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Classifying ecological quality and integrity of estuaries

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Abstract

There is an increasing need in assessing ecological quality and integrity of estuaries and lagoons. This chapter shows the most recent efforts in assessing individual biological elements (from phytoplankton to fishes), together with the integrative tools developed in different geographical areas worldwide. However, reducing complex information from multiple ecosystem elements to a single color or value is a substantial challenge to marine scientists, and requires the integration of different disciplines (chemists, engineers, biologists, ecologists, physics, managers, etc.), to reach agreement on the final assignment of ecological status. Hence, in the near future, emphasis needs to be directed at understanding the complexities of estuarine system functioning rather than simplifying and scaling down the system into smaller components.

Keywords

quality, integrity, estuaries, ecological, classifying

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1.08 Classifying Ecological Quality and Integrity of Estuaries

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Abstract

There is an increasing need in assessing ecological quality and integrity of estuaries and lagoons. This chapter shows the most recent efforts in assessing individual biological elements (from phytoplankton to fishes), together with the integrative tools developed in different geographical areas worldwide. However, reducing complex information from multiple ecosystem elements to a single color or value is a substantial challenge to marine scientists, and requires the integration of different disciplines (chemists, engineers, biologists, ecologists, physics, managers, etc.), to reach agreement on the final assignment of ecological status. Hence, in the near future, emphasis needs to be directed at understanding the complexities of estuarine system functioning rather than simplifying and scaling down the system into smaller components.

1.08.1 Introduction

1.08.1.1 Estuarine Management and the Need for Classifying Ecological Quality

Marine environments, in general, and transitional waters (estuaries and lagoons) in particular, are facing increasing and

significant impacts, which include physical and chemical transformation, habitat destruction, and changes in biodiversity (Halpern et al., 2007, 2008). The causes include land reclamation, dredging, pollution (sediment discharges, hazardous substances, eutrophication, etc.), unsustainable exploitation of estuarine resources (shellfishing, fishing, aggregate

extraction, etc.), unmanaged tourism, introduction of alien species, and climate change (see Halpern et al., 2007).

Being areas where rivers meet the sea, estuaries not only are highly variable environments but also are often the focus of human activities. Anthropogenic activities have induced modifications in the physical characteristics of estuaries through dredging, land reclamation, harbor and industrial development, as well as recreational and tourist development; the water quality of these environments is also affected by discharges of pollutants such as domestic and industrial effluents. Biological components have also been subject to human influence through commercial harvesting of certain species as well as the introduction of alien species (either species which compete directly for resources or through the introduction of parasite and disease organisms). Estuaries are also affected by activities in the catchment such as water abstraction as well as pollutants from agricultural runoff and industrial activities.

To manage these pressures and impacts on transitional environments, legislative instruments approved worldwide (i.e., Clean Water Act, in USA, or the Water Framework Directive (WFD), in Europe) address the need to assess their ecological or environmental status (see Borja et al. (2008a) for an overview). The concept of environmental or ecological status takes into account the structure, function, and processes of marine ecosystems bringing together natural physical, chemical, physiographic, geographic, and climatic factors, and integrates these conditions with the anthropogenic impacts and human activities in the area concerned (Borja et al., 2009b).

As commented in Borja et al. (2008a), the above concept defines quality in an integrative way, by using several biological parameters (from phytoplankton to mammals) together with physico-chemical and pollution elements. Rogers et al. (2007) reviewed the selection of the ecosystem components, adding to the above-mentioned structural components other ecosystem attributes such as food web dynamics, species diversity, and the distribution of life histories, which are not direct biological properties but functions of the entire ecosystem. They are important because they provide information about the functioning and status of the ecosystem, and have been widely perceived as additional and potentially useful indicators of estuarine environmental status. This approach is intended to allow an assessment of the ecological status at the ecosystem level ('ecosystem-based approach' or 'holistic approach' methodologies) more effectively than can be done at a species or chemical level (i.e., quality objectives).

'Ecosystem-based management' emphasizes four common principles, namely that effective management must (Boesch, 2006; Elliott et al., 2006): (1) be integrated among components of the ecosystem and resource uses and users; (2) lead to sustainable outcomes; (3) take precaution in avoiding deleterious actions; and (4) be adaptive in seeking more effective approaches based on experience.

Hence, an ecosystem-based approach should explicitly account for the interconnections within the estuarine ecosystem, recognizing the importance of interactions among many target species or key services and other nontarget species; acknowledge interconnections among ecosystems, such as air, land, and sea; and integrate ecological, social, economic, and institutional perspectives, recognizing their strong interdependencies (Boesch, 2006).

However, following Borja et al. (2009b), not all integrative tools currently available are able to respond to these requirements of the ecosystem-based approach. Hence, several well-established, integrative techniques, such as sediment quality triad (SQT), weight of evidence (WOE), and ecological risk assessment (ERA) (see Chapman, 2009) focus more on assessing pollution (at ecotoxicological level) than assessing integrity of the ecosystem (Borja et al., 2009b).

Hence, methods for classifying estuaries and lagoons according to anthropogenic stress include those centered on the primary community structural variables (abundance, species richness, and biomass) and derived community structural variables (such as diversity indices, abundance (A/S) and biomass (B/A) ratios, and evenness indices) (see Gray and Elliott (2009) for references and details). They also include functional analyses such as those involving feeding guilds (as in the infaunal trophic index (ITI), by Word, 1990) and their responses to elevated organic levels (as in the AZTI's marine biotic index (AMBI), by Borja et al. (2000), and the benthic quality index (BQI), by Rosenberg et al. (2004), among others; see many other examples in this chapter). For example, detritus and deposit-feeding dominance are reflected in any assessment of trophic analysis. As indicated in Gray and Elliott (2009), there are well-defined numerical methods which aim to detect and reflect stress in benthic communities. For example, species-abundance-biomass (SAB) curves (Pearson and Rosenberg, 1978), abundance-biomass comparisons (ABCs, Warwick, 1986), AMBI (Borja et al., 2000), and diversity indices. (e.g., Rosenberg et al., 2004; Borja et al., 2007). Given the estuarine quality paradox (see this concept in the next section), however, the main problem is that some of these methods detect naturally as well as anthropogenically stressed areas, thus decreasing the ability of detecting and maximizing the signal (anthropogenic change)-to-noise (natural change) ratio.

Finally, though successful, the advances made in recent times in assessing estuarine and lagoonal quality were probably only a first step and many challenges remain to be addressed in the future, including the development of reliable methods to integrate multiple physico-chemical and biological elements into a single evaluation of estuarine system condition (Borja et al., 2008a, 2009a). This integration should be made by using different elements, different media, and results from different locations within the same estuarine water body. The challenge is not only to integrate indicators for single ecosystem elements, but also to include measures of ecosystem structure, function, and processes. Hence, the ecological integrity of an estuary or a lagoon should be evaluated using all information available, including as many biological ecosystem elements as is reasonable, and using an ecosystem-based assessment approach (Borja et al., 2008a, 2009e). Hence, this chapter focuses on this challenging issue, as an overview of the current situation worldwide.

1.08.1.2 The Estuarine Quality Paradox and Environmental Homeostasis

The ecological components of estuaries have long been known to follow a well-defined set of characteristics – for example, estuaries are characterized by having a few dominant species of stress-tolerant, euryoecious, small-form (low individual biomass), short-lived organisms which occur in high abundances and are tolerant of organic-rich areas; intraspecific competition

is more likely than interspecific competition (e.g., [McLusky and Elliott, 2004](#)). The forms are more likely to be r-strategists rather than k-strategists ([Gray and Elliott, 2009](#)). These typical estuarine species are tolerant of high variability in environmental master factors; for example, those such as tubificid oligochaetes and certain nereid polychaetes, which occur mainly in estuaries, will not be stressed by variable salinity.

Hence, estuaries and lagoons have long been regarded as environmentally naturally stressed areas because of the high degree of variability in their physico-chemical characteristics. In particular, most environmental variables such as temperature, salinity, dissolved oxygen (DO), turbidity, and bed sediment widely vary along spatial and temporal variables. Accordingly, the biota is adapted to such changes, thus ensuring they are naturally stress tolerant and hence resilient to change. This then suggests that they can absorb stress without showing adverse effects – the so-called ‘environmental homeostasis’ ([Elliott and Quintino, 2007](#)). Under these conditions, many species are not present (e.g., stenohaline marine forms which cannot tolerate widely varying salinities) and so only stress-tolerant species are found. Hence, stress for one species, by causing it to be absent, becomes a subsidy (i.e., a benefit) for another species which can then capitalize on the lack of interspecific competitors and thus be more successful. [Odum \(1985\)](#) first discussed the stress–subsidy continuum and indicated that one organism’s stress (adverse effect) is another organism’s subsidy (benefit).

Estuaries and their biota are classified extensively according to the response to anthropogenic stress, especially the way

in which the biota respond at the individual, population, and community levels of biological organization. In many cases, this relates to the structure of those biological elements, especially community structure. These features have been shown in the fish ([Breine et al., 2007](#)) and macrobenthos ([Gray and Elliott, 2009](#); [Table 1](#)); the latter of which under anthropogenic stress is characterized by small organisms, r-strategists and the replacement of k-strategists, high abundances of few organically tolerant species, and low diversity and low individual biomass organisms with the potential to produce high biomasses. In addition, they have a high turnover and biological productivity (as shown by an increase in the production to average biomass ratio, P/B) and a dominance by oligochaetes and oligochaetiform polychaetes which tolerate adverse environmental conditions such as low oxygen and low and variable salinity. It is emphasized that these are expected characteristics of many stressed ecosystems not only pertaining to the marine and estuarine environment ([Odum, 1969, 1985](#)) ([Table 2](#)).

These features also apply to the floral community (e.g., [Wilkinson et al., 2007](#)) wherein polluted estuarine and lagoonal areas, especially those influenced by organic discharges, sewage runoff, and industrial effluent, become dominated by opportunistic green algae, occasionally forming mats. Despite this, large concentrations of ephemeral green filamentous algae are naturally occurring in transitional water bodies, which naturally have large nutrient inputs and retain these nutrients ([Wilkinson et al., 1995](#)). Estuaries naturally

Table 1 Conceptual basis and assumptions inherent in macrobenthic impact studies

A. Natural State

- 1) A natural macrobenthic assemblage either tends toward or is in an equilibrium state.
- 2) Under nonimpacted conditions, there are well-defined relationships (which therefore may be modeled) between faunal and environmental (abiotic) variables.
- 3) In approaching the normal equilibrium state, the biomass becomes dominated by a few species characterized by low abundance but large individual size and weight.
- 4) Numerical dominance is of species with moderately small individuals; this produces among the species a more even distribution of abundance than biomass.
- 5) The species are predominantly k-selected strategists.

B. Moderate pollution

- 6) With moderate pollution (stress), the larger (biomass) dominants are eliminated, thus producing a greater similarity in evenness in terms of abundance and biomass.
- 7) Also with moderate pollution, diversity may increase temporarily through the influx of transition species.

C. Severe pollution

- 8) Under severe pollution or disturbance, communities become numerically dominated by a few species with very small individuals.
- 9) Those small individuals are often of opportunist, pollution-tolerant species which have r-selected strategies.
- 10) Under severe pollution, any large species that remain will contribute proportionally more to the total biomass relative to their abundance than will the numerical dominants.
- 11) Thus, under severe pollution, the biomass may be more evenly distributed among species than is abundance.
- 12) However, under severe pollution, species with large individuals may be so rare as to be not taken with normal sampling.
- 13) The change in assemblage structure with increasing disturbance is predictable, follows the conceptual models and is amenable to modeling and significance testing.

D. Recovery

- 14) Opportunists are inherently poor competitors and may thus be outcompeted by transition species and k-strategists if conditions improve.

[McManus and Pauly \(1990\)](#) also consider that under normal conditions:

- 1) The biomass dominants will approach a state of equilibrium with available resources.
- 2) The smaller species are out of equilibrium with available resources.
- 3) The abundances of the smaller species are subject to more stochastically controlled variation than the larger species.

Modified from [Warwick, R.M., 1986. A new method for detecting pollution effects on marine macrobenthic communities. Marine Biology 92, 557–562](#) and [McManus, J.W., Pauly, D., 1990. Measuring ecological stress: variations on a theme by R.M. Warwick. Marine Biology 106, 305–308.](#)

Table 2 Trends expected in stressed ecosystems: the estuarine features as applied to topics summarized by Odum (1985)

Feature	Odum (1985)	Estuarine feature	
Energetics	1. Community respiration increases	Yes, in general:	Higher respiration in larger populations of small organisms and organic rich sediments; possibly with osmoregulatory stress caused by salinity change
	2. P/R (production/respiration) becomes unbalanced	Unknown	Possibly due to higher respiration caused by salinity stress
	3. P/B and R/B (maintenance:biomass structure) increase	Yes, in general:	Higher P/B in smaller and shorter lived organisms, e.g., dominance by oligochaetes and small polychaetes; high turnover organisms
	4. Importance of auxiliary energy increases	Depends on meaning:	Increase in allochthonous energy input as well as relatively high autochthonous production
	5. Exported or unused primary production increases	Depends on meaning:	Export of material to adjacent sea areas but also import from catchment
Nutrient cycling	6. Nutrient turnover increases	Yes, but:	Because of physical characteristics – high nutrient inputs and use
	7. Horizontal transport increases and vertical cycling of nutrients decreases (cycling index decreases)	Partly the case:	Both horizontal and vertical cycling is high, depending on flushing characteristics and residence time; importance of material movement from pelagic to benthic system
	8. Nutrient loss increases	Yes, but:	Because of the physical characteristics – high nutrient loss through flushing and export through predators
Community structure	9. Proportion of r-strategists increases	Yes:	High abundances of few, short-lived stress-tolerant species
	10. Size of organisms decreases	Yes:	High abundances of small organisms dominant in benthos; low megafaunal populations
	11. Lifespans of organisms decreases	Yes, in general:	On average, benthic and planktonic community composed of short-lived organisms; planktonic organisms adapted to prevent flushing of populations
	12. Food chains shorten because of reduced energy flow at higher trophic levels and/or greater sensitivity of predators to stress	Not necessarily:	Food chains can be not only very short (macrophytes-herbivorous ducks) but also very long because of the opportunistic nature of many predators; while marine predators (stenohaline marine fishes) may be reduced there are many other fish and bird predators
	13. Species diversity decreases and dominance increases; if original diversity is low, the reverse may occur; at the ecosystem level, redundancy of parallel processes theoretically declines	Yes (first part); unknown (second part):	Classic estuarine community in all components of few species; exacerbated with distance landward in the estuary; competition between species may be less than competition within species
General system-level trends	14. Ecosystem becomes more open (i.e., input and output environments become more important as internal cycling is reduced)	Not necessarily so:	Internal cycling is important even though nutrients and organic matter are delivered from external sources
	15. Autogenic successional trends reverse (succession reverts to earlier stages)	Unknown:	
	16. Efficiency of resource use decreases	Not necessarily so:	While there may be an excess of organic resources, leading to export, much is used within the system to support high predator populations
	17. Parasitism and other negative interactions increase, and mutualism and other positive interactions decrease	Not shown:	
	18. Functional properties (e.g., community metabolism) are more robust (homeostatic-resistant to stressors) than are species composition and other structural properties	Yes:	Ability of the system to withstand stressor effects without adverse impacts

show the transition from a highly diverse marine flora, with many red and brown macroalgae as well as green macroalgae in the outer regions, to an upper estuarine algal flora dominated by the Chlorophyceae. Under high organic and nutrient loading, the features produce macroalgal mats with the latter often displacing seagrass beds (de Jonge and Elliott, 2002).

Such an estuarine stress is detected not only at the community level but also at the physiological level of biological organization. Methods such as scope for growth (SFG) have long been used to indicate anthropogenic stress in marine and estuarine areas (e.g., Widdows and Johnson, 1988). However, Navarro (1988) and Guerin and Stickle (1992) both indicate the way in which salinity stress, through natural freshwater inputs, reduces energetic budgets.

Therefore, although estuaries and lagoons are exposed to high degrees of anthropogenic stress, they are also naturally stressed, highly variable ecosystems. The difficulty of separating these causes of change (natural and anthropogenic stress) in estuaries in relation to the usual structural features in estuaries has given rise to the estuarine quality paradox (Dauvin, 2007; Elliott and Quintino, 2007). This can be defined as: "the characteristic by which the dominant estuarine faunal and floral community is adapted to and reflects high spatial and temporal variability in naturally highly stressed areas but the natural community has features very similar to those found in anthropogenically-stressed areas thus making it difficult to detect anthropogenically-induced stress in estuaries. Furthermore, as estuaries are naturally organically-rich areas then the biota has similarities to anthropogenically-organic rich areas. Because of this, there is the danger that any indices which are based on those features and used to plan environmental improvements are flawed."

The main difficulty posed by the estuarine quality paradox is that any technique used for classifying areas using the characteristics shown by both natural estuarine and unnatural anthropogenic features will erroneously show estuaries to be stressed. Hence, there is the difficulty in using the techniques in estuaries for detecting and separating anthropogenic stress from natural stress.

1.08.2 Classifying Biological Quality Elements

Biological quality elements (i.e., phytoplankton, macroalgae, macroinvertebrates, angiosperms, or fishes) in transitional waters are supported by several physico-chemical elements, such as temperature, salinity, nutrients, dissolved oxygen, suspended solids, and turbidity. These elements, together with geomorphological and tide characteristics, determine the type of estuary (Harrison, 2004), and, to some extent, the biological communities that the transitional waters can support. Hence, they have been used, under some legislation, as key elements in assessing ecological quality, that is, oxygen (Best et al., 2007a), nutrients (Devlin et al., 2007), or suspended solids (Bilotta and Brazier, 2008; Devlin et al., 2008).

Most of these variables are related to eutrophication, and normally are studied in the same context as phytoplankton or macroalgae (see below, and Loureiro et al., 2006; Azevedo et al., 2008; Giordani et al., 2009). However, some legislation, such as the WFD, seeks for independent assessments. This led to the proposal for methods to assess physico-chemical elements alone (Borja et al., 2004b; Bald et al., 2005).

1.08.2.1 Plankton

1.08.2.1.1 Phytoplankton are good indicators of eutrophication

The first thing to occur with nutrient enrichment of coastal water bodies is uptake by and stimulation of phytoplankton growth. Excessive amounts of nutrients may, under the right conditions, cause overgrowth of phytoplankton leading to low dissolved oxygen conditions as the bloom dies and the biomass decays, as well as reduced water transparency which may lead to losses of seagrasses (Figure 1). Additionally, nutrient additions may cause changes in natural nutrient ratios and/or speciation leading to blooms of opportunistic species, many of which are harmful or toxic (e.g., *Aureococcus anophagefferens*; Glibert et al., 2007). Because of their direct link and sensitivity to nutrient loading, phytoplankton growth is considered a direct effect (e.g., OSPAR, 2002; WFD methods) or a primary

Figure 1 Progression of eutrophication and impact evaluation starting with increased primary production reflected as increased phytoplankton biomass (Chl *a*) and macroalgal abundance, leading to low dissolved oxygen, losses of seagrasses, and changes in community composition to include nuisance and toxic blooms. From Bricker, S.B., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C., Woerner, J., 2007. Effects of Nutrient Enrichment in the Nation's Estuaries: A Decade of Change, National Estuarine Eutrophication Assessment Update. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD, 322 pp. <http://ccma.nos.noaa.gov/news/feature/Eutrouupdate.html> (accessed July 2010).

symptom (e.g., the assessment of estuarine trophic status (ASSETS) method, in Bricker et al., 2003), indicative of the beginnings of eutrophication problems. As such, phytoplankton is a good indicator for nutrient-related impacts. Typically, measurements of chlorophyll *a* (Chl *a*) are used to represent phytoplankton biomass in coastal systems. Other measures that are used to evaluate the status of phytoplankton populations include the abundance and species composition of the phytoplankton community and changes in the frequency and duration of blooms.

Several methods have been developed to evaluate the status of phytoplankton in coastal and estuarine water bodies for use as an indicator of nutrient-related water-quality impairment (Table 3). The methods include different metrics, some use only Chl-*a* concentration (e.g., Trophic State Index (TRIX), Environmental Protection Agency's National Coastal Assessment (EPA NCA), and Institut Français pour l'Exploration de la Mer (IFREMER)), whereas others combine additional characteristics such as duration and spatial coverage of bloom concentrations (e.g., ASSETS), or weighting factors that represent the relative contribution to overall water quality (e.g., transitional water quality index (TWQI) lake water quality index (LWQI); Table 3). This highlights that the Chl-*a* indicators, while able to stand on their own, are typically part of larger multivariable indices designed to accurately evaluate overall eutrophic conditions. This is reflective of the fact that they indicate the first signs of nutrient-related problems, whereas other indirect problems such as low dissolved oxygen and losses of seagrasses are indicative of more significant nutrient-related impairment. Additionally, although phytoplankton (i.e., Chl *a*) may increase with increased nutrient additions in some types of systems, in others, biomass of the macroalgal (e.g., *Enteromorpha* and *Ulva*) component will increase, rather than phytoplankton. It is important to recognize these situations (i.e., lagoons; Nobre et al., 2005) and, as an integrative approach, to consider including macroalgal biomass as a complementary primary eutrophication indicator just as other indicators (e.g., dissolved oxygen and losses of seagrasses) are included in indices of overall eutrophic condition and are a required variable within many mandated monitoring programs (e.g., WFD, Oslo Paris Convention for the Protection of the North Sea (OSPAR), and EPA NCA) in order to track the more severe problems associated with nutrient enrichment.

The intent here is to examine phytoplankton indicators to see how well they reflect the water quality of estuarine and coastal/transitional systems, at the same time understanding that they typically are combined with other indicators to give a full picture of eutrophication.

1.08.2.1.2 Phytoplankton indices

Chl-a concentrations and reference conditions. The simplest of the phytoplankton indicators uses only the concentration of Chl *a* as a measure of phytoplankton biomass. Although most of the methods listed in Table 3 include Chl-*a* concentration, there are different ways of determining the status of Chl *a* dependent upon the time frame and spatial scales of sampling, the statistical measure used to determine the representative concentration (e.g., mean annual, index period mean, and/or maximum), and the reference concentration and scale that determines the final status.

Although all the Chl-*a* indices are included in a multiparameter index, TRIX is the only one for which the Chl-*a* indicator cannot stand alone because it is integrated with three other variables that make up the index (Table 3).

The EPA NCA uses comparison of samples from an annual index period (June – October) to reference conditions determined from national studies to determine the rating of poor, fair, or good (where poor >20 µg l⁻¹, fair 5–20 µg l⁻¹, and good 0–5 µg l⁻¹). The samples are taken one time per year based on a random statistical design and provide 90% confidence in the rating for a region (USEPA, 2001a). Because of the sampling design, the EPA NCA cannot make determinations for individual water bodies.

The TWQI/LWQI method uses nonlinear functions to transform annual average Chl-*a* concentrations from sites representative of the system into a quality value (0 = worst, 100 = best) which is then multiplied by a weighting factor (here, 15% of total water quality is attributed to Chl *a*) that accounts for the relative contribution to the overall index (Giordani et al., 2009). The Chl-*a* quality value scores range from optimal conditions of 6 µg l⁻¹, for a quality value of 100, to a low quality value of 0 at a concentration of 30 µg l⁻¹ or greater. These ranges are consistent with conditions in Mediterranean lagoons and continental estuaries (Giordani et al., 2009).

The Helsinki Convention (HELCOM) uses HEAT (HELCOM Eutrophication Assessment Tool) as mean summertime concentrations of samples that are spatially representative of a water body combined with reference values that are basin specific and are determined from historical data, empirical modeling, or state-of-the-art hydrodynamic or ecological modeling for pristine conditions. The boundary between good and moderate status is the reference concentration +50%, which is equal to an ecological quality ratio (EQR, *sensu* the WFD; see Borja, 2005) of 0.67. The threshold between good and moderate EQR is used to determine the extent of deviation of the sample EQR from the reference EQR and from good status (Andersen and Laamanen, 2009).

ASSETS uses the 90th percentile of annual values for Chl *a* combined with the spatial coverage of high values and the frequency of occurrence of blooms to determine the Chl-*a* condition within each salinity zone (tidal fresh 0–0.5 ppt, mixing zone 0.5–25 ppt, and seawater zone >25) within a system. The ratings for each zone are combined as an area weighted sum to determine the final Chl-*a* rating for the system (high, moderate, and low; Bricker et al., 1999, 2003, 2007).

The Chl-*a* assessment under WFD guidance and the OSPAR Comprehensive Procedure (OSPAR COMPP) are similar in that they both use mean summertime/growing season concentrations for samples that are spatially representative of the water body, and the maximum summertime Chl-*a* concentration is also used (OSPAR, 2002; ECOSTAT, 2003). In some cases, under the WFD guidance, the 90th percentile of annual Chl-*a* measured values are also used (i.e., Revilla et al., 2009). Both the WFD guidance and OSPAR COMPP require development of reference conditions for comparison to measured values from reference sites or historical data. The WFD assessment results in ratings for high, good, moderate, poor, and bad for which thresholds between categories were developed during WFD intercalibration exercises and reflect the location of the assessment (i.e., Basque coast in the Revilla et al., 2009 study).

Table 3 Methods to evaluate the status of phytoplankton in coastal and estuarine water bodies

Method/ approach	Country/ region using method	Biomass			Other characteristics	Community composition	Abundance	Indicators in overall eutrophication index
		Chl <i>a</i> thresholds and ranges (ug/l)	Sample timeframe	Statistical measure				
EPA NCA ¹	USA	Poor > 20 Fair 5–20 Good 0–5 ----- Lower for sensitive systems	Index period (June–Oct)	Concentration, % of coastal area in poor, fair and good condition based on probabilistic sampling design for 90% conf in areal result		No		Chl <i>a</i> , water clarity, DO, DIP, DIN
TRIX ²	EU	No thresholds, integrated with other index variables		Concentration		No		Chl <i>a</i> , DO, DIN, TP
TWQI /LWQI ³	EU	Good QV100 = 6 Bad QV0 = 30	Annual	Chl concentration mean annual or seasonal modified by weighting factor		No		Chl <i>a</i> , seagrasses, macroalgae, DO, DIN, DIP
HEAT ⁴	Baltic	Deviation from ref EQR <6.7 No dev from ref EQR >0.67	Summer (June–Sept)	Mean summer concentration	Increases in concentration, frequency and duration	Indicator spp.	X	Chl <i>a</i> , phytoplankton, nutrients, water transparency, SAV, DO, benthic invertebrates, summertime bloom intensity index
ASSETS ⁵ (eutrophic condition component only)	USA, EU, Asia, Australia	High >20 Mod 5–20 Low 0–5 ----- Lower for sensitive systems	Annual	90th percentile Chl concentration of annual data	Spatial coverage, frequency occurrence	Nuisance and toxic bloom occurrence, frequency, duration		Chl <i>a</i> , macroalgae, DO, seagrasses, nuisance/toxic blooms

(Continued)

Table 3 (Continued)

Method/ approach	Country/ region using method	Biomass			Other characteristics	Community composition	Abundance	Indicators in overall eutrophication index
		Chl a thresholds and ranges (ug/l)	Sample timeframe	Statistical measure				
WFD ⁶	EU	Cantabrian coast: Bad >14 Poor 10.5–14 Moderate 7–10.5 Good 3.5–7 High 0–3.5	Summer	Summer Chl concentration mean, max and sometimes 90th percentile annual data	Increases in concentration, frequency and duration	Indicator spp.	X	Chl a, phytoplankton, macroalgae, microphytobenthos, seagrasses, DO, nutrients, algal toxins
OSPAR COMPP ⁷	EU	Good QV100 = 6 Bad QV0 = 30	Growing season	Growing season Chl concentration mean, max	Increases in concentration, frequency and duration	Indicator spp.	X	Chl a, phytoplankton, macroalgae, microphytobenthos, seagrasses, DO, nutrients, algal toxins
IFREMER ⁸ (lagoons)	France	>30 Red 10–30 Orange 7–10 Yellow 5–7 Green 0–5 Blue	Annual	Mean annual Chl concentration		Phytoplankton abundance of <2 um, >2 um	X	Chl a, Phytoplankton counts (<2, >2 um), macrophytes (biomass, diversity), macrobenthos (richness, diversity), water (DO, Chl, Chl/phaeo, turbidity, SRP, TP, TN, NO ₂ , NO ₃ , NH ₄), sediment (OM, TN, TP)

¹USEPA (2008b);²Vollenweider et al. (1998);³Giordani et al. (2009);⁴Andersen and Laamanen (2009);⁵Bricker et al. (2003, 2007);⁶Devlin et al. (in prep);⁷OSPAR (2002);⁸Souchu et al. (2000).

The reference conditions for OSPAR COMPP are developed from reference sites and final ratings are determined as: (1) problem area if the measured Chl *a* is greater than the reference condition +50%; (2) potential problem area if it is between the reference concentration and +50%; and (3) non-problem area if it is equal to or less than the reference site concentration.

The IFREMER method for lagoons uses mean annual or mean seasonal data compared to a fixed scale to determine the status for Chl *a* with final ratings being color coded to match the WFD scaling from best (blue) to worst (red; Souchu et al., 2000). It is interesting to note that the thresholds and ranges used by IFREMER, determined from historical studies such as Organization for Economic Cooperation and Development (OECD; Souchu et al., 2000), are roughly consistent with the scales reported for TWQI/LWQI, EPA NCA, and ASSETS (Table 3). It is also important to note that although the reference conditions for EPA NCA and ASSETS are determined from national studies rather than on a case-by-case basis, each of these methods has a different scaling for systems that are sensitive (i.e., Florida Bay).

Phytoplankton abundance and indicator species, duration and frequency of blooms. Most of the assessment methods recognize that high Chl-*a* concentrations are only one of the potential impacts of nutrient enrichment. To provide a complete picture of eutrophic conditions, other characteristics should be included such as changes in community composition to include more nuisance and toxic species that result from changes in nutrient ratios, and increased duration and frequency of blooms which result from increases in nutrient loads.

All but three of the methods from Table 3 include some measure of changes in community composition to potentially harmful or toxic species which usually includes changes in frequency and duration of blooms. The methods HEAT and OSPAR COMPP monitor for toxic bloom species looking at changes in specific groups (i.e., dinoflagellates, diatoms, and cyanobacteria). The ASSETS nuisance and toxic bloom index uses a combination of observations of nuisance and toxic blooms and the frequency and duration of the blooms to determine the status. The WFD approach, used by Revilla et al. (2009), and the IFREMER method use abundance of phytoplankton larger and smaller than 2 µm as an indicator of the potential presence of toxic bloom organisms.

1.08.2.1.3 How well do these phytoplankton indices assess eutrophication impacts?

Examination of the different phytoplankton indices is enlightening. In cases where thresholds denoting the boundaries indicating small or large impacts are named, there is rough consistency among the named thresholds. As these measures have been determined independently, this suggests that there is consistency among water bodies globally in the response to nutrient loads as well as a global understanding of undesirable levels of Chl *a*. In the case of the EPA NCA and ASSETS, there is recognition that some systems, such as Florida Bay (a sensitive carbonate-based system), require thresholds and ranges that are lower but overall there is general agreement on the thresholds of Chl-*a* concentrations that indicate high-level impacts. It is also interesting to note that the concentration ranges appear

to be useful in both lagoons and estuaries (Giordani et al., 2009).

Most methods are attentive to the spatial scales of sampling to assure results that are representative of the system. This is important as typically one part of the system is more heavily impacted than others and thus the methods must include some way to recognize the patchy spatial nature of these impacts. Most of the methods use an average of samples from different sites, whereas the EPA NCA looks at the ratio of stations above a threshold and can estimate the area of the region that is impaired given the statistical sampling design. The time frames of sampling vary from annual to summertime/growing season (Table 3). While it is typically during the growing season that extreme Chl-*a* concentrations are observed, in some cases, the restriction to an index period may under- or overestimate concentrations due to the variability in bloom timing as a result of variable climate and freshwater inflow.

The other most common phytoplankton indicator is the observation of changes in community reflected in the increase in nuisance and toxic species, including increases in bloom frequency and duration. In some cases, this is done on a species-by-species basis using indicator species as the measure of change, such as the method used by HEAT. In other cases, the relative abundance of size categories is used with the smaller forms indicative of possible toxic forms (e.g., WFD and IFREMER). This has been used for the assessment of the ecological status of phytoplankton in the Basque coast under the guidance of the WFD. In this case, it is important to note that using different definitions of the groups and thresholds that are included in the size analysis resulted in different final assessments of conditions (Revilla et al., 2009). The different approaches also have different requirements for analyses, that is, greater taxonomic expertise as well as different criteria for monitoring. For sustainability of a monitoring and assessment program, these differences must be taken into consideration.

A measure of the success of the phytoplankton indicator is whether it accurately reflects the extent and significance of nutrient-related impacts for the system where it is applied. Here we see that these methods are able to capture changes in biomass and species composition and, as such, are successful in noting nutrient-related/induced changes. However, changes in phytoplankton are an indicator only of the start of nutrient-related problems and do not highlight more significant nutrient-related issues. As such, it is recommended that the phytoplankton indicators be used in conjunction with additional indicators of nutrient enrichment so that a complete picture of nutrient-related impairments can be constructed.

1.08.2.2 Macroalgae

Methods for assessing macroalgae are not as well developed as assessment methods for other biological elements, that is, phytoplankton. Despite this, macroalgae in estuarine and coastal marine waters have been used extensively as indicators of marine quality. This is because macroalgae, together with seagrasses, are

1. an important resource, as extensive primary producers in estuaries;
2. a food source for waterfowl;
3. a habitat and nursery area for commercially and recreationally important fish species;

4. a protection against erosion; and
5. a buffering mechanism for excessive nutrient loadings (Gibson et al., 2000).

As photosynthetic sessile organisms, they respond directly to abiotic and biotic environmental factors, thus representing sensitive bioindicators of natural and anthropogenic changes (Orfanidis et al., 2001, 2003). However, due to this double response, they do not stand alone as an indicator of ecosystem condition, and additional parameters (e.g., water column nutrient concentrations and light penetration) are required to interpret macroalgae data (Gibson et al., 2000).

Despite this, methods based upon macroalgae in assessing environmental pollution are more developed in coastal areas than in estuarine habitats (Wilkinson and Rendall, 1985). Probably this is because of the complex natural gradients within estuaries, exemplified by the estuarine quality paradox explained earlier. Hence, Wilkinson et al. (1995, 2007) identified three algal zones (A, B, and C), within British estuaries. Zone A is the sheltered open coast at the mouth of the estuary and the lowermost part of the estuary, with 50–100 seaweed species; zone B contains the lower estuarine flora (fucoid-dominated), in the more saline part of the estuary, with richness reduced to 10–40 species; and zone C contains the upper estuarine flora, with mat-forming opportunists, and fucoids completely absent, with only 0–10 species.

In this section, methods including macroalgae together with other biological quality elements (i.e., phytoplankton and benthos) have not been considered. Some examples of this approach are the TWQI (Giordani et al., 2009), ASSETS (Bricker et al., 2003, 2007), and the Chesapeake Bay health index (Williams et al., 2009). Methods using macroalgae alone include different metrics or indicators in the quality assessment approach (Table 4). Hence, most of the methods include some measurement of richness (even as the presence/absence) and abundance (generally as percentage of cover, and also as biomass). Several methods use the ecological or functional groups (see Orfanidis et al., 2001) or the presence of indicator species (opportunistic or sensitive) as a way of detecting disturbances in the studied area. Very few methods use other metrics, such as the algae penetration into the estuary (Wilkinson et al., 2007), the depth range (Selig et al., 2007), or the Rhodophyceae/Chlorophyceae ratio (Sfriso et al., 2007) (Table 4).

All of these methods have been developed after the year 2000, most of them being proposed within the European WFD. Following Borja and Dauer (2008), any index developed for

quality assessment should be validated, including (1) testing of the index using an independent data set, different from the index development data set (calibration data set); (2) setting *a priori* correct classification criteria; and/or (3) presentation of a strong *a posteriori* justification for use based upon best professional judgment. However, very few of the above-mentioned methods have been validated or used out of the countries using the method. Hence, the ecological evaluation index (EEI) has been used in other countries of the Mediterranean (as an example, see Orlando-Bonaca et al., 2008 in Slovenia); the opportunistic macroalgal blooms approach has been used in Portugal (Patricio et al., 2007); and the rapid quality index (R-MaQI) has been validated in Venice Lagoon (Sfriso et al., 2009). Probably this lack of validation, due to the incipient use in estuarine quality assessment, is one of the most important weaknesses of such indices.

All these methods link eutrophication pressure and macroalgae response, because responses to other anthropogenic pressures, such as hydromorphological changes within the estuary, harmful substances discharge, etc., are less evident (but, see Melville and Pulkownik, 2006). Krause-Jensen et al. (2008) studied the relationships linking distribution and abundance of marine vegetation to eutrophication, by collating 73 relationships originating from 38 publications from the period 1982 to 2007 and covering a wide range of ecosystems (both coastal and estuarine). Of the 73 relationships, 38 link vegetation responses significantly to eutrophication pressure as expressed by nutrient richness or water transparency, 18 link the responses to combinations of eutrophication pressure and ecosystem characteristics, and nine link the responses to ecosystem characteristics alone. The compilation demonstrates that macroalgae generally respond quantitatively to changes in eutrophication pressure by growing deeper, being more abundant, and more widely distributed in clear waters with low nutrient concentration as compared with more turbid and nutrient-rich ecosystems.

1.08.2.3 Angiosperms

Seagrasses are marine angiosperms that form meadows in near-shore brackish or marine waters, in temperate and tropical regions. The four European seagrass species grow from the intertidal (*Zostera noltii*) and down to 5–15 m depth in North European waters (*Zostera marina*), but seagrasses may be found even deeper than 50 m in clear Mediterranean waters (*Cymodocea nodosa* and *Posidonia oceanica*). In the

Table 4 Methods to evaluate the status of macroalgae in estuarine water bodies

Method/approach	Country using method	Indicators/metrics	Reference
Macrophyte assessment	USA	Richness; cover (%); density, biomass	Gibson et al. (2000)
Ecological evaluation index (EEI)	Greece	Mean abundance (%); two ecological (functional) groups	Orfanidis et al. (2001, 2003)
Macroalgae assessment	Spain	Richness; presence of pollution indicator species; cover (%); ratio between green algae and the rest	Borja et al. (2004a)
Fucoid presence	United Kingdom	Presence/absence; algae penetration into the estuary	Wilkinson et al. (2007)
Opportunistic macroalgal blooms	United Kingdom; Ireland	Biomass; cover (%); opportunistic	Scanlan et al. (2007)
Total and opportunistic algal cover	Denmark	Macroalgal cover (%); opportunistic fraction	Krause-Jensen et al. (2007)
Depth distribution	Germany	Presence; depth range (m); cover (%)	Selig et al. (2007)
Rapid quality index (R-MaQI)	Italy	Presence/absence; Rhodophyceae/Chlorophyceae ratio	Sfriso et al. (2007)

The proposal from Selig et al. (2007) is for Baltic coasts; but has been used in habitats similar to estuarine areas.

Mediterranean Sea, *P. oceanica* beds cover between 25 000 and 50 000 km² of the coastal areas corresponding to 25% of the sea bottom at depths between 0 and 40 m. Additional details on seagrasses from North and South America can be found in Short et al. (2006) and SeagrassNet website.

As a function of complex biogeochemical processes (Harris, 1999; Hansen et al., 2000; Eyre and Ferguson, 2002), seagrass meadows are extremely productive systems, which exhibit high biodiversity, and support complex food webs, constituting a habitat refuge for a number of organisms (Orth, 1992; Boström and Bonsdorff, 2000; Borum et al., 2005). Changes in seagrass areas are often a symptom of major changes in environmental characteristics, and therefore constitute an important indicator for assessing the state of the environment.

Seagrasses respond to natural variations in light availability, nutrient and trace element availability (Duarte, 1995), grazing pressure, marine pests and pathogens (Giesen et al., 1990), weather patterns, and episodic floods and cyclones. Due to their high minimum light requirements, the most widespread and pervasive cause of seagrass decline is a reduction in available light, which may be due to turbidity events during floods, enhanced suspended sediment loads, and elevated nutrient concentrations (Boström et al., 2002). Besides, phytoplankton and fast-growing macroalgae are better competitors for light than benthic plants, and their biomass can shade seagrasses during progressive eutrophication (Fourqurean and Zieman, 1991; Frost-Christensen and Sand Jensen, 1992; Walker and McComb, 1992; Dennison et al., 1993; Terrados et al., 1999). Likewise, seagrass species can suffer from toxic effects of trace metal contamination (Prange and Dennison, 2000) or, in other cases, bioaccumulate it, which can have ramifications for grazers. Seagrasses can also change in response to physical disturbances (e.g., ports and marinas, temperature raise, and fishing pressure represented by commercial and recreational harvesting of fish and shellfish) which may cause direct damage to plants. Finally, removal of forage or predator species can also have detrimental effects that cascade through food chains.

The causes for changes in seagrass areas may therefore be natural or anthropogenic, and it is often difficult to differentiate what changes are attributable to human activities. Nevertheless, marine angiosperms have been showing to be robust ecological indicators of water and sediment quality, and because of their susceptibility to human disturbances were included as one of the biological quality elements to take into account in terms of the implementation of the European WFD.

To establish reference conditions for marine angiosperms is rather difficult because meadows are extremely variable in terms of extension, abundance, and species composition. Moreover, these characteristics depend on the geographic location and the hydrodynamic regime, and therefore reference conditions must be defined taking into account the typology of the habitat where the meadow occurs (Foden and Brazier, 2007). To accomplish these complex tasks, three methods can be used:

1. *Historical data.* When existing, these are normally relatively easy to obtain, although they may be heterogeneous and more often not corresponding to a metrics. Data quality may therefore be poor or even unknown. Of course, in many cases, such data do not exist.
2. *Expert judgment.* This is usually the way to integrate and interpret historical data, the main inconvenience being the expert subjectiveness.
3. *Modeling.* This is region specific and requires large data sets for calibration and validation.

Although all these approaches have some limitations, their common idea is that a reference point cannot be a disturbed point (Gerritsen, 1995; Moreno et al., 2001; Reynoldson et al., 2001), and therefore the determination of such points must be done based on the prevalence of high ecological quality parameters. The metrics required are usually: (1) abundance, expressed as density in g m⁻² of dry weight, possibly distinguishing between rooted and aerial parts of the plants; (2) cover, expressed as the area in square meter occupied by the meadow; and (3) taxonomic composition.

Aerial photography, satellite imagery, and systematic towed video surveys can be used to map the extent of seagrass coverage in some coastal waterways. Besides, there have been significant advances in the determination of seagrass properties other than coverage (e.g., species composition and biomass) from satellite imagery, although local ground confirmation of the taxonomy to the genus level is advised. Nevertheless, high levels of turbidity may constitute a constraint to the application of this methodology in tide-dominated coastal waters (e.g., deltas, estuaries, and tidal creeks; Larkum et al., 2007).

Only recently, seagrasses have been used as quality indicators under accepted protocols. Hence, established in 2001, SeagrassNet is a monitoring program for seagrasses worldwide, which uses a standardized protocol for detecting change in seagrass habitat to capture both seagrass parameters and environmental variables (Short et al., 2006). This program is designed to statistically detect change over a relatively short time frame (1–2 years) through quarterly monitoring of permanent plots. Currently, SeagrassNet operates in 18 countries at 48 sites; at each site, a permanent transect is established and a team of people from the area collects data, which are sent to the SeagrassNet database for analysis (Short et al., 2006). Moreover, after the publication of the WFD, plenty of methods have been published in Europe for angiosperm quality assessment. Some of them focus on coastal species (Romero et al., 2007; Montefalcone, 2009), but many others have been implemented for transitional and low-salinity coastal waters (Krause-Jensen et al., 2005; Best et al., 2007b; Cabaco et al., 2007; Foden and de Jong, 2007; Selig et al., 2007; García et al., 2009).

1.08.2.4 Macroinvertebrates

The estuaries (and some types of lagoons) are transitional systems between freshwater and marine systems where the species are affected by natural and anthropogenic stressors. The main stressor is salinity, which organized the species distribution into three main groups: (1) in the upper part of the estuaries, freshwater species colonized this area affected only by the dynamic tidal range; (2) conversely, in the lower part of the estuaries, that is, the polyhaline zone, marine species occurred as in the more marine salted zone; and (3) between these two extremes, a specific estuarine group of species is considered as adapted to life in variable salinity and in a salinity range intermediate between the upper and lower part of the estuaries. This principle was mainly expressed by the Remane's diagram

(Remane, 1934). In addition, Attrill and Rundle (2002) introduced the concept of ecocline (gradient of change) for estuarine ecosystems. They suggested that the estuary represents a two-ecocline model, with a first ecocline from sea to mid-estuary where there is a salinity decrease overlapping with an ecocline from river to mid-estuary where salinity increases. Hence, there is a double gradient of decreasing species from the river and the marine part to the estuaries. In the mesohaline stretch, but mainly in the oligohaline zones, only the marine-derived species live at the edge of their range along the sea-estuary ecocline and freshwater-derived species at the limit of the river-estuary ecocline. Such organisms were considered the 'true estuarine organisms' represented in the Remane's diagram (Figure 2), but if the estuary is seen as a two-ecocline system, this category of organisms disappears to be replaced by two gradients, running from river or sea into the mid-estuary (Figure 2) (Attrill and Rundle, 2002). The oligohaline zone appears as the poorest, whereas the mesohaline shows intermediate values; the polyhaline zone shows generally high species richness more or less impoverished from the marine surrounding zone (de Paz et al., 2008b). The tidal freshwater zone appears also more or less impoverished in comparison to the tributary rivers or from the upper part of the estuary in the typical freshwater part of the river. The biodiversity changes in relation to the size of the estuary, the largest being the richest and the smallest the poorest, but also in relation to the number of other natural biotope variety, that is, diverse type of substratum, tidal range, depth range from the intertidal zone to the deep subtidal zone in the channels, and hydrodynamism. The variety of biotopes creates, in turn, a variety of ecological conditions which overlap those of single salinity; typically, estuaries show a mosaic of different biotopes (Escaravage et al., 2004).

There is a high level of spatial and temporal variability among observations of macroinvertebrates; however, the estimation of species richness depends on the number of observations and the level of taxonomic identification. For example, among the oligochaetes, most of researchers do not identify them to species level, but richness can reach more than 20 species in anthropogenically modified estuaries, such as the Seine estuary (Dauvin and Ruellet, 2009). An inventory of all aquatic invertebrates of the estuary and the region of freshwater

influence, within the eastern part of the Bay of Seine, was performed to establish the biodiversity pattern (Ruellet and Dauvin, 2008). Such an inventory shows that the region is highly diversified: 1485 taxa of aquatic invertebrates were encountered, including 77 genera, five families, and one subclass. The compiled data show that the distribution of the invertebrate species in the Seine River estuary follows a two-ecocline model, as proposed by Attrill and Rundle (2002).

In northern European waters, two main benthic communities occur in soft bottom in the marine part of the estuaries: (1) an *Abra alba* muddy fine sand community, which impoverished from the marine pole to the mesohaline zone, and (2) a *Macoma balthica* mud community, common on the tidal flat in the mesohaline and oligohaline part, and also at subtidal depth. Common species, such as the polychaete *Hediste diversicolor*, the bivalve *Macoma balthica*, the amphipod *Corophium* spp. and spionid polychaete such as *Pygospio elegans* and *Streblospio* spp., and cirratulid such as *Aphelocheata* spp., showing a very large latitudinal distribution within the Northern Hemisphere, are dominant in this community. In turn, within southern European estuaries, there is a shift in the composition of the species, with those typical of colder water disappearing and those related to warm waters becoming more prevalent (see Borja et al. (2004a), for the composition of southern estuarine species).

Invasive species is a common problem in estuaries worldwide, and European estuarine communities have also been colonized by numerous introduced species, some of which show proliferation, such as the bivalve *Dreissena* spp., *Corbicula* spp., the shrimp *Palaemon macrodactylatus*, and the bristle worm *Hypania invalida*, and, in the upper estuary, *Crangonyx pseudogracilis* and *Dikerogammarus villosus*. In general, community vulnerability to invasions is ascribed to combinations of several factors such as the presence of vacant niches, habitat modification, and disturbances before and during invasions. Although the link between the biodiversity of communities and their vulnerability to invasions remains to be proved, invasibility is known to increase if a community lacks certain species, which ought to be present under normal conditions. A new hypothesis linking the various explanations of increased invasibility is that of fluctuating resource

Figure 2 Remane's diagram redrawn following the two-ecocline model. The estuarine species are removed; the number of freshwater species is lower than that of marine species. From Attrill, M.J., Rundle, S.D., 2002. Ecotone or ecocline: ecological boundaries in estuaries. *Estuarine, Coastal and Shelf Science* 55, 929–936.

availability such as an increased amount of unused resources (Davis et al., 2000).

The estuarine communities are also characterized by high contrast between zones with poor macroinvertebrate abundance and therefore biomass and production and zones with high abundance, biomass, and secondary production (Dauvin and Desroy, 2005). Heip et al. (1995) underlined the positive relationship between pelagic primary production and macrobenthic biomass in estuaries. In general, benthic production represents a high source of food for fish and birds, and is among the more productive zone for the aquatic ecosystem. The conservation and restoration of tidal flats, due to their high importance in terms of estuarine functionality, are often underlined in estuarine management.

The implementation of the European WFD has provoked a large debate on the use of benthic bio-indicators and indices to determine the quality of the estuarine (transitional) and coastal waters in terms of the establishment of ecological quality status (Borja, 2005; Borja et al., 2009c). Nevertheless, assessing estuarine quality by macroinvertebrates remains difficult due to the high variability of natural conditions in such ecosystems; moreover, estuaries are highly modified in many countries. There are numerous definitions of indicators but, essentially, they are quantitative or qualitative parameters, attributes or variables which characterize the environmental status and/or the pressures which may affect that status; they are selected according to their ability to characterize the overall state of an ecosystem, thus simplifying an extremely diverse range of parameters to a small group of indicators (Aubry and Elliott, 2006). Moreover, indicators could provide a valuable means of communication to stakeholders and policymakers (Aubry and Elliott, 2006). Furthermore, ecological indices are used as quantitative tools in simplifying, through discrete and rigorous methodologies, the attributes and weights of multiple indicators with the intention of providing broader indication of a resource, or the resource attributes, being assessed (Pinto et al., 2009). A clear distinction between indices and indicators must be made. Any measure that allows the assessment and evaluation of a system's status (descriptive indicators, environmental quality indicators, and performance indicators), as well as assessment of any management actions for conservation and preservation that occur in the ecosystem, is considered an indicator; indices are considered one possible measure of a system's status (Dauvin, 2007).

The advantages of using macroinvertebrates to assess ecological quality are multiple: (1) these organisms are relatively sedentary, meaning that they cannot avoid deteriorating water/sediment quality conditions; (2) they have relatively long life spans; (3) they comprise diverse species that exhibit different tolerances to stress; and (4) they play an important role in cycling nutrients and materials between the underlying sediments and the overlying water column (Dauvin, 2007). Still, Rakocinski and Zapfe (2005) have underlined several disadvantages of the existing benthic indices:

1. they represent a static expression of an ecological condition;
2. they are not explicitly linked to changes in ecological function;
3. they may not be specific with respect to different kinds of stressors;

4. they are subject to underlying taxonomic changes across estuarine gradients;
5. their use can be labor intensive; and
6. they are not applied consistently across biogeographic provinces.

Univariate indices were largely developed for marine or coastal ecosystems; however, new functional and multimetric indices are being developed for macroinvertebrates (see Díaz et al., 2004; Borja and Dauer, 2008; ICES, 2008; Pinto et al., 2009). The benthic indicators and indices can be classified into four categories: (1) based on diversity, (2) based on ecological groups, (3) based on trophic groups, and (4) indices synthesizing several other indicators. Most of the recently developed indices in the second category are based on dividing soft benthic species into previously defined ecological groups and then determining the respective proportion of the different groups in the benthic communities (Borja et al., 2000). They provide information about the relative abundances of the sensitive species faced with increasing organic matter in the sediment and those of the species that are resistant or indifferent to such increases, or even favored by such conditions (e.g., the opportunistic species that proliferate when the sediment is rich in organic matter; on the contrary, sensitive species disappear). However, the main problem is that most of the indices, which aim to determine anthropogenic stress, relate to abundances of stress-tolerant species, which may also be tolerant of natural stressors such as in estuaries (see Section 1.08.1.2). Similarly, many indices described relate to anthropogenically organic-rich systems, whereas estuaries are naturally organic-rich systems.

Weisberg et al. (1997) developed a multimetric benthic index of biotic integrity (B-IBI) (for acronyms within this section, see Table 5) which was based on 17 candidate measures. These included primary and derived community indices (species richness, abundance, diversity, etc.) as well as the percentage of abundance of different functional groups. By accommodating differences in salinity and substratum, comparing test and reference areas, and by using a rank-scoring system for the deviation by different metrics to reference conditions, Weisberg et al. (1997) were able to separate stressed benthic areas from reference conditions.

Other multicriteria methods have been developed in European waters, those based upon AMBI and BQI (see Borja et al., 2009c) being the most successful. As an example, Quintino et al. (2006) analyzed data from the Portuguese coasts to produce univariate indices used for the WFD: abundance, species richness, biomass, Margalef index, Pielou evenness index, Shannon-Wiener index, Simpson's Index, AMBI and its reciprocal (1/AMBI), EQR (calculated according to the UK multimetric approach), BQI, A/S, and B/A. They found that some of the indices gave an underrepresentation and others an overrepresentation of the ecological status. They cautioned that this was not merely of academic interest as misclassifying a good status area as being of moderate status could result in a large expense to make unnecessary remedial work (Gray and Elliott, 2009). It was particularly notable that many of the indices needed refining to cope with the naturally low diverse areas in estuaries and other transitional areas. Hence, when these indices were applied to highly stressed natural estuarine or lagoonal environments, the low species

Table 5 Indices for assessing environmental quality based on the structure of macroinvertebrates in transitional waters

Univariate indices

Descriptors

Number of species (species richness), abundance (A), biomass (B)

Indices of diversity

- Shannon–Wiener diversity index (H') (Shannon and Weaver, 1949)
- Simpson's indices of dominance, diversity and evenness (Simpson, 1949)
- Brillouin indices of diversity and evenness (Brillouin, 1956)
- Pielou evenness index (J') (Pielou, 1966)
- Margalef's index (Margalef, 1968)
- Hurlbert index (Hurlbert, 1971)
- Hill's diversity numbers and evenness measures (Hill, 1973)
- BPI: Benthic pollution index (Leppäkoski, 1975).
- Taxonomic diversity index and Taxonomic distinctness (Warwick and Clarke, 1995)

Graphical methods

- RFD: Rank-frequency diagram (Frontier, 1977)
- K-dominance curves (Lamshead et al., 1983)
- ABC curves (Warwick and Clarke, 1994).

Ecological groups

- Indice Annélidien de Pollution (Bellan, 1980)
- Biotic index (Hily, 1984)
- MMI: Macrofauna monitoring index (Roberts et al., 1998)
- AMBI: AZTI's marine biotic index (Borja et al., 2000)
- BENTIX (Simboura and Zenetos, 2002)
- ISI: Indicator species index (Rygg, 2002)
- IE2C: Indice Biotique et Indice d'Evaluation de l'Endofaune Côtière (Grall and Glémarec, 2003).
- BOPA: Benthic opportunistic polychaetes/amphipods ratio (Dauvin and Ruellet, 2007)
- BITS: Benthic index based on taxonomic sufficiency (Mistri and Munari, 2008)
- Benthic opportunistic annelids/amphipods ratio (Dauvin and Ruellet, 2009)

Functional indices

- ITI: Infaunal trophic index (Word, 1979)
- EQI: Ecofunctional quality index (Fano et al., 2003)

Multimetric indices

- Pollution coefficient (Satsmadjis, 1982)
- BQI: Biological quality index (Jeffrey et al., 1985)
- Organism sediment index (Rhoads and Germano, 1986)
- RTR: Infauna ratio-to-reference of sediment Quality Triad (Chapman et al., 1987)
- BIEC: Benthic index of estuarine condition (Weisberg et al., 1993)
- B-IBI: Benthic index of biotic integrity (Ranasinghe et al., 1994)
- BCI: Benthic condition index (Engle et al., 1994)
- BHQ: Benthic habitat quality (Nilsson and Rosenberg, 1997)
- VPBI: Virginia province benthic index (Paul et al., 2001)
- NQI: Norwegian quality index (Rygg, 2002)
- IEI: Index of environmental integrity (Paul, 2003)
- BQI: Benthic quality index (Rosenberg et al., 2004)
- IQI: Infaunal quality index (Prior et al., 2004)
- INES: Fuzzy index of environmental integrity for transitional environments (Mistri et al., 2005)
- MarBIT: Marine biotic index tool (Meyer et al., 2006)
- DKI: Danske Kvalitet Indeks (Borja et al., 2007)
- BEQI: Benthic ecosystem quality index (Van Hoey et al., 2007)
- BBI: Brackish water benthic index (Perus et al., 2007)
- DAPHNE (Forni and Occhipinti-Ambrogi, 2007)
- FINE: Fuzzy index of ecosystem integrity (Munari and Mistri, 2008)
- MISS: Macrobenthic index for semi-sheltered systems (Lavesque et al., 2009)

Multivariate approaches, packages and models

- PLI: Pollution load index (Jeffrey et al., 1985)
- BRI: Benthic response index (Smith et al., 2001)
- M-AMBI: Multivariate-AMBI (Borja et al., 2004a; Muxika et al., 2007)
- PRC: Principal response curves (Pardal et al., 2004)
- TICOR: Typology and reference conditions for portuguese and coastal waters (Bettencourt et al., 2004)
- Combination of indices: B-IBI & TICOR (Chainho et al., 2008)
- APBI: Acadian province benthic index (Hale and Helshe, 2008)
- P-BAT: Portuguese-benthic assessment tool (Pinto et al., 2009)

richness and dominance of a few tolerant species in these systems presented a challenge to the application of the biotic indices (Puente and Díaz, 2008). Similarly, based on the B-IBI, a multimetric approach was developed for Arcachon Bay, an Atlantic lagoon, in an attempt to correctly assess the benthic ecological status of an area that was physically perturbed by a deposit of dredged sediment. This approach, called macrobenthic index for sheltered systems (MISS), took into account the natural variability of a set of variables describing the biological integrity of the reference conditions. These are classified into three groups: (1) community structure (abundance, biomass, number of species, Shannon's diversity, and evenness); (2) trophic composition (grazer, selective deposit feeder, non-selective deposit feeder, suspension feeder, and carrion feeder); and (3) pollution indicators (AMBI, benthic opportunistic polychaetes/amphipods ratio (BOPA), W-statistic, and abundance of sensitive, and tolerant and opportunistic species) (Lavesque et al., 2009). The MISS approach clearly proved that no single biotic index can correctly assess the ecological status of a given benthic invertebrate community.

Table 5 shows an updated list of benthic indices found in the literature (from Díaz et al., 2004; Borja and Dauer, 2008; ICES, 2008; Pinto et al., 2009; Borja et al., 2009c; Ducrottoy et al., 2010). Hence, there is a plethora of univariate and multivariate indices and methods to assess the status or the integrity of estuaries using macroinvertebrates, sometimes associated with other environmental or biological variables. Most of them, as the indices of diversity, are ancient, whereas others have been published recently and often used in a limited number of cases. AMBI and multivariate AMBI (M-AMBI), BQI (and its various adaptations), B-IBI, and infaunal trophic index (ITI) are among the more universal methods used in several geographical zones, not only in European and American waters but also in other coastal and transitional zones around the world, that is, Mediterranean Sea, Indian Ocean, south- and north-American estuaries, Asia, etc. (see ICES, 2008). Some supplementary developments on macroinvertebrates take into account the species level as indicators. Ducrottoy et al. (2010) highlighted that indicator species can be considered to reflect the quality status of their habitat, whether they are present or absent from samples. There are many examples of such indicator species in the marine environment, particularly those living in sediments, that is, the polychaete *Capitella capitata* is often considered as a good indicator of organic enrichment. In this scope, dominant, keystone, sentinel, introduced, invasive species can be used as possible indicators. A specific group can be also used as the spionid polychaetes as environmental indicators in the Tampa Bay, Florida (Dix et al., 2005).

Nevertheless, a quantum of research can be continued to reexamine and adapt the different index thresholds for transitional waters (Borja et al., 2009c). Moreover, some additional research is needed in the coming years (see Borja et al., 2009e), including the need to take physical disturbances into account (e.g., dynamic forcing of the systems) and to favor multicriteria approaches, including the indices that are based on the structure and production of the communities, in the development of an operating report. Moreover, testing indices with different human pressures are being undertaken in some cases (Calabretta and Oviatt, 2008; Chainho et al., 2008; de Paz et al., 2008a; Josefson et al., 2008; Thrush et al., 2008; Borja et al., 2009d; de-la-Ossa-Carretero et al., 2009), but more tests are needed. Finally, when using indicators and indices, investigators should be as pragmatic as possible in making them environmentally sustainable, economically viable, technologically feasible, socially desirable/tolerable, legally permissible, and politically expedient.

1.08.2.5 Fishes

The survival and development of healthy estuarine fish communities require good environmental (physical, chemical, and biological) conditions (Marchand et al., 2002). Fishes therefore can provide a good indication of estuarine health or condition. The use of fishes includes assessments at various levels of biological organization – the subcellular level, cellular level, organ level, individual level, population level, and community level (Table 6).

In general, measures restricted to lower levels of biological organization, such as molecular, biochemical, cellular, or physiological changes that occur at the subcellular, cellular, and organ levels are referred to as 'biomarkers', whereas changes that occur at higher levels of biological organization (individual, population, and community) are more usually referred to as 'bioindicators' (Lam and Gray, 2003). Biomarkers typically are used in the detection of pollution or contaminants; they respond rapidly to environmental conditions and as such are effective early warning systems of potential problems before they appear at higher levels of biological organization.

1.08.2.5.1 Subcellular

In most monitoring programmes, an assessment of anthropogenic inputs of pollutants into estuaries usually includes chemical analyses of the water and sediments. However, the biological uptake (bioaccumulation) of these pollutants is probably of more interest than the absolute concentration in the water or sediments; this is particularly relevant if the species

Table 6 Levels of biological organization in fishes and examples of measures used at each level

Biological organization	Example
Subcellular	Bioaccumulation of contaminants (e.g., metals and PAHs)
Cellular	Enzyme activity, stress proteins, DNA integrity
Organ	Liver histopathology
Individual	Condition, disease, physical deformities, parasite load
Population	Abundance, age/size structure, indicator species
Community	Species diversity, species abundance/biomass, guild composition

in question is used for human consumption (Marchand et al., 2002). Chemical contaminants may enter fishes in several ways: either by direct uptake from the water (via the gills), consumption of contaminated sediment, or via the food chain by consumption of contaminated prey. Contaminants that cannot be excreted tend to accumulate in the animal's tissues. Analyses of fish tissues, therefore, can be used to directly measure the occurrence and levels of various chemical compounds such as metals and organic pollutants. Many marine juvenile fishes utilize estuaries as nursery areas and contamination by pollutants (metals, pesticides, etc.) may occur during this intensive feeding and growth period (Marchand et al., 2002). As many of these species are commercially exploited, their health, quality, and survival are of direct concern to man. The chemical analyses of pollutants in the tissues of fishes can provide an indication of actual or potential problems.

The accumulation of chemical pollutants and their bioaccumulation, however, may differ between organisms, the individual tissues of an organism, and between chemicals. A variety of factors can affect the biological uptake of pollutants and these include physiological factors (variations in reproduction and nutrition), environmental factors (salinity, temperature, and pH of the water, the presence of other chemicals/metals in solution, and the geochemistry of the local area), and the chemical nature of binding of the pollutant (Elliott et al., 2002; Marchand et al., 2002). Chemical pollutants, particularly heavy metals, may also be present in the tissues of nonpolluted organisms, and in order to assess the level of bioaccumulation of metals in organisms, baseline data are necessary on metal concentrations in individuals from clean areas (Marchand et al., 2002).

1.08.2.5.2 Cellular

Although the analysis of fish tissues can directly measure the bioaccumulation of various pollutants (or xenobiotics), exposure to such compounds may trigger certain biochemical responses which serve either to metabolize the chemical, or to store it, thereby preventing interference with essential biochemical reactions within the cell (Elliott et al., 2002). Some chemicals bind proteins or enzymes that are concerned with their metabolism and biotransformation (Lam and Gray, 2003). The measurement of these enzymes or intermediates can signal the presence of certain chemicals or even toxic effects. Many of these enzymes are also specific to certain classes of compounds such as the induction of metallothionein synthesis by exposure to heavy metals (mercury, cadmium, lead, zinc, etc.). The mixed-function oxygenases (MFOs) are involved in the biotransformation and elimination of chlorinated and aromatic hydrocarbons such as polychlorinated biphenyls (PCBs), dioxins, and polycyclic aromatic hydrocarbons (PAHs) (Vasseur and Cossu-Leguille, 2003). Ethoxyresoufin O-deethylase (EROD) enzyme activity is widely used to detect exposure to PCBs and PAHs. The inhibition of acetylcholinesterase (AChE) enzyme activity also represents a specific marker of exposure to agricultural pesticides such as organophosphate and carbamate insecticides (Corsi et al., 2003; Lam and Gray, 2003; Vasseur and Cossu-Leguille, 2003). Other cellular-level biomarkers may also include parts of cells confined to certain tissues, or nucleic acids or specific regions of protein within nerve synapses or cell membranes

(Lam and Gray, 2003). Lipid peroxidation, a process resulting in the degradation of cell membranes, can be observed when antioxidant and detoxifying systems are deficient, whereas permanent changes in DNA structure are biomarkers of exposure to mutagens or carcinogens (Vasseur and Cossu-Leguille, 2003). The measurement of stress proteins can also be used to provide a general indication of overall stress within an organism (Lam and Gray, 2003).

Cellular biomarkers serve as early indicators of disturbance and act as an early warning of possible perturbations at the individual, population, and community levels, which may deteriorate over time. Although cellular-level biomarkers can serve as early warning systems for exposure to and/or toxicity of certain compounds, several biomarkers are also influenced by other factors such as hormones, growth factors, reproductive stage, and stress as well as other chemical compounds; they can also be tissue- and species-dependent (Vasseur and Cossu-Leguille, 2003). The influence of season, sex, age, reproductive stage, and environmental conditions therefore must be known for these tools to be effective for environmental monitoring (Corsi et al., 2003; Vasseur and Cossu-Leguille, 2003; Sánchez and Porcher, 2009).

1.08.2.5.3 Organ

The ability to detoxify pollutants is essential to fish; pollutants entering an organism may be either metabolized or stored in particular organs within the body. In fish, the liver is the main storage organ and is also the site of detoxification (Elliott et al., 2002). If the stress persists or if the detoxifying mechanisms fail, then this can result in cell damage and physiological changes (Corsi et al., 2003; Vasseur and Cossu-Leguille, 2003). Fish liver structure or histopathology can provide a good indication of exposure to xenobiotics, and can also indicate the ability of an individual to resist future insults (Stentiford et al., 2003). In addition to the liver, histopathological studies can include other organs such as the kidneys, gills, and ovaries.

Although organ structure can be used to indicate stress or exposure to xenobiotics, they are not a diagnostic tool and cannot be used to determine the particular pollutant that caused the alterations. Furthermore, organ structural alterations may be a result of exposure to different combinations of pollutants.

1.08.2.5.4 Individual

At the whole-organism level, pollution or stress can result in either mortality or indirectly by causing changes in behavior such as impairing feeding and reproduction; these effects on individuals can reduce population growth, which, in turn, will result in effects at the community level (Elliott et al., 2002). Measures of the health or condition of individual fish can include some measure of the weights of individual body organs or tissues (Elliott et al., 2002). A general condition index, which is a measure based on the relationship between weight and size (length), can provide a coarse measure of the food intake and nutritional health of an organism. The gonadosomatic index (GSI) compares the mass of the gonad with the total mass of the animal and assumes that an ovary increases in size with increasing development; the liver somatic index (LSI) relates liver weight to whole body weight and can also provide an indication of the health of an

individual; if an organism is not feeding, then liver weight will decrease (Elliott et al., 2002; Corsi et al., 2003). Any effect by contaminants or stress on the integrity of the gonads and/or liver will affect the GSI and LSI.

Morphological disorders, disease, and parasite infestation can also be used to assess individual fish health (Elliott et al., 1988). Morphological disorders can include blemishes, lesions, lymphocystis-type nodules, fin rot, eye deformity, mouth ulceration, and skeletal deformities (Hemingway and Elliott, 2002). Stressed fishes are more susceptible to disease and parasite infestation, which can affect the growth and body condition of that organism (Elliott et al., 2002).

It should be noted that condition factors or indices often vary between life stages within a population and also with feeding status; emaciation created by spawning, poor food conditions, or overwintering will be reflected by these indices as well as environmental stress (Elliott et al., 2002).

1.08.2.5.5 Population

Population parameters are typically restricted to a single species and include measures of abundance, biomass, length and year classes, and distribution patterns. Estimates of abundance will give an indication of the size of the populations of the species, and temporal variations will show seasonal and annual cyclical patterns (Hemingway and Elliott, 2002). The accuracy of measures of abundance, however, depends on the sampling methods used and assumes that these samples are representative of the whole population. Cohort analysis is based on catch data obtained from different age or size groups of the population. Most populations exhibit polymodal size distributions and an analysis of size-frequency information can be used to determine recruitment success, growth, and population changes (Elliott et al., 2002).

Indicator species include sensitive taxa that have fairly narrow water-quality and habitat requirements. Monitoring these populations is a useful indicator of environmental quality as they are often the most sensitive to environmental change and will be the first to disappear when conditions deteriorate. As the sensitive fauna is eliminated, they are often replaced by more tolerant species, which may thrive and become more abundant (Elliott et al., 2002). Indicator species may also include rare or threatened species, which are of conservation value. Because rare species are fragile, they may become endangered or even locally extinct with increasing anthropogenic stress (Costello et al., 2002). However, it should be noted, that the status of some fish species might vary geographically. For instance, a particular species may be abundant in one region, but threatened in another; this is because some rare fishes are at the limits of their geographic distribution (Seegert, 2000; Costello et al., 2002). It should be noted that changes in the distribution range and abundance of certain species could also be a result of global factors such as climate change rather than local conditions. The occurrence, distribution, and abundance of populations of exotic or introduced species also represent a potential threat to naturally occurring taxa through competitive exclusion and predation (Marchand et al., 2002).

1.08.2.5.6 Community

Fish communities can be described according to the number of species present (richness) and the distribution of individuals or biomass among those species (Elliott et al., 2002). These

variables can be used to derive other measures such as diversity indices; the Shannon–Wiener index (H'), for example, gives a measure of species richness and evenness within a community. Another means of interpreting fish community structure uses the concept of functional groups or guilds; these can be based on the ecological preferences of a species, their reproductive strategies, or their feeding modes (Elliott et al., 2007). However, some authors (Selleslagh et al., 2009) suggest that the guild approach may not be useful to provide valuable information on the ecological status of small estuaries. Because fish communities respond to a variety of environmental factors (physical, chemical, and biological) they provide an integrated measure of estuarine conditions or health. A fish community has the ability, to a limited extent, to absorb change within the various levels of biological organization; it is able to compensate for short-term localized stress such as disease or low dose and/or infrequent pollution events. However, if this stress is prolonged or too intense, the biological community will change from a relatively diverse and complex community to one that is relatively simple and species poor (Whitfield and Elliott, 2002). Trends in fish community attributes can be described and compared with data from other systems or with some reference condition and these results be used to assess the overall condition of the ecosystem (Elliott et al., 2002).

Community attributes such as species richness, abundance, and diversity indices, however, are heavily influenced by sampling effort. Furthermore, different combinations of species and abundance can yield the same diversity (H') value. It is also important to be able to distinguish between natural variations and anthropogenic impacts when assessing fish community structure and function (Martinho et al., 2008). Although fish communities can provide a good integrated measure of ecosystem condition, their response to disturbance or impact can only be diagnosed only after degradation or impact has occurred (Vasseur and Cossu-Leguille, 2003). Furthermore, because fish communities integrate environmental conditions, it is difficult to assign particular causes responsible for degradation.

1.08.2.5.7 Biomarkers and bioindicators in environmental monitoring and assessment

Most estuarine and coastal monitoring programs have the objective of measuring the quality of the environment and fishes are a useful group for such biomonitoring programs. With any survey of fishes in estuaries, information may be required at any one or several levels of biological organization; for example, information may be required at the cellular, individual, population, community, or ecosystem level. With a progression through each level, the speed of response to environmental change decreases and the inherent variability in the ecosystem increases; for example, an individual fish will respond rapidly to a change in water quality, whereas the community will take longer to show changes (Elliott et al., 2002). Biomarkers can also complement chemical and bioindicator assessments in that they can provide an early warning signal of contamination and help establish relationships between chemical quality, fish health, and ecological status (Sánchez and Porcher, 2009). Ideally, biomonitoring programs should include measures of stress at the subcellular, cellular, and organ levels (biomarkers), as well as at the individual,

population, and community levels (bioindicators). Such approaches, however, are difficult to apply in large monitoring networks due to practical and economical constraints.

Fish biomarkers have been used in a number of monitoring programs to assess the environmental condition of coastal and estuarine waters. In the USA, a fish contaminants index, which examines the bioaccumulation of contaminants (e.g., metals, dichlorodiphenyltrichloroethane (DDT), dieldrin, lindane, PAH, and PCB) in fish tissues, has been developed as part of a national estuary monitoring program (USEPA, 2001). Part of a national monitoring program for coastal and estuarine waters within the UK includes a range of fish biomarkers (e.g., bioaccumulation, metallothionein, vitellogenin, EROD activity, bile metabolites, DNA adducts, and liver pathology) (DEFRA, 2005).

Fish bioindicators have also been used to monitor and assess estuaries. One approach has been to include a number of attributes (or metrics) into a single, integrated measure. Such a multimetric approach has been developed and applied in the United States (Deegan et al., 1997; Hughes et al., 2002), South Africa (Harrison and Whitfield, 2004, 2006), Spain (Borja et al., 2004b; Uriarte and Borja, 2009), Belgium (Breine et al., 2007), and the United Kingdom (Coates et al., 2007). Many of these multimetric approaches include metrics at various levels of biological organization from individual (e.g., number of diseased fishes), population (e.g., indicator species), to community (e.g., species richness, dominance, resident species, and piscivorous species) measures. Recently, some of them have been evaluated to determine the efficacy in assessing ecological status (Henriques et al., 2008; Martinho et al., 2008) and its response to human pressures (Uriarte and Borja, 2009).

Key to any biomonitoring program, whether based on biomarkers, bioindicators, or both, is the ability to define the normal (natural) situation, measure any departure from this situation, assess whether any departure is significant, and explain the cause and effect (Hemingway and Elliott, 2002).

1.08.3 Integrating Multiple Compartments of the Ecosystem in Assessing Ecological Quality

1.08.3.1 North America

There are many large spatial scale assessments of aquatic environmental condition in the United States (Table 7), but most are designed to address single types of environmental stress. For instance, there are national assessments of contaminant accumulation (National Status and Trends (NS&T), Mussel Watch, Kimbrough et al., 2008), bacterial concentrations on beaches (Dorfman and Stoner, 2007), and nutrient effects in estuaries (Bricker et al., 2007, 2008). There are also biotic assessments of fishery condition (NMFS, 2008), endangered species (NMFS, 2006), and coral reefs (Waddell and Clarke, 2008).

There are only three programs that integrate across the types of stressors for marine environments at the national level. One of these is the National Coastal Condition Report (NCCR), which includes many components of the coastal ecosystem and also considers at a lesser level the connections to human uses and human health. The NCCR is led by the US EPA with collaboration from the National Oceanic and Atmospheric Administration (NOAA), the US Fish and Wildlife Service

(FWS), and the US Geological Survey (USGS). The report also includes case-study contributions from states and tribes. It is a comprehensive report on the condition of the nation's estuarine regions, coastal wetlands, seagrass meadows, coral reefs, mangrove and kelp forests, upwelling areas, and coastal fisheries that together present a broad baseline picture of conditions within the coastal ecosystem. The assessment combines five primary indicators: water quality (nutrient related), sediment quality index (inorganic and organic pollutants), benthic (benthic population and communities), coastal habitat (wetland loss rate), and fish tissue contaminants (fish and shellfish tissue contaminant concentration) into a rating for the overall condition of the coastal ecosystem. Coastal monitoring data from programs such as EPA's National Coastal Assessment Program, NOAA's NS&T Program, FWS's National Wetland Inventory, and data from the Great Lakes National Program Office (GLNPO) are used to develop these indices of condition.

The NCCR primary indices focus on ecological conditions, showing that overall conditions are rated fair and have improved slightly since the initial NCCR in 2001. The worst ecological conditions are observed in the Northeast, Gulf of Mexico, Great Lakes, and Puerto Rico regions, and the best in South Central Alaska and Hawaiian regions as they do not report by individual system. The report also includes data capturing the human use and human health aspects of the coastal ecosystem. For instance, it includes information on fish stock and catch, fish consumption advisories, and beach advisory statistics. The report includes data collected through EPA's Clean Water Act Section 305(b) Program, the National Listing of Fish and Wildlife Advisories Program, and the Beaches Environmental Assessment, Closure and Health Program databases, as well as NOAA's National Marine Fisheries Service statistics on fish populations and fishing catch. Although these data sets are not as robust as the data sets supporting the ecological indicators, they highlight the extent of human use of the coastal ecosystem as well as human health risks associated with polluted beaches and contaminated fish, providing a more complete/integrated picture of the ecosystem condition. As an example, the results show that fish consumption advisories have been issued for an estimated 77% of the US coastal waters for a total of 23 individual chemical contaminants, although four primary contaminants (PCBs; mercury; DDT and its degradation products; and dioxins and furans) were responsible for 92% of all fish consumption advisories in 2003.

The Heinz Center's State of the Nation's Ecosystems (Heinz, 2008) and the EPA's Report on the Environment (USEPA, 2008a) also provide a broad view of conditions nationally across many ecosystems, but both of these reports are broader in scope than the NCCR, including terrestrial in addition to coastal ecosystems. The Heinz report provides results for coasts and oceans, farmlands, forests, freshwaters, grasslands and shrublands, and urban and suburban areas, whereas the EPA report includes evaluation of air, water, land, human exposure and health, and ecological condition. Like the NCCR, these reports are multi-agency collaborations.

The Heinz report (Heinz, 2008) provides condition indicators for each of the major ecosystem types and 10 core national indicators that provide a broad perspective on national trends and conditions. A list of 108 indicators describes 10 major ecosystem characteristics within five groups: physical

Table 7 North American methods in assessing ecological quality using multiple compartments of the ecosystem

<i>Method/report name</i>	<i>Organization</i>	<i>Stressor</i>	<i>Ecosystem and/or organism</i>	<i>Human use/ human health</i>	<i>References</i>
<i>Single stressor/ecosystem/organism methods</i>					
ASSETS/NEEA (Assessment of Estuarine Trophic Status / National Estuarine Eutrophication Assessment)	NOAA NOS	Nutrients	Estuaries, coastal waters	No	Bricker et al. (1999, 2003, 2007) http://www.eutro.org
NS&T (National Status and Trends Program) Mussel Watch	NOAA NOS	Inorganic and organic contaminants	Estuaries, coastal waters, mussels	No	Kimbrough et al. (2008)
IBIs	USACE, IBIs	Contaminants	Streams, coastal waters, benthic invertebrates	No	Karr (1981, 1991)
Annual Beach Report Card	Heal the Bay	Bacteria	California beaches, coastal waters	Use, health	http://www.healthebay.org/brc/annual/2006/counties/la/analysis.asp http://www.healthebay.org/brc/annual/2006/execsumm.asp
Testing the waters: A guide to water quality at vacation beaches	NRDC	Bacteria	National beaches	Use, health	http://www.nrdc.org/water/oceans/ttw/titinx.asp http://www.nrdc.org/water/oceans/ttw/ttw2008.pdf
Our living oceans: Report on the status of U.S. living marine resources	NOAA NMFS	Fishing pressure	Nearshore fish species and marine mammals and sea turtles	Use, health	NMFS, 1999
Status of US fisheries	NOAA NMFS	Fishing pressure	US fish stock	Use	NMFS, 2008
Recovery program for threatened and endangered species	NOAA NMFS	Fishing pressure	Endangered species	Use	NMFS, 2006
SQT (Sediment Quality Triad)	USEPA, NOAA NOS, Environment Canada	Inorganic and organic contaminants	Sediment chemistry and sediment toxicity, benthic community composition	No	Chapman (1986); Chapman et al. (1987)
EMAN (Ecological Monitoring and Assessment Network)	Environment Canada, citizen monitoring	Difficult to tell	Also difficult to tell	???	www.eman-rese.ca
The state of coral reef ecosystems of the United States and Pacific freely associated states: 2008	NOAA NOS	Climate change and coral bleaching, coral disease, tropical storms, coastal development/runoff, tourism/recreation, commercial fishing, subsistence and recreational fishing, vessel damage, marine debris, aquatic invasive species	Coral, reef fish populations, macroinvertebrates	Use	Waddell and Clarke (2008)

(Continued)

Table 7 (Continued)

<i>Method/report name</i>	<i>Organization</i>	<i>Stressor</i>	<i>Ecosystem and/or organism</i>	<i>Human use/ human health</i>	<i>References</i>
<i>Integrative Methods</i>					
National Coastal Conditions Reports	EPA (lead) with NOAA, FWS, USGS	Nutrients, contaminants	Estuaries, coastal waters, wetlands, fish, some socioeconomics	Uses, health (i.e., consumption advisories, beach closures)	USEPA (2001); USEPA (2005); USEPA (2008)
The State of the Nation's Ecosystems: Measuring the lands, waters, and living resources of the United States	The H. John Heinz III Center for Science, Economics and the Environment	Nutrients, carbon, oxygen, chemical contaminants, physical conditions (i.e., temperature)	Coasts and oceans, farmlands, forests, fresh waters, grass and shrub lands, plants and animals, socioeconomics? Leaves to others the analysis of pressures and the effects of actions taken to reduce stressors	Uses, health (i.e., beach closures)	The H. John Heinz (2008)
EPA Report on Environment	EPA	Inorganic and organic contaminants, nutrients, oxygen, climate change	Air, water (fresh and sea), wetlands, land, human exposure and health, ecological condition, fish, economics specifically excluded	Health	USEPA (2008)
State of the Maryland Coastal Bays	Maryland DNR	Nutrients, sediment toxicity	Stream and lagoon, water quality, sediment quality, sea grass, wetlands, benthic community, fish and shellfish	Uses	Wazniak et al. (2004)

dimensions, chemical and physical conditions, biological components, and human uses. Of the 108 indicators, 63% have some or all data and can be reported nationally, whereas 37% cannot be reported nationally due to inadequate data or needed further development of the indicator. Although the report does not include an in-depth economic analysis, it includes highlights of the relative economic significance of resources for human uses as well as events such as blooms of toxic algae that can cause economic losses. However, this report does not include a combined overall evaluation for condition of any one ecosystem or combined multi-ecosystem assessment. Data sources are national in scope, derived mostly from federal agency reports and databases. Results do not address individual water bodies but rather larger regions. As an example, the 2008 report shows that streams and groundwater in farmland areas have higher concentrations of nitrate than streams in forested or urban and suburban areas, most likely from nitrogen fertilizer applied by farmers. Between 1992 and 2003, 20% of groundwater wells had nitrate concentrations that exceeded the federal drinking water standard, and between 1992 and 2001, 13% of streams in farmland areas had nitrate concentrations that exceeded the standard.

EPA's Report on the Environment uses a set of indicators to answer 23 questions about stressors to air, water, and land, their effects on human health and the environment, and the condition of the environment. These questions are related to EPA's five strategic goals: clean air, clean and safe waters, healthy land, healthy communities and ecosystems, and stewardship and compliance, and also focus on protection of human health and the environment. The answers are provided by 86 indicators of environmental and human health conditions, but there are no integrated assessments provided for any of the ecosystems. There is, however, discussion of critical indicator gaps that prevent the questions from being fully answered and the challenges to filling these gaps. Data used are mostly national in scope (a few are regional), from federal and state agencies and from nongovernmental organizations. Socioeconomic indicators are excluded, as is the use of indicators such as number of permits issued and enforcement actions taken. Results for hazardous waste indicator show that municipal solid waste generation rose from 1960 to 1990; however, since 1990, the daily per capita generation of municipal solid waste has been relatively constant, showing that the total increase in waste may be primarily a function of population growth. Hazardous waste, generated primarily through industrial processes, decreased from 1999 to 2005, although there was a small rise between 2003 and 2005.

These reports are all limited by a lack of national sampling programs that provide comprehensive supporting data sets; however, there are a number of regional or water-body-specific reports which are supported by dedicated monitoring efforts. One such example is the State of the Maryland Coastal Bays Assessment (Wazniak et al., 2004). The report contains 13 indicators of water quality, living resources, and habitat to evaluate the overall health of the coastal bays and to track changes over time. The component indicators are combined to give an overall assessment of the ecosystem which integrates across ecosystem components and stressors, and includes impacts to human uses and human health. The report also goes beyond that of the national reports in relating the outcomes to management objectives for ecosystem components

(e.g., reduce and control invasive/exotic species, increase seagrass abundance, and reduce nitrogen loading to streams), which provides insight and information to guide a management framework.

Many of the regional and water-body-specific assessments in the United States (e.g., State of the MD Coastal Bays, Wazniak et al., 2004; State of Santa Monica Bay, Santa Monica Bay Restoration Commission 2004; and State of Barnegat Bay, Barnegat Bay Estuary Program 2005) have been associated with the various national estuary programs, which were created to provide integrated management units. Although the national reports can provide the larger perspective, management is typically done on the local scale. The National Estuary Program water body assessments and recommendations for management are good examples of the use of integrated assessments at the local scale.

1.08.3.2 Europe

In recent years, increasing pressures and impacts within European estuaries have led to the approval of a series of laws which focus on water management, the WFD being the most important (see details in Borja, 2005 and Borja et al., 2008a). This Directive emphasizes the increasing need to protect European estuarine and lagoonal ecosystems and to move toward marine integrative management. The main objective of the WFD is to achieve a good ecological status, for all European water bodies, by 2015.

To achieve such an objective, the WFD requires the development of tools and methodologies to assess the status of several elements of the ecosystem, including physico-chemical and biological elements (phytoplankton, macroinvertebrates, macroalgae, phanerogams, and fishes), as shown above. However, the WFD, instead of using all these elements in assessing environmental quality in an integrative way, uses a simple approach known as the 'one out, all out' principle (Heiskanen et al., 2004; Borja, 2005; Tueros et al., 2009). This principle takes the final quality of a water body from the worst-rated element, which may be a useful starting point, but eventually should be avoided, due to the problems that arise in the final classification (Borja et al., 2004b; Moss, 2008; Tueros et al., 2009).

Hence, some authors propose to integrate all physico-chemical and biological elements into a single assessment of the ecosystem (Borja et al., 2004b, 2008a), and some guiding principles have been developed at national levels (e.g., in the UK; Rogers et al., 2007). The approach of Borja et al. (2004b) was further detailed in aspects, such as physico-chemical (Bald et al., 2005), chemical (Rodríguez et al., 2006; Tueros et al., 2008, 2009), phytoplankton (Revilla et al., 2009), macroinvertebrates (Muxika et al., 2007), and fishes (Uriarte and Borja, 2009). Finally, it consists in weighting some elements which have scientifically sound methods, and are appropriately sampled, validated, and intercalibrated (Borja et al., 2004b, 2008a, 2009a).

Despite the importance of transitional waters (both estuaries and lagoons) in Europe, the recent fourth assessment of Europe's environment (European Environment Agency, 2007) makes mention of these waters only 5 times, mainly in the context of eutrophication, conservation, or climate change. However, the ecological status of these important ecosystems,

in terms of goods and services, is not assessed. Probably this is due to the complex European political organization, being easier to find this information at the national (even regional level) rather than at the Union's level.

Hence, some countries, such as Ireland or the United Kingdom, among others, have some classification of status. Ireland uses, as one of the indicators, the eutrophication of estuarine and coastal water bodies. The 67 water bodies from 20 estuarine and coastal areas around Ireland were assessed for the period 2001–05, showing that 14.9% were classed as eutrophic, 7.5% as potentially eutrophic, 37.3% as intermediate, and 40.3% were unpolluted (Environmental Protection Agency, 2006). However, the eutrophic water bodies are decreasing throughout time (Figure 3). France uses, in Mediterranean lagoons, the principle of 'one out, all out', in the assessment of eutrophication, which includes physico-chemical elements in water and sediment, phytoplankton, macroalgae, and macroinvertebrates (Souchu et al., 2000; Ifremer, 2008) (Figure 4).

Another integrated estuarine quality assessment is undertaken in the Basque Country (northern Spain), by Borja et al. (2004b, 2009a); however, in this case, without using the 'one out, all out' principle. The ecological status is determined by including physico-chemical, chemical, hydromorphological, and biological (phytoplankton, macroalgae, macroinvertebrates, and fishes) elements, by integrating all the elements at the water body scale. This is made using a decision tree in the integration (see Borja et al., 2008a, 2009a). From this analysis, the Basque estuaries show a progressive improvement in their ecological status, reducing both bad and poor status (from 25% and 56%, in 1995, to 0% and 16%, in 2007, respectively) and increasing moderate and good status (from 19% and 0%, in 1995, to 59% and 25%, in 2007, respectively), especially after 2001 (Figure 5). In recent times, around 30% of the transitional water bodies are consistent with the WFD objective in achieving good status, by 2015.

This balance in water quality is in response to both negative pressures (dredging, land reclamation, discharges of polluted

Figure 3 Ireland estuarine and coastal water quality, in terms of percentage of water bodies in each class (1995–2005). From Environmental Protection Agency, 2006. Water Quality in Ireland 2005. Key Indicators of the Aquatic Environment. EPA, Wexford, 23 pp.

Figure 4 Results from the eutrophication investigation, undertaken in 2007, within French Mediterranean lagoons. Red: very good status; green: good; yellow: moderate; orange: poor; red: bad. From <http://rsl.cepralmar.com/bulletin.html> and Ifremer (2008).

Figure 5 Evolution of the ecological status, shown as percentage of estuarine stations, within the Basque Country (northern Spain), for the period 1995–2007. Modified from Borja, A., Bald, J., Franco, J., Larreta, J., Muxika, I., Revilla, M., Rodríguez, J.G., Solaun, O., Uriarte, A., Valencia, V., 2009a. Using multiple ecosystem components in assessing ecological status in Spanish (Basque Country) Atlantic marine waters. *Marine Pollution Bulletin* 59 (1–3), 54–64.

waters, engineering works, etc.), or positive actions (water treatment, recovery of degraded wetlands, etc.), resulting finally in a response of biological and physico-chemical elements. This is reflected, ultimately, when integrating all of the data into a unique ecological status assessment, as shown in Figure 5. This is coherent and coincident with the expert judgment on the current status of the Basque marine waters but, in addition, with different contributions showing this positive trend over recent times (García-Barcina et al., 2006; Borja et al., 2009a; Uriarte and Borja, 2009).

1.08.3.3 South Africa

The South African coastline extends approximately 3000 km from the Orange (Gariep) River (28°38' S; 16°28' E) on the west coast to Kosi Bay (26°54' S; 32°53' E) on the east coast (Figure 6). The west coast of South Africa is bordered by the Atlantic Ocean and is influenced by the north-flowing Benguela Current of upwelled inshore waters. The east coast borders the Indian Ocean and is influenced by the south-flowing Agulhas Current; being tropical in origin, the waters of the Agulhas Current are relatively warm but as it flows south it tends to cool. The estuaries of South Africa cover three biogeographic regions, a cool-temperate west coast, a warm-temperate south coast, and a subtropical east coast (Harrison, 2002). Some 300 coastal outlets have been identified along the coast of South Africa and these include relatively large, permanently open estuaries, small estuaries that are often closed to the sea by the formation of sand barriers at the mouth, very small coastal streams, and even dry riverbeds that only occasionally contain water (Whitfield, 2000).

Regional assessments of South African estuaries have been concerned with either establishing ecological importance or assessing ecological health. Ecological importance is an expression of the contribution of an estuary to the maintenance of ecological diversity and the provision of goods and services at regional and national scales. Measures of health are used to describe an estuary's condition and how well a particular system is fulfilling its ecological function relative to undisturbed or natural conditions.

1.08.3.3.1 Estuarine importance

The importance of an estuary can be measured in terms of a number of zoological, botanical, physical, and socioeconomic factors such as the presence of rare or endangered species, well-developed and diverse plant communities, unique hydrological features, and important recreational or amenity value. All these factors contribute to the overall importance of an estuary.

Several measures of estuarine importance have been developed and applied to South African estuaries. Coetzee et al. (1996) developed a botanical importance rating system, which incorporates factors such as plant community area cover, plant community condition (degree of impact), plant community importance within the estuary (dependence), and plant community richness. Systems that contain a greater area cover of a plant community have fewer impacts associated with it, and have a greater number of communities (community richness) will have a high importance score. The botanical importance rating system has been applied to estuaries in the Western and Eastern Cape systems (Coetzee et al., 1997; Colloty et al., 2000).

Turpie (1995) used estuarine water birds for prioritizing South African estuaries for conservation. South African estuaries were ranked according to single- and multiple-criteria indices, which included measures of diversity, abundance, rarity, and conservation status. The value of certain estuaries for overwintering Palaearctic migrant waders was also emphasized.

Maree et al. (2003) developed a fish importance rating (FIR) system to prioritize South African estuaries in terms of their importance to estuarine-associated fishes. The FIR is based on a scoring system of seven criteria that are considered to reflect the importance of estuaries to estuarine-associated fishes. The seven criteria were divided into two components: species importance and estuarine importance. Species importance incorporates three criteria: species richness, endemic species richness, and exploitable species richness, whereas estuarine importance included four physical measures of estuarine type, estuarine size (area), estuarine condition, and estuarine isolation. The FIR has been applied to estuaries spanning the entire South African coast.

Turpie et al. (2002) assessed the conservation priority of South African estuaries using a number of attributes, which

Figure 6 Map of South Africa.

included estuarine size, rarity of estuarine type, habitat diversity, and biodiversity in terms of plants, invertebrates, fish, and birds. The biodiversity component was restricted to a species rarity measure for each biotic component. South African estuaries were then allocated a conservation importance score based on the weighted sum of the various attributes. The above index incorporates components that are linked to both ecological quality and integrity, and the index is therefore of value to both scientists and managers. In contrast to the physical and habitat information, however, the biodiversity component is less comprehensive with many estuaries requiring more information for certain taxa.

1.08.3.3.2 Estuarine health

Estuaries are productive systems that provide a valuable supply of goods and services such as the provision of nursery areas for important fish and invertebrate species, feeding and overwintering sites for birds, nutrient recycling, flood attenuation, and human recreation and development. Many human activities, however, carried out in estuaries and their catchments impact on estuarine biodiversity and resources. In order to manage

estuaries effectively, some measure of their health is often necessary.

The community degradation index (CDI) represents the earliest attempt to assess the condition of South African estuaries. The CDI was originally developed by Ramm (1988) and was adapted and applied to South African estuaries along the eastern KwaZulu-Natal coast (Ramm, 1990). This index is based on a physical-hydrologic classification of estuaries and the development of reference fish species lists for each estuary type. The fish species richness of each estuary from survey data was then compared with the relevant reference richness to produce a CDI value, which ranges from 0 (undegraded) to 10 (degraded).

The estuarine health index (EHI), which was also applied to east coast KwaZulu-Natal estuaries, is a multidisciplinary index that includes physical (geomorphology), biological (fishes), water-quality, and aesthetic parameters, and condenses these into a single, composite measure of overall estuarine health (Cooper et al., 1994). The biological health index (BHI) component is based on the CDI approach and utilizes a geomorphological classification system to group estuaries into similar typologies, based on a number of physical

geomorphological variables. A reference list of fish related to each group of estuaries is then developed and the survey species lists are compared with each reference. Whereas the CDI measures the degree of dissimilarity (or degradation) between a fish assemblage a reference condition, the BHI was modified to reflect the degree of similarity (or health) to reference conditions. The water quality index (WQI) of the EHI comprised three subcategories: suitability for aquatic life (dissolved oxygen, oxygen absorbed, and ammonia), trophic status (nitrate, ortho-phosphate, and chlorophyll *a*), and suitability for human contact (fecal coliforms). The aesthetic health index (AHI) component was designed to assess the state of development in and around a particular estuary and included parameters such as floodplain land use, shoreline development, development in the floodplain surrounds, the presence of bridges, as well as additional factors such as smell, water clarity, the presence of exotic vegetation, solid waste and litter, and the presence of algal blooms or aquatic nuisance plants. The composite EHI is calculated as the sum of the three separate indices (Cooper et al., 1994).

The EHI was further developed and applied to the entire South African coast as part of a national state of the environment report for the Department of Environment and Tourism (Harrison et al., 2000). This assessment included all the major components of the EHI (geomorphology, ichthyofauna, water quality, and aesthetics) but with some modifications. The water-quality component, for example, included only six parameters: dissolved oxygen, oxygen absorbed, ammonia (suitability for aquatic life), nitrate, ortho-phosphate (trophic status), and fecal coliforms (suitability for human contact). The biological (fish) component was also modified to include a multimetric approach that was based on measures of fish species richness, species composition, and relative abundance (Harrison et al., 2000). Some 250 estuaries spanning the entire South African coast were assessed using this approach.

Harrison and Whitfield (2004) further developed the multimetric approach to produce the estuarine fish community index (EFCI). This index is based on 14 fish community attributes or metrics that include measures of richness and composition (species richness and species composition), abundance (species abundance and number of species >90% of the catch), estuarine dependence (number of estuarine-resident species, number of estuarine-dependent marine species, abundance of estuarine-resident species, and abundance of estuarine-dependent species), and trophic composition (number of benthic invertivore species, number of piscivorous species, abundance of benthic invertivore species, and abundance of piscivorous species). Using data collected during the state of the environment survey, the EFCI has been applied to 190 South African estuaries (Harrison and Whitfield, 2006).

The above bioindicator of estuarine ecological integrity (or health) operates at the community rather than the individual or cellular level. Recent use of fish biomarkers has shown that pollution and other perturbations can influence the health of individuals within a population without being detected at the community level. Richardson (2008) successfully combined the use of fish biomarkers and bioindicators to provide a biomonitoring tool that can be applied to fish assemblages in South African estuaries. This new index can provide an early warning to managers, indicating which estuaries are under

threat of imminent ecological collapse due to adverse anthropogenic influences.

1.08.3.3.3 Resource-directed measures

The National Water Act (Act 36 of 1998) in South Africa requires that the nation's water resources be protected, used, developed, conserved, managed, and controlled in an equitable, efficient, and sustainable manner. This Act requires the implementation of resource-directed measures (RDM), which involves the determination of the water quality and quantity required to meet basic human needs and for the protection of aquatic ecosystems (Adams et al., 2002). Provision is made for a water reserve to be established prior to the authorization, through licensing, of water use (e.g., for agriculture, domestic and industrial uses). The ecological reserve is the quality and quantity of water required to maintain a desired level of ecosystem structure and function and this is defined by assigning each estuary to an ecological management class. The determination of the ecological management class is based on a combination of measures of estuarine health and estuarine importance (DWAF, 1999).

Estuarine health in this process is determined in terms of both abiotic and biotic components. Abiotic measures include hydrology (seasonal river inflow patterns and floods), hydrodynamics and mouth condition, water quality (axial and vertical salinity gradients, nutrients, suspended solids, dissolved oxygen, and toxic substances), physical habitat (intertidal sediment structure and distribution, bed or channel modification and canalization, migration barriers, bridges, weirs, bulkheads, training walls, jetties, and marinas), and human disturbance (degree of nonconsumptive use, e.g., walking and boating, and consumptive use, e.g., fishing and bait-collecting activities). The biotic components include microalgae, macrophytes, invertebrates, fish, and birds; this comprises measures of species richness, abundance, and community composition for each group. The present ecological status is defined by six categories (A–F), each representing a broad degree of deviation from reference (natural) conditions (A) and where a highly degraded system would be classified in category F. Estuary importance is established following the procedure developed by Turpie et al. (2002), which includes measures of estuarine size, estuarine type rarity, habitat diversity, biodiversity importance, and functional importance. The ecological management class represents the desired future condition of an estuary and is based on a combination of estuarine health and importance (DWAF, 1999). The measures required to improve the health of a particular estuary are part of the RDM process and provision is also made for a monitoring program to document the changes in the estuary over time.

The National Water Act represents a major shift in emphasis from water resource development to resource management; the natural environment is no longer regarded as a competitive user of water but rather the base from which the resource is obtained and which must therefore be protected and managed (Adams et al., 2002).

1.08.3.4 Australia

When the combined state, territory, and federal governments of Australia adopted a national strategy for ecological sustainable development (ESD) in 1992 (COAG, 1992), one of the key

objectives was to develop a system of State of the Environment (SoE) reporting to monitor the condition (or health) of the environment. 'Estuaries and the sea' became a major theme for the early Australian SoE reports (DEST, 1994; Zann, 1994) and these reports soon identified significant gaps in information and data concerning suitable indicators on which to base spatial and temporal comparisons of estuary condition (SEAC, 1996). As a result, Ward et al. (1998) reviewed 61 possible indicators for Australian estuaries and the seas, making recommendations and identifying possible data sources. Despite this early work, and a host of related scientific studies (Fairweather, 1997; Deeley and Paling, 1999; Harris and Silveira, 1999; Kennedy and Jacoby, 1999; Edgar and Barrett, 2000; Ward, 2000; Melville and Pulkownik, 2006; Hirst and Kilpatrick, 2007; Scanes et al., 2007; Birch and Olmos, 2008), a clear set of national guidelines for monitoring Australian estuaries and marine waters have not been developed. A major obstacle in developing a consistent approach to monitoring of natural resources in Australia has been the complex and often confusing process of coastal zone management, which is well documented (e.g., Zann, 2000). Other issues include the size

of the Australian continent, with over 1000 estuaries; the large climatic variations across the nation; the huge diversity of estuaries between major regions; and the institutional and jurisdictional arrangements that often provide little clarity of key responsibilities. This has made the adoption of a single set of indicators particularly onerous and perhaps irrelevant. As a result, when a major assessment of Australia's catchments, rivers, and estuaries was carried out in 2002 (NLWRA, 2002), it relied heavily on a qualitative assessment of a set of general health criteria by an expert group (Table 8). The audit covered 972 water-bodies and concluded that the majority (482) were near pristine (Table 9), although estuary condition varied greatly between the populated and unpopulated Australian states (NLWRA, 2002).

Although no Australian standards exist for classifying ecological quality and integrity of estuaries, a long list of possible health indicators has been developed and most states have selected elements to monitor (Table 10). In addition, SoE reporting, including reporting on the condition of estuaries, has been carried out by national, state, and local governments, in most cases for over a decade, and usually provides

Table 8 Criteria used in the initial assessment of Australian estuary condition

	<i>Near-pristine condition</i>	<i>Largely unmodified condition</i>	<i>Modified condition</i>	<i>Extensively modified condition</i>
	<i>Catchment natural cover > 90%</i>	<i>Catchment natural cover ~ 65–90%</i>	<i>Catchment natural cover < 65%</i>	<i>Catchment natural cover < 35%</i>
Land use	Limited roads and disturbance to natural conditions and processes	No known gross impacts from land use, e.g., sediments to waterways and estuary	Documented impacts from land use (e.g., sediments and nutrients to waterways)	Documented impacts from land use throughout waterways and into estuary
Catchment hydrology	No dams or impoundments, virtually nil abstraction	No dams or significant impoundments, some abstraction	Dams and impoundments, significant abstraction modifying natural flows	Dams and impoundments, significant abstraction modifying natural flows
Tidal regime	No impediments to tidal flow, changes from natural morphology (e.g., training walls, barrages, bridges, and causeways)	No significant impediments to tidal flow or changes from natural morphology	Impediments to tidal flow and/or changes from natural morphology (e.g., training walls, causeways, and artificial opening of entrance)	Major changes to tidal flow and/or major changes from natural morphology
Floodplain	Wetlands intact in vegetation and hydrology, no alterations to flood pattern	Wetlands mostly intact in vegetation and hydrology, no alterations to flood pattern	Wetlands mostly cleared in vegetation an/or changes in hydrology (e.g., drains, tidal barrages, and levees)	Wetlands mostly cleared in vegetation an/or changes in hydrology (e.g., major losses in fresh to brackish wetlands)
Estuary use	Extractive activities limited to Indigenous or limited and sustainable commercial and recreational fishing, no aquaculture	Extractive activities limited to sustainable commercial and recreational fishing, minor aquaculture	Extractive activities include dredging, extensive aquaculture, habitat modifying fishing methods (e.g., prawn trawling)	Extractive activities include dredging, extensive aquaculture, habitat modifying fishing methods (e.g., prawn trawling)
Pests and weeds	Minimal impact on estuary from catchment weeds and limited pests and weeds within estuary	Minimal impact on estuary from catchment weeds and limited pests and weeds within estuary	Significant impact on estuary from catchment weeds and impact on estuary ecology from pests and weeds within estuary	Significant impact on estuary from catchment weeds and impact on estuary ecology from pests and weeds within estuary
Estuarine ecology	Ecological systems and processes intact (e.g., benthic flora and fauna)	Ecological systems and processes mostly intact (e.g., some changes to benthic flora and fauna)	Ecological systems and processes modified (e.g., loss of benthic flora and fauna)	Ecological systems and processes degraded (e.g., major changes to habitats or species assemblages)

Reproduced from NLWRA, 2002. Australian Catchment, River and Estuary Assessment 2002. National Land and Water Resources Audit, 2002. Commonwealth of Australia, Canberra.

Table 9 Condition of Australia's estuaries by process type

<i>Class</i>	<i>Subclass</i>	<i>Near-pristine</i>	<i>Largely unmodified</i>	<i>Modified</i>	<i>Extensively modified</i>	<i>Total</i>
Wave	Estuary	28	41	62	25	156
	Strandplain	36	13	10	1	60
	Other	40	30	22	17	109
Tide	Estuary	57	25	9	4	95
	Tidal flat/creek	210	43	16	15	284
	Other	40	17	23	9	89
River	Wave-dominated delta	28	24	30	12	94
	Tide-dominated delta	36	16	11	9	72
Not classified		9	1	3	0	13
Total		484	210	186	92	972

Reproduced from NLWRA, 2002. Australian Catchment, River and Estuary Assessment 2002. National Land and Water Resources Audit, 2002. Commonwealth of Australia, Canberra.

summaries of major environmental changes in many key estuarine environments (e.g., Kirkham, 1997). The primary responsibility for monitoring estuary quality and integrity in Australia lies with the state (=provincial) governments. For example, in New South Wales (NSW), the task of monitoring estuary health has recently been divided between the state government and a regional framework of Catchment Management Authorities (CMAs). A list of indicators has been recommended for monitoring estuarine health at the state level, derived from the national guidelines (Table 10). Each NSW CMA has developed its own set of indicators depending on local priorities and resources (Table 9) and will report on the quality of each estuary in a report-card format. This new framework builds on a previous system involving environmental reporting at local and state government authorities (e.g., NSWDECC, 2006) and is in an early stage of development. A major issue to be resolved relates to the standardization of methodologies. Although guidelines exist for many of the physico-chemical measures used in these health assessments, such as water analyses for nutrients, pollutants, and turbidity (e.g., ANZECC and ARMCANZ, 2000), ecological measures are often not standardized and a range of practices have been selected. For example, the extent of estuarine wetland communities (e.g., seagrasses) is an ecological indicator adopted by most regional, state, and national authorities (Table 10); however, no nationwide standard methods of mapping (e.g., scale or resolution) have been developed or adopted, a situation that may lead to inconsistencies in the future.

Despite this slow progress in developing a consistent approach across jurisdictions, a particularly successful and well-planned estuary monitoring program, the South East Queensland Healthy Waterways Program (Abal et al., 2000, 2006), has been established in the Moreton Bay region through a cooperative approach between the national, state, and local governments. A range of environmental and ecological quality parameters in approximately 18 major estuaries in SE Queensland are assessed, with the main ecological parameters relating to seagrasses (depth range and distribution), coral cover, and riparian condition. Detailed methods for this monitoring program are available in annual technical reports (e.g., EHMP, 2008). The parameters are used to develop a biological health rating (BHR). In addition, a suite of water-quality parameters is used to provide an EHI. A single BHR and a single EHI

value are generated based on the number of sites within in each estuary that comply with established standards. These values are reviewed by an expert panel and combined into a report-card format for each estuary, providing a condition indicator for each estuary.

In summary, despite a long history of environmental reporting and some excellent local and regional examples of successful programs, the general framework for quantitatively classifying the ecological quality and integrity of estuaries in Australia is not well developed. Existing nationwide assessments of estuary condition have relied on qualitative criteria. Currently, there are a large number of emerging projects and programs that are likely to fill this gap in the coming years.

1.08.3.5 International Methodologies and Comparison across Geographies

The goal of methods developed to evaluate ecological condition is to reduce or summarize environmental indicators to a number that will provide adequate information to form the basis for management decisions. The more integrated methods allow for assessment at the ecosystem level, rather than only an ecosystem component. Ideally, an assessment of ecological status will provide results showing the level of ecological impairment and the dominant source(s) and level of pollutant that has caused observed impairment so that management measures can be targeted for maximum effectiveness. Continued monitoring and application of the assessment method allow for tracking of management success through time. To ensure their usefulness in the development of successful management measures, assessment methods must fulfill several requirements. One important aspect is to include pollutant sources and loads along with biological and physico-chemical indicators. In this way, the level of pollutant load can be associated with the level of impairment and from this relationship successful management criteria can potentially/better be developed. Both biological and physico-chemical components should be used to provide an accurate evaluation of conditions. Using nutrients as an example, although there may be no problems with dissolved oxygen, which would indicate no significant nutrient-related problems, there may be losses of seagrasses and excessive algal blooms (micro- and macroalgae) which are indicators of the early stages of nutrient enrichment.

Table 10 Examples of estuarine quality and integrity indicators proposed at the different jurisdictional levels within Australia

<i>National estuary indicators</i>	<i>Examples of state estuary indicators</i>	<i>Examples of regional estuary indicators</i>
<p><i>Recommended Australian estuarine health indicators</i></p> <ul style="list-style-type: none"> • Algal blooms • Animal disease/lesions • Animal kills • Animal or plant species abundance • Animals killed or injured by litter (entanglement, starvation, suffocation) • Benthic microalgae biomass (in intertidal sand/mudflat communities) • Biomass, or number per unit area, of epiphytes (in seagrass or mangrove communities) • Biomass, or number per unit area, of macroalgae (in rocky shore, rocky reef or coral reef communities) • Chlorophyll <i>a</i> • Coral bleaching • Death of marine mammals, endangered sharks and reptiles caused by boat strike, shark nets or drum lines • Dissolved oxygen (DO) • Estuary mouth opening/closing • Extent/distribution of key habitat types • Extent/distribution of subtidal macroalgae • Occurrence of imposex • Pest species (number, density, distribution) • pH • Presence/extent of litter • Salinity • Seagrass: depth range • Sedimentation/erosion rates • Targeted pathogen counts • Total nutrients in the sediment with dissolved nutrients in the sediment • Total nutrients in the water column with dissolved nutrients in the water column • Toxicants in biota • Toxicants in the sediment • Turbidity/water clarity • Water-current patterns • Water soluble toxicants in the water column • Water temperature 	<p><i>NSW estuarine health indicators</i></p> <ul style="list-style-type: none"> • Extent of mangroves, saltmarsh, seagrass and macrophytes • Freshwater inflow • Fish assemblages • Stress biomarkers • Pelagic chlorophyll <i>a</i> • Estuaries Baseline Data Collection Program • Hydrography survey • CMA regional monitoring <p><i>QLD estuarine health indicators</i></p> <ul style="list-style-type: none"> • Extent and diversity of estuarine habitats • Estimated wild fish stocks • Algal blooms in estuarine and marine environments • Exceedences of marine and estuarine water quality guidelines • Number of hotspot areas causing acidified waterways • Sea level rise • Introduced species <p><i>TAS estuarine, coastal & marine indicators (draft)</i></p> <ul style="list-style-type: none"> • Physical–chemical condition (water quality parameters incl. toxicants) • Biological condition (e.g., species distributions & abundance; algal blooms; chlorophyll-<i>a</i>; pest species, mass mortality events; litter, etc.) • Habitat extent (extent/distribution of key habitat types, subtidal, inter/supratidal) 	<p><i>NSW northern rivers CMA</i></p> <ul style="list-style-type: none"> • Water quality suite: pH, DO, salinity, conductivity, turbidity, temperature • Secchi disc • Benthic light (light loggers) • Total nitrogen • Total phosphorous • $\delta^{15}\text{N}$ sewage plume mapping using oyster meat or aquatic plants • Chlorophyll <i>a</i> • Habitat assessment (health & presence of riparian, intertidal and subtidal habitats) • Seagrass depth range • Waterwatch and/or bugwatch <p><i>NSW hunter central rivers CMA</i></p> <ul style="list-style-type: none"> • Chlorophyll <i>a</i> • Seagrass & macro-algae extent • Seagrass depth • Modelled catchment nutrient loads • Water quality suite: Secchi disc, temperature, salinity, DO, pH • Saltmarsh and mangrove extent • Seagrass condition, including epiphytic growth • Phytoplankton composition • Macro-algal blooms • Growth rates of sediment fans • Extent of mudflats • Shell fishery closures • Faecal coliforms and/or enterococci <p><i>Southern rivers CMA</i></p> <ul style="list-style-type: none"> • Seagrass, mangrove, saltmarsh habitat extent • Seagrass depth limits • Water clarity/turbidity • Water quality suite (temperature, salinity, conductivity, pH) • Secchi depth • Chlorophyll <i>a</i> • Catchment nutrient and sediment loads • Faecal coliforms • Shellfish closures • Fish kills • Invasive species (i.e., <i>Caulerpa taxifolia</i>)

Adapted from Fraser, M., 2008. The development of an ecosystem health monitoring program for the estuaries and coastal lakes in the southern Catchment Management Authority region. Southern Catchment Management Authority, Department of Environment and Climate Change, NSW Government, Sydney, Australia.

Because ecological degradation is a global problem, many methods have been developed worldwide to try to evaluate ecological status (i.e., see section 1.08.2). Here, methods developed for eutrophication are used as examples in the discussion of integrated methods. Only screening models are considered as these are most useful for resource managers (Table 11). Dynamic models are not considered, despite their potential to help in understanding details of nutrient-related problems,

because typically they are very complex with rigorous data requirements that are not necessarily needed for the screening process. The eutrophication assessment methods described here highlight a commonality among ecological assessment methods; they typically focus on a single stressor/pollutant – here, nutrients. A true integrated assessment method would include additional stressors; however, single stressors are typically the focus because of the complexity, with respect to study

Table 11 Method of eutrophication assessment, the biological and physico-chemical indicators that are used by the method, whether the method integrates nutrient load with ecological condition assessment and whether the method formulation results in one integrated rating

Method name	Biological indicators	Physico-chemical indicators	Nutrient load related to impairments	Integrated rating
TRIX ¹	Chl	DO, DIN, TP	No	Yes
EPA NCA ² Water Quality Index	Chl	Water clarity, DO, DIN, DIP	No	Yes
ASSETS ³	Chl, macroalgae, seagrass, HAB	DO	Yes	Yes
LWQI/TWQI ⁴	Chl, macroalgae, seagrass	DO, DIN, DIP	No	Yes
OSPAR COMPP ⁵	Chl, macroalgae, microphytobenthos, seagrass, HAB	DO, TP, TN, DIN, DIP	No	Yes
WFD ⁶	Chl, macroalgae, microphytobenthos, seagrass, HAB	DO, TP, TN, DIN, DIP	No	Yes
HEAT ⁷	Chl, seagrass, benthic invertebrates, HAB	DIN, DIP, TN, TP, DO	No	Yes
IFREMER ⁸	Chl, seagrass, macrobenthos, HAB	DO water clarity, SRP, TP, TN, DIN, sediment organic matter, sediment TN, TP	No	Yes
AMBI ⁹	Soft bottom macrobenthic community (five classes)		No	Yes
BENTIX ¹⁰	Soft bottom macrobenthic community (three classes)		No	Yes
ISD ¹¹ (lagoons)	Benthic community biomass size classes		No	Yes
B-IBI ¹²	Benthic community species diversity, productivity, indicator spp., trophic composition		No	Yes

¹Vollenweider et al. (1998);²US EPA (2008);³Bricker et al. (2003) and Bricker et al. (2007);⁴Giordani et al. (2009);⁵OSPAR (2002);⁶Devlin et al. (in prep);⁷Andersen and Laamanen (eds.) (2009);⁸Souchu et al. (2000);⁹Borja et al. (2000, 2007);¹⁰Simboura and Zenetos (2002);¹¹Reizopoulou and Nicolaidou (2007);¹²Weisberg et al. (1997).

design and resource allocation, related to identifying and examining synergistic impacts of multiple stressors and addressing multiple stressors through management. Another commonality is the emphasis on the status of ecological condition without the linkage to pollutant source and load (Table 11). In these cases, although the ecological status is determined, the relationship of pollutant source/load and impairment level cannot be determined, and thus the analysis is of limited usefulness toward development of management measures to address pollutant issues.

Although all methods have been developed with the intent of accurately evaluating eutrophic conditions, several important questions arise with the recognition that the methods have different formulations to determine the level of impairment (Table 11). Will they all give the same result if applied to the same water body? If not, does one or another do a better job of determining the extent of nutrient-related conditions? Because reference conditions are determined differently, does this mean that thresholds for undesirable conditions are different among different places? If so, can fair comparisons be made among results derived from different methods? What are the implications to management and how should the selection of a method be made?

Some of these questions can be answered in comparisons of multiple model application to the same system. This

comparison was made for the ASSETS, EPA NCA, and OSPAR COMPP methods using Barnegat Bay, New Jersey, USA, as the test system (Bricker et al., 2005; Table 12). Another comparison was made of the ASSETS, TRIX, EPA NCA, OSPAR COMPP, and WFD methods using the Thames River and Medway estuaries as the test sites (Devlin et al., in prep; Table 13). These comparisons, the first applied to a shallow microtidal lagoon system (Barnegat Bay, average depth 1.5 m, tidal range 0.24 m) and the other to a deeper macrotidal estuarine system (Medway Estuary, average depth 10 m, tidal range 5 m), highlight some of the differences among the methods. For example, in the application to Barnegat Bay, the two methods that include submerged aquatic vegetation, macroalgae, and harmful algal blooms (HABs) both result in a rating of problem or bad, whereas the EPA NCA, for which only water column indicators are used, shows the system to be in fair or moderate condition.

There is agreement among methods for most indicators, the exception is Chl *a*, for which ASSETS gives a rating of high (worst) and the others as good and no problem. The difference is that ASSETS uses the 90th percentile of annual data, whereas the OSPAR COMPP and EPA NCA use growing season/summertime values. In this system, Chl-*a* concentrations may reach a maximum beyond the limit of the index period used by the other two methods; thus, the chlorophyll

Table 12 Comparison of three eutrophication assessment methods results from application to Barnegat Bay, New Jersey, USA

<i>Element</i>	<i>ASSETS¹</i>	<i>EPA NCA WQI²</i>	<i>OSPAR COMPP³</i>
Nutrient load	High (OHI = 90% from watershed)	Not used	Problem*
DIN concentration	Not used	Fair (0.17 mg l ⁻¹)	No problem (12.5 uM)
DIP concentration	Not used	Fair (0.031 mg l ⁻¹)	No problem (0.25 uM)
N:P ratio	Not used	Not used	No problem (16.5)
Chlorophyll <i>a</i>	High (90th% = 9.67 ug l ⁻¹)	Good (4.74 ug l ⁻¹)	No problem (3.64 ug l ⁻¹)
Macroalgae	Problem	Not used	Problem
Submerged aquatic vegetation	Problem	Not used	Problem
Dissolved oxygen	No Problem (10th% = 5.8 mg l ⁻¹)	Good (6.29 mg l ⁻¹)	No problem (min. = 3.51 mg l ⁻¹)
HABs	Problem	Not used	Problem
Water clarity	Not used	Poor (0.56 WCI)	Not used
Overall waterbody classification	Bad	Fair	Problem area

¹ Chl *a* values are 90th percentile, DO values are 10th percentile based on NJDEP 2002–03 data, other indicator assessments based on Kennish (2001a,b); Hunchak-Kariouk and Nicholson (2001); Olsen and Mahoney (2001) and Seitzinger et al (2001).

² Values are averages for samples taken in Aug and Sept of 2002–03. This includes samples from 43 NJDEP stations (all 1600, 1700 and 1800 series plus 1506A and R08 – R20) 76 samples for Chl *a* and DO and 104 for N and P during August and September of 2002–03. Secchi depth = 0.835 m giving a value of 0.56 for the WCI based on the conversions Secchi depth * K_d = 1.45 and Light at depth/Incident light = $\exp(-K_d)$ from Batiuk et al. (2000).

³ Data are from NJDEP oceanic station data 1989–03. Reference values are winter means (Dec. 21–April 21) for DIP (0.46 uM) and NO₃ (9.15 uM), annual means for DO, and means for growing season for Chl *a* (Feb 1–July 31). Values in table, for stations detailed in ² above, are winter means for DIP, NO₃, annual mean for DO and growing season for Chl *a*.

* No reference value was available for nutrient inputs and the rating of problem for this component was determined from the OHI value calculated in the NEEA/ASSETS method. The color coding is consistent with the EU WFD color coding for ecological condition (high (best) – blue, good – green, moderate – yellow, poor – orange, bad (worst) – red).

Table 13 Comparison of results of application of five eutrophication assessment methods to Medway Estuary, UK, for eutrophic condition only

<i>Element</i>	<i>WFD</i>	<i>ASSETS</i>	<i>EPA NCA WQI</i>	<i>OSPAR COMPP</i>	<i>TRIX</i>
DIN	Moderate	Not used	Poor	+	*
DIP	Not used	Not used	Poor	+	*
Chlorophyll <i>a</i>	Good	High	Good	+	*
Dissolved oxygen	Good	No Problem	Good	-	*
Water clarity	Poor	Not used	Poor	Not used	Not used
HAB/algal toxin	Not used	No Problem	Not used	Not used	Not used
PP indicator spp.	High	Not used	Not used	-	Not used
Macroalgae	Moderate	Moderate	Not used	+	Not used
Submerged aquatic vegetation	Not applicable***	Not Applicable***	Not used	Not used	Not used
Zoobenthos/fishkills	Not used	Not used	Not used	-	Not used
Waterbody eutrophic condition	Moderate	Moderate	Poor	Potential problem area	Mesotrophic

*SPM used as modifier in the nutrient assessment,

**No individual score for TRIX, four parameters combined/integrated to derive rating,

***No submerged aquatic vegetation is observed in this system.

The color coding is consistent with the EU WFD color coding for ecological condition (high (best) – blue, good – green, moderate – yellow, poor – orange, bad (worst) – red).

indicator is not accurately captured by using limited temporal data. This can be especially problematic in cases when the sampling is done over only 1 or 2 days per year during the index period as is done in the EPA NCA program (USEPA, 2001a). This may also be a problem in cases where only water column indicators are used; for example, using macroalgae as an indicator in lagoons together with Chl *a* may be particularly important as it may be the macroalgal component that grows rather than the phytoplankton in this type of water body (Nobre et al., 2005).

The comparison of results among the five methods applied to the Medway estuary also shows discrepancies in the ratings for Chl with only ASSETS giving a high (worst) rating which is

also related to the time frames of sampling – growing season/index period versus annual data. In this case, in the deeper water system, the use of submerged aquatic vegetation is not as important as there is none observed, but the macroalgae component is important. It is important to note that despite some discrepancies, in both comparisons (Tables 12 and 13), the integrated outcome for the water body is fairly consistent, with the Barnegat Bay ratings fair to bad and Medway ratings moderate to poor and bad. In the recent Marine Strategy Framework Directive, there is recognition that the five quality classes of the WFD are largely un-implementable with respect to meaningful type-specific thresholds (Borja et al., 2010). The focus has been narrowed to a more practical approach that

includes two classes of environmental status (i.e., good or otherwise) rather than the five ecological status classes of the WFD. In this case, both systems (Barnegat Bay and Medway estuary) would be subject to the monitoring, assessment and management requirements that systems below good status are required to undergo by WFD legislation.

Although these eutrophication assessment methods use an integrated approach by combining biological and physico-chemical indicators, there are other integrative methods that deserve mention. In the case of the methods that use soft-bottom benthic community analysis (i.e., AMBI, BENTIX, B-IBI, and ISD; Table 11), the data provide a result that is integrative but not in the same way as the other methods. The result is in itself a reflection of integrated conditions with ratings ranging from pristine/unpolluted to extremely polluted (see Borja et al., 2000; Simboura and Zenetos, 2002; Reizopoulou and Nicolaidou, 2007; Weisberg et al., 1997; or Zaldivar et al., 2008 for comparative discussion). In this case, application of the methods provide an integrated result; however, it is not possible to identify the primary stressor(s) that are causing the impairment and thus it has the opposite problem of the single stressor focus of the other methods. Although it has been shown that AMBI is reflective of dissolved oxygen conditions (Borja et al., 2006, 2009d) and thus reflects one of the eutrophication indicators, it would be an interesting exercise to apply the AMBI and other benthic analysis methods to the Barnegat Bay and Medway estuary and see if the results would be the same.

1.08.4 Discussion

Monitoring programs worldwide seek integrative methodologies for assessing estuarine environmental or ecological status. However, following Díaz et al. (2004), rather than developing such integrative methods, we are assisting a tautological development of new indices for particular biological elements (e.g., phytoplankton and macroinvertebrates), which appear to be endemic, self-propagating, and rarely justified. This recent increase in the number of aquatic habitat quality indices suggests that there is little acceptance of any specific metric by environmental managers or scientists.

The growth in the number of these indices has been fueled by management's desire for a reductionist approach to the assessment of habitat quality (Díaz et al., 2004). Basically, the final outcome is the integration of multivariate data into a single site-specific numeric value that can be interpreted by a nonspecialist within a good- versus-bad gradient, often to meet a minimum legislative requirement (i.e., the Clean Water Act or the WFD). However, it is generally agreed that the ecological assessment methodologies should respond to the drivers–pressure–state–impacts–response (DPSIR) paradigm, requiring (1) assessment of ecological integrity; (2) evaluation of whether significant ecological degradation has occurred (in relation to anthropogenic disturbance); (3) identification of the spatial extent and location of ecological degradation (in relation to recent historical changes and/or reference conditions); and (4) determination of causes of unacceptable degradation in order to guide management actions (Borja and Dauer, 2008).

Most of the indices presented in this chapter that were developed for a particular biological element have similar merits: (1) multiple attributes (i.e., richness, diversity, opportunistic/

sensitive species, etc.) are combined into a single measure designed to maximize the ability to distinguish between degraded versus nondegraded condition; (2) they are developed with an appropriate methodology that accounts for biological variability that is associated with natural estuarine controlling factors such as latitude, salinity, and sediment particle size; and (3) they allow the comparison of values that reflect the degree to which component measures of key biological attributes at one location deviate from corresponding optimum values expected under undisturbed or reference conditions (Díaz et al., 2004). However, there is an urgent need to test the response of this plethora of indices to individual and mixed human pressures, within different geographies, as it has been done for some of them (Chainho et al., 2008; Henriques et al., 2008; Martinho et al., 2008; Borja et al., 2009d; Uriarte and Borja, 2009).

Although some of the reports described above (e.g., US Heinz Center Report, Report on the Environment, and Australian Ecosystem Health Monitoring Program) improve upon assessments limited to individual stressors, there remain numerous opportunities for further improvements in how integrated assessments are conducted (see Borja et al., 2009a). The biggest shortcoming is the lack of integration of the indices for different biological elements into an overall evaluation of ecosystem health. The difficulty is mostly the lack of an agreed-upon methodology, rather than a lack of intent. As an example just for eutrophication assessment, ASSETS uses matrices to combine characteristics of the pressure–state–response components and uses a matrix to combine the results of the three components into a single rating. The OSPAR COMPP and WFD assessment use a 'one-out all-out' process to determine the overall status of conditions but do not include the pressure or response components. Additionally, the integration into an overall evaluation should also include human use and socio-economic concerns so that the costs of environmental degradation can be highlighted. For instance, ASSETS considers physico-chemical and biological components and the interconnectedness between the watershed and coastal waters (i.e., land-based nutrient sources; e.g., Whittall et al., 2007) and recognizes the economic impact of nutrient-related damage (e.g., Bricker et al., 2006; Lipton, 2007), but ASSETS is still a single-issue focus (nutrients). Although present assessment methods are limited in the guidance they can provide to managers, development of multistressor assessment methods, albeit needed, will be complex and most effective if developed at the local level. However, a major issue is determining methods that are accurate and acceptable on a large-scale basis that can then be applied at a smaller scale (and need to be intercalibrated; Borja et al., 2007).

A second concern is that most of the integrated assessments are based on biological community endpoints; however, in most cases, the biological indices used to conduct these assessments have been developed regionally and differ substantially in their formulation (Díaz et al., 2004; Borja and Dauer, 2008). Regional development makes sense, as species composition and reference expectations for community parameters change naturally with ecoregion and habitat, but there is little assurance that regional indices are all calibrated to the same scale. Although biological assessments provide many advantages because they integrate the effects of multiple stressors over time, common scaling of the indices is essential for national

assessments intended to accurately compare condition across regions (Borja et al., 2009e).

Another concern with the use of biological indicators is the difficulty in relating observed effects back to a particular stressor that is causing impairment (see Borja et al., 2009d). While the integrative response that biological indicators offer is valuable, the actions taken by managers is typically directed to individual stressors. Distinguishing whether biological impairment results from habitat perturbation, invasive species, or pollutant stress, as well as which pollutant among many candidates, is essential to directing appropriate corrective actions. Some of them pose a technical challenge, as the methods for developing stressor attribution are still in development and are more advanced for stream environments than for estuaries or lagoons. An important issue, however, is structuring the assessment reports so that both stressors and response indicators are included and linked such that management recommendations can be made. Heavy metal contamination levels are generally higher in estuaries than in the open sea, but there was a wide variability of benthic responses to contamination probably due to the high spatio-temporal heterogeneity of the estuaries (Dauvin, 2008).

Beyond these technical challenges, most integrated assessment reports do not have a well-defined audience and are not well linked to management activities. This is particularly problematic at the national level, for example, the US Congress is often listed as the target audience. Although there have been legislative requests for such reports, the US Congress has not adopted them and used them as a focal point for hearings or triggers for large-scale directional changes. One of the difficulties is that management is best done at the local level and most of the national reports do not provide results by water body but rather by region. Additionally, it is a challenge to identify the causative influences for environmental degradation as there is often a synergistic effect. Moreover, monitoring, assessment, and management of a single stressor is very resource intensive, and thus typically it is the priority stressor or issue that receives the most scientific study and management attention, whereas other stressors that are not as easily linked to impacts, or might be more difficult to manage, are not given priority treatment. Thus, although challenging at the national level, integrative reports more effectively stimulate management actions at the local level, for example, many of the US National Estuary Programs use integrative reports to focus their management priorities. The national reports can be used to highlight a priority stressor of concern and gain momentum to manage that one stressor at the local level. It is also easier to develop management plans at the local level given that scientific study and report development is more participatory among groups with the same management goals, and there is potentially less disconnect between the scientists and the users of the report.

One challenge in making the management linkage is that the present reports focus on historically important stressors and do not provide much information on emerging issues. This is a natural outcome of emerging issues being too new to have yet been incorporated into large-scale monitoring programs and therefore the data sources to make them a focal point of a national assessment are lacking. However, the result is that the reports focus on legacy issues for which management actions have already been undertaken, rather than on issues which managers are contemplating action. Theoretically

emerging issues such as climate change or emerging contaminants are integrated into the biological responses that are key to these reports; however, relating the biological responses back to the stressors on which managers are considering action would substantially enhance the value of the reports.

Finally, the managers' goal is to provide the public with understandable maps integrating transitional water condition information from the different elements, presenting condition (quality) categories using simple colors. Reducing complex information from multiple ecosystem elements to a single color is a substantial challenge to marine scientists. Assessing marine quality will require the integration of different disciplines (chemists, engineers, biologists, ecologists, physicists, managers, etc.), to reach an agreement on the final assignment of ecological status (Borja et al., 2009e). Hence, emphasis needs to be directed at understanding the complexities of estuarine system functioning rather than simplifying and scaling down the system into smaller components (Díaz et al., 2004).

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