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Application and sensitivity testing of a eutrophication assessment method on coastal systems in the United States and European Union

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Abstract

The Assessment of Estuarine Trophic Status (ASSETS) screening model has been extended to allow its application to both estuarine and coastal systems. The model, which combines elements of pressure, state and response, was tested on four systems: Maryland Coastal Bays and Long Island Sound in the United States and The Firth of Clyde (Scotland) and Tagus Estuary (Portugal) in the European Union. The overall scores were: Maryland Coastal Bays: Bad; Firth of Clyde: Poor; Tagus Estuary: Good. Long Island Sound was modelled along a timeline, using 1991 data (score: Bad) and 2002 data (score: Moderate). The improvement registered for Long Island Sound is a consequence of the reduction in nutrient loading, and the ASSETS score changed accordingly. The two main areas where developments are needed are (a) In the definition of type-specific ranges for eutrophication parameters, due to the recognition that natural or pristine conditions may vary widely, and the use of a uniform set of thresholds artificially penalizes some systems and potentially leads to misclassification; (b) In the definition and quantification of measures which will result in an improved state through a change in pressures, as well as in the definition of appropriate metrics through which response may be assessed. One possibility is the use of detailed research models where different response scenarios potentially produce changes in pressure and state. These outputs may be used to drive screening models and analyze the suitability of candidate metrics for evaluating management options.

Keywords: Eutrophication assessment; Screening model; Pressure; State; Response; ASSETS; Estuary; Coastal zone management

1. Introduction and objectives

1.1. Background of eutrophication and need for assessment methods and models

Eutrophication is a natural process in which the addition of nutrients to coastal waters from the watershed and ocean stimulates algal growth. In recent decades, nutrient loadings have increased to many times natural levels on account of human activities such as fossil fuel burning and agricultural use of fertilizers. These changes have caused a variety of impacts, such as high levels of chlorophyll *a* (Boynton et al., 1982), overgrowth of seaweed and epiphytes, occurrences of anoxia and hypoxia (Gerlach, 1990; CENR, 2000), nuisance and toxic algal blooms (ORCA, 1992; Rabalais et al., 1996), and losses of submerged aquatic vegetation (SAV, Twilley et al., 1985; Burkholder et al., 1992; Fig. 1). It is clear that nutrient-related eutrophication is a significant problem worldwide (Table 1; Chiaudani et al., 1980; Kelly and Naguib, 1984; Gillbricht, 1988; Hodgkin and Hamilton, 1993; Joint et al., 1997; Okaichi, 1997).

Potential consequences of eutrophication range from nuisances to serious human health threats. Hypoxia or anoxia may lead to fish kills (Glasgow and Burkholder, 2000), and loss of seagrass habitat (Burkholder et al., 1992; Twilley et al., 1985) and benthic organisms (Rabalais and Harper, 1992) may lead to long-term reductions in abundance, diversity, and harvest of fish in eutrophic systems (Breitburg, 2002). Algal toxins are a threat if ingested in fish and shellfish tissue or inhaled directly

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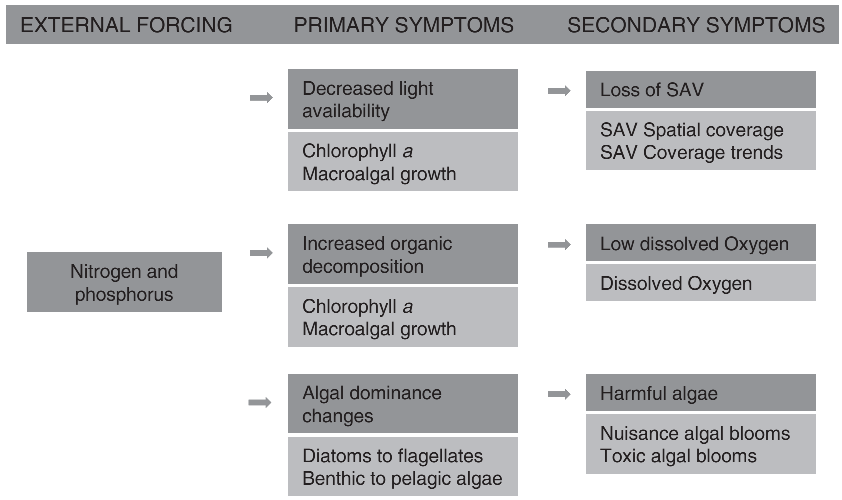


Fig. 1. Conceptual model of eutrophication.

Table 1
Example locations of observed eutrophic symptoms

Eutrophication symptoms	System or country of observation
Toxic algal blooms	Pamlico and Neuse River Estuaries (Burkholder et al., 1999) NE Coast of UK (Joint et al., 1997)
Nuisance algal blooms	Lower Laguna Madre (Whitledge and Pulich, 1991) Southern North Sea (Gillbricht, 1988) Baltic Sea (Bonsdorff et al., 1997) Mediterranean Sea (e.g. Lac de Tunis: Kelly and Naguib, 1984) Northern Adriatic (Chiaudani et al., 1980), Australia (Hodgkin and Hamilton, 1993) Japan (Okaichi, 1997)
Depleted dissolved oxygen	Mississippi River Plume (Rabalais et al., 1996) Chesapeake Bay (Cooper and Brush, 1991)
Loss of SAV	Chesapeake Bay (Orth and Moore, 1983) Laguna Madre (Onuf, 1995)

(Anderson et al., 2000). Measurable socio-economic costs include economic losses to seasonal tourism and the seafood industry (Anderson et al., 2000), while indirect and non-use values are more difficult to determine (Turner et al., 1999).

Predicted increases in problems are of particular concern as coastal populations and the use of fertilizers and fossil fuels continue to increase (Bricker et al., 1999; NRC, 2000). Research efforts have advanced the understanding of processes contributing to eutrophication, and have produced recommendations to reduce and prevent problems (e.g. NAS, 1969; NRC, 2000). Models have been developed to explore cause/effect relationships (e.g. NOAA and EPA, 1988; Dettmann, 2001), and symptoms-based assessment methods have been developed (Bricker et al., 1999, 2003; OSPAR, 2001). Research, modelling, and assessment efforts, taken together with United States federal mandates and European Union Directives, have the potential to significantly reduce future problems (Table 2). These efforts must include the analysis of economic dimensions of eutrophication, in combination with traditional approaches, to maximize the effectiveness of future management strategies (Folke et al., 1994; Segerson and Walker, 2002).

1.2. State of the art for simple models of eutrophication

Eutrophication models were historically developed for freshwater environments such as lakes and reservoirs (Dillon and Rigler, 1975; Vollenweider, 1975; Jørgensen, 1976). Simple time-varying or statistical approaches reflected the established relationships between pelagic algae and nutrient loading, using proxies such as chlorophyll *a*, total nitrogen or phosphorus. Increases in particulate matter in the water column were easily assessed using turbidity, and secondary symptoms of eutrophication such as low dissolved oxygen (DO) could be predicted from this parameter set. More recently, increased pressure on coastal

Table 2
Key legislation to address eutrophication in the United States and in the European Union

United States	European Union
Clean Water Act of 1972	EU Water Framework Directive
Air Pollution Prevention and Control Act of 1977	Urban Waste Water Treatment Directive
Coastal Zone Management Act of 1972	Nitrates Directive
Harmful Algal Bloom and Hypoxia Research and Control Act of 1998	OSPAR, HELCOM and Barcelona conventions

systems has highlighted the need for eutrophication modelling of estuarine and coastal systems, which has led to the adaptation of approaches used for freshwater, often with limited success. Partly, this is because in coastal environments there is often no clear relationship between nutrient forcing and eutrophication symptoms—systems with similar pressures show widely varying responses (e.g. Tett et al., 2003).

The reaction has often been to increase modelling complexity (e.g. Radach and Moll, 1989; Baretta et al., 1995; Lung et al., 1993; Chau and Jin, 1998) but a general model for coastal areas must incorporate a diversity of factors including tidal range effects, toxic algal species, benthic symptoms of eutrophication, and top-down control of phytoplankton by filter-feeders. This has led to the successful development of partial models, focusing for example on phytoplankton dynamics (e.g. Le Gall et al., 2000) or on opportunistic seaweeds (e.g. Solidoro et al., 1997; Alvera-Azcárate et al., 2003). In parallel, simple modelling schemes have achieved a measure of success, by combining a range of factors for assessing eutrophication effects. Examples of these are the CSTT (1997), National Estuarine Eutrophication Assessment (NEEA) (Bricker et al., 1999), Fjordenv (Stigebrandt, 2001) and OSPAR (2001) screening models. Whilst these models are not designed for detailed management, they do provide an effective assessment of system state, and in some cases also include measures of pressure. The challenge is currently to test, compare, cross-validate and improve simple models of this nature, and to make use of complex modelling to fill data gaps and to explore specific scenarios. Results from research models may then be distilled into these screening models, using a simplified set of parameters. Some of these may not be measurable in the field, but derived as an index or ratio from the results of complex models.

1.3. Objectives

The purpose of this paper is to outline and test the Assessment of Estuarine Trophic Status (ASSETS) eutrophication model, which is a development of the United States NEEA methodology. The NEEA has been refined by means of a more quantitative approach, and extended by combining the three diagnostic tools, briefly described below, which build up the core methodology into an overall assessment index. The ASSETS eutrophication model is tested on four systems in the United States and European Union. The aim was to apply the model across a range of different coastal types (Maryland Coastal Bays, Firth of Clyde and Tagus Estuary), and also to evaluate its application to a long-term data series for Long Island Sound, in order to test its sensitivity to policy-driven changes.

2. Methodological developments of the ASSETS model

The general characteristics and implementation of the ASSETS model have been described in Bricker et al. (2003) and Nobre et al. (2005), and will only synthetically be referred to herein. The model uses a Pressure-State-Response framework to assess eutrophication by means of the following three indices:

- Overall human influence (OHI) on development of conditions (Pressure),
- Overall eutrophic conditions (OEC) within a water body (State), and
- Determination of future outlook (DFO) for conditions within the system (Response).

The three separate indices are combined to provide a single eutrophication assessment rating for an estuary, based on a heuristic approach which includes 94 possible combinations (Bricker et al., 2003).

Several adaptations have been made to the ASSETS methodology to increase its applicability to different types of water bodies (e.g. coastal lagoons) and to improve its assessment capability. These adaptations focus on the pressure and state (OHI and OEC) indices, respectively.

2.1. Determination of pressure—OHI

ASSETS uses a simple model to combine human pressure and system susceptibility. The loading-susceptibility model, initially developed for estuaries with regular river flow (Bricker et al., 2003) has been generalized to any coastal system, including other types of estuaries, bays and lagoons. The general equation is

$$\frac{\mathrm{d}M\mathrm{w}}{\mathrm{d}t} = M_{\mathrm{in}} + M_{\mathrm{ef}} - M_{\mathrm{out}} + M_{\mathrm{ex}}, \tag{1}$$

where M_{w} is the mass of nutrient in the coastal system (kg), t the time (s), M_{in} the nutrient loading to the coastal system from river sources ($\mathrm{kg\,s^{-1}}$), M_{ef} the nutrient loading to the coastal system from other anthropogenic sources (effluents and diffuse sources) ($\mathrm{kg\,s^{-1}}$), M_{out} the nutrient discharge from the coastal system due to advection ($\mathrm{kg\,s^{-1}}$) and M_{ex} the nutrient exchange between the ocean and the coastal system due to dispersion ($\mathrm{kg\,s^{-1}}$).

Expressing M_{out} as the product of v_{out} ($\mathrm{m^3\,s^{-1}}$) and m_{out} ($\mathrm{kg\,m^{-3}}$), where v_{out} is the advective outflow and m_{out} is the outflow nutrient concentration, and dividing through by the volume of the coastal system yields:

$$\frac{\mathrm{d}m\mathrm{w}}{\mathrm{d}t} = \frac{Qm_{\mathrm{in}}}{V} + \frac{M_{\mathrm{ef}}}{V} - \rho m_{\mathrm{out}} + \frac{\varepsilon T_{\mathrm{p}}}{VT}(m_{\mathrm{sea}} - m_{\mathrm{w}}), \tag{2}$$

where m_{w} is the nutrient concentration in the coastal system ($\mathrm{kg\,m^{-3}}$), Q is the river flow ($\mathrm{m^3\,s^{-1}}$), m_{in} is the nutrient concentration in river water ($\mathrm{kg\,m^{-3}}$), ρ the advective flushing rate v_{out}/V ($\mathrm{s^{-1}}$), ε the fraction of water leaving on the ebb which does not return in the flood tide, which is a proxy for re-entrainment, T_{p} the tidal prism ($\mathrm{m^3}$), T the tidal period (s) and m_{sea} the concentration of nutrient in the offshore water ($\mathrm{kg\,m^{-3}}$).

The model developed for OHI in estuarine systems (Bricker et al., 2003) has been modified based on Eq. (2) to derive the following two equations for a steady-state system (a precautionary approach is taken with regard to river-borne nutrient discharge, i.e. that it is derived from anthropogenic sources):

(a) For the hypothetical case where there is no human input, the background nutrient concentration m_{b} in the system can be expressed as

$$m_{\mathrm{b}} = \frac{\varepsilon T_{\mathrm{p}} m_{\mathrm{sea}}}{\varepsilon T_{\mathrm{p}} + QT}. \tag{3}$$

(b) Conversely, if there is no ocean input, i.e. considering only the river loading and the nutrient load M_{ef} in the effluent discharged into the coastal system, the nutrient concentration m_{h} in the system due to anthropogenic activity is

$$m_{\mathrm{h}} = \frac{T(Qm_{\mathrm{in}} + M_{\mathrm{ef}})}{QT + \varepsilon T_{\mathrm{p}}}. \tag{4}$$

For the special case where there are no river inputs, Eq. (2) may be simplified as

$$\frac{\mathrm{d}m\mathrm{w}}{\mathrm{d}t} = \frac{M_{\mathrm{ef}}}{V} + \frac{\varepsilon T_{\mathrm{p}}}{VT}(m_{\mathrm{sea}} - m_{\mathrm{w}}). \tag{5}$$

As before, the modified OHI model based on Eq. (5) yields the following at steady state:

(a) For the hypothetical case where there is no human input, the background nutrient concentration mb in the system can be expressed as

$$m_{\mathrm{b}} = m_{\mathrm{sea}}, \tag{6}$$

where m_{sea} is the offshore concentration.

(b) Conversely, if there is no ocean input, i.e. considering only the nutrient concentration M_{ef} in the effluent discharge into the coastal system:

$$m_{\mathrm{h}} = \frac{M_{\mathrm{ef}} T}{\varepsilon T_{\mathrm{p}}}. \tag{7}$$

Finally, OHI is calculated using the expression:

$$\mathrm{OHI} = \frac{m_{\mathrm{h}}}{(m_{\mathrm{h}} + m_{\mathrm{b}})}. \tag{8}$$

2.2. Determination of state—OEC

The five variables below were selected from the original 16 (Bricker et al., 1999) to use for determination of OEC. These are divided into two groups (Bricker et al., 2003):

1. Primary symptoms: Chlorophyll *a* and macroalgae—excessive concentration or abundance indicates early stages of eutrophication. The epiphytes symptom was removed from the list, due to difficulties in interpretation and application within the model.
2. Secondary symptoms: Low DO, losses of SAV, and occurrence of nuisance and/or toxic algal blooms are indicators of well-developed problems.

The NEEA used information and data syntheses provided by experts working within the estuaries (Bricker et al., 1999). Where possible, ASSETS employs data rather than "expert knowledge", for instance by using statistical criteria for determination of status for chlorophyll *a* (90th percentile) and for DO (10th percentile). Improvements have been made by applying a Geographic Information System (GIS) to the definition of salinity zones, and using GIS-based spatial weighting to determine parameter values. All modifications are fully described in Bricker et al. (2003).

Additional improvements have been proposed for the evaluation of phytobenthos symptoms (macroalgae and SAV). The approach is based on the calculation of potential area of colonization compared to the effective colonized area to determine a percentage value for the spatial coverage classification. Potential colonization areas may be estimated using GIS, by crossing bathymetry and substrate information with physico-chemical data such as non-nutrient-related light extinction due to other types of suspended particulate material. The most suitable areas for colonization by benthic plants are then determined using the GIS layers and expert knowledge and compared to observed areas of colonization.

2.3. Determination of response—DFO

Response is based on an assessment of the susceptibility of the system, the capacity of a system to dilute and/or flush nutrients, in combination with foreseeable changes in nutrient loads. Predictions of nutrient loading (increase, decrease, unchanged) are based on expected population increase, planned management actions and expected changes in watershed uses (see e.g. Boesch, 2002). A matrix is used for DFO (Bricker et al., 2003).

2.4. Synthesis—grouping of pressure, state and response indicators

The OEC, OHI and DFO are combined into a single overall score to provide a grade in one of five categories: high, good, moderate, poor or bad. These categories are color-coded following the convention of the EU Water Framework Directive (2000/60/EC), and provide a scale for setting eutrophication-related reference conditions for different types of transitional (estuarine) and coastal waters (Bricker et al., 2003).

3. Validation for different system types and historical datasets

A detailed assessment for three contrasting systems in the United States and European Union (Table 3) is given below, followed by a comparative application of ASSETS to Long Island Sound using datasets from 1991 and 2002.

3.1. Maryland Coastal Bays

The Maryland Coastal Bays system consists of two interconnected embayments, Isle of Wight Bay and Assawoman Bay, and several smaller subsystems, that extend along the Atlantic coast behind barrier islands adjacent to the Chesapeake Bay. The system is small ($52\,km^2$) and shallow (mean depth 1.1 m) with very low freshwater inflow (Boynton et al., 1996). Circulation is tidally dominated (mean range 0.7 m) and the limited freshwater input combined with restricted ocean exchange results in residence times of 10–21 days (Wazniak and Hall, 2005). The Maryland Coastal Bays are classified in the NEEA as highly susceptible to nutrient inputs due to a low capacity to dilute and flush nutrients (Bricker et al., 1999). Land use is about one quarter urban with the remainder about equal parts forest and mixed agricultural (Fig. 2a; Maryland Department of the Environment, 2002). The most important agricultural use is animal feeding operations, which is the largest contributor of nitrogen to this system (Fig. 2b, Boynton et al., 1996). There is also a seasonal component to nutrient loading due to the summertime watershed population, which increases by an order of magnitude (Boynton et al., 1996).

A total nitrogen load of $2.20 \times 10^3\,ton\,yr^{-1}$ was calculated using demographic and livestock census data for the estuarine watershed (USBC, n.d.; USDA, 1992). Application of the modified ASSETS OHI model, which includes a conservative re-entrainment estimate of 50%, gives a result of 92%, which falls in the High category. High loading and high susceptibility give an OHI rating of High.

Water quality data collected monthly from 21 stations were used for the application of ASSETS and represent more than 1400 samples for 2001–2002 (MD DNR, n.d.), over a yearly period. The chlorophyll *a* percentile 90 for Maryland Coastal Bays is $42.0\,\mu g\,L^{-1}$ resulting in a rating of High (20–$60\,\mu g\,L^{-1}$), though three stations have a percentile 90 in the Hypereutrophic ($>60\,\mu g\,L^{-1}$) range. Spatial coverage for high and hypereutrophic concentrations is high and frequency of occurrence is periodic. Macroalgal abundance is a problem, particularly for nutrient responsive species, with high spatial coverage and periodic frequency (Wazniak et al., 2004). The rating for macroalgae is High (Table 3). DO percentile 10 falls in the biologically stressful range (2–$5\,mg\,L^{-1}$) for nine

Table 3
Summary of system characteristics and ASSETS results

System	Type	Area (km^2)	Mean depth (m)	Tidal Range (m)	Susceptibility	OHI	OEC: Primary symptoms	OEC: Secondary symptoms	OEC: Overall	DFO	ASSETS
Maryland Coastal Bays	Small, limited FW lagoon	52	1.1	0.7	H	H	H	H	H	IL	**Bad**
Firth of Clyde	Fjord	2500	40	<2	H	H	L	H	MH	IL	**Poor**
Tagus estuary	Estuary	320	6	2.6	L	L	M	L	ML	IH	**Good**
Long Island Sound 1991	Large, high FW coastal embayment	3400	19	1.9	M	M	H	H	H	IL	**Bad**
Long Island Sound 2002					M	M	H	L	M	IL	**Moderate**

H—High, MH—Moderate High, M—Moderate, ML—Moderate Low, L—Low, IL—Improve Low, IH—Improve High.

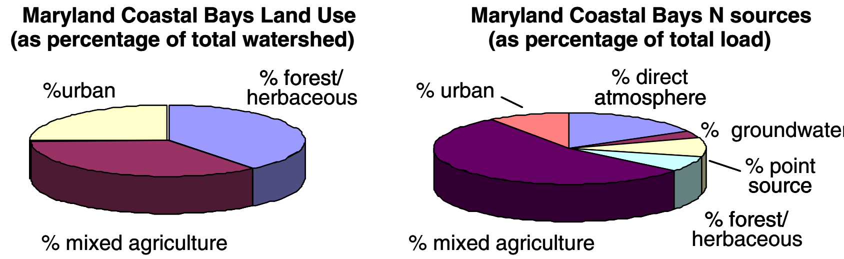


Fig. 2. (a) Watershed land use and (b) sources of nitrogen to Maryland Coastal Bays.

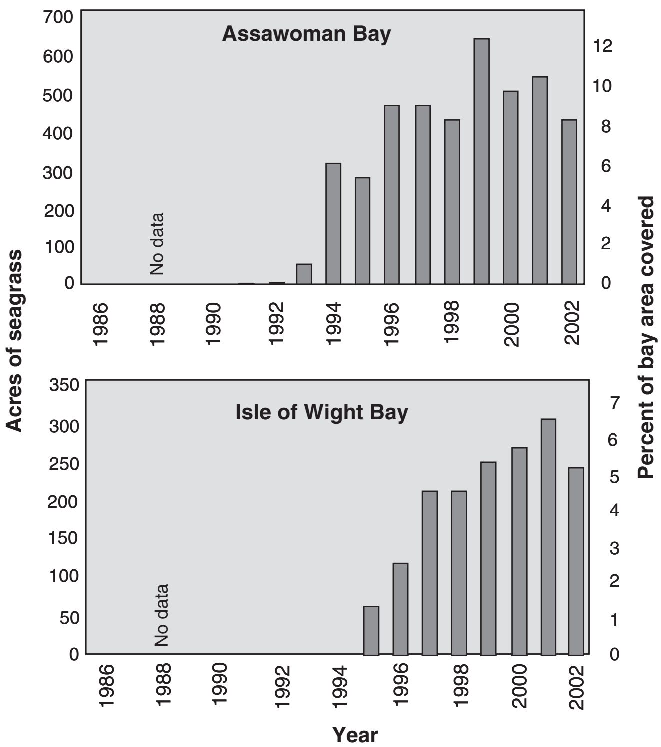


Fig. 3. SAV trends in Maryland Coastal Bays (Assawoman, St. Martin River and Isle of Wight). Adapted from Wazniak et al. (2004).

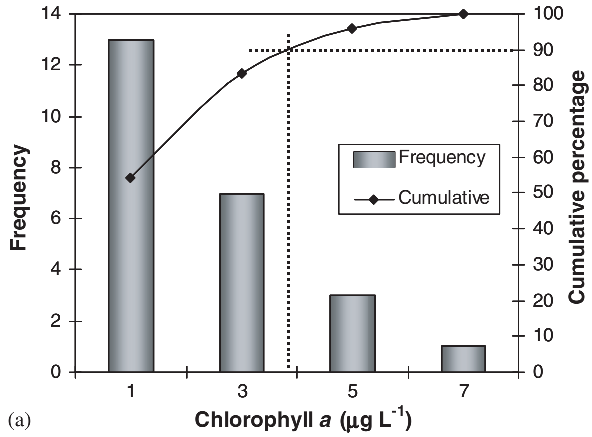
stations, the spatial area is high and the frequency of occurrence is periodic. This gives an overall rating of Moderate for DO. Much of the submerged grass was lost in the 1930s and 1940s due to wasting disease and effects of a 1933 hurricane (Orth, 2002). Since the 1980s, there have been steady increases in seagrass spatial coverage (Wazniak et al., 2004; Fig. 3). This variable is given a rating of Increasing SAV.

Several species responsible for harmful and toxic blooms are known to occur—however, there is no evidence of toxic episodes (Wazniak et al., 2004). In 1999, 2000, 2002 and 2003 a nuisance brown tide species, *Aureococcus anophagefferens* occurred in abundances considered to be a problem ($>35\times10^3$ cells ml^{-1}, Gastrich. and Wazniak, 2002), with high spatial coverage and a periodic frequency of occurrence. Based on these data, the rating for nuisance blooms is High. The final classification for state (OEC), which accounts for primary and secondary eutrophication symptoms, is High (Table 3), indicating significant nutrient-related problems. The expected response of this system was examined by considering future changes in nutrient loading from population growth and land use changes, and existing and planned management measures. Whilst the population is expected to increase, there are plans for implementation of nutrient management measures, and management plans are legally required for all Maryland agricultural land. Reductions of nutrient loads from agricultural uses (especially poultry farms) will be particularly important (Fig. 2). Given the plans for nutrient reductions, it is expected that nutrient loads will decrease in the future (Table 3). The combination of high pressures from the watershed and serious nutrient-related problems results in an overall synthesis rating of Bad for Maryland Coastal Bays, despite the expectation that conditions are expected to improve in the future.

3.2. Firth of Clyde

The Clyde Sea is a deep (mean depth 40 m) broad (2500 km^2) fjord on the west coast of Scotland (Tett et al., 2003) with productive fisheries for Norway Lobster (Mason and Fraser, 1986) and until recently, for herring (Bailey et al., 1986). The Firth of Clyde contains a deep basin separated from the North Channel by a sill. Thermal stratification is strong throughout the summer while salinity-driven stratification is variable but persists throughout the year, and deep waters of the Clyde are replaced during the winter.

Data used for the application of ASSETS are from Turrell and Slesser (1992), Bock et al. (1999), Tett and Edwards (2003) and from a database containing about 560 records collected from four stations in 1993–1994 (J.Y-Lee, pers. comm.). Nutrient concentration-depth profiles for the Clyde Sea, Inner Firth and lochs such as Striven suggest that hypernutrification is widespread in the surface waters where salinities are lower than 33. The high susceptibility of the Clyde to nutrient inputs is due to the low capacity of the system to dilute or flush nutrients (Grantham and Tett, 1993). The Clyde estuary receives domestic and industrial waste from a population of nearly 2.5 million, together with natural, agricultural and industrial nutrients from the catchment area which includes the river Clyde and its tributaries, that drain much of western central Scotland



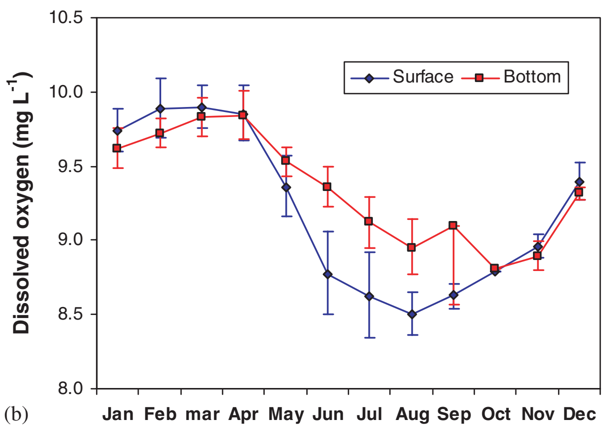


Fig. 4. (a) Chlorophyll *a* percentile 90 of the Firth of Clyde. (b) Dissolved oxygen concentrations in the Firth of Clyde.

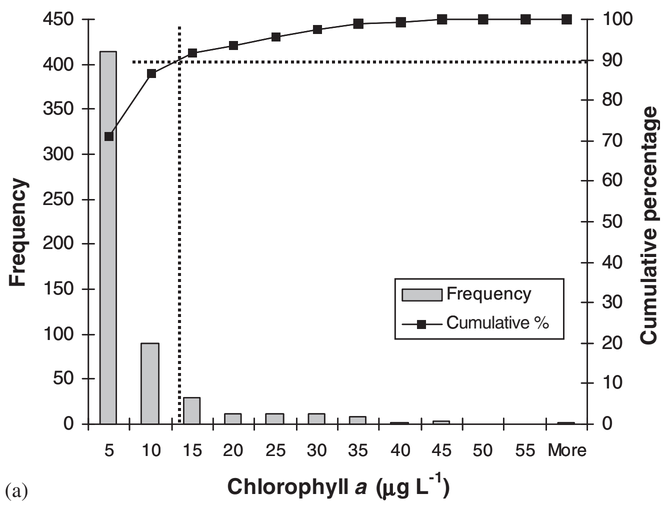
(Haig, 1986; Grantham and Tett, 1993). The second major source of nutrients is the inflow across the Great Plateau that separates the Outer Firth from the North Channel of the Irish Sea (Edwards et al., 1986). Chlorophyll *a* concentrations in the Firth of Clyde are Low ($<5\,\mu g\,L^{-1}$) during all the year, with a percentile 90 of $3.8\,\mu g\,L^{-1}$ (Fig. 4a). No information was found for macroalgae in the Clyde estuary, and the percentile 10 for DO is $8.7\,mg\,L^{-1}$ overall (Fig. 4b). Although this value was calculated with mean monthly values obtained for the lower reaches of the system, minimum values never fall below $8\,mg\,L^{-1}$, which indicates that there are no problems with low DO.

Although the inner Clyde estuary zone supports a rich and varied biological community, which includes eelgrass (*Zostera* spp.) no information on the dynamics of SAV was found. In 1980, fish kills in the Firth of Clyde were associated with a bloom of the red tide organism *Gyrodinium*. Additionally, colonies of *Phaeocytis* sp. continue to occur in waters at Millport (Tett and Edwards, 2003). This suggests that these nuisance blooms are probably periodic, although no data on the periodicity are available—the duration of blooms was classified as seasonal. High secondary symptoms indicate serious problems, but low primary symptoms indicate that other factors such as the morphological characteristics of the system may also be involved in symptom expression. Future nutrient inputs from agriculture depend upon the adoption of good agricultural practices in the use of fertilizers and animal slurries (*sensu* EC "nitrates" Directive 91/676/EEC). Measures taken to reduce pollution risks include the ongoing provision and upgrading of sewage treatment plants for the major cities and towns, which is likely to produce a significant improvement in waters receiving such outfalls (OSPAR, 2000). A decrease in nutrient inputs is thus expected. The Firth of Clyde is classified in the ASSETS category of Poor, which reflects a range of undesired pressure and state conditions despite the recognition that these are expected to improve in the future. Table 3 summarizes the ASSETS results obtained for each classification step.

3.3. Tagus estuary

The Tagus River is the largest of the Iberian Peninsula, flowing into a large ($320\,km^2$, mean depth 6 m, tidal range 2.6 m) tidal estuary. Forty percent of the estuarine area is intertidal, covered by salt marshes and mudflats. Data for the Tagus estuary were compiled in a relational database containing about 80 000 records. These cover 146 sampling stations distributed throughout the system, over a 40-yr period. The estuary has a low susceptibility to nutrient inputs due to the combination of low average depth, strong tidal currents, and low input of river water, which promote vertically homogenous conditions in the water column. The nitrogen load to the system is classified as moderate and its combination with the low susceptibility to nutrient inputs gives a final pressure (OHI) rating of Low (Table 3). High eutrophic conditions are obtained for chlorophyll *a* in the tidal freshwater and mixing zones while in the seawater zone the percentile 90 falls within the Medium (5–20 $\mu g\,L^{-1}$) category (Fig. 5a). High values are observed every year and the spatial coverage of these values is also high. Although macroalgal colonization occurs in old oyster shell substrates in the intertidal part of the mixing and seawater zones, no problems with excessive growth are detected. SAV is not present in the Tagus and thus was not considered in the assessment of secondary symptoms. The percentile 10 values for DO in the tidal freshwater and seawater zones are above the biological stress threshold ($5\,mg\,L^{-1}$). In the mixing zone, the percentile 10 value of $5\,mg\,L^{-1}$ is mainly influenced by a few episodic low values covering part of the zone (Fig. 5b).

Nuisance and toxic bloom occurrences were not observed over an extended period of time (20 yr). The Moderate rating for primary symptoms indicates the potential onset of primary symptoms, although no secondary symptoms are observed. Improvements in the wastewater treatment level, capacity and nutrient removal efficiency are projected for the current wastewater treatment plants (WWTP) that discharge to the Tagus Estuary. Furthermore, the construction of new WWTP is expected as the watershed population grows. Thus, it is considered



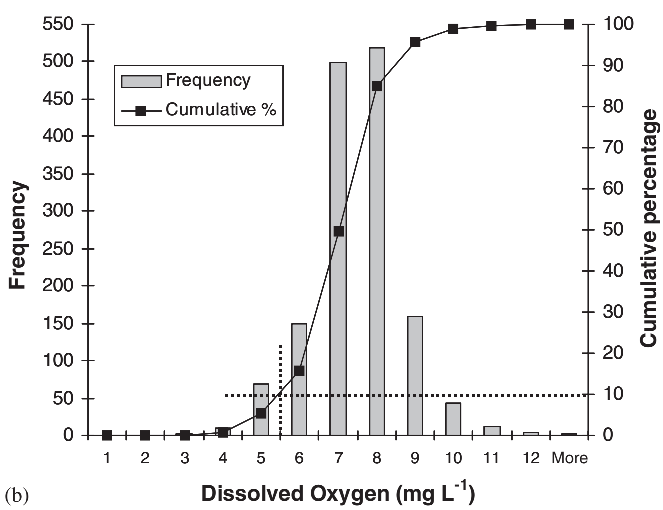


Fig. 5. (a) Percentile 90 for chlorophyll *a* values and (b) percentile 10 for the dissolved oxygen values, in the seawater zone of the Tagus estuary.

that future nutrient loading to the estuary will be significantly reduced. The final ASSETS grade for the Tagus estuary is Good, due to the relatively low pressure, good conditions within the system, and the expected continuation of good conditions in the future as inputs decrease. Table 3 summarizes the ASSETS results obtained for each classification step.

3.4. Long Island Sound

Long Island Sound is a large (3400 km^2) estuary, has an average tidal range of about 2 m, and connections to the ocean at its western end via Block Island Sound and via the East River and New York Harbour to the east. The major tributaries enter from the north with the Connecticut River accounting for about 70% of total freshwater inflow (Wolfe et al., 1991). The East river promotes stratification in the western sound particularly during the spring runoff period (Bricker et al., 1997). The ASSETS method was applied to Long Island Sound to examine whether there have been noticeable changes between 1991 and 2002, a decade after the implementation of management measures designed to reduce nitrogen inputs to the Sound.

Water quality data used for the application of ASSETS to Long Island Sound are from the Long Island Sound Study and represent more than 111 monthly samples for seven stations in 1991 and 387 monthly samples for 17 stations in 2002. Nutrient loading estimates are from NYSDEC and CTDEP (2000). The most significant feature of this system is its location adjacent to the New York metropolitan area and Bridgeport and New Haven, two of Connecticut's largest cities with a total watershed population of over 8 million (USBC, 2002). The total nitrogen loading to Long Island Sound is $60.7 \times 10^3\,\mathrm{ton\,yr^{-1}}$, primarily from point sources (NYSDEC and CTDEP, 2000). Since 1990, about 25 of the 105 WWTP in Connecticut and New York have been upgraded to biological nutrient removal of nitrogen and more are under construction or planned. These upgrades have led to a 30% decrease in nitrogen loading from WWTPs since 1990 (LISS, 2003) and it is expected that these improvements will continue (NYCDEP, 2000; NYSDEC and CTDEP, 2000). The combination of high dilution potential and low flushing potential gives this system a susceptibility rating of Moderate. Application of the loading-susceptibility model using a conservative re-entrainment value of 50% gives a human influence of 59% in 1991 and 51% in 2002. Despite the significant decrease in loadings, the rating of OHI is Moderate for both years. The sensitivity of the OHI component is presently being tested and could potentially be modified since load reductions such as these should be reflected in changes of OHI category. With moderate inputs and moderate susceptibility, the rating for OHI is Moderate for both years (Table 3).

Chlorophyll *a* data for all Long Island Sound stations shows a decrease in the percentile 90 concentration from 18 to 9 $\mu g\,L^{-1}$ between 1991 and 2002, and in average winter/spring bloom concentrations from 17 to about 2 $\mu g\,L^{-1}$ in Western Long Island Sound (LISS, 2001). For both years, the frequency of occurrence is periodic. The spatial coverage in both years is high and the overall rating for chlorophyll *a* is High for both years. Macroalgae were identified as a high-level problem in the early 1990s (Bricker et al., 1999); however, there are no data for comparison to conditions in 2002, and this variable is not used in the assessment. DO percentile 10 for all stations shows an increase from 3.9 $mg\,L^{-1}$ in 1991 to 6.4 $mg\,L^{-1}$ in 2002. However, biologically stressful concentrations are seen in both years with a spatial coverage of High for 1991 and Moderate for 2002. This is concurrent with an observed decrease in hypoxic area from almost 800 km^2 in 1987 to about 330 km^2 in 2002 (LISS, 2003; Fig. 6a). Though the duration is highly variable, there is also a decreasing trend in the duration of low DO events over the same time period (Fig. 6b). The rating for DO is Moderate for 1991 and Low for 2002.

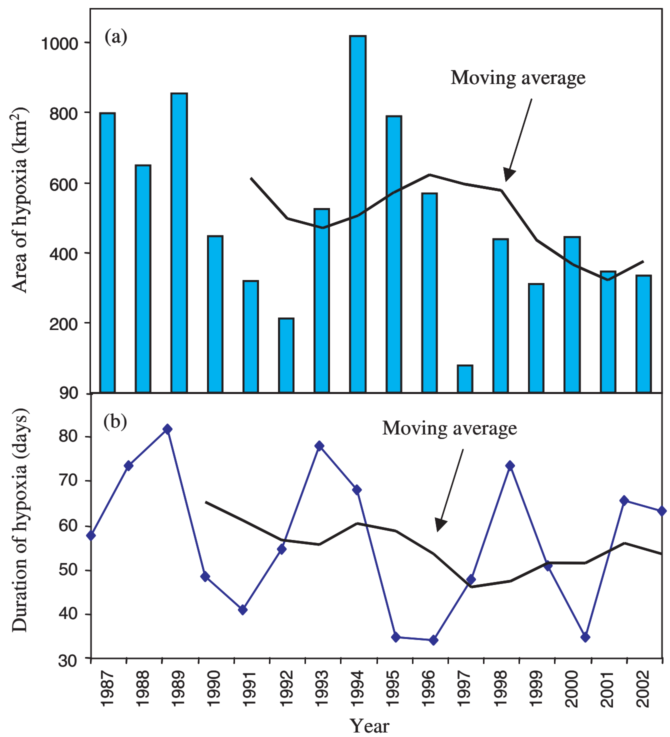


Fig. 6. Areal extent and duration of hypoxia in Long Island Sound 1987–2002.

Nuisance and Toxic Blooms were identified as a moderate-level problem in the early 1990s (Bricker et al., 1999) but there are no data for 2002 for comparison, so this variable was not used in the assessment. SAV was lost in the 1970s and 1980s due to high chlorophyll *a* concentrations in the water column (LISS, 2003). SAV spatial coverage is very low in 1991 and 2002; however, there has been a small increase in SAV from 1991 to 2002. In Mumford Cove, Connecticut eelgrass has increased by 20 ha from 1987 to 2002 (LISS, 2003). The rating for SAV for 1991 is High (worse) and the rating for 2002 is Low (better). The OEC for Long Island Sound is High for 1991 and Moderate for 2002 (Table 3). Although the population is expected to increase in the Long Island Sound watershed over the next 20 yr, the EPA approved TMDLs and the agreement to reduce nitrogen by 58.5% by 2014 (LISS, 2003) are likely to result in continued declines in loadings. The expected decrease in inputs combined with the moderate susceptibility gives a response rating of Improve Low for expected eutrophic conditions in Long Island Sound in the future for both 1991 and 2002 (Table 3). The combination of pressure-state-response scores in 1991 results in an ASSETS rating of Bad. The improvements in conditions within the system that resulted from the decreases in loadings during the 1990s are reflected in the ASSETS score of Moderate for 2002. Although the OHI grade remained unchanged, the assessment method is clearly sensitive to the consequences of lower nutrient pressure on symptom expression (state)—this is evident in the secondary symptom scores.

4. Conclusions

The results obtained for the various systems show that the assessment methodology appears to be sufficiently robust to allow its application to a range of different types. For the Pressure component, the susceptibility may be determined for estuarine systems or for coastal areas with negligible freshwater input, and the sensitivity of the ASSETS method to load reductions will be improved. The various symptoms considered in the assessment of State make this approach suitable for a wide range of system types, where eutrophication may be expressed through phytoplankton blooms, species shifts and occurrence of HAB, or in the benthos by the development of algal mats. The main areas of development of ASSETS are in (i) Improvements to the Definition of Future Outlook, which as the Response component of the method is the key link to management; and (ii) the definition of type-specific reference conditions.

4.1. *Definition of future outlook*

This area presents a challenge in the definition and quantification of measures, which will result in an improved state through a change in pressures, as well as in the definition of appropriate metrics through which response may be assessed. Some socio-economic indicators such as fishery yields are potentially ambiguous, since nutrient enrichment may result in an increase in food availability, whilst simultaneously reducing habitat due to low DO concentrations and loss of shelter, e.g. due to SAV losses. The development of research models which may be used to drive screening models of this type (Nobre et al., 2005) is one approach for detailed assessment of the effect of response measures on the state of a particular system.

4.2. *Type-specific reference conditions*

Typology is a key development area for ASSETS. The NEEA report (Bricker et al., 1999) made it clear that the use of uniform ranges for eutrophication symptoms may lead to misclassification. Expert evaluation of Florida Bay, for instance, considered it to be heavily impacted at chlorophyll *a* values in the Low category. Equally, naturally occurring HABs, which develop in offshore frontal systems and are advected inshore, were responsible for misclassification of Narraguagus Bay. These events also occur in upwelling systems during periods of relaxation, and have been widely reported in the Northwest United States coast, the Iberian Atlantic and the Benguela upwelling. Eutrophication symptoms vary within and across types: for instance, the phytoplankton species composition in an estuary has been shown to be linked to flushing time (Ferreira et al., 2005, 2006), and high (poor) ASSETS grades tend to be associated with slower flushing, reflecting both higher chlorophyll *a* values and increased

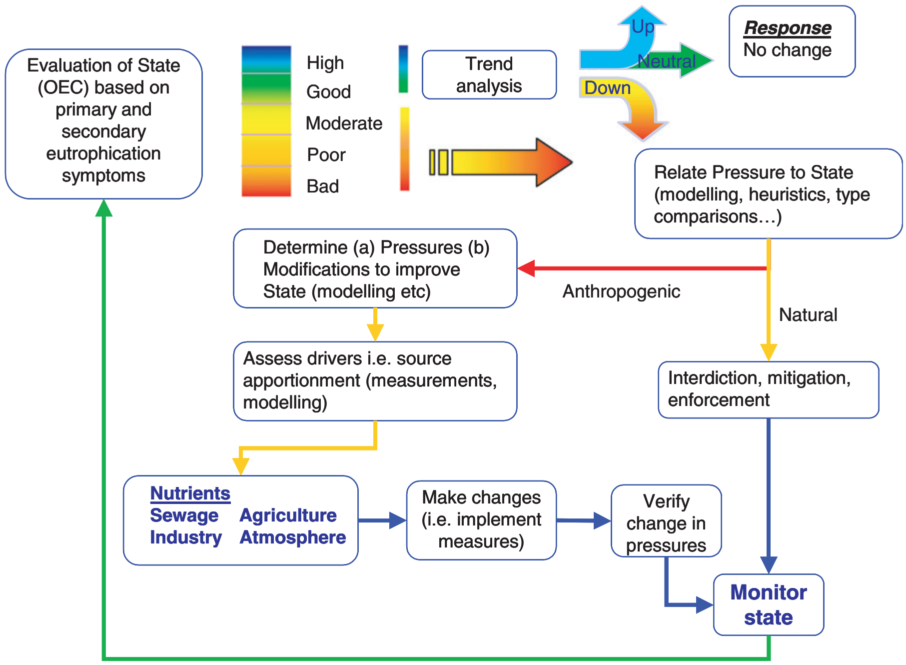


Fig. 7. Conceptual approach to coastal eutrophication management, distinguishing between anthropogenic and natural pressures.

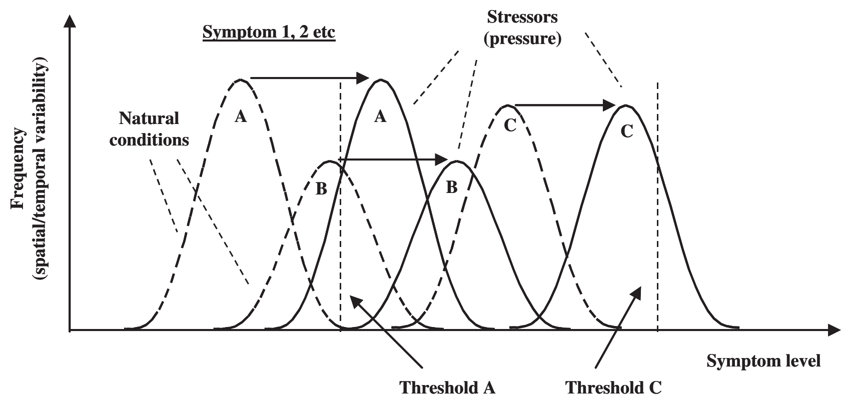


Fig. 8. Type-specific variability in natural and stressed conditions, and respective threshold shifts. The dotted curves represent natural conditions and the full curves show increased eutrophication symptoms due to pressure.

occurrence of HAB. This trend emerges even without considering the variation in pressure among estuaries.

Since ASSETS is a management-oriented screening procedure, it must have the capability to distinguish between a classification where management measures will result in a modification of state, and situations which are naturally occurring and cannot be managed except operationally, e.g. by early warning and timely fisheries interdiction in the case of HABs. A general scheme for interpretation and management response to these different pressures is presented in Fig. 7.

The fact that natural conditions for eutrophication symptoms (and other quality assessment reference conditions) vary among different water body types is normative in the European Union Water Framework Directive, and is recognized by NOAA and EPA in the United States. The consequences of this for the definition of symptom thresholds are shown in Fig. 8. It is for instance possible for a system belonging to a particular type to have naturally low DO, associated with limited mixing, or high water temperature or salinity. The development of ASSETS is thus closely linked to typology definitions for the United States and European Union. Typology should be based on physical characteristics, rather than metrics such as chlorophyll *a*, since systems belonging to the same type will usually exhibit a range of eutrophication symptoms, depending on human use. The DISCO package (Smith and Maxwell, 2002) derived from LOICZVIEW, has been successfully used to cluster United States estuaries into a few types as the first step in an update to the 1999 US National Eutrophication Assessment (Bricker et al., 2004), and top-down approaches based on heurisitics have been applied in Europe (Ferreira et al., 2005, 2006).

The definition of thresholds for eutrophication symptoms such as chlorophyll *a* or DO must be based on robust criteria, related to ecological sustainability and type-specific reference conditions, i.e. a quasi-pristine situation. Additionally, the assessment of state may include facultative variables, which will depend on type. The number of categories to be used should be type-independent, as should the normalized scores, which are processed in the ASSETS OEC matrix. However, those normalized scores will be associated with type-specific thresholds, so that the results obtained provide a more balanced classification and provide decision-makers with a more reliable approach for management.

The ASSETS screening model, together with the detailed classification of these and many other coastal systems is available at http://www.eutro.org.

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