

Review

Evaluating marine ecosystem health: Case studies of indicators using direct observations and modelling methods

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ABSTRACT

A major challenge in ocean and coastal management is to find simple ways to evaluate the health of such complex ecosystems. This task may prove complicated as selection criteria needs to be established for choosing appropriate indicators and evaluation tools which do not mask or leave out inherent ecosystem properties and dynamics. Here, we review some empirical analyses and modelling techniques which can be used to derive environmental health indicators. With a series of case studies ranging from the combined use of structural and functional attributes of the system, to modelling outputs that integrate the biological and physical environments, we illustrate the usefulness and complementarities of these methods to assess ecosystem health. The choice of relevant indicators will depend on the ecological questions raised as well as the biological and habitat components considered which can range from a single level (individual or population) to multiple levels (community or ecosystem-based indicators) in the ecosystem. Each method has its own capabilities and limitations that may render it useful or insufficient in some cases. We suggest, however, that, whenever possible, the combination of ecological attributes and tools should be used to improve our knowledge and assessment of marine ecosystems for better management and conservation in the future.

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1. Introduction

1.1. Why define and develop ecological indicators?

In 2011 at the University of Oxford (United Kingdom), a unique international workshop attended by marine scientists was convened by the International Panel on the Status of the Oceans (IPSO) to provide a framework for assessing the cumulative impact of multiple human-induced stressors on the ocean. These scientists concluded that the world's ocean is facing unprecedented challenges leading to a high risk of biological extinction and loss of vital habitats. In the case of marine ecosystems, human-mediated degradations include pollution (Beman et al., 2005), habitat fragmentation or destruction (Airoldi and Beck, 2007), introduction of invasive species (Bax et al., 2003), overexploitation of marine resources (e.g. Coll et al., 2008; Pauly et al., 1998), hydroclimatic changes (Greene and Pershing, 2007), and acidification (Orr et al., 2005). It is clear that humans can have profound effects on a marine ecosystems' health and many of these effects can act cumulatively (Dube, 2003), and synergistically (Kirby and Beaugrand, 2009). Given the vast array of threats facing marine ecosystems, simple management solutions are likely to be unsuitable (Villanueva et al., in press).

It appears that the resilience of marine species and ecosystems to anthropogenic pressures is weaker than previously thought (Hughes et al., 2005; Malakoff, 1997). Although present extinction rates in the oceans do not appear as evident as those on land, marine extinctions do occur and they may have been simply overlooked due to the difficulty of investigating the marine environment (Malakoff, 1997). Classical examples where overexploitation have led to marine species' extinctions include that of the Steller's sea cow (*Hydrodamalis gigas*) and the great auk (*Pinguinis impennis*), which were both hunted to extinction in the 19th century (Roberts and Hawkins, 1999). In contrast to vertebrates, most marine invertebrates have large effective population sizes and, as a consequence, are often considered to be relatively less vulnerable to extinction (Carlton, 1993; McKinney, 1998). However, population loss can occur even in invertebrates as seen in the benthic limpet *Lottia alveus*, which was extirpated during the post-Pleistocene period as a result of habitat destruction (Carlton, 1993).

Although anthropogenic-related effects on marine systems are now widely recognized, they are often difficult to quantify (Vitousek et al., 1997). Coupled with the complex and unpredictable characteristics of these systems, an accurate assessment of the state of an ecosystem can be a useful decision-making tool for management and conservation (Jørgensen et al., 2010; Rice and Rochet, 2005). As a result, a scientific interest in indicators has grown exponentially over the last two decades fuelled recently, by the call for "indicators of sustainable development" at the Rio Earth Summit in 1992, and by society's growing demand for regular and periodic accounts of environment health (Hayes et al., 2008). In Europe, the publication of the Water Framework Directive (WFD, 2000/60/EC)

and the Marine Strategy Framework Directive (MSFD, 2008/56/EC) have stimulated the search for more effective indicators to assess the quality of the European water bodies leading to an increase in the number of papers describing indicators and the organization of several workshops and conferences on this topic (see Shin and Shannon, 2010).

1.2. Definition of ecological indicators

Whilst ecosystem health can be expressed by its ecological aspects (Costanza, 1992), broader definitions that include economic and human dimensions of the system are now being adopted (White et al., 2010). According to Xu and Mage (2001), for example, ecosystem health can be defined as "the system's ability to realize functions desired by society and maintain them over a long time". In this case, both functional (activities and processes) and structural characteristics (components) of the ecosystem are considered in context of ecological and societal needs (White et al., 2010). To date, more than 200 indicators exist to describe marine ecosystem health (Rice, 2003). These indicators, which can extend from the cellular to the community level, convey information on the state of the ecosystem in a simplified form to a range of stakeholders with diverse interests and backgrounds (scientists, policy makers, the media, and the general public; Jørgensen et al., 2010). Ideally, any indicator should be (1) sufficiently sensitive to provide an early warning of change, (2) distributed over a broad geographical area or otherwise widely applicable, (3) capable of providing a continuous assessment over a wide range of stress, (4) relatively independent of sample size, (5) easy and cost-effective to measure, collect and calculate, (6) able to differentiate natural cycles or trends from those induced by anthropogenic stress, and (7) relevant to ecologically significant phenomena (Noss, 1990).

Today, most environmental management plans favour the adoption of an ecosystem-based approach. 'Ecosystem-based management' postulates that effective management must (1) be integrated among components of the ecosystem and resource uses and users; (2) lead to sustainable outcomes; (3) take precaution in avoiding deleterious actions; and (4) be adaptive in seeking more effective approaches based on experience (Borja et al., 2009). An ecosystem based approach is often difficult to implement however, due to gaps in our data and knowledge in terms of measuring ecosystem functioning using biological indicators (Van Hoey et al., 2010). While current measurements based upon distributions, ratios, biodiversity indices and classification schemes can provide a snapshot of an ecosystem's structural properties, they provide little or only partial information about its functioning (Borja et al., 2008). As these metrics are often based on monitoring data or quality assessment of a single factor (e.g., species or chemical state), a possible way of assessment is to reduce complexities into simplified forms representing key ecosystem factors such as: (1) identification of key functional groups, such as zooplankton or benthic invertebrates, (2) modification of ecological interactions and energy flow

through the system, and (3) an integrative approach linking the habitat to biological attributes.

Another major challenge for the environmental management of marine systems is to identify simple indicators that capture the inherent complexity of the ecosystem. Since it is unlikely that any individual biological measurement will adequately capture both the complexity and the dynamical functioning of any marine ecosystem, a combination of different indicators should be used (Buchs, 2003). In this context, both “descriptive” indicators related to ecosystem structure (e.g. diversity, species composition, abundance) and “functional” indicators that measure ecosystem activities (e.g. productivity, nutrient cycling, ecosystem metabolism) can provide complementary information on ecosystem health (e.g. Bunn and Davies, 2000; Elliott and Quintino, 2007; Xu et al., 1999). Indicators based on these metrics should be carefully selected however, so that each represents a different aspect of the system (e.g. Salas et al., 2006). Community indicators that integrate both structural and functional aspects of key functional groups such as when considering phytoplankton diversity and productivity can provide useful information on both the physico-chemical environment and ecosystem dynamics. At a community or an ecosystem scale, assessments may be based on ecological responses such as changes in feeding interactions and energy flows. Here, emergent properties such as adaptation and resilience to perturbations can be used as an ecosystem health indicator which can be determined using a combination of techniques for species feeding interaction analyses (stable isotope, stomach/gut contents analysis and fatty acid trophic markers) and/or modelling techniques.

1.3. Tools to derive indicators

Empirical and modelling approaches can provide indicators of ecosystem state and dynamics, respectively. Empirically estimated indicators are currently used as specific and concrete proxies to indicate community response to change, such as the state of fish stocks for fisheries management (Rice and Rochet, 2005), or benthic community structure for habitat quality assessment (Borja et al., 2008). In contrast, model-based indicators are applied primarily to resolve ecosystem management questions and so the properties of these indicators depend upon the model's assumptions and algorithms. While the empirical approach is common, modelling approaches to derive indicators are less employed due mainly to the lack of sufficient data, thus, restricting modelling approaches mostly to sub-systems (Borja et al., 2003) and their application for ecosystem management purposes. Model-based and empirical indicators further differ in that they consider distinct spatial, temporal and ecological scales; dynamic models usually cover a longer time horizon, for example, and may also integrate ecological scales from the individual to the community (Pelletier et al., 2008).

In order to illustrate the various tools (from direct measurements to modelling techniques) that can be applied to derive indicators to evaluate ecosystem health of marine ecosystems, a series of case studies ranging from the combined use of structural and functional attributes of the system to modelling outputs which integrate the biological and physical environment are discussed. Results from these case studies provide concrete examples that can show the usefulness and applicability of various measurement tools and analytical methods as indicators. Each method described in case studies considers a different degree of integration of biological and habitat components of the ecosystem (Fig. 1), ranging from a single (species and population indicators) to multiple level(s) (community and ecosystem indicators) (Table 1). We have not included the use of top predators in this paper since indicators based upon these groups and their trophic interactions are

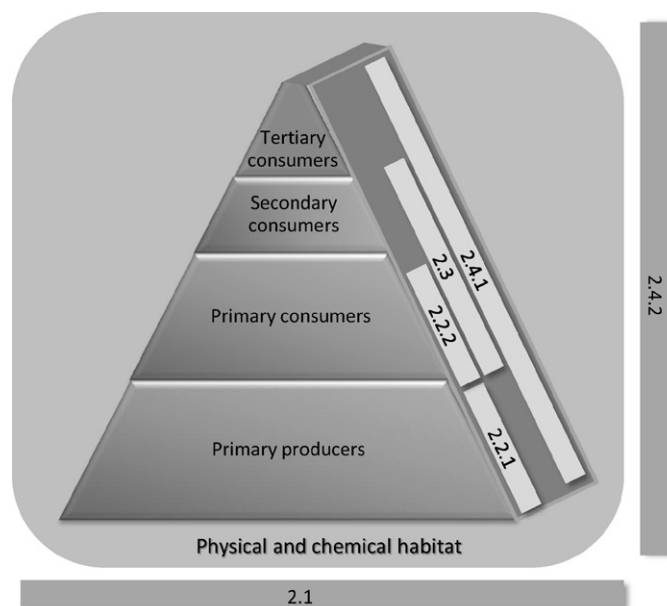


Fig. 1. Schematic diagram of the number (length of arrow) of biological components (trophic levels) and/or habitat considered for calculating an indicator in each case study. The number indicated in each arrow corresponds to the case study discussed in the main text.

already well described in the literature (e.g. Boyd et al., 2006 and references therein).

2. Case studies

2.1. Indicators of the state of the physical and chemical environment: the use of numerical techniques

The anthropogenic use of both freshwater and land systems has increased greatly over time leading to an intensification in nutrient loadings from the continent to the sea (Beman et al., 2005). Changes in nutrient availability can be sufficient to alter natural biogeochemical cycles and it can affect the whole ecosystem (e.g. reduction of biological diversity; Vitousek et al., 1997). While a substantial knowledge on the current dynamic equilibrium of systems is a prerequisite to detect changes, the absence of a relative reference state is one of the main problems encountered by many environmental scientists (Goberville et al., 2011).

A new multivariate non-parametric statistical technique, the Procedure to Establish a Reference State for Ecosystems (PERSE) has been developed recently to identify change in natural systems in relation to a reference state (Goberville et al., 2011). The PERSE method is based on the Non-Parametric Probabilistic Ecological Niche (NPPEN) model (Beaugrand et al., 2011). The technique can be applied to calculate the probability that an observation statistically belongs to the centroid of a relative reference state (here each year, see Fig. 2; Goberville et al., 2011). For example, a probability close to 0 (e.g. the year 2001, see Fig. 2) indicates a perturbation where the observation lies outside the relative reference state. The NPPEN is insensitive to missing data which is a common feature found in many ecological databases and it can be transposed easily to any marine or terrestrial dataset to give a rapid evaluation of the ecosystem state and provide a structured framework to improve future ecosystem management strategies.

When this new technique was applied to a dataset describing both physical and chemical variability in French coastal waters collected as part of the monitoring programme Service d'Observation en Milieu Littoral (SOMLIT), the NPPEN provided both a rapid

Table 1
Summary of tools to derive empirical data and model-based indicators described in the case studies. The time span relates to the period over which the indicator value provides information (“short” is less than a year, long is more than a year). MTI = Marine Trophic Index; FIB = fishing-in-balance; PCI = Phytoplankton Community Index, EEI = Ecological Evaluation Index, RICQI = Rocky Intertidal Community Quality Index.

Tools	Ecological attribute	Examples of indicators	Operational indicators	Spatial scale	Time span
Indicator species/functional groups	Population abundance	Anthropogenic disturbance	e.g. BO2A ratio	Local	Short
	Community composition	Anthropogenic disturbance, quality of the environment, hydroclimatic influences	e.g. PCI, EEI, RICQI	Local	Short
	Community size structure	Hydroclimatic influences, quality of the environment	Not identified	Regional	Long
Isotope profile analysis	Trophic structure	Overexploitation	e.g. MTI	Local, regional	Long
	Transfer of organic matter	Diet	Not identified	Local, regional	Short, long
Fatty acid trophic markers	Transfer of organic matter	Habitat use, water quality, hydroclimatic influences Diet, habitat use	Not identified	Local	Short
Stomach/gut contents' analysis	Transfer of organic matter	Diet, habitat use	Not identified	Local	Short
Mass-balanced models	Trophic structure and dynamics	Overexploitation, trophic transfer efficiencies, ecosystem functioning	e.g. MTI, FIB, Omnivory Index Ecosystem Integrity	Regional	Long
End-to-end models	Ecosystem structure and dynamics	Overexploitation, hydroclimatic influences, ecosystem functioning, effectiveness of management actions	e.g. MTI, FIB	Regional	Long
Numerical analyses	Hydrology	Anthropogenic disturbance	Not identified	Local, regional	Short

evaluation of the state of coastal systems and evidence of anthropogenic disturbance (Goberville et al., 2011). The results showed that some coastal systems in France have been altered by nitrates and phosphate over-enrichment from 1997 onwards. Furthermore, one prominent mode of disturbance resulted from human activities that are known to mobilize the nutrient elements through land

clearing, production and the application of fertilizers, creating a serious environmental issue.

2.2. Indicators based on key functional groups/indicator species (single trophic level)

2.2.1. Primary producers

2.2.1.1. Indicators using phytoplankton groups. Phytoplankton can provide sensitive and quantifiable indications of ecological changes and environmental perturbations in marine surface waters, due to their fast growth rates and response to low levels of pollutants (Paerl et al., 2003). Phytoplankton succession and community composition reflects the environmental conditions of the ecosystem (see Smayda and Reynolds, 2001 and references therein), among which nutrient availability plays a significant role in structuring that community (Beman et al., 2005). For that reason among others, phytoplankton is a key-element in defining water quality in coastal and marine systems as referred to in the European Union Water Framework Directive (WFD, 2000/60/EC) and more recently, in the Marine Strategy Framework Directive (MSFD, 2008/56/EC; Ferreira et al., 2011).

Ecological assessments using biological indicators are often simplified and expressed as numerical scales, e.g. the Ecological Quality Ratio (EQR; Van de Bund and Solomini, 2007), to determine the ecological status and ensure comparability of different monitoring protocols employed in many Northwestern Atlantic systems. In the case of phytoplankton, integrated indices (Devlin et al., 2007) take into account several factors such as the chlorophyll *a* concentration, the frequency of elevated phytoplankton counts (individual species and/or total cell counts), the frequency of noxious species blooms (Harmful Algal Blooms) and/or a seasonal progression of functional groups.

The Phytoplankton Community Index (PCI) is one example of a structural indicator based on the abundance of “life-forms” such as “pelagic diatoms” or “medium-sized autotrophic dinoflagellates” (Tett et al., 2008). The method aims to define an area in Cartesian space (the so-called “doughnut” in Tett et al., 2008), which

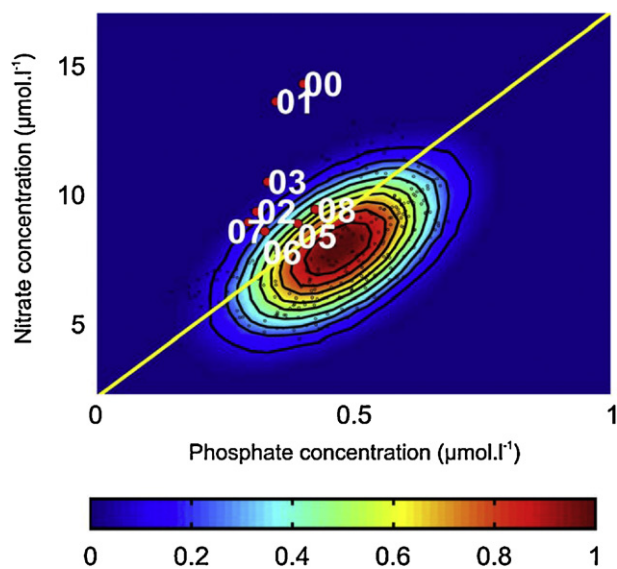


Fig. 2. Probabilities for the observations to belong to the relative reference state for each year (from 2000 to 2008) in February (here for the offshore site of Roscoff, France). The position of the reference state was calculated in an Euclidean space of two dimensions, the first dimension (*x*) being represented by the phosphate concentration and the second (*y*) being represented by the nitrate concentration. The greater the distance between an observation and the centroid of high probability, the less likely that the observation belongs to the reference state. The line represents the Redfield ratio (N:P = 16).

From Goberville et al. (2011).

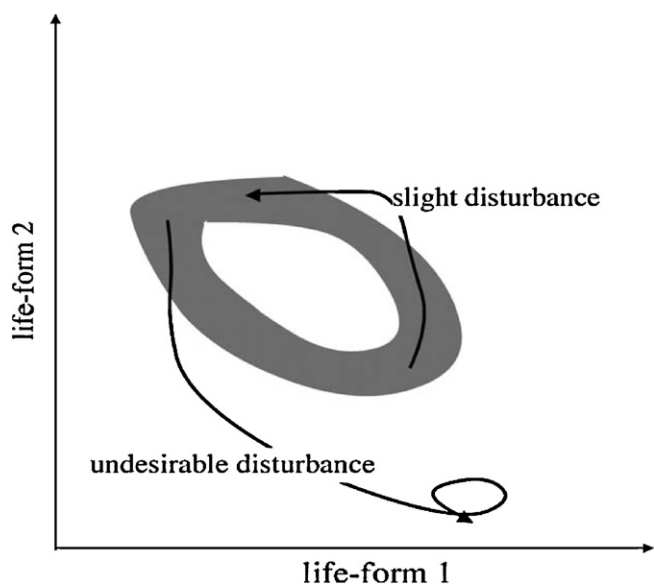


Fig. 3. Schematic diagram of the doughnut-shape of the envelope encompassing the state space of two life-forms representing the planktonic community experiencing natural seasonal and interannual variation and the potential excursions of these states upon disturbances caused by external drivers.

From Tett et al. (2008).

includes all the ecosystem states that are “normal” for type-specific conditions at varying spatio-temporal scales. A sustained movement away from this area is considered to be an undesirable disturbance (Fig. 3). Boundaries of these areas are defined from empirical multivariate analysis either to reduce the data to a few key dimensions of representative groups of regularly co-occurring species or, as an alternative, the identification of individual “life-forms” based on function and taxonomy (Devlin et al., 2007). The PCI can be particularly useful as an indicator of biological quality and ecosystem health in the context of eutrophication. Tett et al. (2008) further suggest that the development of PCI for management purposes could be facilitated by linking a theory that predicts reference envelopes in relation to both the physical environment and to undisturbed levels of nutrients. In that regard, the PERSE method (Goberville et al., 2011) described previously in Section 2.1, “Indicators of the state of the physical and chemical environment: the use of numerical techniques” could be an appropriate candidate.

2.2.1.2. Indicators using phytobenthic groups. Macrobenthic assemblages in general are sensitive indicators of changes in marine environments as they are long-lived and in most cases, sessile organisms (Borja et al., 2008; Levine, 1984; Van Hoey et al., 2010). Effects seen in macrobenthic communities as a result of acute and chronic disturbances include changes in species richness or diversity, total abundance and total biomass, which in turn may lead to modifications in trophic structure and functioning (Borja et al., 2011; Dauvin, 1998; Pearson and Rosenberg, 1978; Warwick and Clarke, 1994).

Common macroalgal indices developed for water quality assessment use univariate metrics such as species richness (including presence/absence), total biomass and total abundance (Levine, 1984). However, newer methods fulfil the requirements of current legislation needs, such as the WFD, where “ecological status” refers to the quality of the structure and functioning of an aquatic ecosystem requiring the use of indicator species (opportunistic or sensitive), as well as ecological or functional groups.

One of these is the Ecological Evaluation Index (EEI) that quantifies changes in the structure and function of transitional and coastal

waters (Orfanidis et al., 2003). The EEI evaluates shifts in alternative ecological states (pristine or degraded) by classifying marine benthic macrophytes into two ecological state groups based on their abundances, “late-successional” and “opportunistic”. Opportunistic groups are dominant in degraded conditions when EEI values are lower than 6 and indicate that the ecosystem should be restored. Since the EEI is assumed to be closely related to ecosystem functioning and processes, such as nutrient cycling and fish production, high values of the EEI indicate the existence of high ecologically and economically valued communities (Orfanidis et al., 2003).

Indices calculated in hard substrata communities, in particular, benefit from incorporating information on macrofauna (Rogers and Greenaway, 2005) since deviations from the natural system may be more evident if macroflora and macrofauna are combined (Bishop et al., 2002). Díez et al. (2011) have proposed the Rocky Intertidal Community Quality Index (RICQI) to assess the quality of coastal waters along the Atlantic Iberian coasts. This index combines metrics derived from Basque coast rocky intertidal macrofauna and macroflora assemblages comprising: indicator species abundance, morphologically complex algae cover, species richness, and faunal cover. The RICQI index was used to detect the effects of disturbance caused by a wastewater treatment plant in Plentzia Bay (Basque Coast, Spain) and was found to distinguish different degrees of disturbance more accurately than other methods (Díez et al., 2011).

2.2.2. Primary consumers

2.2.2.1. Indicators using zooplankton groups. Indicators using zooplankton data can be based upon: (1) the abundance of individual taxa; (2) functional attributes; (3) species assemblages; and (4) links with higher trophic levels (Beaugrand, 2005). Zooplankton abundance, distribution and composition are being monitored over an extensive spatial scale in the North Atlantic using the Continuous Plankton Recorder (CPR). Unique multi-decadal datasets on phytoplankton groups, zooplankton species and oceanographic parameters can be obtained from this approach. The abundance of plankton species has been used to document abrupt ecosystem shifts in different regions of the world (Weijerman et al., 2005) and the effects of global warming and its consequences on regional hydrodynamics (Beaugrand, 2009). Functional attributes of the ecosystem such as plankton body size (Li et al., 2004; Beaugrand et al., 2010) can also be related to climate and ecosystem functioning.

Community body size largely determines the types and strengths of energy flows in ecosystems thereby affecting ecological networks and ecosystem function (Woodward, 2009). Beaugrand et al. (2010) reported that climate-induced northward shifts in copepods led to a more diverse community but a smaller mean community body size; they suggested that this reorganization of the North Atlantic planktonic ecosystem has had negative consequences on the drawdown of biological carbon. Beaugrand et al. (2010) argued that the slower sinking of smaller fecal pellets lead to an increase in the particulate carbon residence time in the epipelagic zone. Organic carbon that resides longer in surface waters could be processed by smaller-sized zooplankton and, thus, dissipated through more complex food webs (Li et al., 2004). Further effects were suggested to include a deepening of the nutricline due to increased stratification, in turn leading to a shift in the phytoplankton community from diatoms (major exporters of carbon to depth) to coccolithophorids (Cermeño et al., 2008). It is likely that this change in plankton diversity will propagate to other trophic levels (e.g. fish), mostly in extra-tropical regions where both the mean and seasonality of temperature limit diversity (Hiddink and Hofstede, 2008).

There are pitfalls with indicators that are based solely on functional attributes as they attempt to capture an ecosystem state in a single value and, thus, may fail to provide enough information to understand the nature of the observed changes. Species

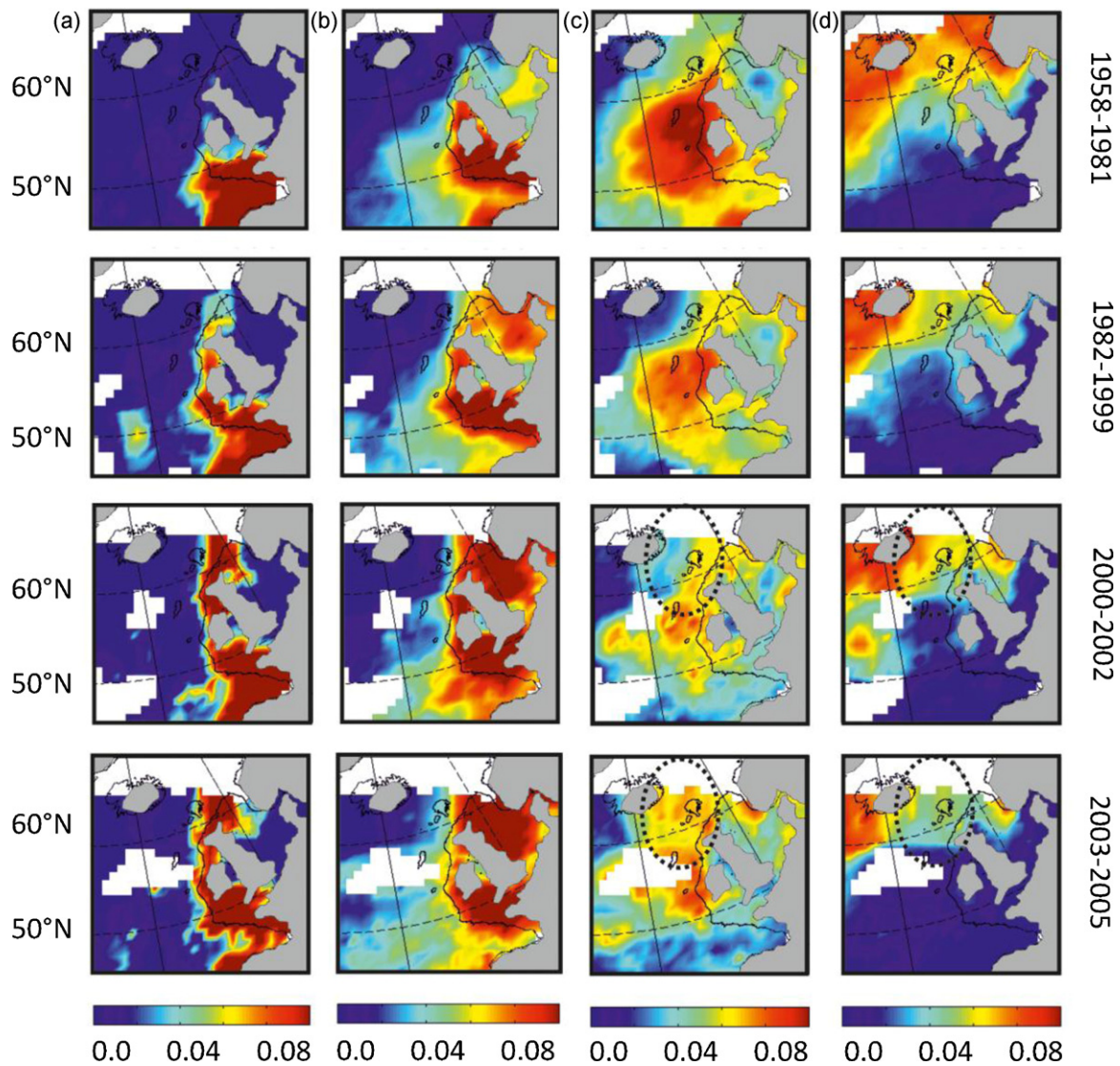


Fig. 4. Long-term changes in the mean number of (a) warm temperate pseudo-oceanic species, (b) temperate pseudo-oceanic species, (c) cold-temperate mixed-water species, and (d) subarctic species, from 1958 to 2005. The period 1958–1981 was a period of relative stability and the period 1982–1999 was a period of rapid northward shifts. Black dotted circle or oval denotes areas where pronounced changes have been observed.

From Beaugrand et al. (2002).

assemblage indicators could bring additional information to better understand the resilience (Holling, 1973) of pelagic ecosystems and allow future changes to be anticipated. Using species assemblages, rapid biogeographical shifts in calanoid copepods in the north-eastern part of the North Atlantic Ocean has been documented in several studies (Fig. 4; Beaugrand et al., 2002; Beaugrand, 2009). An increase in warm-water assemblages northwards was associated to a reduction in the number of cold-water species assemblages. These biogeographical movements reached up to 23 km per year on average for the period of 1958–2007 and were explained mainly, although not exclusively, by the northward movements of isotherms.

2.2.2.2. Indicators using zoobenthic species. Structural characteristics (e.g. diversity and abundance) of macrofauna can be useful to monitor the impacts and persistence of environmental stress in coastal systems (e.g. Dauvin, 2007). As an example, a 20-year survey (1977–1996) of a fine sand associated benthic community of *Abra alba* and *Hyalinoecia bilineata* was used to assess the impacts of the Amoco Cadiz oil spill tragedy in 1978 at Pierre Noire, France

(Dauvin, 1998). For this purpose, the benthic opportunistic amphipod/annelid ratio (BO2A) and the Shannon diversity (H') index was evaluated. The BO2A ratio increased just after the oil spill where amphipod abundance and frequency decreased (Dauvin, 1998). Six months following the spill, a change in the community structure was also observed (local BO2A maximum value of 0.11) despite the apparent good quality of the environment. Two main BO2A ratio trends were observed during the 19-year post oil spill period: (1) high values during the first 8 years (1979–1987) with a peak of the opportunistic polychaete *Pseudopolydora pulchra* abundance in the summer of 1982 and 1984 and (2) lower values from 1988 to 1996 indicating amphipods recovery, in particular *Ampelisca* spp., which originally dominated the communities (Dauvin, 1998). Similarly, macrobenthic community diversity (H'), another structural metric index, showed concurrent changes during the 20-year survey. Diversity rapidly increased with the disappearance of the dominant *Ampelisca* species and decreased following its re-colonisation by the end of the survey (1990–1996).

In addition to structural characteristics, functional attributes of benthic communities should also be considered in detecting

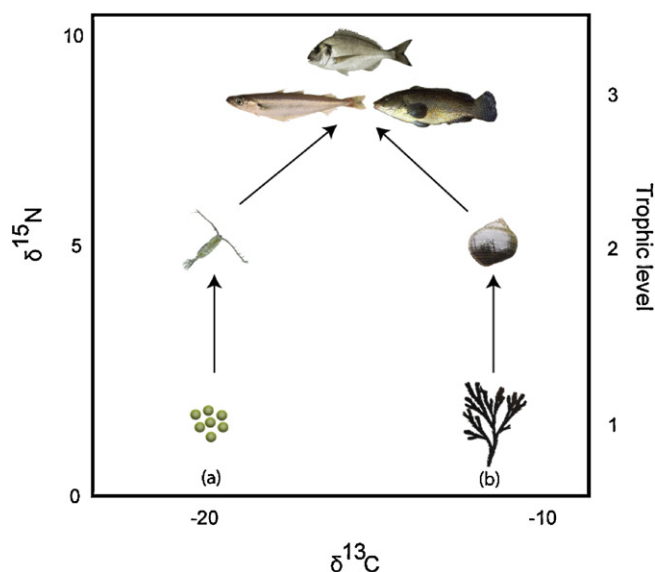


Fig. 5. Schematic diagram of producers and consumers isotopic ratios (in ‰) in the (a) pelagic and (b) benthic food webs. Stable isotope signatures of carbon (X-axis) allow to determine trophic pathways, while nitrogen isotope ratios (Y-axis) can be used to indicate organism trophic level (Z-axis).

Images provided by P. Porcher, O. Barbaroux, R.R. Kirby and O. Glippa.

environmental perturbations, especially in estuarine habitats (Dauvin, 2007; Elliott and Quintino, 2007). Since estuaries are low diversity ecosystems characterized by the accumulation of organic matter, the detection of anthropogenic stress can be more difficult (i.e. Estuarine Quality Paradox, see Dauvin, 2007; Elliott and Quintino, 2007) and the use of taxonomy-based indicators alone is insufficient. Recently, Rakocinski (2012) proposed the use of macro-invertebrate functional indicators to assess organic enrichment and hypoxia in three central Gulf of Mexico estuarine systems. The suggested indicators reflected important ecosystem functions related to both trophic transfer (expressed as secondary production) and the potential for biogeochemical processing, the latter expressed as community maturity. In contrast to species-specific indicators, these benthic process indicators are taxonomically independent and therefore they are more easily compared across habitats, environmental gradients and geographic regions (Basset et al., 2004; Rakocinski, 2012). Unfortunately, before these indicators can be used for estuarine management purposes, several practical challenges, such as establishing operational thresholds, need to be resolved.

2.3. Stable isotopes profiles as environmental and trophic indicators (two or more trophic levels)

Stable isotopes help to follow and trace details of element cycling and so are useful in studying organic dynamics. The ecological applications of isotope tracers are highly diverse ranging from the origins of human diets to the sources and fates of gases involved in greenhouse warming (Dawson and Siegwolf, 2007; Fry, 2006). Ecologists often use isotopes as a support tool to study feeding habits, nutrient tracking, species migrations and habitat use of species and communities in the ecosystem (Fig. 5; Thompson et al., 2005). In marine systems, long-lived primary consumers with a moderate tissue turnover rate, such as bivalves, can be useful indicators of available organic matter sources, and so are good habitat indicators (Fukumori et al., 2008).

Sedentary and omnivorous suspension feeding bivalves that occupy the sediment-water interface, for example *Crassostrea gigas*, which feeds on a mixture of organic living and decaying

material are useful bio-indicators of benthic–pelagic coupling between organic matter sources within an ecosystem (Grangeré et al., 2012; Lefebvre et al., 2009). In a study involving cultured oysters from the same origin transplanted into several ecosystems, Lefebvre et al. (2009) combined measurements of stable carbon (C) and nitrogen (N) isotope ratios within the adductor muscles with measurements of hydro-biological variables within the water column. This combination of indicators made it possible to discriminate two groups of coastal ecosystems that differed in hydrology, nutrient inputs, and size of their respective watersheds at the same time providing a more holistic view of ecosystem functioning. Specifically, the study revealed that a higher input of nutrients in coastal ecosystems can increase the contribution of benthic food sources (microphytobenthos and macroalgae) to the oyster's diet and, therefore, a significant change in the organic matter available at the base of the food web. Recently, a similar approach was applied to disentangle the role of climatic and anthropogenic factors impacting coastal ecosystems in the same region. Grangeré et al. (2012) found that climatic factors act in synergy with anthropogenic factors (i.e. nutrient enrichment) in low-eutrophicated marine ecosystems while climatic factors appear to be dampened by anthropogenic factors in high-eutrophicated ones.

Stable nitrogen profiles in bioindicator species can also complement standard chemistry assessments of water quality. While chemical indicators are useful for measuring eutrophication levels they cannot indicate the source of causative nutrients to nitrogen loading, for example. Combined with the $\delta^{15}\text{N}$ signatures of bioindicator species, it is possible to distinguish whether nitrogen originates from multiple anthropogenic sources including sewage effluents system or from chemically synthesized fertilizers (Cohen and Fong, 2005). Fertig et al. (2009) have observed increased levels of $\delta^{15}\text{N}$ values in macroalgae (*Gracilaria* sp.) and oyster mussels in the coastal bays of Maryland, USA. Combined with conventional water monitoring, elevated $\delta^{15}\text{N}$ profiles inferred that human wastes were the primary contributors to eutrophication in some areas of the coastal zone.

In a fisheries management context, the $\delta^{15}\text{N}$ assay can be useful to demonstrate, albeit indirectly, the impacts of fishing pressure on marine food webs (Fry, 2006). Fishing removes large fish from the oceans selectively, thereby reducing the mean trophic level creating a phenomenon known as “fishing down the food web” (Pauly et al., 1998). Since increases in trophic level from a plant to herbivore or a herbivore to carnivore have been estimated to involve $\delta^{15}\text{N}$ increases of 2.2–3.4‰ in the consumer versus its diet (Fig. 5), $\delta^{15}\text{N}$ measurements can be used for estimating trophic levels in field conditions (Fry, 2006). This technique, often referred to as the nitrogen isotope “trophometer”, has been used to detect changes in the mean trophic level of marine food webs as a result of increasing fishing pressure (Fry, 2006). Moreover, the estimated trophic level from isotope analysis showed a strong correlation with estimates from modelling outputs, i.e. Ecopath with Ecosim and Ecospace (EwE) models (Dame and Christiansen, 2008; Kline and Pauly, 1998; cfr. 4.1 Dynamic trophic models). As an example, the trophic structure of several representative species present in the food web of the South Catalan marine ecosystem (Northwestern Mediterranean) was analysed using the Ecopath ecosystem modelling tool combined with the stable isotope approach (Navarro et al., 2011). Whereas a clear correlation between the trophic levels estimated by the Ecopath model and the $\delta^{15}\text{N}$ values was found, the two estimators of trophic width (the omnivore index and the total isotopic area) were only related for some species. This discrepancy is possibly due to the difference in performance of the indicators that is related to the feeding strategy of the predators (Navarro et al., 2011).

Complementary assessments of an organism's diet using stomach/gut content analyses alongside C and N isotopic profiles can

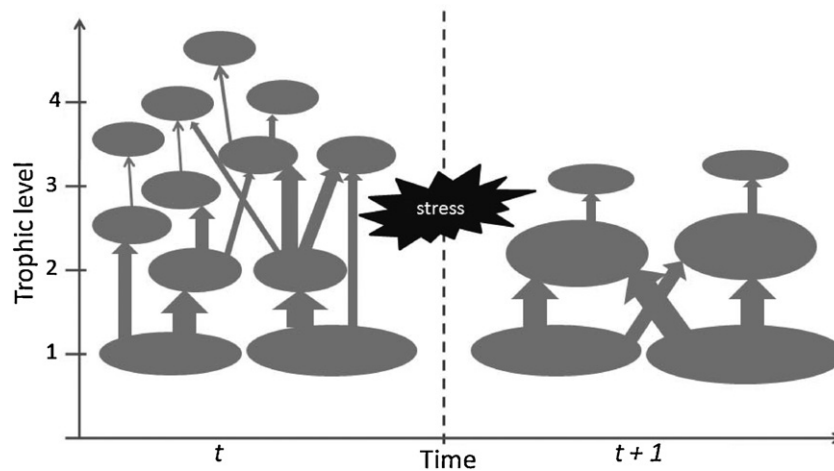


Fig. 6. Schematic diagram showing trophic network connectivity before (left) and after (right) a perturbation or stress. The depletion of higher-trophic level species simplifies the ecosystem structure while strengthening the connectivity and increasing the abundance of less-predated lower-trophic level species. Biomass (low-sized species) is represented by circle size. Flows from one trophic level to another are represented by arrows, while predation strength is represented by arrow size.

Modified Villanueva (2004).

provide substantial temporal and spatial information for studies of food web structure (Fry, 2006). The combination of these tools not only facilitates the interpretation of processed data (Evans-White et al., 2001) but provides a more complete dietary history of an organism (Kline, 1999; Pasquaud et al., 2007). Recently, the feeding habits of an invasive species in French watersheds, the European catfish *Silurus glanis*, and its potential impacts on the recipient food web were investigated using stable isotopes and gut content analyses (Syväranta et al., 2009, 2010). For this purpose, mean $\delta^{13}\text{C}$ and $\delta^{15}\text{N}$ values were assessed for the catfish, their potential prey and their competitors. Catfish were found to be flexible predators with large individuals being able to utilize both terrestrial and marine-derived prey, while the smaller individuals were rather feeding on the abundant crayfish and smaller prey fish species (Syväranta et al., 2009, 2010; Martino et al., 2011). In this way, catfish are likely to avoid intra- and inter-specific competition and consequently, enhance its success for invasion. Moreover, since they appear to utilize resources differently, they will not necessarily have strong adverse impacts on the native fish community. The combined analyses of organisms' isotopic values and stomach content' analysis thus provided more detailed information on the trophodynamics of French watersheds and the potential impacts of an invasive species, in particular.

Ever since the pioneering work carried out by Jeffries (1970) and Lee et al. (1971), fatty acid composition has also been used to trace food sources through multiple food web linkages (e.g. Parrish, 2009). Whilst the concept of Fatty Acid Trophic Markers (FATMs) is well established to characterize the diets of pelagic groups such as zooplankton (Brett et al., 2009; Dalsgaard et al., 2003; Gonçalves et al., in press; Sargent and Falk-Petersen, 1988), fish (Piché et al., 2010) and marine mammals (Budge et al., 2006), its application appears more challenging in benthic food webs since benthic consumers may have a highly mixed diet (Kelly and Scheibling, 2012). Although the source of fatty acids may be difficult to establish in benthic taxa, the combination of FATM's with stable isotope profile analysis can help identify organic matter sources and trophic interactions (Alfaro et al., 2006; El-Sabaawi et al., 2009; Kelly and Scheibling, 2012; Volkman, 2006). As an example, Spilmont et al. (2009) used a combination of these techniques to identify the food sources of coexisting and potentially competing ghost shrimp (*Trypaea australiensis*) and soldier crab (*Mictyris longicarpus*) populations on a sandflat in south-eastern Queensland, Australia. The analysis of FATMs indicated a potential competition between the two organisms for the same food source, namely benthic diatoms.

Additionally, the isotopic signatures of the crabs suggested that meiofauna could partly represent a trophic intermediate between diatoms and the soldier crab. In this case, it is clear that the complementary use of tools has provided more accurate information on trophic interactions than would have been possible using either technique alone.

2.4. Ecosystem-based indicators: the output of modelling techniques to determine ecosystem health (more than three trophic levels)

Many factors affect the health of marine ecosystems and their biodiversity, but arguably, commercial fishing is the strongest. The Johannesburg Plan of action recognizes overfishing, as a global challenge that can only be overcome by sustainable exploitation. The growing concern for the oceans' declining productivity and biodiversity has brought about the development of integrated approaches in resource management (Johnson et al., 2003). Since single-species models are inadequate, ecosystem modelling is used to study ecosystem responses to different stressors, and explore alternate scenarios for sustainable ecosystem-based fisheries management (Christensen et al., 2007). Under the assumption that one model can represent the state of an ecosystem and the processes underlying it, an ecosystem model can be used to compute a set of indicators describing the physical attributes, lower trophic levels (phyto- and zooplankton mainly in pelagic systems), higher trophic levels, and ecosystem integrity. It can also be used to simulate future ecosystem states of the ecosystem under various scenarios, and provide information about the values an indicator would take under any given scenario.

2.4.1. Dynamic trophic models

Ecosystem-based indicators derived from mass-balance models, such as EwE, focus on trophodynamics by measuring ecological interaction strengths and how they may be modified following perturbations such as fishing (Christensen et al., 2005; Cury et al., 2005). Usually, in strongly exploited ecosystems low trophic-level species proliferate at the expense of higher-trophic levels and this food-web stunting requires a certain ecological adaptive response, such as a redirection of predation fluxes to biological communities. Consequently, ecosystem structure and functioning will be modified with feedback effects on exploitation (Villanueva, 2004). Fig. 6 shows the effects of increases in the biomass of lower-trophic level species as a result of the depletion of higher-trophic

level species (Villanueva, 2004). In such a situation, ecosystem networks are modified due to energy flows and connectivity changes. Other ecosystem-based indicators measuring transfer efficiencies between trophic levels have been proposed such as the ‘Marine Trophic Index’ (MTI), fishing-in-balance (FiB) index and primary production required (Cury et al., 2005) based on criteria specified by the International Council for the Exploration of the Sea (ICES, 2001).

2.4.2. End-to-end models

End-to-end models combine the entire food chain, the physical environment and the potential effects of human pressures into an integrative framework to yield multi-level indicator values of ecosystem state (Fulton, 2010). End-to-end models are also useful for simulating ecosystem responses to forcing factors. This modelling technique has been applied to the southern Benguela upwelling region to quantify ecosystem state using various indicators such as fish-induced mortality on zooplankton, plankton seasonal dynamics, fish biomass and food web structure (Travers et al., 2009). The ecosystem response to overfishing has also been studied by comparing this model’s outputs with those resulting from an EwE model of the same ecosystem (Travers et al., 2010). Indicators including fish species biomass, mean community and catch trophic levels, predator–prey ratio, and biomass per trophic level can be used for this purpose. Whereas these indicators are easily computed from end-to-end models, most cannot be derived from sampling because ecosystems are constituted by too many components.

End-to-end models also test an indicator’s sensitivity to forcing factors and its response under different scenarios. For example, Travers et al. (2006) used the OSMOSE (Object-oriented Simulator of Marine eCOSystem Exploitation, Shin and Cury, 2004) model to test the sensitivity of size-based, species-based, and trophodynamic indicators to fishing. Sensitivity of these indicators were found to depend on the target species on which the fishing scenario is applied, the fishing intensity of the scenario (when a species is too highly overfished for example, some indicators no longer vary), and the way indicators are computed (species or community indicators, weighted by abundance or biomass, derived from individual metrics or distribution, etc.).

The Atlantis modelling framework (Fulton, 2010 and references therein) is a comprehensive ecosystem-based approach incorporating several components: biophysical model, fishing effects emulator model, management model, and a sampling model simulating realistic time-series monitoring data to produce a time-series of ecosystem-based indicators. This framework was built initially to evaluate different management strategies of fisheries in marine ecosystems along the Australian coast. Other applications however, can include exploration of single or cumulative ecosystem effects arising from multiple anthropogenic factors such as pollution, habitat loss, species invasion and climate change.

3. Difficulties in the practical use of ecological indicators

It is useful to highlight some encountered challenges from previous attempts of using monitoring programs to evaluate ecosystem health: (1) characterizing the features of a healthy ecosystem in an objective way, (2) availability of historic data to examine long-term trends, (3) the restricted spatial extent of studies, (4) separating natural fluctuations in the system from long-term trends, and (5) the lack of appropriate analytical methods (see White et al., 2010 and references therein). Furthermore, identifying and justifying baseline reference points for ecosystem indicators are turning out to be complex (Rice, 2003). Setting limits and trends is often based on expert knowledge, which can be highly subjective, and often,

these limits are inconsistent between ecosystems. In the absence of any well-defined and well-established reference levels for ecosystem indicators, reference directions are generally preferred in order to provide an assessment of recent ecosystem trends (Shin et al., 2010).

Transforming “conceptual” indicators proposed by the scientific community into “operational” indicators used for management purposes may not always prove as a straightforward task. Varieties of benthic indices are currently available and applied in environmental management in order to measure the status of ecological conditions and trends in succession in marine benthic systems. Here, the practical use of benthic indices is provided as an example to highlight some general shortcomings in the implementation of ecological indicators. A “good indicator” as defined by Salas et al. (2006) should be “applicable in extensive geographical areas and in the greatest possible number of communities or ecological environments”. Current benthic indicators do not fulfil this essential requirement as they are usually developed for a specific habitat and/or geographic area and are difficult to transpose to other habitats (Teixeira et al., 2010); for example, the B-IBI (Benthic Index of Biotic Integrity: Weisberg et al., 1997) was developed specifically for the Chesapeake Bay, whereas the MISS (Macrobenthic Index of Sheltered Systems: Lavesque et al., 2009) was proposed as an adaptation to *Zostera noltii* communities. There are many other examples (see Pinto et al., 2009). Also, the use of several indicators in the same area to compare their performance often gives contradicting results, questioning the general applicability of these indicators across habitats (e.g. Blanchet et al., 2008).

The second major drawback of current benthic indicators opposes another essential characteristic of a “good indicator”, which is the “handling easiness” (Salas et al., 2006). Most benthic indices currently in use are multi-metric because they offer the advantage, contrary to univariate indices, of being insensitive to seasonal variability of the macrofauna, especially those based on general life history traits (Reiss and Kröncke, 2005). However, in addition to requiring expert knowledge in the determination of benthic species, the calculation of multivariate indices relies on several metrics (e.g. 11 for the B-IBI) that are not always clearly defined or objective, and can sometimes be site-dependent (Dauvin et al., 2010; Pinto et al., 2009). Moreover, some operational limits are far from intuitive, such as the five digit values used for the BOPA and BO2A index (Dauvin, 1998).

Regulation plans, such as the WFD in Europe, has led to inflation in the number of benthic indicators, rendering the selection of the optimal suite of indicators even more difficult. Recent indices, based on the functioning of benthic communities rather than their species composition, might be more useful, at least as complementary indicators (cfr. Section 2.2.2.2; Elliott and Quintino, 2007). In this context, the use of stable isotopes (cfr. Section 2.3; Lefebvre et al., 2009), fatty acids (cfr. Section 2.3; e.g. Cheung et al., 2010) and the monitoring of CO₂ fluxes (Klaassen and Spilmont, 2012) also offer promising opportunities, especially in the intertidal environment, which was underlined recently as an interesting target for the assessment of the ecological quality of coastal areas (Fitch and Crowe, 2010).

4. Considering the complexity of marine systems

Understanding the responses of marine ecosystems to human and environmental pressures requires knowledge on the dynamics of perturbed systems (Hughes et al., 2005). Since marine ecosystems are complex adaptive systems (CAS) in which macroscopic dynamics emerge from numerous nonlinear interactions at smaller hierarchical (spatio-temporal) scales (Levin and Lubchenco, 2008), multiple responses can occur when pressures

are exerted on the system. Often, the state of the dynamic system under pressure will initially show little obvious change until a critical threshold is reached at which point a sudden shift to a contrasting dynamical regime takes place (Scheffer, 2009). For example, Beaugrand et al. (2008) and Kirby and Beaugrand (2009) have shown that a small increase in temperature can trigger abrupt ecosystem shifts across multiple trophic levels in marine systems in the North Atlantic and North Sea. An outcome of their studies was to demonstrate how biodiversity and the carrying capacity of an ecosystem like the North Sea, could be altered rapidly when multiple stresses acted synergistically, in this case when hydroclimatic change and overfishing were combined.

Identifying measures that capture the inherent complexity of marine systems remains a major challenge in ecosystem-based management plans. Presently, it seems unlikely that any simple “indicator” will embrace the dynamical functioning of marine ecosystems. Ideally, indicators of the state of marine systems should integrate linkages across different spatio-temporal scales. Within the set of CAS characteristics to sound management identified by Levin and Lubchenco (2008), it appears that fundamental elements to monitor the state(s) of marine systems should at least include diversity and heterogeneity (adaptive capacity of a system), redundancy and degeneracy (*sensu* Edelman and Gally, 2001; that is functional redundancy), modularity (in time, space and in organizational structures) and the tightness of feedback loops, and obviously, these elements are interrelated. In the light of this statement, several questions emerge: what is an ecological quality status/condition in a CAS? What is the meaning of a reference state/point or an ecological integrity and how can we quantify this?

One possible way to define an “indicator” of ecosystem state in a complex theory framework is to evaluate ecological resilience *sensu* Holling (1973), i.e. the magnitude of disturbance that can be tolerated before a system moves into a different region of state space and a different set of controls. Based on this interpretation, resilience has the following three properties: (a) the amount of change the system can undergo (and implicitly, therefore, the amount of extrinsic force the system can sustain) and still remain within the same domain of attraction (that is, retain the same controls on structure and function); (b) the degree to which the system is capable of self-organization (versus lack of organization, or organization forced by external factors); and (c) the degree to which the system can build the capacity to learn and adapt.

Allen et al. (2005) used discontinuities in size and functional groups to assess relative resilience in ecosystems since according to Peterson et al. (1998), systems with a diversity of function within scales and redundancy across scales are the most resilient. According to Carpenter et al. (2001) resilience measures differ in two important ways from traditional indicators: they apply to the entire system (its functioning and organization) and they focus on variables that underlie the capacity of the system to provide ecosystem services. As resilience implies that ecological systems do not simply track environmental forcing, the adoption of this term into management will de-emphasize treating environmental stress as a simple dose-dependent problem (Thrush et al., 2009). A recent whole-system field experiment suggests that early warning indicators related to variability, autocorrelation, and recovery time can be used to detect regime shifts in aquatic food webs (Carpenter et al., 2011). Whilst these statistical indicators can be useful tools for empirical studies of nonlinear dynamics of ecosystems or other complex systems, operational measures for ecosystem management based on the conceptual framework of adaptive cycles are still under development.

5. Discussion

There have been five major extinction events on Earth over geological time, and on each occasion the marine environment has experienced losses in biodiversity; the greatest of these occurred during the end-Permian mass extinction. Currently, there is a loss of biodiversity on Earth that is believed to be occurring at a rate unprecedented in Earth's history. This new Holocene extinction is often referred to as the sixth great extinction and it is considered to be solely due to anthropogenic causes including habitat destruction, hunting and climate change (Barnosky et al., 2011). It is therefore very timely to monitor the current state of marine ecosystems from the local to the global scale (Pereira and Cooper, 2006) in order to anticipate potential changes and to take appropriate management actions to ensure its sustainable use in the future.

The case studies in this paper show some examples of tools that can be used to derive indicators of ecosystem state and function. In the context of an ecosystem approach to marine management, the combination of attributes and tools that consider different aspects of the system could provide a more integrative view of its health. For example, indicators using both structural and functional attributes of key groups (Sections 2.2.1 and 2.2.2), may provide useful information on the quality of the physico-chemical environment as well as ecosystem health. Considering the complexity of systems, the study of trophic interactions and energy flows through the system (Sections 2.3 and 2.4) can be achieved by using a combination of techniques (stable isotope and stomach/gut content' analysis, Fatty Acid Trophic Markers (FATMs) and/or modelling techniques) to integrate information on several trophic levels giving an holistic view of the structure and the functioning of the system. While models mainly offer a simplified picture, they remain powerful tools, especially for large-scale analyses, and could provide an integrated view linking the environment to biotic interactions, as is the case for end-to-end models in particular. However, their use to make forecasts for management should be done with care until their performance and robustness are adequately tested (Rose et al., 2010).

Ironically, the emergence of ecosystem-based management coupled with increased awareness of human impacts on ecosystems has driven the need for organismal data that have greater resolution and accuracy (Hofmann and Gaines, 2008). While various metrics can be derived from low resolution models using functional groups for example, there is still the need to collect and analyse high resolution data. In effect, direct measurements can provide complementary information when combined with ecological modelling techniques. For example, empirical analyses provide ground truth data for validating dynamic models (Pelletier et al., 2008); abundance indices collected during monitoring programmes can be compared to abundances calculated from models to help model validation. In this respect, it is also important to ensure that the scales of data collection and model resolution are compatible. Since empirical approaches mostly pertain to local assessments, large-scale surveys and regional approaches should be adopted in order to facilitate their integration with models (Pelletier et al., 2008). Furthermore, empirical measurements can be used to test and calibrate the proposed indicators, a process often ignored in practice (Bockstaller and Girardin, 2003).

In order to produce a reliable assessment of ecosystem health for management purposes, a necessary trade-off will occur between capturing the complexity of the system and preserving the conceptual simplicity of the assessment. Combining several indicators where each individual indicator will represent a different aspect of ecosystem health may offer a solution not only to evaluate the ecosystem but also to reduce the level of uncertainty in the results (Dauvin, 2007). As an example, “Costanza's overall index of ecosystem health” (Costanza, 1992) is based on six attributes

of the system: (1) homeostasis; (2) absence of disease; (3) diversity or complexity; (4) stability or resilience; (5) vigour or scope of ecosystem growth; and (6) balance between system components. Even though Costanza's concept appears attractive, the practicality and usefulness as an indicator in ecosystem-based management has not yet been fully demonstrated.

6. Conclusions

Given the complexity of managing marine ecosystems in the face of potentially interacting stressors, the need to assess the health and resilience of marine ecosystems is increasing (Hofmann and Gaines, 2008). In response to this demand, appropriate indicators and evaluation tools which do not mask or leave out inherent ecosystem properties and dynamics should be selected. In the case studies described previously, empirical and modelling techniques and, in particular, their complementary use to derive population, community and ecosystem indicators is illustrated and further developments along this line should be encouraged to obtain operational indicators for ecosystem-based management in the marine environment.

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