Marine Pollution Bulletin 56 (2008) 1519–1537

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Review

Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide

Angel Borja [a,*], Suzanne B. Bricker [b], Daniel M. Dauer [c], Nicolette T. Demetriades [d], João G. Ferreira [e], Anthony T. Forbes [d], Pat Hutchings [f], Xiaoping Jia [i], Richard Kenchington [g], João Carlos Marques [h], Changbo Zhu [i]

[a] AZTI-Tecnalia, Marine Research Division, Herrera Kaia, Portualdea s/n, 20110 Pasaia, Spain
[b] NOAA-National Ocean Service, National Centers for Coastal Ocean Science, 1305 East West Highway, Silver Spring, MD 20910, USA
[c] Department of Biological Sciences, Old Dominion University, Norfolk, VA 23529, USA
[d] Marine and Estuarine Research, P.O. Box 417, Hyper by the Sea 4053, South Africa
[e] IMAR–Institute of Marine Research, Centre for Ocean and Environment, FCT-UNL, Portugal
[f] Marine Invertebrates, Australian Museum, Sydney, NSW 2010 Australia
[g] Australian National Centre for Ocean Resources and Security, University of Wollongong, NSW 2522, Australia
[h] IMAR–Institute of Marine Research, University of Coimbra, Portugal
[i] South China Sea Fisheries Research Institute, CAFS, PR China

ARTICLE INFO

Keywords:
Ecological integrity
Integrative assessment
Ecological status
Environmental quality
Marine parks

ABSTRACT

In recent years, several sets of legislation worldwide (Oceans Act in USA, Australia or Canada; Water Framework Directive or Marine Strategy in Europe, National Water Act in South Africa, etc.) have been developed in order to address ecological quality or integrity, within estuarine and coastal systems. Most such legislation seeks to define quality in an integrative way, by using several biological elements, together with physico-chemical and pollution elements. Such an approach allows assessment of ecological status at the ecosystem level ('ecosystem approach' or 'holistic approach' methodologies), rather than at species level (e.g. mussel biomonitoring or Mussel Watch) or just at chemical level (i.e. quality objectives) alone.

Increasing attention has been paid to the development of tools for different physico-chemical or biological (phytoplankton, zooplankton, benthos, algae, phanerogams, fishes) elements of the ecosystems. However, few methodologies integrate all the elements into a single evaluation of a water body. The need for such integrative tools to assess ecosystem quality is very important, both from a scientific and stakeholder point of view. Politicians and managers need information from simple and pragmatic, but scientifically sound methodologies, in order to show to society the evolution of a zone (estuary, coastal area, etc.), taking into account human pressures or recovery processes.

These approaches include: (i) multidisciplinarity, inherent in the teams involved in their implementation; (ii) integration of biotic and abiotic factors; (iii) accurate and validated methods in determining ecological integrity; and (iv) adequate indicators to follow the evolution of the monitored ecosystems.
While some countries increasingly use the establishment of marine parks to conserve marine biodiversity and ecological integrity, there is awareness (e.g. in Australia) that conservation and management of marine ecosystems cannot be restricted to Marine Protected Areas but must include areas outside such reserves.

This contribution reviews the current situation of integrative ecological assessment worldwide, by presenting several examples from each of the continents: Africa, Asia, Australia, Europe and North America.

1. Introduction

The marine environment presents high levels of complexity, diverse habitats and supports a high level of biodiversity. These provide goods and services that support different uses which should be undertaken in a sustainable way. However, the marine, and particularly estuarine, environments are facing increasing

* Corresponding author.
E-mail addresses: aborja@pas.azti.es (A. Borja), suzanne.bricker@noaa.gov (S.B. Bricker), ddauer@odu.edu (D.M. Dauer), ticky@mer.co.za (N.T. Demetriades), joao@hoomi.com (J.G. Ferreira), Pat.Hutchings@austmus.gov.au (P. Hutchings), jiaxiaoping53@163.com (X. Jia), richard.kenchington@netspeed.com.au (R. Kenchington), jcmimar@ci.uc.pt (J.C. Marques), changbo@ecowin.org (C. Zhu).

0025-326X/$ - see front matter © 2008 Elsevier Ltd. All rights reserved.
doi:10.1016/j.marpolbul.2008.07.005

and significant impacts, which include physical and chemical transformation, habitat destruction and changes in biodiversity (Halpern et al., 2007, 2008). Causes include land reclamation, dredging, pollution (sediment discharges, hazardous substances, litter, oil-spills, eutrophication, etc.), unsustainable exploitation of marine resources (sand extraction, oil and gas exploitation, fishing, etc.), unmanaged tourism, introduction of alien species and climate change (see Halpern et al., 2007). These are driven by economic and social pressures for development and access to marine resources and activities through *i.a.* commercial fishing, aquaculture, tourism, recreation and maritime transport.

In order to resolve these problems, policy-makers world-wide seek to develop strategies to protect, conserve and manage the marine environment. The United Nations Convention on Law of the Sea (UNCLOS, 1982) is the international basic legal framework that governs the uses of the oceans and seas. UNCLOS establishes an international obligation to protect and use the resources of the marine environment sustainably as does the 1992 Convention on Biological Diversity (CBD, 2000), as highlighted by Parsons (2005).

At a national or regional level, several initiatives have been developed recently: (i) in December 1998, Australia released an Oceans Policy (Commonwealth of Australia, 1999, 2006); (ii) the Canadian Parliament passed the Oceans Act, which came into force in January 1997, being Canada's Oceans Strategy released in 2002 (Parsons, 2005); (iii) in the USA, the Pew Oceans Commission, created in 2000, and the US Commission on Ocean Policy, created by the Oceans Act of 2000, reported in 2004 (Granek et al., 2005); (iv) in Europe, the Water Framework Directive (WFD), which promotes the protection of continental, estuarine and marine waters, was released in 2000 (Borja, 2005), and the European Marine Strategy (EMS) Directive, was presented in 2005 (Borja, 2006; COM, 2005a, b, c); (v) in South Africa the National Water Act of 1998 (www.dwaf.gov.za/documents/publications) and the developing Coastal Management Act are presently in the form of the Integrated Coastal Management Bill (www.deat.gov.za); and (vi) in the People's Republic of China (PRC) a substantial body of legislation exists to address environmental protection (laws on Water (1988/01/21) and Environmental Protection (1989/12/26): Sea Water Quality GB 3097-1997, Environmental Quality for Surface Water GB 3838-2002, and Provisions for Monitoring of Marine Culture and Propagation Areas (2002/04/01)).

The objectives of these initiatives are to protect and/or restore the corresponding seas by ensuring that human activities are carried out in a sustainable manner, to provide safe, clean, healthy and productive marine waters. In summary, they try to promote the sustainable use of the seas and conserve marine ecosystems. Hence, the main objective of these legislative measures and policies is to maintain a good environmental or ecological status for marine waters, habitats and resources. The concept of environmental 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 activities in the area concerned.

The above concept defines quality in an integrative way, by using several biological parameters together with physico-chemical and pollution elements. This approach is intended to allow an assessment of the ecological status at the ecosystem level ('ecosystem-based approach' (EBA) or 'holistic approach' methodologies (Browman et al., 2004; Nicholson and Jennings, 2004; Rudd, 2004; Foster et al., 2005; Jennings, 2005; and Apitz et al., 2006)), more effectively than can be done at a species (e.g. mussel biomonitoring or Mussel Watch) or chemical level (i.e. quality objectives). The EBA is defined as: "a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use in an equitable way. The application of the EBA will help to reach a balance of the conservation, sustainable use, and the fair and equitable sharing of the benefits arising out of the utilization of genetic resources" (CBD, 2000). However, there are various interpretations of the EBA and its application almost always brings about confrontations and resistance among managers, proponents, and stakeholders (Morishita, 2008).

Following this approach, increasing attention has been paid to the development of tools for different physico-chemical or biological (phytoplankton, zooplankton, benthos, algae, phanerogams, and fishes) elements of the ecosystems. However, very few methodologies integrate all the elements into a unique evaluation of status and performance of an aquatic system. The need for such integrative tools to assess the ecosystem quality is very important, both from a scientific and stakeholder point of view. The scientific challenge is to develop robust simple, pragmatic, but scientifically sound methodologies, which can provide communities and decision-makers with tools to define and monitor the evolution, current condition and biological performance of marine ecosystems and bioregions.

These approaches include: (i) multidisciplinarity, inherent in the teams involved in their implementation; (ii) integration of biotic and abiotic factors; (iii) accurate and validated methods for determining ecological integrity; (iv) accurate and validated methods for determining the extent and effect of human uses and impacts; (v) adequate indicators to follow the evolution of the monitored ecosystems; and (vi) the use of protected areas as means of conserving and managing viable representative examples of marine environments especially coastal areas where greatest anthropogenic inputs occur. Finally there should be some early warning systems for abrupt changes in environmental conditions.

Arising out of the above, a special session on 'Integrative tools and methods in assessing ecological integrity in estuarine and coastal systems' was organised to discuss all the abovementioned topics at the 'EcoSummit 2007–Ecological Complexity and Sustainability' conference in Beijing (China), in May 2007, The debate among the attendees of this session resulted in this contribution, which reviews the current situation of the integrative ecological assessment worldwide, by presenting several examples from several continents e.g. Africa, Asia, Australia, Europe, and North America.

2. Current situation in North America

2.1. Legislative framework

Canada's legislative framework and application have been discussed by Foster et al. (2005); O'Boyle and Jamieson (2006), and Canessa et al. (2007). In the United States (USA) the main legislation for prevention and study of pollution is based on the Clean Water Act (CWA) of 1972, Air Pollution Prevention and Control Act of 1977, Coastal Zone Management Act of 1972, Harmful Algal Bloom and Hypoxia Research and Control Act of 1998, and, most recently, the Oceans Act of 2000.

Responsibility for monitoring and assessment of water quality in the USA is shared by federal agencies, primarily the Environmental Protection Agency (EPA) and the National Oceanic and Atmospheric Administration (NOAA; Fig. 1). The EPA is charged with regulating most aspects of water quality under the federal CWA (USEPA, 2003). This establishes that, wherever possible, water quality must provide for the protection and propagation of fish, shellfish, and wildlife, for recreation in and on the water and/or protection of the physical, chemical, and biological integrity of those waters. States and tribes designate uses for their waters in consideration of CWA goals and establish water quality criteria to

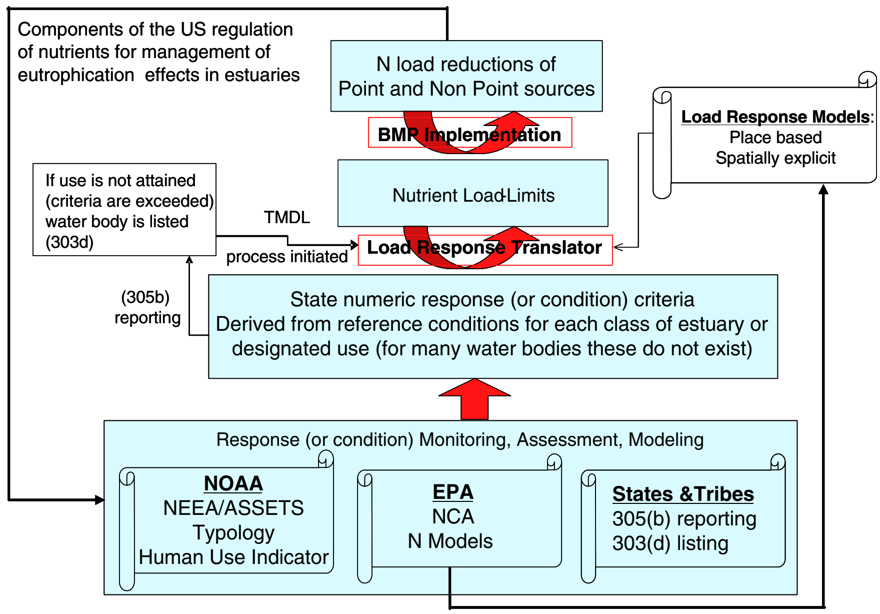

Fig. 1. Roles, responsibilities and interactions of US Agencies for estuarine and coastal environments (EPA, NOAA, States).

protect integrity and uses. The CWA Sections 305(b) and 303(d) state reporting requirements require regular monitoring designed to identify waterbodies that do not meet criteria for designated uses (Keller and Cavallaro, 2008). These waterbodies are included on the Section 303(d) list of impaired waters which establishes protocols that must be followed to mitigate pollution induced impacts (USEPA, 2003; regarding impairment between the CWA, Keller and Cavallaro (2008) can be consulted).

Responsibility for implementing standards and criteria, and for monitoring to assess attainment, is generally delegated by EPA to state water management authorities. States and tribes are required to report periodically to the EPA on water quality conditions, and to develop plans to remedy impacts when they occur. EPA and NOAA support regulatory decisions by providing research and assessment results, and they share some management responsibilities (e.g. Coastal Zone Management Act Reauthorization Amendments (CZARA) Section 6217 coastal non-point pollution control program; CZMA, 1996).

Using nutrients as an example, EPA in 1998 developed the National Strategy for the Development of Regional Nutrient Criteria (USEPA, 1998, 2001c). This strategy detailed EPA's intention to develop technical guidance manuals for four types of waters (lakes, rivers, estuaries/coastal waters, wetlands), which can be seen at: http://www.epa.gov/waterscience/criteria/nutrient/guidance/marine/. The approaches described in the manuals have been applied by EPA and resulted in publication of 26 ecoregional nutrient criteria documents for freshwaters. To date, there have been no such criteria established for estuaries, however, there is a guidance manual (USEPA, 2001a).

2.2. Tools and methodologies used in assessing ecological integrity

Several methods are used by US EPA and NOAA to evaluate ecological integrity or the condition status of coastal waters. The NOAA's National Status and Trends Program (NS&T) gauges the spatial distribution and temporal trends of chemical contamination and develops indicators to evaluate environmental contaminant exposure. Data from NS&T fixed sampling sites are used to assess the distribution, concentration and extent of chemical impacts at a given point and over time, and are important for planning future resource management and restoration activities. The NS&T includes the Mussel Watch Project, Bioeffects Assessments, and the National Estuarine Eutrophication Assessment/Assessment of Estuarine Trophic Status (NEEA/ASSETS; http://ccma.nos.noaa.gov/stressors/pollution/nsandt/) and is designed to address requirements of the CWA.

The EPA's National Coastal Assessment (NCA) Program also surveys the condition of the nation's coastal resources. The Program is implemented through a federal–state partnership and is designed to fulfil section 305(b) of the CWA, which requires EPA to report periodically on the condition of the nation's waters (USEPA, 2003). Data from NCA sites are selected through a statistical random sample design and used together with site specific data from NOAA's NS&T Program and from other national programs to provide regional and national results for five primary indices: Water Quality (WQI), Sediment Quality (SQI), Benthic (BI), Coastal Habitat (CHI), and Fish Tissue Contaminants (FTCI; NCCR1, NCCR2; USEPA, 2001a, 2001b, 2005). These indices provide information on both ecological condition and human use of estuaries.

Results of the two methods for eutrophication assessment are compared here and the EPA NCA BI is also highlighted. For additional benthic indices (e.g. Index of Biological Integrity (IBI)) consult Díaz et al. (2004). All are designed to evaluate conditions and some also address causes of impacts with the intent to inform management.

2.2.1. NEEA/ASSETS

NOAA's eutrophication assessment examines nutrient related water quality problems at individual system, regional and national scales (Bricker et al., 1999; NOAA, 1996, 1997a, 1997b, 1997c, 1998). The recent update examines changes that have occurred since the early 1990s (Bricker et al., 2007). The NEEA is complementary to the National Research Program for Nutrient Pollution in Coastal Waters (Howarth et al., 2003), it interacts with the EPA NCA, and it supports efforts by US states and the European Commission (EC) member states to fulfil requirements of the CWA section 305(b) and the EU WFD (e.g. COAST, 2003; OSPAR, 2002), respectively. The method is described here in brief (for details see Bricker et al., 1999, 2003, 2007; Ferreira et al., 2007b; Scavia and Bricker, 2006, www.eutro.org, www.eutro.us).

(i) Pressure-Influencing Factors (IF) are determined by a matrix that combines the magnitude of nutrient inputs from the watershed with a measure of the system's ability to dilute or flush the nutrient inputs (i.e. susceptibility). The magnitude of loads is determined by a model that compares anthropogenic loading, from monitoring data or model estimations (e.g. USGS SPARROW model, Smith et al., 1997, and WATERSN model, Castro et al., 2001; Whitall et al., 2003, 2004), with natural background concentrations. The model factors in possible oceanic sources providing insight to the success of potential watershed-based management measures.

(ii) State-Overall Eutrophic Condition (OEC) is based on five variables that are divided into two groups: (1) *primary symptoms* that indicate early stages of eutrophication (chlorophyll *a* (Chl) and macroalgae); and (2) *secondary symptoms*, indicative of well-advanced problems (low dissolved oxygen (DO), losses of submerged aquatic vegetation (SAV), and occurrence of nuisance and/or toxic algal blooms (HABs)). An area-weighted-estuary-wide value for each variable is determined based on concentration, spatial coverage, and frequency of occurrence of problem conditions. The overall OEC, falling into one of five categories (i.e. High, Moderate High, Moderate, Moderate Low or Low) is determined by a matrix that combines the average score of *primary symptoms* and the highest score (worst impact) of the three *secondary symptoms*, thus giving the *secondary symptoms* a higher weighting in a precautionary approach.

(iii) The expected Response-Future Outlook (FO) or future condition (worsen, no change, improve) is determined by combining susceptibility of the system with expected changes in nutrient loads. Predictions of future loading (increase, decrease, unchanged) are based on predicted changes in population and watershed uses, mitigated by planned management actions.

(iv) ASSETS Synthesis: IF, OEC and FO are then combined into a single rating for the estuary resulting in a rating of: Bad, Poor, Moderate, Good or High.

Modifications to the NEEA/ASSETS include development of a type classification based on physical and hydrologic characteristics that is expected to improve assessment accuracy and management effectiveness. The EPA has also worked to develop a classification (Burgess et al., 2004) which resulted in 11 groupings or types, compared to the 10 NEEA/ASSETS groups (Chapter 6 in Bricker et al., 2007; Kurtz et al., 2006). A human use indicator has also been developed to complement the NEEA/ASSETS water quality indices. Despite its importance, few previous studies have looked at the social and economic costs of eutrophication. A variety of potential human-uses (e.g. fishing, swimming, boating, tourism) could be considered, although, adequate data are not available for most activities (Bricker et al., 1999; USEPA, 2005). Fishing is important in most estuaries and is usually impacted directly by eutrophication; data are available through the US National Marine Fisheries Service (NMFS), Marine Recreational Fisheries Statistics Survey (MRFSS) which regularly conducts surveys of recreational fishing activity and success in most US estuarine systems. MRFSS fish catch data can be combined with water quality data to determine whether recreational fishing catch rates are related to eutrophic conditions (e.g. Lipton and Hicks, 1999, 2003; Bricker et al., 2006). With additional analysis, potential lost economic value can be estimated using techniques such as travel cost and random utility models (Herriges and Kling, 1999), or by benefits transfer (Walsh et al., 1992).

2.2.2. EPA National Coastal Assessment: water quality and benthic indices

(i) The WQI is the NCA indicator that describes nutrient related conditions. The WQI combines the status of five indicators: dissolved inorganic nitrogen (DIN), dissolved inorganic phosphorus (DIP), Chl, water clarity, and DO. Samples are taken once per year at randomly selected statistical sites during a summer index period (June–October; USEPA, 2001a). The WQI is intended to characterize acutely degraded conditions within coastal regions during the index period and is not expected to capture site-specific detail. By comparison, NEEA/ASSETS evaluates systems, then synthesizes individual results to regional and national levels.

Each WQI indicator is assessed for each site. The five indicators are given equal weight and are combined to give an overall rating for the site, compared to the NEEA/ASSETS which gives *secondary symptoms* a higher weight. A regional rating for each NCA indicator is developed based on combined results for individual sites within the region. A national picture is then developed from regional results.

(ii) The NCA BI is a set of regionally-based or site-specific benthic indices of estuarine environmental condition that reflect changes in diversity and population size of indicator species to distinguish degraded from undegraded benthic habitats (Engle et al., 1994; Weisberg et al., 1997; Engle and Summers, 1999; Van Dolah et al., 1999). The indices reflect changes in benthic community diversity and the abundance of pollution-tolerant and pollution-sensitive species. A high BI rating is indicative of a wide variety of species, a low proportion of pollution-tolerant species, and a high proportion of pollution-sensitive species. A low BI rating indicates that benthic communities are less diverse, have more pollution-tolerant species, and fewer pollution-sensitive species than might be expected.

2.3. Some examples of integrative assessment

2.3.1. NEEA/ASSETS

The NEEA/ASSETS method was applied to 141 individual systems, though not all had adequate data for complete analysis (Bricker et al., 2007). The majority of systems that were assessed (36 of 64) had high IF ratings indicating that inputs of nitrogen from human related activities were large compared to the capacity to dilute or flush nutrients. High nitrogen loads were largely attributed to the influence of dense coastal populations.

Eutrophication is a widespread problem with the majority of systems assessed (64 of 99 or 78% of assessed estuarine area) rated as having moderate to high levels of eutrophication (Fig. 2; Chapter 4 in Bricker et al., 2007). OEC and symptom expressions were geographically variable, through the mid-Atlantic (a region of greatest population density), which was most impaired. The most frequently noted causes of impacts were agricultural activities (crops and animal husbandry), urban runoff, wastewater treatment plants and atmospheric deposition. Comparisons of results from early 1990s and 2004 showed no appreciable change in assessed systems with moderate, moderate high and high level eutrophication impacts (Fig. 3), despite a national population increase in coastal counties of 13% between 1990 and 2003 (Crossett et al., 2004). In 1999, 68% (84 of 124 assessed systems) had moderate to high levels of eutrophication, in the 2004 study 65% (64 of 99 assessed systems) had moderate to high levels of eturophication. The increase in unknowns is primarily a result of the way the data were collected with personal visits and workshops held in the 1999 study and self-reporting at a website, with minimal personal contact in the follow-up study.

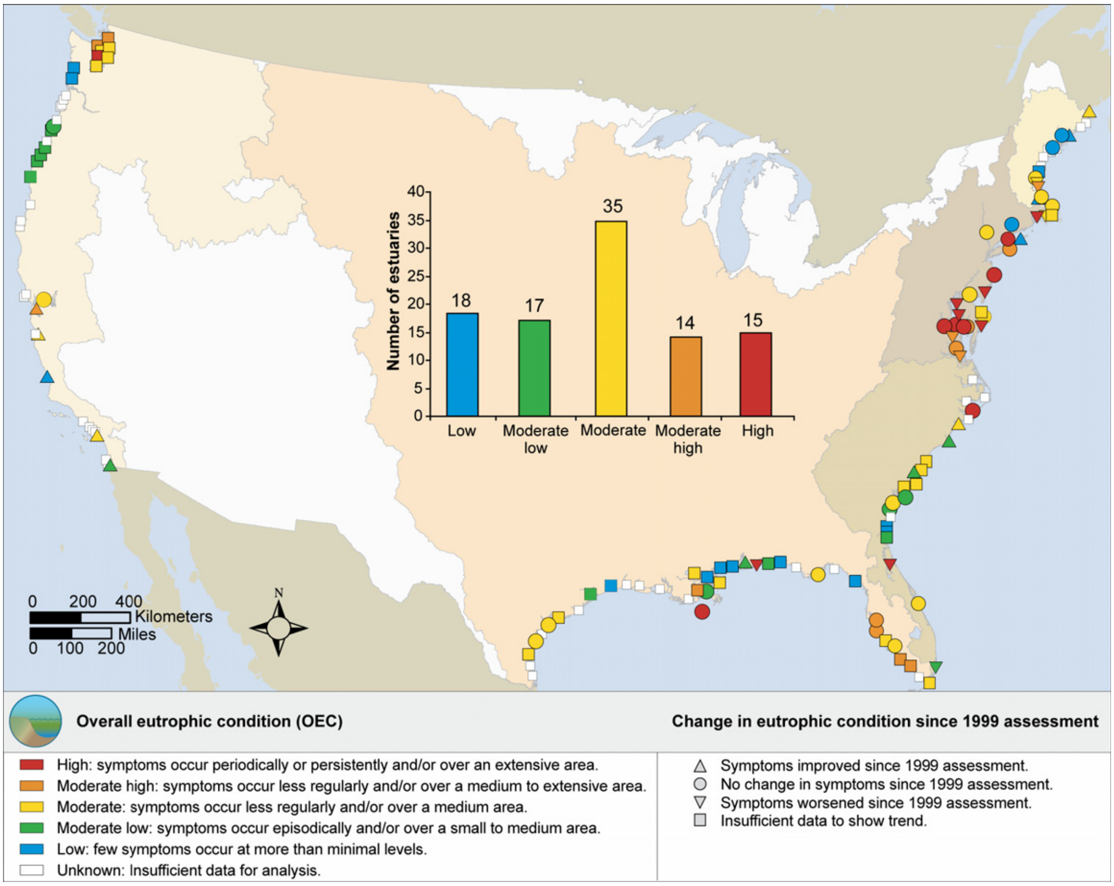

Fig. 2. Overall eutrophic condition on a US national scale.

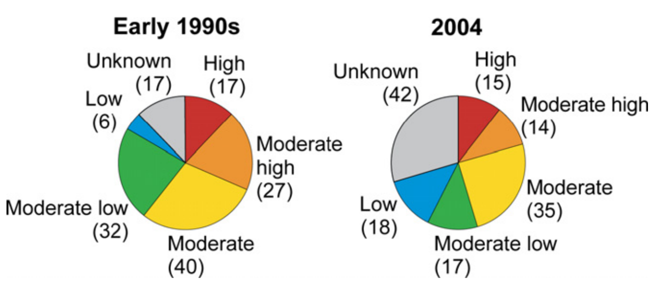

Fig. 3. Number of estuaries in each eutrophication category in the early 1990s (1999 assessment; Bricker et al., 1999) and 2004 (Bricker et al., 2007), in US.

The FO predicts worsening conditions by 2020 for 65% of the estuaries, and improvements for 20%. This is largely based on national population estimates that suggest increases of 12% by 2020 (Crossett, pers. comm.), a bleak outlook for the nation's estuaries; future outlook was not determined for 67 systems, illustrating uncertainty in these conclusions.

Adequate data were available for determination of an ASSETS rating for 48 systems. Only one system (Connecticut River) was rated as high quality, while five were rated as good, 18 as moderate and the remainder rated as poor or bad. In addition to USA systems, this method has been applied internationally (www.eutro.org). The intent is to share lessons learned and encourage pro-active approaches for protection and maintenance of estuarine health globally.

A human use indicator was developed and applied to Barnegat Bay, New Jersey; an excellent candidate for the application of this indicator due to extensive recreational fishing activity. Salinity, temperature, and DO data for Barnegat Bay were averaged by month and year and then matched to the month and year of fishing trips from the NOAA MRFSS database. Summer flounder, the most sought after species in Barnegat Bay, is a good indicator of the human use impacts of eutrophication. The solid line in Fig. 4 shows the average actual catch of summer flounder per month for the period 1997–2002. The statistical model was used to predict summer flounder catches under improved water quality conditions. Specifically, an upper limit on Chl concentrations was set so that sample averages could not exceed 7.12 $\mu g\,L^{-1}$, and a lower limit on dissolved oxygen was set at 6.51 $mg\,L^{-1}$. The dashed line in Fig. 4 represents predicted summer flounder catches under improved water quality conditions. The distance between the two lines is the impairment due to eutrophication. Overall, the catch of summer flounder is reduced by water quality impacts from the predicted average of 1.25 fish per trip to 0.92 fish per trip, a 26% reduction. Using net value costs for mid-Atlantic fisheries determined by McConnell and Strand (1994), it is estimated that eutrophication impacts cost Barnegat Bay fishermen an average of $25.4 million per year in net benefits for this species alone (Chapter 6 in Bricker et al., 2007).

2.3.2. EPA National Coastal Assessment

The NCCR2 summarizes results by region to show that the overall condition of estuaries in the US is fair (Fig. 5). Only the

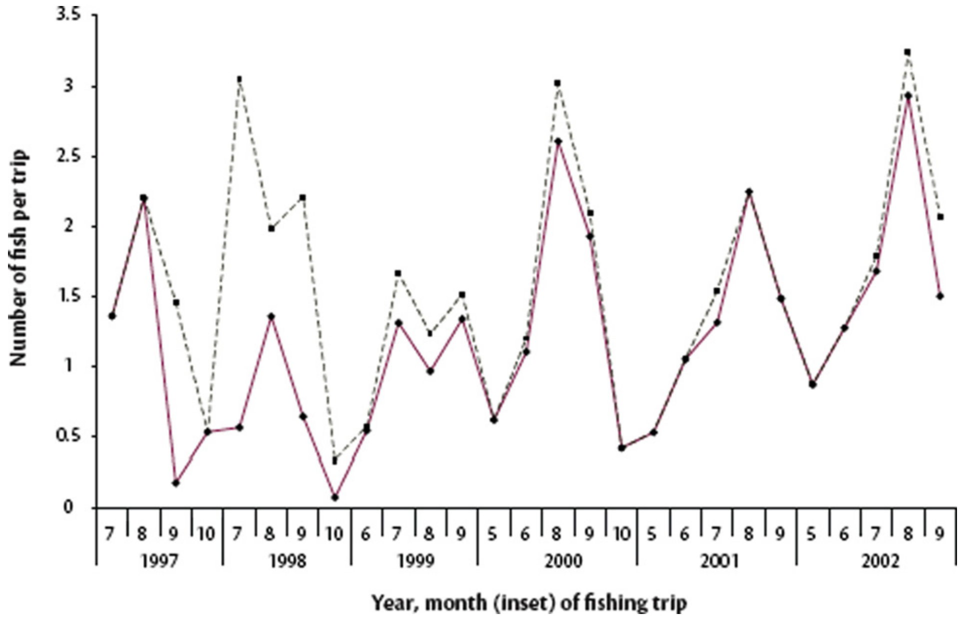

Fig. 4. Barnegat Bay (US) monthly average summer flounder actual catch per recreational fishing trip (solid line), and predicted catch rates under improved water quality (WQ) conditions (dashed line).

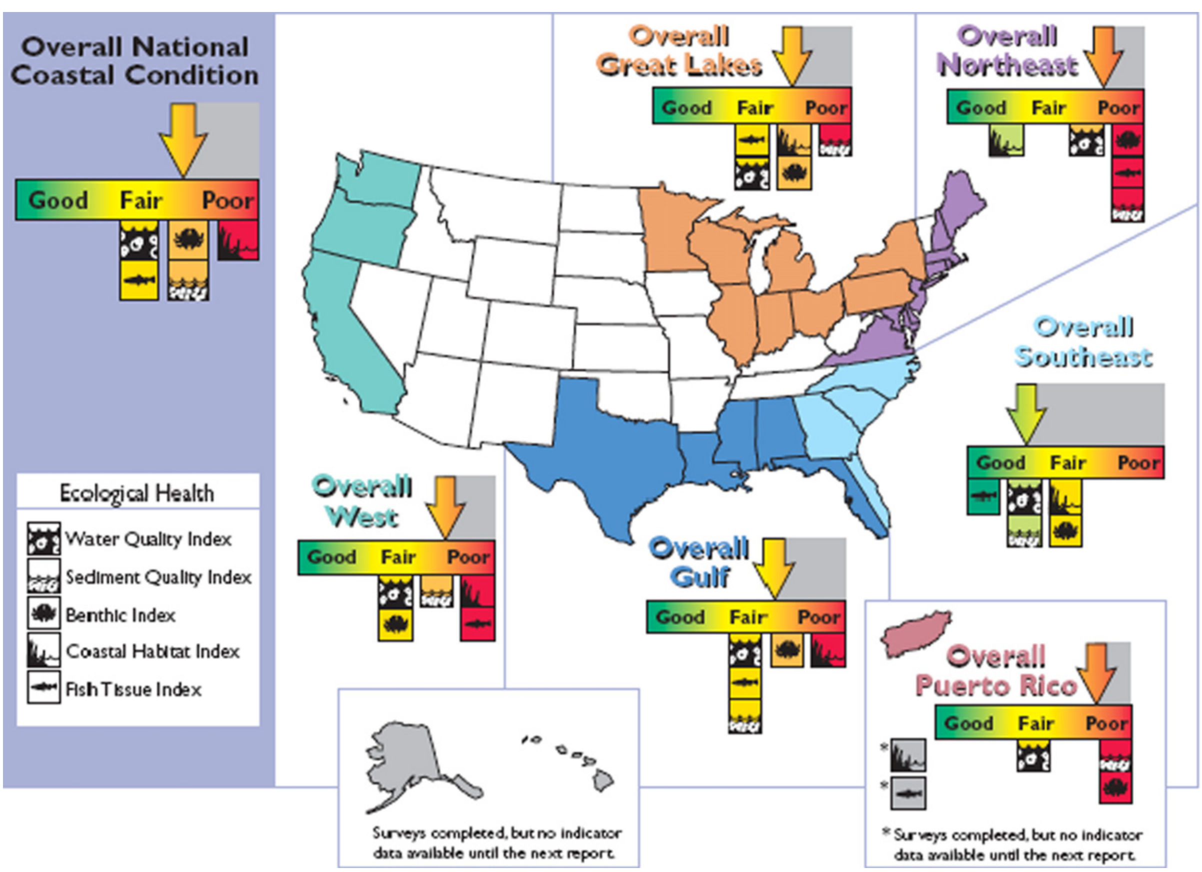

Fig. 5. Overall US national and regional coastal condition between 1997 and 2000 (USEPA, 2005).

CHI received a poor overall rating and only the FTCI was rated good for any region. The WQI, equivalent to the OEC of the NEEA/ASSETS, the BI and SQI were rated fair to poor. About 28% of estuarine area is impaired for aquatic life use, 22% is impaired for human use, and an additional 44% is threatened for both uses. The EPA NCA reported no significant changes in overall environmental condition on a national basis from the early to the late 1990s, but the WQI was reported to improve over the same time period.

3. Current situation in Africa

3.1. Legislative framework

The most recent report on the current situation of the overall African continent was published by the United Nations Environment Programme (UNEP), the situation of coastal and marine environments was reported by Arthuron and Korateng (2006). As other regions worldwide, coastal populations in Africa continue to grow, and pressures on the environment from land-based and marine human activities increased within the last 50 years; coastal and marine living resources and their habitats are being lost or damaged in ways that have diminished biodiversity and thus decreased opportunities for livelihood and aggravating poverty (Arthuron and Korateng, 2006). They have identified key concerns over the continent, including natural disasters, poverty, overexploitation of offshore fisheries, exploitation of non-living resources (oil, diamonds, etc.), modification of river flows to the coast by damming and irrigation, and other pollution from land, marine and atmospheric sources. One of the conclusions is the need for developing and promoting integrated coastal management plans, with strong inter-sectoral and international linkages, including those with catchment management authorities with responsibilities for Integrated Water Resource Management.

On these regards, one of the most advanced countries implementing such policies within the continent is South Africa. This is a dry country with an average rainfall of less than 500 mm year^{-1}, well below the global average, and with expectations of declines associated with climate change over the next century. Reviews of environmental legislation in South Africa, particularly relating to aquatic resources and the coastal zone, are associated with reviews by the Council for the Environment (1989, 1991) which began proposing policies for coastal zone management. The status of coastal management was subsequently reviewed by Sowman (1993), preceding the development of a green paper (Department of Environmental Affairs and Tourism, 1998) focussed on sustainable coastal development, followed by a white paper (Department of Environmental Affairs and Tourism, 2000) on the same topic. The appearances of these policy documents were paralleled by publications by Glazewski (1997) and Glavovic (2000a, 2000b) aimed at converting the policies and concerns articulated in the white paper into an integrated coastal management bill and ultimately a national Coastal Management Act (www.deat.gov.za) which is presently in the process of ratification.

On the aquatic resources side, including freshwater, estuarine and marine environments, the National Environmental Management Act 107 of 1998 was superceded/complemented by the National Water Act of 1998 (www.dwaf.gov.za/documents/publications). This new act represented a radical digression from the philosophy inherent in the historical approach to the management of aquatic resources, i.e. that aquatic environments, particularly fresh water and estuarine systems, were granted a legal *persona* in that the dependence of the functionality of these systems on a minimal level of freshwater flow was given a legal status which had to be taken into account when any water abstraction was contemplated. The arguable premise that aquatic systems, such as wetlands, rivers and estuaries, are ultimately dependent on minimal levels of freshwater availability, beyond which their functionality will be impaired, clearly generates the question as to what is this level and how it might be established. This aspect will be dealt with in the next section.

3.2. Tools and methodologies used in assessing ecological integrity

In the present context the emphasis will be on the determination of the freshwater requirements of estuaries (Department of Water Affairs and Forestry, 2004), henceforth referred to as the "reserve" although the procedure is described as part of a package dealing also with reserve requirements of rivers and wetlands (http://www.dwaf.gov.za/documents/policies/wrpp). Other methodologies have been published for assessing quality using fish (Harrison and Whitfield, 2004, 2006), estuarine health (Cooper et al., 1994; Harrison et al., 2000), or conservation significance (Turpie et al., 2002) as indicators.

3.3. Some examples of integrative assessment

An example of the above where an assessment of the current status was followed by remedial action and the institution of a monitoring system to check on the effectiveness of the measures used is provided by the Mhlanga estuary (29°42′S; 31°6′E) on the northern outskirts of the city of Durban, east coast of South Africa. This small system with an estuarine area of barely 12 ha (Begg, 1978) is nevertheless typical of many of the 73 systems which occur along the 570 km of the KwaZulu-Natal coastline and further south into the Eastern Cape Province. The major physical and chemical features of these systems are determined by the seasonal rainfall, and consequently variable river flow, coupled with strong wave action as well as longshore sand transport which typically result in the closure of these systems during winter low flow periods. Under these conditions tidal action is lost and with it the organisms dependent on an intertidal habitat. Salinities typically fall due to sustained low levels of fresh water input and outward seepage through the bar, but layering may develop if the bar is low enough for overwash to occur during high wave conditions. Water levels behind the bar will rise, depending on the height of the bar, and can result in substantial backflooding such that the overall extent of the aquatic environment, in terms of water column and benthic habitat, increases well beyond that associated with high tides during periods with an open mouth. Under natural conditions this bar would naturally be breached during summer high flow periods but historically (Begg, 1984) this pattern has been disrupted by artificial breaching to prevent flooding of cultivated land or infrastructure in the backflooded areas.

The Mhlanga estuary has over the last 25 years become one of the better known of the smaller KwaZulu-Natal systems by virtue of studies including general surveys of the system carried out in 1980–1981 (Begg, 1984), as well as a more intensive focus on the fish fauna producing information on trophic relationships within the fish community (Whitfield, 1980a), fish distribution in relation to food resources (Whitfield, 1980b) and factors affecting the recruitment of juvenile fish into the estuary (Whitfield, 1980c). Harrison et al. (2000) produced a nationwide assessment of the state of South African estuaries based on the geomorphology, ichthyofauna, water quality and aesthetics. The latter three parameters were rated on a scale of poor, moderate or good. The fish fauna was assessed on the basis of species richness and community composition, the water quality on suitability for aquatic life in terms of dissolved oxygen, ammonia, faecal coliforms, nitrate nitrogen and *ortho*-phosphate and the aesthetics on a "visual appraisal of the state of development in and around the estuary" incorporating *i.a.* any type of anthropogenic influence, algal blooms, odours, noise or invasive plants. The fish fauna and aesthetics of the Mhlanga estuary were rated as good but the water quality as poor. The poor water quality reflects the vulnerability of these small systems during the closed mouth periods when water exchange is minimal and tidal effects non-existent. In

2002–2003 the South African Water Research Commission sponsored a multi-disciplinary study incorporating mouth dynamics, physico-chemical conditions, nutrient conditions, phytoplankton and microphytobenthos, zooplankton, benthos, fish and birds (Perissinotto et al., 2004). This study aimed at contributing to the implementation of measures for reserve determinations for estuaries. In the local context the project focussed on the ‘‘responses of the biological communities to flow variation and mouth state in temporarily open/closed estuaries”, one of which was the Mhlanga.

In summary, the study supported perceptions and interpretations developed some 20 years earlier (Begg, 1984), that the broad natural cycle of summer breaching and winter closure due to the seasonal rainfall pattern was a major driving force in the functioning of these temporarily open/closed systems. Although these estuaries became non-tidal and salinities dropped to virtual freshwater levels during closed periods, with a consequent effect on benthic invertebrate diversity, the fish fauna, which tended to consist largely of juveniles recruited to these nursery grounds during open mouth periods or through overwash, appeared able to handle these low salinities. Retention and accumulation of water behind the bar also resulted in an expanded aquatic and benthic environment relative to that existing under high tide conditions. The increased and stable water column permitted the development of phytoplankton and in turn the development of a zooplankton and planktivorous fish fauna, while the benthos was able to expand in abundance although not in diversity. Optimisation of these processes depended on regular seasonal cycles of breaching or overtopping, allowing fish or invertebrate migration, followed by periods of closure which allowed the accumulation of biomass, both plant and animal, before the next exchange. Disruption of this cycle by artificial breaching and draining of the estuary during winter when water levels normally peak would disrupt this cycle. Additional effects would be imposed by nutrient inputs from agricultural runoff or urban pollution resulting in algal blooms, eutrophication and oxygen depletion.

In the Mhlanga, records of behaviour of the mouth, coupled with historical observations (Begg, 1978, 1984), calculations of the pristine mean annual and monthly runoff and the present situation indicated that outflow of treated water from a sewage works situated upstream of the estuary significantly increased the total flow into the estuary and the frequency of mouth breaching, resulting in the type of impacts described above.

At low input levels the variable quality of the treated effluent was such as to generate localised periodic low or anoxic conditions resulting in fish kills. The increase in water inflow into the river from the sewage works resulted from the fact that the water used in the catchment was derived from other catchments and resulted in an overall increase in the Mhlanga flow. In this situation, the impacts on the estuary arose from the rather unusual situation of excess flow rather than the more common problems arising from water abstraction.

The provisions of the reserve determinations allow for either the maintenance of an existing acceptable ecological status or the implementation of measures to improve the ecological status of an estuary. In this case the measures that have been implemented by the local municipality involve the installation of a pipeline to transport the excess water to an adjacent catchment which has been subject to significant abstraction as well as improved treatment of the waste water from the sewage works. A monitoring operation has been implemented to assess the success or negative effects of the reduction in water input. A closed circuit camera is being used to monitor mouth dynamics including the possibility of anthropogenic interference.

4. Current situation in Asia

4.1. Legislative framework

In Asia, China possesses comprehensive laws and regulations dealing with coastal areas, including over 25 legislative instruments (Zhijie, 1989; Cao and Wong, 2007) addressing issues such as regulations on dumping (1985, 1992), Marine Protected Areas (1994, 1995, 1997), Environmental Impact Assessment (2002) (Lindhjem et al., 2007), and the implementation of the UNCLOS Convention in 1998 (Keyuan, 2001), together with specific dispositions e.g. for fisheries (Keyuan, 2003).

Integrated Coastal Zone Management (ICZM) requires appropriate legislation, and benefits from the existence of strong public participation and independent coordination (Lau, 2005) thus avoiding the twin pitfalls of marginalizing stakeholders and encouraging sectoral management. Participation and coordination issues are not easily achieved in the present-day PRC, however a pilot structure for ICZM exists in Xiamen (Xue et al., 2004; Peng et al., 2006) and is planned for Shanghai (Lau, 2005). Nevertheless, the concept of integrated assessment, as set down e.g. in the European WFD, does not seem to be widely applied.

4.2. Tools and methodologies used in assessing ecological integrity

A review of Chinese literature indicates that progress towards the application of extended tools used to assess ecological integrity is incipient (e.g. Xue et al., 2004; Huang et al., 2006; Leung, 2006). At the same time, there is a growing national concern in regard to shifting from methods based on water chemistry and simple biological diversity metrics to more sophisticated approaches which use ecological indicators of degradation to provide a more robust assessment; two methods are reviewed below: (i) coastal eutrophication assessment, which is compared with the ASSETS model (discussed in the North America section), and (ii) the integrated Comprehensive Index Assessment Method (CIAM), for marine resource assessment.

4.2.1. Coastal eutrophication assessment

The application of the ASSETS index to Chinese coastal waters (Xiao et al., 2007) provided an opportunity to review the methods currently used in China for assessing coastal eutrophication. Historically, this assessment has focused on chemical indices, using techniques such as the Nutrient Index Method (NIM), to study the effects of system loading by nutrients, and may therefore be considered ‘‘Phase I” (*sensu* Cloern, 2001) approaches (Yao and Shen, 2005).

The NIM, proposed by the Chinese National Environmental Monitoring Center, is based on a nutrient index (N_i) in seawater (Lin, 1996), calculated using Eq. (1)

$$\frac{C_{\mathrm{COD}}}{S_{\mathrm{COD}}}+\frac{C_{\mathrm{TN}}}{S_{\mathrm{TN}}}+\frac{C_{\mathrm{TP}}}{S_{\mathrm{TP}}}+\frac{C_{\mathrm{Chla}}}{S_{\mathrm{Chla}}} \tag{1}$$

where: C_{COD}, C_{TN}, C_{TP} *and* C_{Chla} are measured concentrations of Chemical Oxygen Demand (COD), total nitrogen, total phosphorus (in mg L^{-1}) and chlorophyll *a* (in μg L^{-1}) in sea water, respectively. S_{COD}, S_{TN}, S_{TP} and S_{Chla} are standard concentrations of COD (3.0 mg L^{-1}), total nitrogen (0.6 mg L^{-1}), total phosphorus (0.03 mg L^{-1}) and chlorophyll *a* in seawater (10 μg L^{-1}), respectively (Lin, 1996). If N_i is greater than 4 the seawater is considered eutrophic.

While NIM is widely used in Chinese coastal systems, research in recent decades has identified key differences, in nutrient enrichment, between the responses of limnology-originated methods, such as this one, and those of coastal-estuarine ecosystems (Cloern, 2001; Bricker et al., 2003; Ferreira et al., 2007a, 2007b). Partly, this is because coastal environments systems with similar pressures show widely varying responses, so there is often no clear relationship between nutrient forcing and eutrophication symptoms. In particular, nutrient concentrations have often been shown to be poor indicators of eutrophication symptoms (e.g. Tett et al., 2003), since ecosystem responses are modulated by factors such as morphology, tidal range, natural turbidity and water residence time.

4.2.2. Marine resources and ecological quality assessment

Most estuaries and coastal inlets and embayments are also important fishery grounds. The Comprehensive Index Assessment Method (CIAM) was developed and applied to evaluate the ecological quality of the marine fisheries environment for major coastal areas of China (Jia et al., 2003, 2005; Ma et al., 2006). CIAM incorporates four assessment modules: seawater quality, nutrient level, primary production level, and diet organism richness; the index is the mean value of the sub-indices.

Seawater quality assessment in CIAM evaluates water pollution status. The main components of coastal pollution in China include organics (indicated as COD), eutrophication, total hydrocarbons, and heavy metals. The organic pollution status is assessed using the Organic Pollution Index method (the *A* value), while the status of other types of pollution is assessed using Factorial Analysis (*Pi*) according to Fishery Water Quality GB 11607-1989 and Sea Water Quality GB 3097-1997. The classification of sea water quality used in CIAM is given in Table 1. The *A* value is directly used as P_i during the comprehensive ecological quality assessment stage.

Concentrations of dissolved nitrogen (DIN), phosphate and silicate and their ratios are used in the sea water nutrient level assessment using a NIM (the *E* value). The seawater nutrient level is classified as: E = 0–0.5, Grade: 1, Nutrient level: Low; E = 0.5–1, Grade: 2, Nutrient level: Medium; and E > 1, Grade: 3, Nutrient level: Eutrophic. The *E* value is used as P_i in the CIAM.

Primary productivity level and diet organism richness are important indicators for the fishery environment quality status. Since they vary significantly among different areas along the coast, six grades are used to classify the quality status (Table 2). The level for each item is taken as P_i in the final CIAM. The CIAM index is calculated as

$$I_p = \frac{1}{n}\sum_{n}^{i=1} P_i \qquad (2)$$

I_p, comprehensive ecological quality index; *P*, index level assessed for indicator *i* (i.e. indices for seawater quality, nutrient, primary productivity and diet organism richness); *n*, total number of indicators.

The CIAM of the marine fishery environment is classified into six grades according to quality index I_p, excellent, fine, relatively fine, moderate, poor and very poor (Table 1).

4.3. Some examples of integrative assessment

4.3.1. Eutrophication assessment of Jiaozhou Bay, Northeast China

ASSETS was applied to Jiaozhou Bay, and compared with the evaluation of eutrophication status using chemical indices. Jiaozhou Bay is on the west coast of the Yellow Sea (35°57′–36°18′N, 120°06′–120°21′E), it has a surface area of 397 km^2 and a mean depth of 7 m (Editorial Board of "Bays in China", 1993). It is a semi-enclosed body of water, connecting to the Yellow Sea through a 2.5 km channel, and has a mean tidal range of 2.5–3.0 m, but tides can reach 4.2 m, resulting in a well mixed water column (Liu et al., 2004).

The bottom of the bay contains spawning, nursery and feeding grounds for fish, and intensive mariculture. Historically, this has focused on the bay scallop (*Argopecten irradians*) and Pacific oyster (*Crassostrea gigas*), cultivated on longlines. Recently, the longlines have been removed and Manila clam (*Tapes philippinarum*) is now cultivated, with a production of 200,000 t year^{-1}. The main issue in the bay is an increase in both the frequency and magnitude of harmful algal blooms (HABs), since the 1990s, although most events are non-toxic (Han et al., 2004).

Jiaozhou Bay has a volume of 1900×10^6 m^3, which, with a nitrogen load into the bay of 30 ton per day (Wang et al., 2006), results in a *High* rating for the nutrient component of IF (0.933). Strong tidal mixing and high river discharge (8×10^8 m^3 year^{-1}) contribute to moderate flushing and dilution potential (Editorial Board of "Bays in China", 1993). However, the intensive 'top-down' control of the food web has a significant impact, mitigating eutrophic symptoms.

Table 1
Classification of sea water quality used in CIAM, including organic pollution assessment, TPH (Total Petroleum Hydrocarbon), and heavy metal pollution (adapted from Jia et al., 2003); and comprehensive ecological quality grade of marine fishery environment (Jia et al., 2003)

	Quality index (*A* value)	Grades	Quality assessment
Organic pollution assessment	<0	1	Excellent
	0–1	2	Clean
	1–2	3	Relatively clean
	2–3	4	Slight pollution
	3–4	5	Medium pollution
	>4	6	Serious pollution
	P_i	Grades	Quality Assessment
TPH and heavy metal pollution assessment	<0.4	1	Background
	0.4–0.6	2	Clean
	0.6–0.8	3	Relatively clean
	0.8–1.0	4	Slight pollution
	1.0–2.0	5	Pollution
	>2.0	6	Serious pollution
	Index range	Grades	Quality status
Comprehensive ecological quality grade of marine fishery environment	0.2	1	Excellent
	0.2–0.4	2	Fine
	0.4–0.6	3	Relatively fine
	0.6–0.8	4	Moderate
	0.8–1.0	5	Poor
	>1	6	Very poor

Table 2
Grade of primary productivity and diet organism richness (Jia et al., 2003)

Item	Grades					
	1	2	3	4	5	6
Status	Low	Relatively low	Medium	Relatively high	High	Super high
Index level	>1.0	1.0–0.8	0.8–0.6	0.6–0.4	0.4–0.2	<0.2
Primary productivity (mg C mg^{-1} day^{-1})	<200	200–300	300–400	400–500	500–600	>600
Phytoplankton (10^4 ind m^{-3})	<20	20–50	50–75	75–100	100–200	>200
Zooplankton (mg m^{-3})	<10	10–30	30–50	50–75	75–100	>100
Benthos (g m^{-2})	<5	5–10	10–25	25–50	50–100	>100

The susceptibility component of the IF, based on only natural circumstances, is *Moderate* but when shellfish aquaculture is taken into account, the overall susceptibility is *Low*. This is one example of the difficulty in universal application of such methods, since ASSETS must be potentially adapted to incorporate local societal factors. The combination of *High* nutrient load and *Low* susceptibility gives an overall IF rating of *Moderate Low*.

Chlorophyll *a* is the only indicator with information for the primary symptoms. No information exists for macroalgae, which was therefore classified as *Unknown*. Maximum chlorophyll *a* values in Jiaozhou Bay did not exceed the threshold indicated in ASSETS for *Medium* eutrophic conditions. ASSETS uses the 90th percentile value of annual data to provide a typical maximum value for chlorophyll *a*, and in the bay this value is 4–5 μg L^{-1}, i.e. in the *Low* category. Therefore, the rating for primary symptoms is *Low* based on chlorophyll *a*.

Data for dissolved oxygen were collected from various sites over an annual cycle, as a secondary symptom. No information was found for SAV, but due to the historical scale of kelp aquaculture in the bay, the level for this secondary symptom would be at worst *Low*.

Few values below the threshold for biologically stressful dissolved oxygen condition (5 mg L^{-1}) were detected in Jiaozhou Bay. As described earlier, the 10th percentile is applied to provide a more consistent minimum value for dissolved oxygen. In this system, the 10th percentile for annual dissolved oxygen data is between 6 and 7 mg L^{-1}, indicating no problems with regard to this indicator.

Some 69 harmful algal species were observed in Jiaozhou Bay (Han et al., 2004). Toxic blooms are episodic, usually lasting for only a few days (e.g. Huo et al., 2001). Therefore, the symptom of "nuisance and toxic blooms" is rated as *Low*.

The highest level of the three secondary symptoms falls into the *Low* category, and the OEC resulting from the combination of primary and secondary symptoms for this system is *Low*.

The estimate based on the current development scenario gives a 9.3% human population increase over 20 years (P.R.C. National Bureau of Statistics, 2001). In addition, Qingdao (the main land nutrient source, pop. 8 million) is strongly promoting its tourism industry and less space is available for mariculture in the bay. Accordingly, reduced top-down control on primary production could lead to increased eutrophic symptoms. Additionally, Qingdao's preparations to host the Olympic Sailing Regattas in 2008 have focused attention on water quality issues and mitigation of eutrophic symptoms. The government has pledged to build more wastewater treatment plants in the near future, and more restrictive pollutant emission regulations are coming into effect (Wang et al., 2006).

As a whole, nutrient loads are expected to decrease, despite the increase in the urban population, and the water quality is likely to improve. FO can therefore be considered to be *Improve low*. Table 3 summarizes the results obtained from the application of ASSETS to Jiaozhou Bay, which resulted in an overall score of *High Status*, indicative of minimal or no eutrophication problems.

The results are better than expected due to top-down control related to intensive shellfish mariculture. This has important implications for successful management of nutrient-related problems, which are not captured by the Chinese "Phase I" NIM, which classifies the system as *Eutrophic*. Moreover, the NIM cannot by definition be clear indicators for a large system, because there is no allowance for spatial differences in impact level within a waterbody. The evaluation of the systems using salinity zones, as in ASSETS, contributes to a more accurate evaluation of the system and subsystems, necessary to target management efforts. A comparison of various methods for eutrophication assessment is shown in Table 4.

The top-down control of the food web in Jiaozhou Bay suggests a feasible way to manage coastal eutrophication. These control strategies, which have traditionally been used in China, are now being discussed in the EU and the USA (e.g. Lindahl et al., 2005; Ferreira et al., 2007b). Paradoxically, the Chinese, USA and other governments and scientists currently focus mainly on a bottom-up approach in improving water quality, though there is plenty of scope to promote top-down control. Water quality data col-

Table 3
ASSETS application to Jiaozhou Bay

Index	Method	Indicator	Level of expression	Index result	ASSETS score
IF[a]	Susceptibility	Dilution potential	Moderate	Low (due to intense shellfish aquaculture)	
		Flushing potential	Moderate		
	Nutrient inputs		High		
	PSM[c]	Chlorophyll *a*	Low		
		Macroalgae	No problem		
OEC[b]		Dissolved Oxygen	Low	Low	High
	SSM[d]	SAV loss	Low		
		Nuisance and toxic blooms	Low		
FO[e]	Future nutrient pressure		Decrease	Improve low	

[a] IF, influencing factors.
[b] OEC, overall eutrophic condition index.
[c] PSM, primary symptoms method.
[d] SSM, secondary symptoms method.
[e] FO, future outlook index.

Table 4
Summary of comparison among "Phase I/II" methods (adapted after Bricker et al., 2006)

Methods	Temporal focus	Indicator criteria/thresholds	Combination method
Nutrient Index I	Not specified	Modified after Japanese criteria	Sum of four ratios
Nutrient Index II	Not specified	Modified after Japanese criteria	Ratio of three indicators to their threshold values
OSPAR COMPP	Growing season, winter for nutrients	Individually/regionally determined reference condition	Integration of scores for four categories
EPA NCA	Summer index periods	Determined from American national studies	Ratio of indicators: good/fair indicators to poor/missing data
ASSETS	Annual cycle	Determined from American national studies	Average of primary and highest secondary are combined by matrix

lected in Jiaozhou Bay during 1999–2000 were used to estimate the gross removal of phytoplankton by Manila clams. On the basis of reported bivalve stocks, these organisms remove about 627 t yr^{-1} of chlorophyll *a*, which (considering a carbon:chlorophyll ratio of 50 and a Redfield C:N ratio of 45:7 in mass) corresponds to the removal of almost 4,900 t yr^{-1} of nitrogen, i.e 1.5 million population equivalents, or 17% of the population of Qingdao, and to 45% of the estimated 11,000 t yr^{-1} nitrogen load. Along with economic benefits, the introduction of filter-feeders on a reasonable scale thus allows for cost-effective removal of nutrients and mitigation of eutrophic conditions, which is more environmentally-friendly and sustainable for a coastal system (Shastri and Diwekar, 2006).

4.3.2. Marine resources and ecological quality assessment in the South China Sea

To understand the health status and ecological quality of the fishery environment in the northern South China Sea, a comprehensive and systematic survey program was carried out from 1997 to 2002. This included Taiwan bank, East Guangdong, Pearl River estuary, West Guangdong, Southern waters of Hainan, and Beibu Bay. The quality status of the fishery environment in the northern South China Sea was assessed based on the data on seawater quality, sea water nutrient structure and nutrient level, and primary productivity and diet organism level using CIAM (Jia et al., 2005).

The results showed that the overall water quality index was within the criteria limit of Fishery Water Quality GB 11607-1989 and the Grade criteria of Sea Water Quality GB 3097-1997. The organic pollution index (*A* value) range was 0.411–0.237, and the nutrient index (*E* value) range was 0.10–0.34, which indicated the waters were not organically polluted and the nutrient status was low. The primary productivity of the waters, ranging from Grade 5 to Grade 1, with an annual average of Grade 3, was at a the "medium" level. For the richness of diet organisms, the grade of phytoplankton, zooplankton and benthic organisms were 3, 5 and 4 respectively, within a *relatively high* level in general.

The comprehensive quality assessment results (Table 5) showed that the quality indices of 9 factors (including DO, DIN, PO_4^3, *A*, *E*, primary productivity, phytoplankton, zooplankton and benthos) were all lower than 1.0 and the comprehensive quality index was 0.58, indicating the ecological quality of the overall area was relatively fine. However, the comprehensive ecological quality index of Pearl River estuary, East Guangdong waters, West Guangdong waters and Beibu Bay were all over 0.60; the quality status was *moderate*, far worse than the *fine* level, a sign of environmental degradation along the coast of Guangdong Province. This means that due to the continuous and rapid growth of industry and economy of Guangdong, especially in the Pearl River Delta, more attention should be paid to environmental protection and ICZM.

5. Current situation in Australia

5.1. Legislative framework

Australia adopted an Oceans Policy in 1998 and subsequently established a National Oceans Office that initiated a process of 'marine bioregional planning'. These measures reflected requirements and obligations that arose for Australia as a signatory to the United Nations Convention on the Law of the Sea (UNCLOS) which came into force in 1994. Subsequently, the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act) provided the overarching framework for management of Australia's national and international marine environmental responsibilities.

In 2005 the Australian Government brought its program of regional marine planning directly under the scope of the EPBC Act 1999. As a federal nation, however, the situation is complicated by the history of internal jurisdictional responsibilities for the original three nautical mile territorial sea limit that existed prior to the development of arrangements under UNCLOS. In 1975 a decision of the High Court of Australia upheld legislation vesting

Table 5
Quality indices of the fishery environment of northern South China Sea (according to Jia et al., 2005)

Waters	DIN	PO_4^3-P	*A*	*E*	DO	Primary productivity	Phytoplankton	Zooplankton	Benthos	Comprehensive index
Northern South China Sea	0.27	0.55	0.23	0.34	0.72	0.58	0.53	0.88	0.78	0.58 Relatively fine
Taiwan bank waters	0.29	0.52	0.20	0.28	0.67	0.50	0.37	0.75	0.68	0.51 Relatively fine
East Guangdong waters	0.31	0.61	0.21	0.30	0.78	0.54	0.79	0.88	0.81	0.61 Moderate
Pearl River estuary	0.26	0.58	0.23	0.37	0.74	0.68	0.55	1.00	0.91	0.64 Moderate
West Guangdong waters	0.28	0.54	0.25	0.47	0.72	0.58	0.58	0.88	0.77	0.60 Relatively fine
Southern waters of Hainan	0.24	0.50	0.20	0.30	0.73	0.64	0.36	0.87	0.82	0.55 Relatively fine
Beibu Bay	0.24	0.45	0.20	0.30	0.66	0.72	1.00	0.94	0.94	0.66 Moderate

sovereignty, in respect of the territorial sea, in the Federal Government. The implications of this were addressed by the Offshore Constitutional Settlement of 1979 whereby legislative competence and proprietary rights over the three nautical mile territorial sea were transferred to the States and Territories through the Coastal Waters (State Titles) Act (1980). This revalidated pre-existing state legislation, and restored the title, responsibilities and rights of states and territories with respect to the seas, seabed and subseabed within the three nautical mile territorial sea and internal waters.

As a consequence the States and Territories have primary responsibilities for management of marine environments, natural resources and the impacts of human activities within internal waters and the territorial sea limits. The States and Territories have differing approaches to management of marine ecological integrity.

Federal responsibilities relate primarily to areas beyond the new limit although Section 23(2) of the EPBC Act (1999) gives the Federal Government an overarching capacity to address issues that actions taken outside Commonwealth marine areas that have, will have, or are likely to have a significant impact upon them.

Currently there is no standard set of environmental indicators used across Australia by the states and federal governments. After 2001 the implementation of the Australian state of the environment reports (SOE), an attempt was made at the subsequent Australian and New Zealand environment and conservation council (ANZECC) to obtain some uniformity of indicators across Australia. A core set of 75 indicators was established but efforts to reduce this to a smaller core set were unsuccessful. Subsequently for the SOE 2006 process, a data reporting system (DRS) (http://www.deh.gov.au/soe/DRS) was developed. However a brief review of the various SOE's for each state government, for an evaluation of the status of estuarine and inshore coastal waters, revealed no uniformity or consistent pattern of indicators being used between the states. Virtually all the indicators were physical parameters with no biological indicators being used.

In 2004 a comprehensive report was prepared on estuarine, coastal and marine indicators for regional NRM monitoring by the CRC for Coastal Zone, Estuary and Waterway Management (Souter, 2007). This summarised the issues to be targeted such as inland aquatic ecosystems integrity and estuarine, coastal and marine habitat integrity as well as nutrients, turbidity and surface salinity in freshwater aquatic environments, significant native species and ecological communities and invasive species. For each of these items for targeting the report provides a detailed summary of useful indicators. However, there is no attempt to discuss the actions to be taken by the relevant authorities (of which there are many) to act upon the results obtained from any monitoring undertaken. Most important, the situation is confused by the State/Federal boundaries. It is unclear as to whether any real progress has been made in co-ordinating these measures, and then acting upon them.

5.2. Tools and methodologies used in assessing ecological integrity

In parallel with the development of a broader political and legislative framework for marine environmental management in the context of UNCLOS, Australia was engaged in the implementation of the Great Barrier Reef Marine Park Act (1975) which provides for the Great Barrier Reef Region to be managed for conservation and reasonable use.

With an extent of 350,000 km^2 and little physical survey of areas beyond shipping lanes, preparation for ecosystem scale management of the Great Barrier Reef required development of approaches to implement a working understanding of bioregionalisation in a data-lean environment (Kenchington, 1990). The initial tasks included commissioning of a geomorphological classification of reefs and shoals at a scale of 1:250,000. This was followed by detailed surveys of reefs conducted by the Australian Survey Office and subsequently by the development of 1:250,000 rectified maps of the Great Barrier Reef region drawing on LANDSAT imagery to infill ground survey data (Kenchington, 1990). These maps were used for expert consultation on the distribution of ecological communities and fisheries resources. Data on the occurrence and distribution of biological communities were extremely patchy with major data sources being reef research stations and expeditionary studies. The community and resource distribution maps resulting from expert consultation were then used to seek comment and amendment during a phase of public consultation prior to development of a draft zoning plan for a section of the Great Barrier Reef Marine Park (Kenchington, 1990).

The initial zoning of the Great Barrier Reef took account of seabed communities known from fisheries surveys and production and benthic communities such as *Halimeda* algal beds and sponge beds but focussed largely on coral reef communities. In 1998, the Great Barrier Reef Marine Park Authority started a process of review of zoning in the light of experience, new information and changed circumstances of use and management of the Marine Park (Lawrence et al., 2002).

A major element identified in the context of management responsibilities was the need for an adequate network of highly protected (no take) areas representative of all of the bioregions occurring within the Great Barrier Reef World Heritage Area.

There are 30 reef and 40 non reefal bioregions which are characterised by physical and biological features (Day et al., 2002). A decision was made that at least 20% of each of these bioregions was to be declared as a 'no take' zone in order to meet the requirements to maintain World Heritage values. After the complete rezoning, many of these bioregions were represented by more than 20% no take. The location of these no take zones took into account economic and social factors to minimise, as far as practicable, the impact of these zones on the users of the GBR. Ongoing monitoring during the next few years will attempt to obtain data to support the declaration of these no take zones and identify if additional such zones need to be declared, especially in the light of ongoing climate change.

5.3. Some examples of integrative assessment

An example is the Integrated Marine and Coastal Regionalisation for Australia (IMCRA, 1998). The methods developed for assessment of the Great Barrier Reef region were an important input to the process of developing a marine biogeographic regionalisation of Australia which was required to address commitments in connection with the creation of a national system of representative marine protected areas. A workshop in 1994 (Muldoon, 1995) led to the development of an initial interim biogeographic regionalisation (Thackway and Cresswell, 1995), a series of updated versions and most recently in 2006 to an integrated marine and coastal regionalisation of Australia.

The IMCRA has developed best contemporary understanding of provincial bioregions based on regionalisation of demersal fish communities (Last et al., 2005). Nested within this, a meso-scale regionalisation has been developed using finer scale information provided by relevant State and Northern Territory agencies. The third element is a map of the sea bed classified into 14 classes of regions of similar geomorphology.

The IMCRA process is dynamic, providing for updates as new data come to hand indicating need for revisions. Other implementations of integrated oceans management, such as the Australia's south east regional marine plan, can be consulted in Foster et al. (2005) and Vince (2006).

6. Current situation in Europe

6.1. Legislative framework

In recent years the European Union has adopted several Directives for nature protection (e.g. for Habitats and Species, Wild Birds, or Environmental Impact Assessment). However, increasing pressures and impacts within European estuaries and coasts have lead to the approval of a series of laws which focus on water management, including the water framework directive (WFD), the Recommendation on ICZM, the Directive on Marine Strategy, and the Maritime Policy. Details on these Directives and associated implementations can be consulted in many references, such as Borja et al. (2004a, in press), Rice et al. (2005), Borja (2005, 2006), Suárez de Vivero (2007), Fletcher (2007), or a recent special issue in *Marine Pollution Bulletin* (2007: 55(1/6). All these Directives emphasise the increasing need to protect European coastal and estuarine ecosystems and to move towards marine integrative management. The main objective of the WFD is to achieve a Good Ecological Status, for all European water bodies, by 2015. On the other hand, the EMS requires the achievement of a 'Good Environmental Status', for all European seas, by 2021.

6.2. Tools and methodologies used in assessing ecological integrity

The achieving of the abovementioned objectives requires the full development of tools and methodologies suitable to assess environmental quality in an integrative way. The integration of several elements of the ecosystem, including physico-chemical and biological elements (phytoplankton, zooplankton, benthos, algae, phanerogams, fishes), is essential in the assessment (Rogers et al., 2007).

The European scientific community is developing many different methodologies for coastal management (Sardá et al., 2005) and separate assessments of the quality of each of the WFD elements. Some of these methodologies can be consulted in volume 55 (issues 1–6) of *Marine Pollution Bulletin,* together with the report "WFD intercalibration technical report. Part 3–Coastal and Transitional Waters" (March, 2007, http://circa.europa.eu/Public/irc/jrc/jrc_eewai/library?l=/milestone_reports/milestone_reports_2007/coastaltransitional/coast_nea_gig&vm=detailed&sb=Title). Conversely, the efforts addressed until now for the EMS implementation have been focused on fisheries management under the EBA (Browman et al., 2004; Nicholson and Jennings, 2004; Rice et al., 2005; Frid et al., 2006; and Apitz et al., 2006).

Despite the above, very few studies have been published which integrate all physico-chemical and biological elements into a single assessment of the ecosystem (Borja et al., 2004a, in press; Aubry and Elliott, 2006), although some guiding principles have been developed at national levels (e.g. in UK, Rogers et al., 2007). Comparison of methodologies used in the USA and Europe have been undertaken in recent years, both for measurements of eutrophication (Ferreira et al., 2007b) and biological elements, such as benthos (Borja et al., 2008a).

6.3. Some examples of integrative assessment

The approach of Borja et al. (2004a) was further detailed in aspects, such as physico-chemical (Bald et al., 2005), chemical (Borja et al., 2004b; Borja and Heinrich, 2005; Rodríguez et al., 2006), phytoplankton (Revilla et al., 2008), and benthic community structure (Borja et al., 2000; Muxika et al., 2007). The use of ecotoxicological approaches within the WFD investigative monitoring has also been discussed (Borja et al., 2008b).

The WFD established two different quality statuses: chemical and ecological. Chemical status is based upon concentrations of metal and organic compounds, and is determined by comparing monitored concentrations with quality objectives (QO). If concentrations are below QO, the chemical status is met; if concentrations are over QO, the chemical status is not met. The WFD mentions water quality only in assessing the chemical status, but some authors include sediment and biomonitoring components (Borja et al., 2004b, 2006; Borja and Heinrich, 2005), or just sediment (Crane, 2003) (Table 6).

The ecological status integrates physico-chemical, chemical and biological indicators. The physico-chemical indicators used in this assessment are those supporting the biological elements (thermal conditions, salinity, oxygen, nutrients, and transparency). The physico-chemical status is assessed by means of a Factorial Analysis (FA). Hence, the projection of each sampling station, to the line connecting reference conditions of 'high' and 'bad' status, is calculated in the new 3-dimensional space defined by the FA (see Bald et al. (2005), for details). Consequently, those stations located near the high reference would represent a 'high' physico-chemical status, and stations located near the bad reference, would be classified as in 'bad' physico-chemical status. Intermediate stations would be classified in 'good', 'moderate' or 'poor' status.

Pollutant concentrations are also used in assessing the ecological status, but only to determine 'high', 'good' and 'moderate' status (see below). In this particular case, the WFD defines 'high' status, when concentrations of pollutants remain within the range normally associated with undisturbed conditions (*i.e*., below the background level). The concentrations between background levels and QO are in accordance with the WFD 'good status' definition, while 'moderate' status can be considered when concentrations are over QO (for details in this assessment, see Rodríguez et al. (2006).

The metrics used in methodologies implemented for the biological quality assessment within the WFD are very diverse (Table 7), and include multimetric and multivariate approaches (Borja et al., 2004a; Muxika et al., 2007). These authors provide methods to assess the quality of each of the individual biological elements (*i.e.* phytoplankton, macroalgae, benthos and/or fishes). Nevertheless, it is also necessary to integrate these individual results in a unique quality value (Borja et al., 2004a).

Following some interpretations, the classification of the ecological status in the WFD should be based upon the worst of the values in the biological elements. Hence, if the phytoplankton has a moderate value and the remainder of the elements is given a high status, the global classification should be moderate ecological status. Taking into account the spatial and temporal variability of some of the biological elements, and the absence of accurate methodologies in assessing their biological status, Borja et al. (2004a) propose the weighting of those elements, *i.e.* benthos, with contrasted and intercalibrated (Borja et al., 2007) methodologies. Hence, a decision tree permits the derivation of a more accurate global classification, including the physico-chemical and chemical elements (Table 8). Some results of the application of such an approach are presented by Borja et al. (in press).

Most of the methodologies used within the WFD determine the quality at the sampling station level. However, this Directive requires integrating quality at the water body level. One possible way to achieve this is illustrated in Table 9, which concerns a water body with four sampling stations, each representative of a certain surface, within the water body. Having derived the status for each station, this result can be substituted by an equivalent value (or the value of the ecological quality ratio, sensu WFD (see Borja et al., 2004a; Borja, 2005)), which allows weighting the global status, depending on the representativeness of each of the sampling stations or surface. The same approach can be used in a previous step,

Table 6
Decision tree when integrating water, sediments and biomonitors in assessing chemical status, within the Water Framework Directive

Water	Sediments	Biomonitors	Status
All variables meet	All variables meet		Meet
	1 variable does not meet		Meet
	>= 2 variables do not meet		Does not meet
1 variable does not meet	All variables meet		Meet
	1 variable does not meet	All variables meet	Meet
		No data	Meet
		>= 1 variable does not meet	Does not meet
	>= 2 variables do not meet		Does not meet
>= 2 variables do not meet			Does not meet

Note: A variable 'meets' when the concentration is under the quality objectives established by the Directive (Table modified and adapted from Borja et al. (2006)).

Table 7
Metrics used in assessing the biological elements quality, within the Water Framework Directive, after Borja et al. (2004a)

Phytoplankton	Macroalgae	Benthos	Fishes
Chlorophyll a	Richness	Richness	Richness
Species composition	Cover of opportunistic and sensitive spp.	Diversity	Abundance and percentage of resident spp.
Number of blooms	Ratio green algae/other spp.	AMBI	Trophic composition
			Flat fish percentage
			Pollution indicator spp.
			Invasive spp.
			Fish health

AMBI, AZTI's marine biotic index (Borja et al., 2000); spp, species.

when integrating values of biological or physico-chemical elements, as shown in Table 8.

7. Discussion

The marine environment in general, especially estuaries, now faces considerable human impacts from multiple causes (Halpern et al., 2008). These result in physical and chemical transformations, as well as changes in biodiversity, and they occur in a scenario of climate change (Halpern et al., 2007). As illustrated in our review, more or less comprehensive legislation exists at present in North America, Africa, Asia, Australia, and Europe, the intent of which is to assess the ecological quality or integrity within estuarine and coastal systems for use in several management purposes. The objective of this is to promote sustainable use of the seas and conserve marine ecosystems by maintaining marine and estuarine waters in a good environmental or ecological status. This concept of sustainability, applied to ocean governance, has been used more frequently in recent years (Costanza et al., 1998; Christie, 2005).

As a whole, these legislative measures tend to converge in defining environmental water quality in an integrative way by using several biological parameters together with physicochemical and pollution features thereby allowing the ecological status to be assessed at the ecosystem level (Borja, 2005, 2006; Apitz et al., 2006). Although this essential concept is generally widely accepted, the degree of convergence regarding the legislation is variable. For instance, in many countries, most of the data supporting the legislation have been provided by studies carried out in single systems, which do not allow generalisation. In our contribution, the Mhlanga estuary (South Africa) has been proposed as an example of the several inputs, beginning with the hydrologists' assessment of the behaviour of the river mouth, under different flow conditions, followed by the biologists' assessment of the biological responses, the process involved in the determination of the "reserve" (the freshwater needs) for this particular system, as well as for the management response. Other South African studies can be also consulted (Harrison et al., 2000; Harrison and Whitfield, 2006). Conversely, in China, the concept of integrated assessment does not seem to be widely applied and the application of supernumerary tools to assess ecological integrity appears to be incipient (Ma et al., 2006; Cao and Wong, 2007). In addition, in all cases, constraints in applying ecosystem principles arise from territorially based legislation.

Different methods have been designed for evaluating ecological integrity or condition status of coastal waters, focusing on different issues (sediment and benthic organism based toxicity tests, e.g. Hyland et al., 2000; Kiddon et al., 2003; Barnett et al., 2007; Cooksey and Hyland, 2007). Some of the authors address the causes for observed impacts with the intention to inform management (Keller and Cavallaro, 2008). To improve assessment accuracy and management effectiveness, most of the methods propose schemes of classification of the ecological quality status in several categories, based on matrices combining different symptoms or quality elements. Likewise, the need for "early warning" indicators of impending problems or human pressures has been amply recognised (Borja and Dauer, 2008).

In general, we may say that there is an increasing interest in developing assessment tools for different physicochemical or biological ecosystems' elements (*e.g.* for benthic communities see the review of Díaz et al., 2004), although very few methodologies integrate all these elements into a unique evaluation of a water body (Borja et al., 2004a). In practical terms, managers and decision-makers need simple but scientifically well grounded methodologies, capable of demonstrating to the general public the evolution of a zone (estuary, coastal area, etc.), taking into account human pressures or recovery processes (Borja and Dauer, 2008) and capable of guiding the implementation of successful management. In this context, there is a major scientific challenge to develop tools to define adequately the scale and current condition of marine ecosystems and bioregions in terms of biological performance, as well as to monitor changes through time and identify and address through management the causes of observed impairments (Borja, 2005, 2006).

Among such tools, ecological indicators have been widely used to supply synoptic information about the state of ecosystems. Most often they address ecosystem structure and/or functioning,

Table 8
Decision tree in assessing the integrative ecological status, within the Water Framework Directive (modified from Borja et al., 2004a)

BQEs	Do all BQEs meet high status?	Not	Does each BQE meet High or Good quality? Does Benthos meet High quality and from the remainder 1 or 2 meet Moderate quality? (1) Do Benthos and another BQE meet Good quality and the remainder Moderate? (2) Coast: Does Benthos meet Good quality and other Moderate? (3)	Not	Does one BQE meet Moderate quality (except 1,2,3) Does Benthos meet High or Good quality and no one has Bad quality?	Not	Other combinations Does Benthos meet Poor quality?	Not	Do all BQEs meet Bad quality? Does Benthos meet Bad quality?
BQ status	Yes High Then		Yes Good Then		Yes Moderate		Yes Poor		Yes Bad
Ph-Ch conditions	Do Ph-Ch conditions meet High status? Are chemical concentrations <background? Yes	Not	Do Ph-Ch conditions meet Good status? Are chemical quality objectives meet? Yes	Not	Then		Then		Then
H conditions	Do H conditions meet High status? Yes	Not	Then						
Ecological status	High		Good		Moderate		Poor		Bad

BQE, biological quality element; BQ, biological quality; Ph-Ch, physico-chemical; H, Hydromorphological.

Table 9
Example in integrating ecological status of several sampling stations (St) into a single value, for the whole water body (from Borja et al., in press)

Ecological status	St.1	St.2	St.3	St.4	Total
	Poor	Moderate	Good	Good	
Equivalence (E1)	4	6	8	8	
EQR (E2)	0.35	0.55	0.72	0.75	
Surface (km^2)	0.3	0.5	0.7	1.0	2.5
Rate (per one) (R)	0.12	0.20	0.28	0.40	1.0
TOTAL (E1 x R)	0.48	1.20	2.24	3.20	7.12
TOTAL (E2 x R)	0.04	0.11	0.20	0.30	0.65
GLOBAL STATUS					Good

E1: When there are not Ecological Quality Ratio (EQR) values (for terminology, see Borja et al. (2004a)), but only a global quality value; E2: when there are EQR values. *Note*: when using E1: High: E1 (10), E1 × R (8.4–10); Good: E1 (8), E1 × R (6.8–8.39); Moderate: E1 (6), E1 × R (5.2–6.79); Poor: E1 (4), E1 × R (3.6–5.19); Bad: E1 (2), E1 x R (2–3.59).

accounting for a certain aspect or component, for instance nutrient concentrations, water flows, macroinvertebrate and/or vertebrate diversity, plant diversity, plant productivity, erosion symptoms and, sometimes, ecological integrity at a system level (Weisberg et al., 1997; van Dolah et al., 1999; Llansó et al., 2001). In general, the main attribute of a good ecological indicator is the capacity to combine numerous environmental factors in a single value, which would then be useful in terms of management and for making ecological concepts compliant with the general public understanding (Borja and Dauer, 2008). Nevertheless, the application of ecological indicators is not exempt from criticisms, the first of which is that the aggregation of indices results in oversimplification of the ecosystem under observation and is not efficient in capturing qualitative modifications resulting from the emergence of new characteristics arising from self organisation processes (*e.g.* in a global climate change scenario). Moreover, problems arise often from the fact that indicators account not only for numerous specific system characteristics, but also other kinds of factors, *e.g.* physical, biological, ecological and socio-economic.. Indicators must therefore be utilised following the right criteria and in situations that are consistent with their intended use and scope; otherwise they may cause confusing data interpretations.

In view of management needs, what might be the characteristics of a good ecological indicator? What kind of information, regarding ecosystem responses, can be obtained from the different types of biological and physicochemical data usually taken into account in evaluating the state of coastal areas and transitional waters? The analysis of the current situation in different continents shows that these are indeed pertinent questions. Certainly, coastal shallow water ecosystems are extraordinary valuable in terms of goods and services (Costanza et al., 1997; Beaumont et al., 2007), correspondingly subject to strong human impact, and at the same time extremely vulnerable to climate change. Consequently, appropriate, accurate, and efficient information on ecosystem status and trend constitutes a prerequisite for sustainable use of marine ecosystems and resources, including not only environmental protection but also economic growth and social welfare (see for instance the Lisbon Agenda, Szyszczak, 2006).

Measures and indicators presently proposed focus on the living part of aquatic systems and their "supporting" hydro-morphological, chemical and physicochemical elements in order to evaluate aquatic ecosystem health and fitness. These can provide reasonable description of the current status of an ecosystem in terms of actual criteria, for instance as outlined in the European WFD (see the special issue in *Marine Pollution Bulletin* (2007: 55(1/6)). In this sense and for the time being, they fulfil what appear to be the needs of managers and decision-makers.

Despite the above, and with specific regard to coastal and transitional waters, biological elements are limited, within the WFD, to composition, abundance (and biomass) of phytoplankton, other aquatic flora, benthic invertebrate fauna, and fishes. The measures used to quantify these biological elements describe distributions/gradients, ratios, biodiversity indices and classification schemes (Indicators and Methods for the Ecological Status Assessment under the Water Framework Directive, EUR 22314EN). Modelling approaches, for instance, are still rare and limited to sub-systems.

Measures used today may provide "snapshots" of given ecosystem structural properties, but provide little or no information at all about ecosystem functioning. In fact, the causal links between the measured quantities/qualities and underlying ecosystem functioning remain largely uncertain or even unknown. Will this be sufficient in the near future? For instance, will it be good enough to achieve some of the objectives within different marine policies worldwide, which indicate strong demand for more highly integrated and holistic approaches towards sustainable development? What about the magnitude and speed of anticipated changes related to global change combined with the increasing intensity and multitude of marine environment uses by man?

In order to fulfil future needs, environmental science must complement the "static" look at structural ecosystem properties through an approach towards the ecosystem function and dynamics, which can provide a sound and reliable basis for successful management strategies. Common reductionistic approaches can only partially cope with ecosystem complexity that arises from their large number of components, interactions and spatio-temporal dynamics. Inevitably, we must recognize that the whole behaves differently from the sum of its parts, and thus neither examination of a small subsystem nor reduction to simple relationships is an adequate and sufficient approach to understand ecosystem functioning. What happens is that specific qualities/features/properties emerge at the ecosystem level, and these must be related to ecosystem functioning.

Trophic interactions (who eats whom) between and among organisms, although governed and modulated by external boundary conditions, constitute perhaps the most prominent and significant type of ecological relationships. A way to access the ecosystem level is to look upon it as a network of such trophic species-to-species interactions. Through holistic approaches, for instance Ecosystem Network Analysis (ENA), properties of both ecosystem network structure and network flow (of matter and energy) can be explored with respect to aspects of ecosystem function, such as overall system stability and resilience. Comparison of many ecosystems using a standardized approach might enable derivation of principles underlying the relationships between ecosystem network, ecosystem functioning, and ecosystem goods and services, and management of human behaviours that impact upon them.

8. Conclusions

The present worldwide trend is the implementation of legislatively driven measures to assess the ecological integrity of marine systems (estuarine, coastal and offshore). The final aim of all of them is to protect and enhance marine waters, ecosystems and natural resources, promoting a sustainable use of the oceans. Although there are multiple regional and national methodologies, very few can be considered currently as integrative. Our challenge, as scientists, is to develop methodologies and indicators that can summarize and simplify complex data, yet are easily understood by the public, media, resource users, and decision-makers. However, these must be supported by the best scientific knowledge, and take into account that 'ecological integrity' refers to the condition of an ecosystem–particularly the structure, composition, and natural processes, including function and dynamics, of its biotic communities and physical environment.

Acknowledgements

We want to acknowledge the organizers of the 'EcoSummit 2007–Ecological Complexity and Sustainability' Conference, held in Beijing (China), in May 2007, for the acceptance of the session on 'Integrative tools and methods in assessing ecological integrity in estuarine and coastal systems'. We thank also to J. Latimer, USEPA, for use of Fig. 1, and two anonymous referees for their comments. This paper is contribution number 409 from AZTI-Tecnalia (Marine Research Division).

References

Apitz, S.E., Elliott, M., Fountain, M., Galloway, T.S., 2006. European Environmental Management: Moving to an ecosystem approach. Integrated Environmental Assessment and Management 2, 80–85.

Arthuron, R., Korateng, K., 2006. Coastal and Marine Environments, Chapter 5, pp. 155–195. Section 2: Environmental State and Trends: 20-Year Retrospective. In: Africa Environment Outlook 2–Our Environment, Our Wealth (AEO-2). United Nations Environment Programme (UNEP), 540 pp. <www.unep.org/DEWA/Africa/AEO2_Launch>.

Aubry, A., Elliott, M., 2006. The use of environmental integrative indicators to assess seabed disturbance in estuaries and coasts: Application to the humber estuary, UK. Marine Pollution Bulletin 53, 175–185.

Bald, J., Borja, A., Muxika, I., Franco, J., Valencia, V., 2005. Assessing reference conditions and physico-chemical status according to the European Water Framework Directive: A case-study from the Basque Country (Northern Spain). Marine Pollution Bulletin 50, 1508–1522.

Barnett, A.M., Bay, S.M., Ritter, K.J., Moore, S.L., Weisberg, S.B., 2007. Sediment Quality in California Bays and Estuaries. Southern California Coastal Water Research Project, 41 pp.

Beaumont, N.J., Austen, M.C., Atkins, J.P., Burdon, D., Degraer, S., Dentinho, T.P., Derous, S., Holm, P., Horton, T., van Ierland, E., 2007. Identification, definition and quantification of goods and services provided by marine biodiversity: implications for the ecosystem approach. Marine Pollution Bulletin 54, 253–265.

Begg, G.W., 1978. The Estuaries of Natal. Natal Town and Regional Planning Report. 41, 1–657.

Begg, G.W., 1984. The Estuaries of Natal. Part 2 Supplement to NTRP Report 41. Natal Town and Regional Planning Report, 55, 1–631.

Borja, A., 2005. The European Water Framework Directive: a challenge for nearshore, coastal and continental shelf research. Continental Shelf Research 25 (14), 1768–1783.

Borja, A., 2006. The new European Marine Strategy Directive: Difficulties, opportunities, and challenges. Marine Pollution Bulletin 52, 239–242.

Borja, A., Dauer, D.M., 2008. Assessing the environmental quality status in estuarine and coastal systems: comparing methodologies and indices. Ecological Indicators 8 (4), 331–337.

Borja, A., Heinrich, H., 2005. Implementing the European Water Framework Directive; the debate continues. Marine Pollution Bulletin 50 (4), 486–488.

Borja, A., Franco, J., Pérez, V., 2000. A marine biotic index to establish the ecological quality of soft-bottom benthos within European estuarine and coastal environments. Marine Pollution Bulletin 40, 1100–1114.

Borja, A., Franco, J., Valencia, V., Bald, J., Muxika, I., Belzunce, M.J., Solaun, O., 2004a. Implementation of the European Water Framework Directive from the Basque Country (northern Spain): a methodological approach. Marine Pollution Bulletin 48, 209–218.

Borja, A., Valencia, V., Franco, J., Muxika, I., Bald, J., Belzunce, M.J., Solaun, O., 2004b. The water framework directive: water alone, or in association with sediment and biota, in determining quality standards? Marine Pollution Bulletin 49, 8–11.

Borja, A., Bald, J., Belzunce, M.J., Franco, J., Garmendia, J.M., Muxika, I., Revilla, M., Rodríguez, G., Tueros, I., Uriarte, A., Valencia, V., Adarraga, I., Aguirrezabalaga, F., Cruz, I., Laza, A., Marquiegui, M.A., Martínez, J., Orive, E., Ruiz, J.Mª, Sola, J.C., Trigueros, J.Mª, Manzanos, A., 2006. Red de seguimiento del estado ecológico de las aguas de transición y costeras de la Comunidad Autónoma del País Vasco. AZTI-Tecnalia report to Dirección de Aguas, Departamento de Medio Ambiente y Ordenación del Territorio, Gobierno Vasco. 15 Vol, 838 pp. <www.uragentzia.euskadi.net/u81-0003/es/contenidos/informe_estudio/red_osta/es_red_agua/2006.html>.

Borja, A., Josefson, A.B., Miles, A., Muxika, I., Olsgard, F., Phillips, G., Rodríguez, J.G., Rygg, B., 2007. An approach to the intercalibration of benthic ecological status assessment in the North Atlantic ecoregion, according to the European Water Framework Directive. Marine Pollution Bulletin 55, 42–52.

Borja, A., Dauer, D., Díaz, R., Llansó, R.J., Muxika, I., Rodríguez, J.G., Schaffner, L., 2008a. Assessing estuarine benthic quality conditions in Chesapeake Bay: a comparison of three indices. Ecological Indicators 8 (4), 395–403.

Borja, A., Tueros, I., Belzunce, M.J., Galparsoro, I., Garmendia, J.M., Revilla, M., Solaun, O., Valencia, V., 2008b. Investigative monitoring within the European Water Framework Directive: a coastal blast furnace slag disposal, as an example. Journal of Environmental Monitoring 10, 453–462.

Borja, A., Bald, J., Franco, J., Larreta, J., Muxika, I., Revilla, M., Rodríguez, J.G., Solaun, O., Uriarte, A., Valencia, V., in press. Using multiple ecosystem components in assessing ecological status in Spanish (Basque Country) Atlantic marine waters. Marine Pollution Bulletin.

Bricker, S.B., Clement, C.G., Pirhalla, D.E., Orlando, S.P., Farrow, D.R.G., 1999. National Estuarine Eutrophication Assessment. Effects of Nutrient Enrichment in the Nation's Estuaries. NOAA, National Ocean Service, Special Projects Office and National Centers for Coastal Ocean Science, Silver Spring. <http://spo.nos.noaa.gov/projects/cads/nees/Eutro_Report.pdf>.

Bricker, S.B., Ferreira, J.G., Simas, T., 2003. An integrated methodology for assessment of estuarine trophic status. Ecological Modelling 169, 39–60.

Bricker, S., Lipton, D., Mason, A., Dionne, M., Keeley, D., Krahforst, C., Latimer, J., Pennock, J., 2006. Improving Methods and Indicators for Evaluating Coastal Water Eutrophication: A Pilot Study in the Gulf of Maine. NOAA Technical Memorandum NOS NCCOS 20. National Ocean Service, National Centers for Coastal Ocean Science. Center for Coastal Monitoring and Assessment, Silver Spring, MD.

Bricker, S., 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.

Browman, H.I., Stergiou, K.I., Cury, P.M., Hilborn, R., Jennings, S., Lotze, H.K., Mace, P.M., Murawski, S., Pauly, D., Sissenwine, M., Zeller, D., 2004. Perspectives on ecosystem-based approaches to the management of marine resources. Marine Ecology Progress Series 274, 269–303.

Burgess, R., Chancy, C., Campbell, S., Detenbeck, N., Engle, V., Hill, B., Ho, K., Lewis, M., Kurtz, J., Norberg-King, T., Pelletier, P., Perez, K., Smith, L., Snarski, V., 2004. Classification Framework for Coastal Systems. United States Environmental Protection Agency, Office of Research and Development, Research Triangle Park, North Carolina. EPA 600/R-04/061, 66 pp.

Canessa, R., Butler, M., LeBlanc, C., Stewart, C., Howes, D., 2007. Spatial information infrastructure for Integrated Coastal and Ocean Management in Canada. Coastal Management 35, 105–142.

Cao, W., Wong, M.H., 2007. Current status of coastal zone issues and management in China: a review. Environment International 33 (7), 985–992.

Castro, M., Driscoll, C.T., Jordan, T.E., Reay, W.G., Boynton, W.R., Seitzinger, S.P., Styles, R.V., Cable, J.E., 2001. Contribution of atmospheric deposition to the total nitrogen loads to thirty four estuaries on the Atlantic and Gulf Coasts of the United States. In: Valigura, R.A., Alexander, R.B., Castro, M.S., Meyers, T.P., Paerl, H.W., Stacey, P.E., Turner, R.E. (Eds.), Nitrogen Loading in Coastal Water Bodies: An Atmospheric Perspective, Coastal and Estuarine Studies. American Geophysical Union, Washington, DC, pp. 77–106.

CBD, 2000. Ecosystem Approach. Fifth Conference of the Parties to the Convention on Biodiversity. Nairobi, Kenya, May 2000. <www.iisd.ca/biodiv/cop5/>.

Christie, P., 2005. Is integrated coastal management sustainable? Ocean and Coastal Management 48, 208–232.

Cloern, J.E., 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210, 223–253.

COAST, 2003. European Union Common Implementation Strategy Working Group 2.4. Final Draft Guidance on Coastal and Transitional Waters.

COM, 2005a. Proposal for a Directive of the European Parliament and of the Council, Establishing a Framework for Community Action in the field of Marine Environmental Policy. COM (2005), 505 final, SEC (2005), 1290, 31 pp. <http://europa.eu.int/eur-lex/lex/LexUriServ/site/en/com/2005/com2005_0505en01.pdf>.

COM, 2005b. Communication from the Commission to the Council and the European Parliament. Thematic Strategy on the Protection and Conservation of the Marine Environment. COM (2005), 504 final, SEC (2005), 1290, 9 pp. <http://europa.eu.int/eur-lex/lex/LexUriServ/site/en/com/2005/com2005_0504en01.pdf>.

COM, 2005c. Commission Staff Working Document. Annex to the Communication from the Commission to the Council and the European Parliament. Thematic Strategy on the Protection and Conservation of the Marine Environment, and Proposal for a Directive of the European Parliament and of the Council, establishing a Framework for Community Action in the field of Marine Environmental Policy. COM (2005), 504 and 505 final, SEC (2005), 1290, 79 pp.

Commonwealth of Australia, 1999. Australia's Oceans Policy, 2 vols, 104 pp. <http://www.oceans.gov.au/the_oceans_policy_overview.jsp>.

Commonwealth of Australia, 2006. A Guide to the Integrated Marine and Coastal Regionalisation of Australia Version 4.0. Department of the Environment and Heritage, Canberra, Australia. <www.environment.gov.au/coasts/mbp/publications/imcra-4.html>.

Cooksey, C., Hyland, J., 2007. Sediment quality of the Lower St. Johns River, Florida: an integrative assessment of benthic fauna, sediment-associated stressors, and general habitat characteristics. Marine Pollution Bulletin 54, 9–21.

Cooper, J.A.G., Ramm, A.E.L., Harrison, T.D., 1994. The estuarine health index: a new approach to scientific information transfer. Ocean and Coastal Management 25, 103–141.

Costanza, R., D'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Costanza, R., Andrade, F., Antunes, P., van den Belt, M., Boersma, D., Boesch, D.F., Catarino, F., Hanna, S., Limburg, K., Low, B., Molitor, M., Pereira, J.G., Rayner, S., Santos, R., Wilson, J., Young, M., 1998. Principles for sustainable governance of the oceans. Science 281, 198–199.

Council for the Environment, 1989. A Policy for Coastal Zone Management in the Republic of South Africa. Part 1. Principles and Objectives. Pretoria: Joan Lotter Publications. <www.deat.gov.za>.

Council for the Environment, 1991. A Policy for Coastal Zone Management in the Republic of South Africa. Part 2. Guidelines for Coastal Land Use. Academia Publications, Pretoria. <www.deat.gov.za>.

Crane, M., 2003. Proposed development of Sediment Quality Guidelines under the European Water Framework Directive: a critique. Toxicology Letters 142, 195–206.

Crossett, K.M., Culliton, T.J., Wiley, P.C., Goodspeed, T.R., 2004. Population Trends along the coastal United States: 1980–2008. National Oceanic and Atmospheric Administration, National Ocean Service, Management and Budget Office, Special Projects.

CZMA, 1996. Coastal Zone Management Act as amended through P.L. 104-150, The Coastal Zone Protection Act of 1996. <www.eutro.org/documents/czma.pdf>.

Day, J., Fernandes, L., Lewis, A., De'ath, G., Slegers, S., Barnett, B., Kerrigan, B., Breen, D., Innes, J., Oliver, J., Ward, T., Lowe, D., 2002. The Representative Areas Program for protecting biodiversity in the Great Barrier Reef World Heritage Area. In: Kasim Moosa, M. et al. (Eds.), Proceedings of the Ninth International Coral Reef Symposium, Bali, Indonesia, 23–27 October 2000 [Indonesia]: Ministry of Environment, Indonesian Institute of Sciences, International Society for Reef Studies, pp. 687–696.

Department of Environmental Affairs and Tourism, 1998. Coastal Policy Green Paper: Towards Sustainable Coastal Development in South Africa. Wynland Printers. <www.deat.gov.za>.

Department of Environmental Affairs and Tourism, 2000. White Paper for Sustainable Coastal Development in South Africa. Printed for the Government Printer by Formeset Printers, Cape. <www.deat.gov.za>.

Department of Water Affairs and Forestry, 2004. Methodology for the Determination of the Preliminary Ecological Reserve for Estuaries. Version 2. Pretoria. <www.dwaf.gov.za/documents/publications>.

Díaz, R.J., Solan, M., Valente, R.M., 2004. A review of approaches for classifying benthic habitats and evaluating habitat quality. Journal of Environmental Management 73, 165–181.

Editorial Board of "Bays in China", 1993. Bays in China. Ocean Press, Beijing.

Engle, V.D., Summers, J.K., Gaston, G.R., 1994. A benthic index of environmental condition of Gulf of Mexico estuaries. Estuaries 17, 372–384.

Engle, V.D., Summers, J.K., 1999. Refinement, validation, and application of a benthic condition index for northern Gulf of Mexico estuaries. Estuaries 22 (3A), 624–635.

Ferreira, J.G., Hawkins, A.J.S., Bricker, S.B., 2007a. Management of productivity, environmental effects and profitability of shellfish aquaculture–the Farm Aquaculture Resource Management (FARM) model. Aquaculture 264, 160–174.

Ferreira, J.G., Bricker, S.B., Simas, T.C., 2007b. Application and sensitivity testing of a eutrophication assessment method on coastal systems in the United States and European Union. Journal of Environmental Management 82, 433–445.

Fletcher, S., 2007. Converting science to policy through stakeholder involvement: an analysis of the European Marine Strategy Directive. Marine Pollution Bulletin 54, 1881–1886.

Foster, E., Haward, M., Coffen-Smout, S., 2005. Implementing integrated oceans management: Australia's south east regional marine plan (SERMP) and Canada's eastern Scotian shelf integrated management (ESSIM) initiative. Marine Policy 29, 391–405.

Frid, C.L.J., Paramor, O.A.L., Scott, C.L., 2006. Ecosystem-based management of fisheries: Is science limiting? ICES Journal of Marine Science 63, 1567–1572.

Glavovic, B., 2000a. Our Coast, Our Future. A New Approach to Coastal Management in South Africa. Mega Digital. <www.deat.gov.za>.

Glavovic, B., 2000b. Building Partnerships for Sustainable Coastal Development. The South African Coastal Policy Formulation Experience. The Process, Pereceptions and Lessons Learned. Mega Digital. <www.deat.gov.za>.

Glazewski, J., 1997. Towards a Coastal Zone Management Act for South Africa. The South African Journal of Environmental Law and Policy 4 (1), 1–22.

Granek, E.F., Brumbaugh, D.R., Heppell, S.A., Heppell, S.S., Secord, D., 2005. A blueprint for the oceans: Implications of two national commission reports for conservation practitioners. Conservation Biology 19, 1008–1018.

Halpern, B.S., Selkoe, K.A., Micheli, F., Kappel, C.V., 2007. Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. Conservation Biology 21, 1301–1315.

Halpern, B.S., Walbridge, S., Selkoe, K.A., Kappel, C.V., Micheli, F., D'Agrosa, C., Bruno, J.F., Casey, K.S., Ebert, C., Fox, H.E., Fujita, R., Heinemann, D., Lenihan, H.S., Madin, E.M.P., Perry, M.T., Selig, E.R., Spalding, M., Steneck, R., Watson, R., 2008. A global map of human impact on marine ecosystems. Science 319, 948–952.

Han, X.T., Zou, J.Z., Zhang, Y.S., 2004. Harmful algae bloom species in Jiaozhou Bay and the features of distribution. Marine Sciences (in Chinese) 28, 49–54.

Harrison, T.D., Cooper, J.A.G., Ramm, A.E.L., 2000. State of South African Estuaries. Geomorphology, Ichthyofauna, Water Quality and Aesthetics. State of the Environment Series Report. No: 2, 127 pp. Department of Environmental Affairs and Tourism, Pretoria, South Africa.

Harrison, T.D., Whitfield, A.K., 2004. A multi-metric fish index to assess the environmental condition of estuaries. Journal of Fish Biology 65, 683–710.

Harrison, T.D., Whitfield, A.K., 2006. Application of a Multimetric Fish Index to assess the environmental condition of South African Estuaries. Estuaries and Coasts 29, 1108–1120.

Herriges, J.A., Kling, C.L. (Eds.), 1999. Valuing recreation and the environment: revealed preference methods in theory and practice. Edward Elgar, Aldershot, UK.

Howarth, R.W., Marino, R., Scavia, D., 2003. Priority topics for Nutrient Pollution in Coastal Waters: An Integrated National Research Program for the United States. National Ocean Service. NOAA, Silver Spring, MD.

Huang, B., Ouyang, Z., Zheng, H., Wang, X., Miao, H., 2006. Ying Yong Sheng Tai Xue Bao (in Chinese) 17 (11), 2196–2202.

Huo, W.Y., Yu, Z.M., Zou, J.Z., Song, X.X., Hao, J.H., 2001. Outbreak of Skeletonema costatum red tide and its relations to environmental factors in Jiaozhou Bay. Oceanologia et Limnologia Sinica 32, 311–318.

Hyland, J.L., Balthis, W.L., Hackney, C.T., Posey, M., 2000. Sediment quality of North Carolina estuaries: an integrative assessment of sediment contamination, toxicity, and condition of benthic fauna. Journal of Aquatic Ecosystem Stress and Recovery 8, 107–124.

IMCRA, 1998. Integrated Marine and Coastal Regionalisation for Australia: An Ecosystem-based Classification for Marine and coastal Environments. Version 3.3. Environment Australia. <www.environment.gov.au/coasts/mbp/publications/pubs/imcra3-3.pdf>.

Jennings, S., 2005. Indicators to support an ecosystem approach to fisheries. Fish and Fisheries 6, 212–232.

Jia, X., Du, F., Lin, Q., Li, C., Cai, W., 2003. A study on comprehensive assessment method of ecological environment quality of marine fishing ground. Journal of Fishery Sciences of China (in Chinese) 10 (2), 160–164.

Jia, X., Li, C., Gan, J., Lin, Q., Cai, W., Wang, Z., 2005. Diagnosis and assessment on the health status and quality of the fishery ecoenvironment of the northern South China Sea. Journal of Fishery Sciences of China (in Chinese) 12 (6), 757–765.

Keller, A.A., Cavallaro, L., 2008. Assessing the US Clean Water Act 303(d) listing process for determining impairment of a waterbody. Journal of Environmental Management 86, 699–711.

Kenchington, R.A., 1990. Planning the Great Barrier Reef Marine Park. In: Proceedings of Symposium on Marine Heritage and Recreation. University of Waterloo, Canada, pp. 36–54.

Keyuan, Z., 2001. China's exclusive economic zone and continental shelf: developments, problems, and prospects. Marine Policy 25 (1), 71–81.

Keyuan, Z., 2003. Sino-Japanese joint fishery management in the East China Sea. Marine Policy 27 (2), 125–142.

Kiddon, J.A., Paul, J.F., Buffum, H.W., Strobel, C.S., Hale, S.S., Cobb, D., Brown, B.S., 2003. Ecological condition of US mid-Atlantic estuaries, 1997–1998. Marine Pollution Bulletin 46, 1224–1244.

Kurtz, J.C., Detenbeck, N.D., Engle, V.D., Ho, K., Smith, L.M., Jordan, S.J., Campbell, D., 2006. Classifying coastal waters: current necessity and historical perspective. Estuaries and Coasts 29, 107–123.

Last, P., Lyne, V., Yearsley, G., Gomon, M., Ress, T., White, W., 2005. Validation of National Demersal Fish Datasets for Regionalisation of the Australian Continental Slope and Outer Shelf. CSIRO Report to the National Oceans Office, CSIRO Marine Research Hobart.

Lau, M., 2005. Integrated coastal zone management in the People's Republic of China–An assessment of structural impacts on decision-making processes. Ocean and Coastal Management 48, 115–159.

Lawrence, D.R., Kenchington, R.A., Woodley, S.J., 2002. The Great Barrier Reef: Finding the Right Balance. Melbourne University Press, Melbourne. 263 pp.

Leung, W.S., 2006. Ecological water quality indices in environmental management. M.Sc. Dissertation, The University of Hong Kong.

Lin, R., 1996. Review of assessing methods for coastal eutrophication. Marine Environmental Science 15, 28–31.

Lindahl, O., Hart, R., Hernroth, B., Kollberg, S., Loo, L., Olrog, L., Rehnstam-Holm, A., Svensson, J., Svensson, S., Syversen, U., 2005. Improving marine water quality by Mussel Farming: a profitable solution for Swedish Society. Ambio 34 (2), 131–138.

Lindhjem, H., Hu, T., Ma, Z., Skjelvik, J.M., Song, G., Vennemo, H., Wu, J., Zhang, S., 2007. Environmental economic impact assessment in China: problems and prospects. Environmental Impact Assessment Review 27 (1), 1–25.

Lipton, D.W., Hicks, R., 1999. Linking water quality improvements to recreational fishing values: The case of Chesapeake Bay striped bass. In: Evaluating the Benefits of Recreational Fisheries. Fisheries Centre Research Reports 7(2). University of British Columbia.

Lipton, D.W., Hicks, R., 2003. The cost of stress: low dissolved oxygen and recreational striped bass (*Morone saxatilis*) fishing in the Patuxent River. Estuaries 26, 310–315.

Liu, Z., Wei, H., Liu, G., Zhang, J., 2004. Simulation of water exchange in Jiaozhou Bay by average residence time approach. Estuarine, Coastal and Shelf Science 61, 25–35.

Llansó, R., Dauer, D.M., Volstad, J.H., Scott, L.C., 2001. Application of the benthic index of biotic integrity to environmental monitoring in Chesapeake Bay. Environmental Monitoring and Assessment 81, 163–174.

Ma, S., Zhao, J., Chen, B., Chen, J., Cui, Y., Qu, K., Qu, S., 2006. Analysis and synthetic evaluation on environment quality of Laizhou Bay. Marine Fisheries Research (in Chinese) 27 (5), 13–16.

McConnell, K.E., Strand, I.E., 1994. The economic value of Mid and South Atlantic Sportfishing. Vol. 2, Report on Cooperative Agreement #CR-811043-01-0 between the University of Maryland College Park, the Environmental Protection Agency, the National Marine Fisheries Service and the National Oceanic and Atmospheric Administration, 134pp.

Morishita, J., 2008. What is the ecosystem approach for fisheries management. Marine Policy 32, 19–26.

Muldoon, J. (Ed.), 1995. Towards a Marine Regionalisation for Australia. Proceedings of a workshop held in Sydney, New South Wales, 4–6 March 1994. Ocean Rescue 2000 Workshop series, Publication No. 1, Great Barrier Reef Marine Park Authority.

Muxika, I., Borja, A., Bald, J., 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. Marine Pollution Bulletin 55, 16–29.

Nicholson, M.D., Jennings, S., 2004. Testing candidate indicators to support ecosystem-based management: the power of monitoring surveys to detect temporal trends in fish community metrics. ICES Journal of Marine Science 61, 35–42.

NOAA, 1996. NOAA's Estuarine Eutrophication Survey. Vol. 1, South Atlantic Region. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment. Silver Spring.

NOAA, 1997a. NOAA's Estuarine Eutrophication Survey. Vol. 2: Mid-Atlantic Region. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment. Silver Spring.

NOAA, 1997b. NOAA's Estuarine Eutrophication Survey. Vol. 3, North Atlantic Region. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment. Silver Spring.

NOAA, 1997c. NOAA's Estuarine Eutrophication Survey. Vol. 4, Gulf of Mexico Region. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment. Silver Spring.

NOAA, 1998. NOAA's Estuarine Eutrophication Survey. Vol. 5, Pacific Coast Region. National Oceanic and Atmospheric Administration, National Ocean Service, Office of Ocean Resources Conservation and Assessment. Silver Spring.

O'Boyle, R., Jamieson, G., 2006. Observations on the implementation of ecosystem-based management: experiences on Canada's east and west coasts. Fisheries Research 79, 1–12.

OSPAR, 2002. Common Assessment Criteria, their Assessment Levels and Area Classification within the Comprehensive Procedure of the Common Procedure. OSPAR Commission for the protection of the marine environment of the North-East Atlantic. <www.ospar.org/eng/html/welcome.html>.

Parsons, S., 2005. Ecosystem considerations in fisheries management: Theory and Practice. Conference on the Governance of High Seas Fisheries and the UN Fish Agreement Moving from Words to Action. St. John's, Newfoundland and Labrador, 1–5 May 5, 2005. 44 pp.

Peng, B., Hong, H., Xue, X., Jin, D., 2006. On the measurement of socioeconomic benefits of integrated coastal management (ICM): application to Xiamen, China. Ocean and Coastal Management 49 (3), 93–109.

Perissinotto, R., Blair, A., Connell, A., Demetriades, N.T., Forbes, A.T., Harrison, T.D., Iyer, K., Joubert, M., Kibirige, I., Mundree, S., Simpson, H., Stretch, D., Thomas, C., Thwala, X., Zietsman, I., 2004. Responses of the biological communities to flow variation and mouth state in two KwaZulu-Natal temporarily open/closed estuaries (vol. 2). In: Adams, J.B., (Ed.), Contributions to Information Requirements for the Implementation of Resource Directed Measures for Estuaries. Report to the Water Research Commission by The Consortium for Estuarine Research and Management. WRC Report No, 1247/2/04 <www.wrc.org.za>.

P.R.C. National Bureau of Statistics, 2001. Major Figures on 2000 Population Census of China. China Statistics Press, Beijing.

Revilla, M., Franco, J., Bald, J., Borja, Á., Valencia, V., 2008. The assessment of the phytoplankton ecological status in the Basque coast (northern Spain) according to the European Water Framework Directive. Journal of Marine Systems. doi:10.1016/j.seares.2008.05.009.

Rice, J., Trujillo, V., Jennings, S., Hylland, K., Hagstrom, O., Astudillo, A., Jensen, J.N., 2005. Guidance on the Application of the Ecosystem Approach to Management of Human Activities in the European Marine Environment. ICES Cooperative Research Report 273, 1–28.

Rodríguez, J.G., Tueros, I., Borja, A., Belzunce, M.J., Franco, J., Solaun, O., Valencia, V., Zuazo, A., 2006. Maximum likelihood mixture estimation to determine metal background values in estuarine and coastal sediments within the European Water Framework Directive. Science of the Total Environment 370, 278–293.

Rogers, S.I., Tasker, M.L., Earll, R., Gubbay, S., 2007. Ecosystem objectives to support the UK vision for the marine environment. Marine Pollution Bulletin 54, 128–144.

Rudd, M.A., 2004. An institutional framework for designing and monitoring ecosystem-based fisheries management policy experiments. Ecological Economics 48, 109–124.

Sardá, R., Avila, C., Mora, J., 2005. A methodological approach to be used in integrated coastal zone management processes: the case of the Catalan Coast (Catalonia, Spain). Estuarine. Coastal and Shelf Science 62, 427–439.

Scavia, D., Bricker, S.B., 2006. Coastal eutrophication assessment in the United States. Biogeochemistry 79 (1–2), 187–208.

Szyszczak, E., 2006. Experimental Governance. The Open Method of Coordination. European Law Journal 12, 486–502.

Shastri, Y., Diwekar, U., 2006. Sustainable ecosystem management using optimal control theory: Part 1. Deterministic systems. Journal of Theoretical Biology 241, 506–521.

Smith, R.A., Schwarz, G.E., Alexander, R.B., 1997. Regional interpretation of water quality monitoring data. Water Resources Research 33, 2781–2798.

Souter, R. (Ed.), 2007. Building the bridges: seven years of Australia's coastal cooperative research. The Cooperative Research Centre for Coastal Zone, Estuary and Waterway Management (Coastal CRC).

Sowman, M.R., 1993. The status of coastal management in South Africa. Coastal management 21, 163–184.

Suárez de Vivero, J.L., 2007. The European vision for oceans and seas–Social and political dimensions of the Green Paper on Maritime Policy for the EU. Marine Policy 31, 409–414.

Tett, P., Gilpin, L., Svendsen, H., Erlandsson, C.P., Larsson, U., Kratzer, S., Fouilland, E., Janzen, C., Lee, J., Grenz, C., Newton, A., Ferreira, J.G., Fernandes, T., Scory, S., 2003. Eutrophication and some European waters of restricted exchange. Continental Shelf Research 23, 1635–1671.

Thackway, R., Cresswell, I.D. (Eds.), 1995. An Interim Biogeographic Regionalisation for Australia: A Framework for Establishing the National System of Reserves, Version 4.0. Australian Nature Conservation Agency, Canberra.

Turpie, J., Adams, J.B., Joubert, A., Harrison, T.D., Colloty, B.M., Maree, R.C., Whitfield, A.K., Wooldridge, T.H., Lamberth, S.J., Taljaard, S., van Niekerk, L., 2002. Assessment of the conservation priority status of South African estuaries for use in management and water allocation. Water S.A. 28 (2), 191–206.

UNCLOS, 1982. United Nations Convention on the Law of the Sea, signed at Montego Bay, Jamaica, on 10 December 1982, 202 pp. www.un.org/Depts/los/index.htm.

USEPA, 1998. National Strategy for the Development of Regional Nutrient Criteria. Tech. Report No. EPA 822-R-98-002. US EPA, Office of Water, Washington, DC.

USEPA, 2001a. Environmental Monitoring and Assessment Program (EMAP): National Coastal Assessment Quality Assurance Project Plan 2001–2004. United States Environmental Protection Agency, Office of Research and Development, National Health and Environmental Effects Research Laboratory, Gulf Ecology Division, Gulf Breeze, FL. EPA/620/R-01/002. www.epa.gov/emap/nca/html/docs/c2k_qapp.pdf.

USEPA, 2001b. National Coastal Condition Report I. United States Environmental Protection Agency, Office of Research and Development, Office of Water, Washington, DC. EPA-620/R-01/005. www.epa.gov/owow/oceans/nccr/index.

USEPA, 2001c. Nutrient Criteria Technical Guidance Manual: Estuarine and Coastal Marine Waters. Tech. Report No. EPA 822-B-01-003. US EPA, Office of Water, Office of Science and Technology, Washington, DC.

USEPA, 2003. Guidance for 2004 Assessment, Listing and Reporting Requirements Pursuant to Sections 303(d) and 305(b) of the Clean Water Act. United States Environmental Protection Agency, Office of Water, Office of Wetlands, Oceans, and Watersheds.

USEPA, 2005. National Coastal Condition Report II. Office of Research and Development, Office of Water, Washington, DC. EPA-620/R-03/002 December 2004. (www.epa.gov/owow/oceans/nccr2/).

Van Dolah, R.F., Hyland, J.L., Holland, A.F., Rosen, J.S., Snoots, T.T., 1999. A benthic index of biological integrity for assessing habitat quality in estuaries of the southeastern USA. Marine Environmental Research 48 (4–5), 269–283.

Vince, J., 2006. The South East Regional Marine Plan: Implementing Australia's Oceans Policy. Marine Policy 30, 420–430.

Walsh, R.G., Johnson, D.M., McKean, J.R., 1992. Benefit transfer of outdoor recreation demand studies: 1968–1988. Water Resources Research 28 (3), 707–713.

Wang, X.L., Li, K.Q., Shi, X.Y., 2006. The marine environmental carrying capacity of major pollutants in Jiaozhou Bay. Science Press, Beijing.

Weisberg, S.B., Ranasinghe, J.A., Dauer, D.D., Schnaffer, L.C., Diaz, R.J., Frithsen, J.B., 1997. An estuarine benthic index of biotic integrity (B-IBI) for Chesapeake Bay. Estuaries 20 (1), 149–158.

Whitall, D., Hendrickson, B., Paerl, H., 2003. Importance of atmospherically deposited nitrogen to the annual nitrogen budget of the Neuse River estuary, North Carolina. Environment International 29, 393–399.

Whitall, D., Castro, M., Driscoll, C., 2004. Evaluation of management strategies for reducing nitrogen loadings to four US Estuaries. Science of the Total Environment 333, 25–36.

Whitfield, A.K., 1980a. A quantitative study of the trophic relationships within the fish community of the Mhlanga estuary, South Africa. Estuarine and Coastal Marine Science 10, 417–435.

Whitfield, A.K., 1980b. Distribution of fishes in the Mhlanga estuary in relation to food resources. South African Journal of Zoology 15, 159–165.

Whitfield, A.K., 1980c. Factors influencing the recruitment of juvenile fishes into the Mhlanga estuary. South African Journal of Zoology 15, 166–169.

Xiao, Y., Ferreira, J.G., Bricker, S.B., Nunes, J.P., Zhu, M., Zhang, X., 2007. Trophic Assessment in Chinese Coastal Systems – Review of methodologies and application to the Changjiang (Yangtze) Estuary and Jiaozhou Bay. Estuaries and Coasts 30 (6), 1–18.

Xue, X., Hong, H., Charles, A.T., 2004. Cumulative environmental impacts and integrated coastal management, the case of Xiamen, China. Journal of Environmental Management 71 (3), 271–283.

Yao, Y., Shen, Z., 2005. A review on eutrophication research of coastal waters. Marine Sciences (in Chinese) 29, 53–57.

Zhijie, F., 1989. Marine pollution legislation in China: retrospect and prospect. Marine Pollution Bulletin 20 (7), 333–335.