2006) to relate the trend or diagnosis to nutrient enrichment.
5. When there is a such a provisional diagnosis, and the costs of reduction in nutrient loading justify further work, monitor ecosystem health by measurement of primary production (indicating vigour) and of community and species indicators of planktonic and benthic structure.

Step 2 is intended to increase the reliability of the OSPAR screening procedure for identifying *potential problem areas.* The alternative to step 5 is of course to apply the precautionary principle following the provisional diagnosis of steps 3–4, and take steps to reduce nutrient loading without further study. We argue against this as a strategy because the enriched ecosystem may in fact be in or close to an optimum condition, supporting maximum biomass, diversity and fisheries yield.

Fig. 5 summarizes steps 3–5 pictorially, and is meant to imply that a firm diagnosis of undesirable disturbance due to eutrophication follows from the following combination of elements:

- a high GPP that can be shown (by correlation or numerical modelling) to result, in whole or substantial part, from anthropogenic nutrient enrichment;
- NMP a large fraction of GPP, which suggests poor protozoan control of micro-algal growth and hence the potential for exceptional blooms and excess of sinking, potentially oxygen-consuming, organic matter if coupling to mesozooplankton or macrobenthos fails;
- marked deviations from reference conditions in bulk and frequency variables, in particular those which are considered to be particularly important for a given EHD type, such as deep-water oxygen levels in stratified waters;
- marked increases in abundance of EHD-type-specific indicator organisms, such as *Phaeocystis* spp., or opportunistic seaweeds, which are deemed to respond to nutrient loading;
- a significant decrease in the structural health of the pelagic and benthic communities, as shown by changes in the values of appropriate community indices and decreases in the abundance or health of EHD-type specific indicator organisms such as *Nephrops* or seagrasses.

6. Discussion and conclusions

To recapitulate, the theory and methodology set out here for the identification of *undesirable disturbance in the context of eutrophication*, involve:

- the equation of undesirable disturbance with an anthropogenically caused deterioration in *ecosystem health*, recognized in particular by changes in community *structure*;
- the measurement of the primary production component of ecosystem *vigour* (or its proxies) in order to relate disturbance (which may result from several pressures) specifically to nutrient enrichment;
- an ecohydrodynamic typology which: distinguishes the different sensitivities and responses of ecosystems to nutrient enrichment and so allows appropriate indicators to be selected; and guides the identification of type-specific reference conditions for the part of the methodology which concerns change from these conditions.

At the heart of the undesirable disturbance theory is the interaction between the *vigour* and the *structure* of ecosystems. We have supposed that optimum vigour in ecosystems may occur when organic production is greater than that of an oligotrophic reference condition. Figs. 1(b) and 5(c) shows a structural indicator changing slightly, as the optimum is approached, from its value at the zero-pressure or reference condition. Beyond the optimum, excess of vigour leads to *polutrophy*, and structure deteriorates: the ecosystem 'goes over the cliff', either into a state from which recovery may be slow or into a new stable state. It is therefore essential that a monitoring programme be able to detect a trend towards the 'cliff', and it is for this reason that measurements of primary production are, in many cases, essential.

There is a difference in our conceptual framework and that of the WFD in its Annex V. The latter sees all change from a reference condition as a degradation of ecological quality, whereas the concept of ecosystem health implies that some change may be good if it is towards what we have called a (nutrient-driven) optimum. Nevertheless, the two approaches can, we think, be reconciled (Fig. 5(d)). WFD *good* ecological status, which equates with small changes from the reference condition, can be equated with the small changes from reference condition structure which equate in our scheme with the approach to an optimum vigour, during which the ecosystem remains well within its *resistance* to pressure-induced disturbance. We equate the region in Fig. 5(c) that is close to the edge of the structural 'cliff' with WFD *moderate* status: that is, with a system that appears only little changed but is approaching the limits of its resistance to pressure (Fig. 1(b)), and so could easily be sent over the 'cliff' into a degraded state which equates with WFD *poor* or *bad* quality. Some of the EcoQOs in Table 2 explicitly equate an undesirably disturbed state with WFD *poor or bad.*

The basis of the ecohydrodynamic typology that we have presented here is compatible both in principle and practice with that suggested by OSPAR's Strategy, which takes account of *hydrodynamic/physical features* and other aspects such as zooplankton grazing, as *supporting environmental factors*, over a domain that extends from *the point of freshwater penetration at low tide* to the outer edge of the continental shelf. The typology is also compatible in principle with that set out in WFD Annex II system B, although of course the WFD domain currently extends out only to 1 nautical mile (3 nautical miles in some implementations) from the coastal baseline.

We conclude with a more profound question: how robust is the theory we have set out here for undesirable disturbance in the context of eutrophication? The theories of health, ecosystem state, and the effects of disturbance, on which we have drawn, are well-established within the discipline of community ecology, even if remaining a matter of debate. Despite earlier conclusions (Hecky and Kilham, 1988) that eutrophication differs between fresh and marine waters, it has been recently argued that 'regime shift' theory applies as well to marine as to terrestrial and freshwater systems (Scheffer and Carpenter, 2003; Scheffer and van Nes, 2004). Nevertheless, there remains much to be done to test the validity of the undesirable disturbance theory in the tidally energetic waters that characterize the seas around the UK and over much of the NW European continental shelf. Nutrient enrichment, and the risk of eutrophication, is not the only anthropogenic pressure on these ecosystems, which also suffer from toxic pollution, overfishing, mechanical disturbance of the seabed, and climate change (McIntyre, 1995). There is thus both a strong precautionary case for monitoring these seas against undesirable disturbance and a scientific case that such monitoring will test and develop the theory set out here.

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