



# United Nations Environment Programme

UNEP(DEC)/MED WG.268/Inf.14  
2 May 2005

ENGLISH



## MEDITERRANEAN ACTION PLAN

## Seventh Meeting of National Focal Points for SPAs

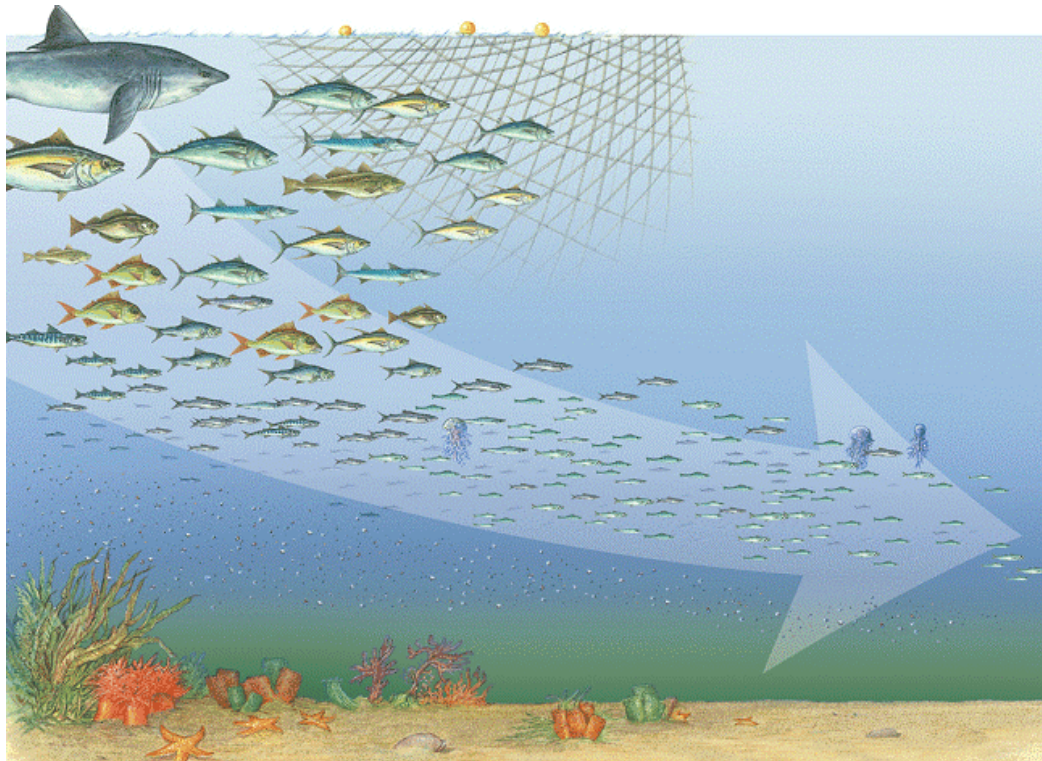
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## APPLICABILITY OF THE CBD INDICATORS FOR DETERMINING TRENDS IN MEDITERRANEAN MARINE BIODIVERSITY

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## **Applicability of the CBD Indicators for Determining Trends in Mediterranean Marine Biodiversity**



## **Applicability of the CBD Indicators for Determining Trends in Mediterranean Marine Biodiversity**

### **Executive Summary**

This report presents a preliminary assessment of the applicability of indicators identified by the Ad Hoc Technical Expert Group (AHTEG) on Indicators for Assessing Progress Towards the 2010 Biodiversity Target, appointed by the secretariat of the Convention on Biological Diversity (AHTEG, 2004-2005) for evaluating the conditions and trends in marine biodiversity throughout the Mediterranean region. The indicators found to have immediate application in the global context include: (a) trends in biomes, ecosystems, and habitats; (b) trends in abundance and distribution of selected species; (c) coverage of protected areas; (d) nitrogen deposition; (e) marine trophic index; (f) water quality in aquatic ecosystems; (g) status and trends of linguistic diversity and numbers of speakers of indigenous languages; and (h) official development assistance provided in support of the Convention (Convention on Biological Diversity). In addition, the following indicators were identified as having potential in the Global Biodiversity Outlook, but need further development: (a) change in status of threatened species; (b) trends in genetic diversity of domesticated animals, cultivated plants, and fish species of major socio-economic importance; (c) area of forest, agricultural and aquaculture ecosystems under sustainable management; (d) numbers and cost of alien invasions; and (e) connectivity and /or fragmentation of ecosystems.

In order to determine the suitability of these indicators for assessing marine biodiversity in the Mediterranean context, the definitions of each of the indicators and the methodologies used to determine values were reviewed. By way of trial testing of applicability, those indicators found to be of particular relevance in the regional context (trends in certain habitats, trends in abundance of selected species, marine protected area coverage, nitrogen deposition, marine trophic index, and water quality) were investigated at the national level in five Mediterranean countries (Spain, Italy, Turkey, Tunisia, and Morocco). These countries present a diverse portfolio of biodiversity resources, a wide range of economic conditions, and somewhat differing approaches to coastal management, and they together represent each of the seven biogeographic regions of the Mediterranean Basin. The constraints in using these indicators at the national level, given current data availability, is discussed.

The report concludes with recommendations on additional indicators that might be used to fully understand condition and trends in Mediterranean marine biodiversity, within the framework of Mediterranean Action Plan (e.g indicators for marine species of Mediterranean concern).

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## **I. Introduction**

### **A. Biodiversity and Marine Biodiversity**

Biological diversity is the variability of living organisms. It includes all plants, animals, microorganisms, the ecosystems of which they are part, and the diversity within species, among species, and of ecosystems. No single component of biodiversity (i.e. genes, species, or ecosystems), is consistently a good indicator of the overall biodiversity as these components can vary independently (Duraiappah et al., 2005).

Over the past few hundred years humans have increased species extinction rates by as much as 1000 times background rates typical over the planet's history. There are approximately 100 documented extinctions of birds, mammals and amphibians over the past 100 years, a rate approximately 50 to 500 times higher than background rates (MA, 2005). Including possibly extinct species, the rate is more than 1000 times higher than background rates (MA, 2005). A range of techniques, including extrapolations from known extinctions and estimations based on modelled impacts of habitat change yield comparable estimates of contemporary species extinction rates.

The distribution of species on earth is becoming more homogenous, i.e. suites of species are becoming more similar across the planet. (Duraiappah et al., 2005). Two factors are responsible for this trend. First, the extinction of species or the loss of populations results in the loss of the presence of species that had been unique to particular regions. Second, the rate of invasion or introduction of species into new ranges is already high and continues to accelerate in pace with growing trade and faster transportation (MA, 2005). Currently documented rates of species introductions to different regions of the world are greater than documented rates of extinction. Thus, while the total number of species on the planet is decreasing due to extinctions, the total number of species on each individual continent is actually increasing. The full ecosystem consequence of homogenization depends on the aggressiveness of the introductions and the services they either bring or impair. However, marine organisms have not been tracked to the same degree as terrestrial ones, masking what may be similarly alarming extinction risks. Genetic diversity has been reduced in commercially important marine fishes, as stocks become reduced or eliminated by heavy fishing pressure (MA, 2005).

While there have been benefits to humans from actions that cause the loss of biodiversity, the benefits of these changes have not been equitably distributed among people and many of the costs and risks of changes in biodiversity have historically not been factored into decision-making (Duraiappah et al., 2005). These costs and risks include:

**Decline of specific ecosystem goods and services:** Many of the changes that have been made in biodiversity and ecosystems have occurred to enhance the production of specific ecosystem services such as food production. However, only four of the 22 ecosystem services examined in the Millennium Assessment have been enhanced: crops, livestock, aquaculture, and (in recent decades) carbon sequestration (MA, 2005). In contrast, 14 other services have been degraded, including capture fisheries, timber production, water supply, waste treatment and detoxification, water purification, natural hazard protection, regulation of air quality, regulation of regional and local climate, regulation of erosion, and many cultural services (spiritual, aesthetic, recreational and other benefits from ecosystems). Modifications of ecosystems to enhance one service generally have come at a cost to other services that the ecosystem provided. The impacts of these trade-offs among ecosystem services affect people in different ways. For example, an aquaculture farmer may gain material welfare from

management practices that increase soil salinization and thereby reduce rice yields and threaten food security for nearby subsistence farmers (Duraiappah et al., 2005).

Even where the net economic benefits of changes leading to the loss of biodiversity have been positive, many people have often been harmed by the changes. In particular, poor people, particularly those in rural areas in developing countries, are more directly dependent on biodiversity and ecosystem services and more vulnerable to their degradation. For example, the ecosystem service with the most significant decline in production – capture fisheries – is also a service that was particularly valuable to poor people, since it provided a cheap source of protein. Richer groups of people are often less affected by the loss of ecosystem services because of their ability to purchase substitutes or to offset local losses of ecosystem services by shifting production and harvest to other regions. For example, as fish stocks have been depleted in the north Atlantic, European and other commercial fishers have shifted to harvest fish from West Africa. Similarly, agricultural intensification may increase production of the main crops while denying poor people, especially the landless, access to food plants – including those regarded as weeds – which make important contributions to household nutrition (Duraiappah et al., 2005).

Many costs associated with changes in biodiversity may be slow to become apparent, may be apparent only at some distance from where biodiversity was changed, or may involve thresholds or changes in stability that are difficult to measure. For example, there is established but incomplete evidence that reductions in biodiversity reduce the resilience of ecological systems. The resilience of a system is a measure of the ability of a system to return and to its pre-disturbance state following a disturbance. The costs associated with this loss of resilience may not be apparent for years until a significant disturbance is experienced. Similarly, a change in biodiversity in one location may have impacts in other locations. The conversion of forest to agriculture in one region for example, can affect the timing and magnitude of river flows in downstream coastal areas far removed from the change in biodiversity.

Threshold effects – abrupt or non-linear changes in a system in response to a more gradual change in a driving force – have been commonly encountered in ecosystems and are often associated with changes in biodiversity (MA, 2005). Overfishing is known to cause abrupt changes in species populations in coastal ecosystems. In tropical coral reefs, loss of herbivorous fish, has contributed to the degradation of reefs to algal-dominated forms. In temperate seas, declines in grazing species have led to reduced biodiversity in urchin barrens (Dayton et al., 1995). Multiple drivers may be involved in these “regime shifts”. The introduction of the carnivorous ctenophore *Mnemiopsis leidyi* (a jellyfish-like animal), in the Black Sea, caused the loss of 26 major fisheries species and is implicated (along with other factors) in the continued growth of the oxygen-deprived “dead” zone. The species was subsequently introduced into the Caspian and Aral Seas, where it is causing similar impacts (MA, 2005).

Tools now exist for a far more complete assessment of the consequences of biodiversity loss for human well-being but most decisions continue to be made in the absence of a detailed analysis of the full costs, risks and benefits (Duraiappah et al., 2005). Despite the existence of these tools, only provisioning ecosystem services are routinely valued. Most supporting and regulating services are not valued at all because the demand curves for these services, which are not privately owned or traded cannot be directly observed or measured. In addition, it is recognized that biodiversity has intrinsic value, which, not being anthropocentric, cannot be valued in conventional economic terms (MA, 2005).

The most important direct drivers of biodiversity loss and change in ecosystem services are habitat change (land use change and physical modification of rivers or water withdrawal from rivers), climate change, invasive alien species, overexploitation, and pollution. For most of these drivers, and for most ecosystems where they have been important, the impact of the

driver is currently remaining constant or growing. Each of these drivers will have important impacts on biodiversity in the 21<sup>st</sup> century (from Duraiappah et al., 2005):

**Habitat transformation, particularly from conversion to agriculture:** Roughly one-third of the earth's terrestrial surface is already cultivated land. Under the MA scenarios, a further 10 to 20% of grassland and forest land is projected to be converted by 2050 (primarily to agriculture). While the expansion of agriculture and the increased productivity of agriculture is a success story of enhanced production of one key ecosystem service, this success has come at high and growing costs in terms of trade-offs with other ecosystem services, both through the direct impact of land cover change and as a result of water withdrawals for irrigation and release of nutrients into rivers. For example, globally roughly 15 to 35 percent of irrigation withdrawals are estimated to be non-sustainable. Habitat loss also occurs in coastal and marine systems, though these transformations are less well seen. Bottom trawling, for instance, can reduce certain diverse benthic habitats into undersea deserts, while destructive fishing and coastal development can completely destroy reefs and seagrass beds.

**Overexploitation, especially overfishing:** For marine ecosystems the most important direct driver of change globally has been over-fishing. Demand for fish as food for people and feed for aquaculture production will expand, and the result will be an increasing risk of major, long-lasting collapse of regional marine fisheries. In some marine systems the biomass of both targeted species, especially larger fish, and those caught incidentally (by-catch) has been reduced up to one or more orders of magnitude compared to pre-industrial fishing levels (Christensen et al., 2003). About three quarters of the world's marine fisheries are either fully exploited or overexploited. More shoreward, the loss of biodiversity in coastal systems is driven by habitat loss and degradation, especially via pollution that leads to eutrophication, over-exploitation, and climate change.

**Biotic exchange:** The spread of invasive alien species and disease organisms has increased because of increased trade and travel, including tourism. Increased risk of biotic exchange is an inevitable effect of globalization. While, increasingly there are measures to control the pathways of invasive species, for example, through quarantine measures and new rules on the disposal of ballast water in shipping, several pathways are not adequately regulated.

**Nutrient Loading:** Since 1950, nutrient loading has emerged as one of the most important drivers of ecosystem change in terrestrial, freshwater and coastal ecosystems, and this driver is projected to substantially increase in the future. Synthetic production of nitrogen fertilizer is been a key driver for the remarkable increase in food production that has occurred during the past 50 years. Humans now produce more reactive nitrogen than is produced by all natural pathways combined. Aerial deposition of reactive nitrogen into natural terrestrial ecosystems, especially temperate grasslands, shrublands and forests leads directly to lower plant diversity while excessive levels of reactive nitrogen in water bodies, including rivers, and other wetlands, and coastal zones frequently leads to algal blooms and eutrophication. Similar problems have resulted from P, the use of which has tripled. Nutrient loading will become an increasingly severe problem, particularly in developing countries and particularly in east and south Asia. Only significant actions to improve the efficiency of nutrient use will mitigate these trends.

**Anthropogenic Climate Change:** Observed recent changes in climate, especially warmer regional temperatures, have already affected biological systems in many parts of the world. There have been changes in species distributions, population sizes, the timing of reproduction or migration events, and an increase in the frequency of pest and

disease outbreaks, especially in forested systems. Many coral reefs have undergone major, although often partially reversible, bleaching episodes, when sea surface temperatures have increased by 1°C during a single season. By the end of the century, climate change and its impacts will be one of the most important direct drivers of biodiversity loss and change of ecosystem services. Climate change increases the rate of species extinction and the loss of genetic diversity; increases the spread of pathogens and incidence of disease in human and non-human populations; directly alter ecosystem services, for example, by causing changes in the productivity and growing zones of cultivated and non-cultivated vegetation; changes the frequency of extreme events, with associated risks to ecosystem services; indirectly affects ecosystem services in many ways, such as by causing sea level to rise which threatens mangroves and other vegetation that now protect shorelines and changing the pH of the oceans; and, increases the difficulty of meeting needs for clean water, energy services and food.

Different interpretations of several important attributes of the concept of biodiversity can lead to confusion in understanding both scientific findings and their policy implications (from Duraipah et al., 2005). Specifically:

The value of the diversity of genes, species, or ecosystems *per se*, is too often confused with the value of a particular component of that diversity. Species diversity in of itself, for example, is valuable because the presence of a variety of species helps to increase the capability of an ecosystem to be resilient in the face of a changing environment. At the same time, an individual component of that diversity, such as a particular food plant species, may be valuable as a biological resource. The consequences of changes in biodiversity for people can stem both from a change in the diversity *per se* and a change in a particular component of biodiversity. Each of these aspects of biodiversity deserves its own attention from decision-makers and each often requires its own management goals and policies.

Second, because biodiversity refers to diversity at multiple scales of biological organization (genes, species, and ecosystems) and can be considered at any geographic scale (local, regional, or global), it is generally important to specify the specific level of organization and scale of concern. For example, the introduction of widespread weedy species to a continent such as Africa, will increase the species diversity of Africa (more species present), while decreasing ecosystem diversity globally (since the ecosystems in Africa then become more similar in species composition to ecosystems elsewhere due to the presence of the cosmopolitan species). Because of the multiple levels of organization and multiple geographic scales involved, any single indicator, such as species diversity, is generally a poor indicator for many aspects of biodiversity that may be of concern for policy makers.

Changes in biodiversity and other environmental changes influence each other in both directions: biodiversity loss can reduce an ecosystem's resilience to environmental perturbation such as brought about by climate change (warming), ozone depletion (increased radiation), and pollution (eutrophication, toxics), while all of these impacts can also result in reduction of biodiversity. In general, diverse natural systems in which neither species, nor population, nor genetic diversity has been severely constricted are better able to adapt to changing environmental conditions. Unaltered coral reefs, for instance, are less likely to experience coral bleaching and subsequent mortality when ocean temperatures increase, and unmodified, diverse grasslands are less susceptible to pest outbreaks when new pest species are introduced (MA, 2005). However, all environmental change has the potential to cause biodiversity loss, especially at the level of genes and populations. The greater the magnitude of change and the more rapid the rate of change, the more likely biodiversity will be affected (and the more likely that subsequent environmental change will lead to greater ecosystem degradation).

Improved, and more widely applicable, measures of biodiversity would aid decision making, at global, regional and national levels (MA, 2005). Existing biodiversity indicators are helping to communicate trends in biodiversity and highlight its importance to human well-being. Improved measures, of biodiversity, agreed by stakeholders, would assist in setting appropriate targets, addressing tradeoffs between biodiversity conservation and other objectives, and in optimizing responses (Duraiappah et al., 2005). Models can and should be developed and used to make better use of limited observational data. Additional effort is required to reduce critical uncertainties, including those associated with thresholds of biodiversity related to the provision of ecosystem goods and services. Given the multiple values of biodiversity, no single measure is likely to be appropriate for all situations.

## **B. Use of Indicators to Measure Changes to and Loss of Biodiversity**

Given the complexity of what constitutes biodiversity, it is clear that actual changes to biodiversity cannot be measured in full in any natural environment, and given logistical considerations, this holds especially true in the marine environment. What is more important is that good metrics are used to measure biodiversity “loss”. The Convention on Biodiversity defines biodiversity loss to be “the long-term or permanent qualitative or quantitative reduction in components of biodiversity and their potential to provide goods and services, to be measured at global, regional and national levels” (CBD COP VII/30). Under this definition, biodiversity can be lost either if the diversity *per se* is reduced (such as through the extinction of some species) or if the potential of the components of diversity to provide a particular service is diminished (such as through unsustainable harvest). The homogenization of biodiversity – that is, the spread of widespread invasive alien species around the world – thus also represents a loss of biodiversity at a global scale (since once-distinct groups of species in different parts of the world become more similar) even though the diversity of species in particular regions may actually increase because of the arrival of new species (Duraiappah et al., 2005).

Biodiversity is defined as, “the variability among living organisms from all sources including, *inter alia*, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems” (United Nations 1992:Article 2). The importance of this definition is that it draws attention to the many dimensions of Earth’s biological diversity. It explicitly recognizes that every biota can be characterized by its taxonomic, ecological, and genetic diversity and by the way these dimensions of diversity vary over space and time is a key feature of biodiversity (Duraiappah et al., 2005). By this definition, for example, a heterogeneous landscape or seascape in which ecosystems are made stable by a web of compensatory interactions among its species is readily recognized as more biodiverse than a homogenous landscape made up of unstable ecosystems that consist largely of non-interacting species. By this definition, the former could readily be considered more biodiverse even if the latter contained twice as many species. Taxonomic diversity is important, but only one component of biodiversity. Thus, only a multidimensional assessment of biodiversity can provide insights into the relationship between changes in biodiversity and changes in ecosystem functioning and ecosystem services (MA, 2005).

In spite of many tools and data sources, biodiversity remains difficult to quantify precisely. However, precise answers are seldom needed to devise an effective understanding of where biodiversity is, how it is changing over space and time, what drivers are responsible for such change, what the consequences of such change are for ecosystem services and human well-being, and what response options are available (Duraiappah et al., 2005). Ideally, to assess the trends and conditions of biodiversity either globally or sub-globally, one must measure the abundance of all organisms, the visible and invisible, over space and time, using taxonomy (e.g., the number of species), functional traits (e.g., the ecological type such as woody versus herbaceous plants), and the interactions among species that affect their

dynamics and function (e.g., predation, parasitism, competition, and facilitation such as pollination, and how strongly such interactions affect ecosystems). Even more important would be to estimate turnover, not just point estimates in space or time. Currently, it is not possible to do this with much accuracy because the data are lacking. Even for the taxonomic component of biodiversity, where information is the best, considerable uncertainty remains about the true extent and changes in diversity, and again, this is especially true in the marine environment.

There are many measures of biodiversity of which species richness represents a single, but important metric that is valuable as the common currency of the diversity of life – but it must be integrated with other metrics to fully capture biodiversity (Duraiappah et al., 2005). Because the multidimensionality of biodiversity poses formidable challenges to the measurement of biodiversity, a variety of surrogate or proxy measures are often used. These include the number of species or species richness of specific taxa, the number of distinct functional types, or the diversity of distinct gene sequences in a sample of microbial DNA taken from the soil (MA, 2005). Species- or other taxon-based measures of biodiversity, however, rarely capture key attributes of biodiversity, such as variability, function, quantity, and distribution, all of which provide insight into the roles of biodiversity. Rareness is an important aspect to biodiversity, and a phenomenon not confined to terrestrial systems – for instance, in a study of 800 Norwegian marine species, over 200 species occurred at only one site (Gray, 1997). Thus, wherever possible, attributes other than taxonomic diversity should and are often measured along with species richness.

Ecological indicators form a critical component of monitoring, assessment, and decision making, but may not serve as adequate indicators of actual changes in biodiversity (Gubbay, 2004; Kabuta et al., 2003). Although ecological indicators are scientific constructs that use quantitative data to measure aspects of biodiversity, ecosystem condition, services, or drivers of change, no single ecological indicator captures all the dimensions of biodiversity (Duraiappah et al., 2005).

While existing data are often insufficient to provide accurate pictures of the extent and distribution of biodiversity, there are many patterns and tools that decision makers or the public can use to derive useful approximations of biodiversity in a given ecosystem. Each country or region will have different information about their biota, north-temperate regions often having the best data, but for some groups, usually vertebrates, plants, and some invertebrates such as butterflies and dragonflies, there is frequently some information. There are also some basic principles about biodiversity distributions or indicators that can be used to supplement existing biotic inventories (Duraiappah et al., 2005). In the marine environment, for example, species richness is associated in part with nutrient availability, temperature, salinity, and water depth, as well as physical heterogeneity of the landscape/seascape.

The difficulties in measuring marine biodiversity extend well beyond scientific understanding of ecosystem structure and function – they have also complicated assessments of the impact of management and response strategies. Existing measures often focus on local biodiversity and do not estimate the marginal gains in regional/global biodiversity values (Duraiappah et al., 2005). For example, biodiversity gains from marine protected areas are typically expressed only as localized species richness, with no consideration of degree of contribution to regional/global biodiversity or trade-offs with commercial fisheries. And even while the bulk of biodiversity remains unknown, better ways are required to estimate the biodiversity importance of different places. Developing a better calculus of biodiversity would enhance integration among strategies/instruments and create opportunities for more effective conservation (Duraiappah et al., 2005).

**C. Marine Biodiversity of the Mediterranean** (excerpted in part from Batisse and Jeudy de Grissac, 1995, except where indicated otherwise)

The Mediterranean lies between Europe, Asia and Africa (about 46°N, 30°N, 6°W and 36°E) and, excluding the Black sea, covers an area of approximately 2.5 million square kilometers, with an average depth of about 1.5 square kilometers and a volume of 3.7 million cubic kilometers. The Mediterranean Sea is comprised of two major basins, western and eastern, that are divided by the relatively shallow strait of Sicily. These two basins are in turn divided into a series of interacting parts and adjacent seas. The Western Mediterranean covers about 0.85 million square kilometers and the Eastern Mediterranean about 1.65 million square kilometers.

There are natural biogeographic subdivisions within the Mediterranean region : foremost being the submarine ridge that runs between the island of Sicily and the African coast at a depth of 360 meters, dividing the Mediterranean into western and eastern sections. Further geographic divisions made by Batisse and Jeudy de Grissac (1995) and others are listed in Table 1.

**Table1.** Main Biogeographic Subdivisions of the Mediterranean Marine Region  
(from Batisse and Jeudy de Grissac, 1995)

Location	Area (square kilometers)	Maximum depth (meters)
Alboran Sea (AL)	69,000	1,375
Algero/Prov. Basin (AP)	700,000	2,000
Tyrrhenian Sea (TY)	247,000	3,000
Ionian Basin (IO)	938,000	5,092
Levantine Basin (LE)	667,000	3,000
Aegean Sea (AE)	214,000	3,543
Adriatic Sea (AD)	131,000	1,324
Total Mediterranean Sea	2,966,000	

The Mediterranean Sea has a negative hydrological balance, with loss through evaporation exceeding the input of water through runoff and precipitation. This deficiency is mainly compensated by the flow of Atlantic surface waters through the Strait of Gibraltar (about 35,000 cubic kilometers). The major feature of the surface current system of the Mediterranean is the movement of water from the Atlantic toward the east combined with numerous spin-off eddies along the way. There is no surface return system from the east to the west, but a return of Mediterranean water takes place by way of intermediate and deep water flowing from east to west and spilling over the sill of Gibraltar into the Atlantic. Such intermediate and deep water results from very pronounced evaporation that gradually transforms surface water with salinity slightly above 36‰ from the Atlantic into denser water with salinity of 38‰ or more, reaching 39.5‰ in the Eastern Basin.

The Mediterranean circulation system also includes strong vertical convection currents that determine the distribution of salinity and provide for vertical recycling of nutrients and other dissolved substances. However, the sea has relatively low concentrations of nutrients even in deeper waters. These chemicals are exported in the flow of deep water through the Strait of Gibraltar that in turn receives nutrient poor surface Atlantic water. No deep nutrient-rich Atlantic waters take part in the Mediterranean circulation, and the input of nutrients is mostly due to river input and agricultural runoff or pollution (Miller 1983).

The uneven distribution of runoff and precipitation along the northern coasts of the Mediterranean Sea, combined with the concentration of population and industrial activity in the north, contributes a waste load of pollutants to northern Mediterranean waters that may eventually spread to other areas. Heavy traffic of oil tankers used to lead to the spread of tar balls on the coasts but this has been considerably reduced through implementation of the Barcelona Convention. Plastic containers and other floating residue from ships and sailing boats remain a nuisance in most areas of the basin. Significant progress has been achieved in the reduction of discharge of urban effluence from major cities although much remains to be done. Relatively high levels of mercury occur in some parts of the basin, but appear to be largely of natural origin and do not dangerously affect fish consumption.

The Mediterranean is naturally adapted to avoid excessive eutrophication since it loses deep water, relatively rich in mineralized or recycled nutrients, and receives low nutrient Atlantic surface water. However, when nutrients are continuously discharged into coastal waters in excess of their self-purification capacity the oxygen balance is disrupted. The oxidation of organic matter then proceeds through anaerobic pathways and coastal waters rapidly become turbid and poisonous to certain forms of marine life. This process can have very negative effects on tourism, as experienced in recent years on the Adriatic coasts (UNESCO 1988; MA, 2005).

Mediterranean waters are oligotrophic except in the vicinity of large rivers, and sediments have in general a low organic carbon content due to the low biological productivity of the waters and the presence of high oxygen concentrations in deepwaters. Local oxygen deficiencies are always connected with eutrophication sources, mostly discharges of raw or treated urban or agricultural effluents. Their distribution around the region is uneven, with a maximum in the northwest and in the Adriatic Sea and a minimum on the southern shores. Owing to the strong stratification of surface waters, eutrophication is more acute in summer when ambient nutrient concentrations are low and oxygen transport through the thermocline is strongly reduced. Winter mixing allows for the required vertical transport of oxygen to keep the deepwaters and the sediments oxidized all over the Mediterranean Sea (Cruzado, 1985).

Approximately 54 percent of the Mediterranean coasts are rocky, yet there are a number of large alluvial plains associated with the deltas of major rivers (Ebro, Rhone, Po and Nile) and those of numerous smaller rivers of the basin, particularly in Tunisia, Greece and Turkey. These rivers drain soils far removed from the coastline and carry very large volumes of

sediment to the sea. Short, often torrential rivers drain small areas on a highly seasonal basis. Thirty-one percent of the soils of the Mediterranean basin lose over 15 tons per hectare per year through erosion, and the loss may reach 250 tons in some parts of Morocco, Italy and Syria. However, the effect of soil erosion on the Mediterranean is not as big an issue as the amount of pollutants carried by these rivers, particularly the Ebro, Rhone and the Po, that drain regions with heavy industrial and agricultural activity.

While it exhibits a low level of biological productivity, the Mediterranean Sea as well as the surrounding lands is characterized by a relatively high degree of biological diversity (UNEP, 1990;1999). The fauna includes many endemic species and is considered richer than that of Atlantic coastal areas (Bianchi and Morri, 2000). The continental shelf is usually very narrow, but the coastal marine area of the Mediterranean, which stretches from the shore to the outer extent of this continental shelf, shelters rich ecosystems and the few areas of high productivity in the sea. Whereas central zones of the Mediterranean are low in nutrients, coastal zones benefit from telluric nutrients that support higher levels of productivity.

The Mediterranean marine vegetation includes about 1,000 macroscopic species, of which about 15 to 20 percent are endemic. *Posidonia oceanica* meadows play a central role in stabilizing the seashore and in maintaining water quality, particularly through oxygen production. The stability of the seashore is maintained by these meadows, and in a number of places the disappearance of sandy beaches has soon followed the disappearance of seagrass meadows. *Posidonia oceanica* meadows are the most important fish nursery areas in the Mediterranean. However, forty species of seagrasses and algae are considered endangered: 38 algae and 2 marine phanerogams (*Posidonia oceanica* and *Zostera marina*). Endemic seagrasses in the northwest Mediterranean are currently threatened by the invasion of an exotic tropical species, *Caulerpa taxifolia*, that was accidentally released in 1984 and has now spread over nearly 2000 hectares, mainly in France but also in Italy and the Balearic Islands (Meinesz et al., 1993).

Mediterranean wetlands and lagoons are of great significance to the conservation of biological diversity and are also highly productive. They perform numerous other functions related to flood control, recreation, tourism, fisheries and agriculture as well as chemical and physical reduction of pollution. They also act as breeding and wintering areas for a great variety of birds and are essential stopover points on the migratory routes of numerous bird species. Nonetheless, a significant number of Mediterranean wetlands have been "reclaimed" over history. Important lagoon systems remain in Spain (Valencia), France (Languedoc and Giens), Italy (Sardinia, Toscana, Pylia, and Venice), Central Greece, Cyprus, Morocco (Nadar), Algeria, in many places in Tunisia, and across the entire Nile delta in Egypt. Estuaries constitute another important and widespread habitat, as there are some 70 sizeable rivers and streams flowing into the Mediterranean. Finally, the region's rocky shores have characteristic biogenic constructions, including platforms with *Lithophyllum licheonides* on steep coasts and vermeted platforms on calcareous coasts.

These and other ecosystems are also important for endangered species. This is the case for the Mediterranean monk seal, which uses caves as habitat, for marine turtles, which use sandy beaches for nesting, seagrasses for feeding and seagrasses or muddy bottoms for wintering, and for marine birds, which use wetlands, rocky shores or islands for nesting and resting (Ramade, 1990).

The biota of the Mediterranean Sea consists primarily of Atlanto-Mediterranean species (62 percent) derived from the adjacent Atlantic biogeographic provinces beyond the Strait of Gibraltar. Many Mediterranean species are endemic (20 percent) while others are cosmopolitan or circumtropical (13 percent) or Indo-Pacific (5 percent). These proportions differ for different major taxonomic groups and also for different parts of the Mediterranean Sea, but the pattern remains essentially the same (Ketchum, 1983).

Within the Mediterranean there is a gradient of increasing species diversity from east to west. The number of species among all major groups of plants and animals is much lower in the

eastern Mediterranean than in the western and central parts of the sea. The southeast corner, the Levant Basin, is the most impoverished area. The benthic and littoral populations show a similar change in species diversity and abundance, which decrease from west to east, and from the northern Adriatic to the south (Ketchum, 1983).

The establishment of a database named "Medifauna" has made it possible to compare the world's marine fauna (about 130,000 described) with that of the Mediterranean (about 8,000 known marine metazoans). Included in the bank are 5,315 species, of which 1,776 are under verification. The Mediterranean Sea includes 6 percent of the world's species for less than 1 percent of the world's ocean area and less than 0.003 of its volume. The number of endemic species is significantly higher than that for the Atlantic Ocean. The percentage of endemism is very high for the sessile or sedentary groups such as ascidians with 50.4 percent, sponges with 42.4 percent, hydroids with 27.1 percent, echinoderms with 24.3 percent, but it is also considerable for the other groups such as decapod crustaceans with 13.2 percent and fish with 10.9 percent. An average of 28% of all species are endemic.

The *Larus audouinii* (Audouin's gull) has reached dangerously low population levels and depends on rocky islands and archipelagoes, free from disturbance, as breeding sites. The Audouin's gull population in the Mediterranean is in the order of 600-800 pairs. Several species of birds typical for the Mediterranean climatological region are threatened in their European, and possibly in the whole of their Mediterranean range, because of the loss of suitable disturbance-free habitat. Of particular note are the endangered species *Pelecanus onocrotalus* (white pelican), *P. crispus* (Dalmatian pelican), *Egretta alba* (great white heron), *Phoenicopterus ruber* (greater flamingo), and *Larus genei* (slender-billed gull). The Mediterranean is of significant importance for migratory birds and twice a year some 150 migratory species cross the narrow natural passages in the regions of Gibraltar, Cap Bon (Tunisia), Messina (Italy), Belen Pass (Turkey), Lebanese coast, and the Suez Isthmus, taking advantage of the wetlands occurring on their way (Ramade, 1990).

The loggerhead (*Caretta caretta*), leatherback (*Dermochelys coriacea*), and green (*Chelonia mydas*) marine turtles are all found in the region. While the loggerhead remains relatively abundant, it seems to have deserted many parts of the Western Basin where it is disturbed by fishing activity. The other two species are becoming increasingly rare. Nesting sites for the herbivorous and migratory green turtle can be found in Cyprus, Turkey, Egypt and Libya. There are only a total of 2,000 nesting females at these sites and this number is declining. The leatherback turtle is rarely seen in the Mediterranean, although there are some breeding records for Israel and Sicily. Important nesting sites for the loggerhead turtle are located on the coast from Turkey to Israel, on a number of Mediterranean islands, and at scattered sites along the North African coast.

Several species of marine mammals have reached dangerously low population levels, and their survival has become questionable unless immediate measures are taken for their conservation. The species in which this is most evident is *Monachus monachus* (Mediterranean monk seal), which depends on rocky islands and archipelagoes free from disturbance as breeding sites. The population of these seals in the Mediterranean is probably less than 300 individuals. Their greatest concentration occurs along the Turkish and Greek coasts and around the Aegean islands. Very small populations are also thought to remain in Morocco, Algeria and Libya.

About 20 different cetacean species has been reported in the Mediterranean Sea, about half of which come Atlantic populations entering the sea only sporadically. Only nine small cetacean species and three large whales species are sighted frequently in the Mediterranean Sea. They are: *Balaenoptera acutorostrata* (Minke whale); *Balaenoptera physalus* (Fin whale); *Delphinus delphis* (Common dolphin); *Globicephala melas* (Long-finned pilot whale);

*Grampus griseus* (Risso's dolphin); *Orcinus orca* (Killer whale); *Physeter macrocephalus* (Sperm whale); *Pseudorca crassidens* (False killer whale); *Stenella coeruleoalba* (Striped dolphin); *Steno bredanensis* (Rough-toothed dolphin); *Tursiops truncatus* (Bottlenose dolphin); and *Ziphius cavirostris* (Cuvier's beaked whale).

The Mediterranean fish fauna is diverse but fisheries are generally declining. Of the 900 or so known fish species, approximately 100 are commercially exploited. Unsustainable catch rates of rays (including the disappearance of certain taxa from commercial catches) and other demersal species are of special concern (Tudela, 2004). Fisheries impacts extend beyond elasmobranchs, finfish, or other target species: longline fishing is a main cause of seabird mortality in the Mediterranean; while longline and other fisheries kill sea turtles incidentally (Tudela, 2004). Longline fleets are a particular threat to the loggerhead turtle population, as are trawlers and small-scale gears in some areas, such as in the Gulf of Gabès. Driftnet fisheries and, to a much lesser extent, small-scale fisheries using fixed nets and purse seine fisheries appear to account for the highest impact on the region's cetaceans and are also responsible for the highest rates of direct human-induced mortality. The population of monk seal in the Mediterranean continues to be at risk from both direct mortality by artisanal fishing gears and an increasing scarcity of food resources driven by overfishing.

Benthic communities including sea mount communities, volcanic vent communities, bryozoans, corals, hydroids and sponges are vulnerable to human disturbance. The mechanical disturbance of marine habitats that occurs with some activities such as trawling, dredging, dumping, and oil, gas and mineral exploration and extraction; can substantially change the structure and composition of benthic communities (Froude, 1998). Like other heavily fished areas of the world, these benthic impacts are apparent in the Mediterranean (Dayton et al., 1995; Thrush et al., 1998). The impact of fishing on the Mediterranean seabed involves the use of bottom-trawling gears, namely otter trawls, beam trawls and dredges, together with some aggressive practices affecting rocky bottoms such as dynamite fishing and fishing for coral and date mussels (Tudela, 2004). Fortunately, the 2005 declaration that bottom trawling in the Mediterranean will be off-limits in depths greater than 1000 meters may significantly protect benthic marine biodiversity if it is enforced (MPA News, 2005). However, trawling also impacts shallow water seagrass beds by both suspending sediments and directly damaging the vegetal mass, and these fisheries are a major threat to *Posidonia* beds (Tudela, 2004).

Thus marine and coastal biodiversity in the Mediterranean is both valuable and at risk. Demographic pressures are severe and mounting (see Figure 1). According to CIESM, Western Mediterranean waters are experiencing a substantial warming trend (+0.2°C in last 10 years), which may have a drastic impact on deep-sea species accustomed to live in the dark at a near-constant temperature of 13°C. Sea level is rising significantly in the Eastern Mediterranean, with an average 12-cm rise registered on the Levantine coast since 1992. The change in marine biodiversity is proceeding at an unprecedented pace, as hundreds of exotic species -- mostly of tropical Indo-Pacific origin -- have settled in recent decades in the Mediterranean Sea. The trend in invasive species appears to be accelerating with the rapid growth of maritime traffic which brings with it alien fauna (introduced via ballast waters or attached to the hull). Intensive agriculture is carried out in the limited coastal plains in many portions of the region, often as a result of wetlands reclamation (EEA, 2000). The rate of coastal erosion resulting from unregulated sand mining for construction, sprawling tourism infrastructure (marinas, hotels) and urbanization, as well as river damming is growing. Yet despite the recognition of the inherent value of Mediterranean coastal and marine biodiversity and acknowledgement of the magnitude of these threats, no effective standardized protocol yet exists for evaluating marine biodiversity in the region. This report reviews the applicability of indicators for monitoring biodiversity that were developed for global use by the CBD, and introduces some new concepts for discussion.

**Figure 1.** Population in coastal areas of the Mediterranean Basin (from EEA, 1999)



## **II. Applicability of Indicators for Determining Trends in Mediterranean Marine Biodiversity**

### **A. Trends in biomes, ecosystems, and habitats**

Given the complexity of what constitutes biodiversity, and the limitations of what can be measured in marine environments, this indicator is the single most comprehensive one of all eight selected by the CBD, at least in theory. The measure describes changes in biodiversity, including interactions between species and habitat heterogeneity (Simboura and Zenetos, 2002). The problem arises when putting theory into practice, as few standardized methodologies have been adopted to determine trends at this highest level of biodiversity.

On land this indicator is typically derived from land cover estimates obtained from remote sensing imagery (satellite images or aerial photographs). While satellite remote sensing imagery is being adapted for more effective use in marine environments as well, the technology is still in its infancy and at this time best used for determining habitat extent in shallow water biomes with little turbidity, such as fringing, patch or barrier coral reefs.

According to Connor et al. (2002) a marine seascape or habitat can be defined as declined if its extent has decreased to 90% or less of its former natural extent or if its distribution within the territory has become significantly reduced. A seascape or habitat can also be defined as declined if its quality, based on change from natural conditions caused by human activities, is negatively affected by a change of its typical or natural components over almost the entire territory or the loss of its typical or natural components in several sub-regions (Derous, 2005). Such judgement is likely to include aspects of biodiversity, species composition, age composition, productivity, biomass per area, reproductive ability, non-native species and the abiotic character of the habitat. A population of a species occurring in the territory is defined as significantly declined if densities show an extremely high and rapid decline in the area over an appropriate time frame, if the species has already disappeared from the major part of its former range in the area or if densities are at a significantly low level due to a long, continuous and distinct general decline in the past. A species has suffered a significant decline in quality if one or more of the following characteristics can be detected: loss of genetic diversity, loss of fecundity or reduction in the number of mature individuals, fragmentation of the population (Connor et al., 2002).

Although the Mediterranean is one of the best-studied bodies of water in the world, even in this region an academic bias towards certain habitats has meant that inconsistencies exist in the level of knowledge about habitat distribution and condition. Boero (2004) claims that soft-bottom benthic communities are relatively well-studied as compared to rocky bottoms and other biotopes. To understand how this skewed perspective can bias sampling and assessment, one needs only to look at the wide variety of other, often overlooked habitat types in the reference list of marine habitat types in the Mediterranean (Table 2).

**Table 2:** Reference List of Marine Habitat Types for the Selection of Sites to be included in the National Inventories of Natural Sites of Conservation Interest  
(as adopted by the Contracting Parties to the Barcelona Convention; from Rais, 2004)

**I. SUPRALITTORAL**

**I. 2. SANDS**

**I. 2. 1. Biocenosis of supralittoral sands**

- I. 2. 1. 5. Facies of phanerogams which have been washed ashore (upper part)

**II. MEDIOLITTORAL**

**II. 1. MUDS, SANDY MUDS AND SANDS**

**II. 1. 1. Biocenosis of muddy sands and muds**

- II. 1. 1. 1. Association with halophytes
- II. 1. 1. 2. Facies of saltworks

**II. 3. STONES AND PEBBLES**

**II. 3. 1. Biocenosis of mediolittoral coarse detritic bottoms**

- II. 3. 1. 1. Facies of banks of dead leaves of *P. oceanica* and other phanerogams

**II. 4. HARD BEDS AND ROCKS**

**II. 4. 1. Biocenosis of the upper mediolittoral rock**

- II. 4. 1. 3. Association with *Nemalion helminthoides* and *Rissoella verruculosa*
- II. 4. 1. 4. Association with *Lithophyllum papillosum* and *Polysiphonia* spp.

**II. 4. 2. Biocenosis of the lower mediolittoral rock**

- II. 4. 2. 1. Association with *Lithophyllum lichenoides* (= entablature with *L. tortuosum*)
- II. 4. 2. 5. Facies with *Pollicipes cornucopiae*
- II. 4. 2. 7. Association with *Fucus virsoides*
- II. 4. 2. 8. *Neogoniolithon brassica-florida* concretion

**II. 4.2.10. Pools and lagoons sometimes associated with vermetids (infralittoral enclave)**

- II. 4. 3. Mediolittoral caves
- II. 4. 3. 1. Association with *Phymatolithon lenormandii* and *Hildenbrandia rubra*

**III. INFRALITTORAL**

**III.1 SANDY MUDS, SANDS, GRAVELS AND ROCKS IN EURYHALINE AND EURYTHERMAL ENVIRONMENT**

**III. 1. 1. Euryhaline and eurythermal biocenosis**

- III. 1. 1. 1. Association with *Ruppia cirrhosa* and/or *Ruppia maritima*
- III. 1. 1. 3. Association with *Potamogeton pectinatus*
- III. 1. 1. 4. Association with *Zostera noltii* in euryhaline and eurythermal environment
- III. 1. 1. 5. Association with *Zostera marina* in euryhaline and eurythermal environment
- III. 1. 1. 8. Association with *Halophytis incurva*

**III. 2. FINE SANDS WITH MORE OR LESS MUD**

**III. 2. 2. Biocenosis of well sorted fine sands**

- III. 2. 2. 2. Association with *Halophila stipulacea*

**III. 2. 3. Biocenosis of superficial muddy sands in sheltered waters**

- III. 2. 3. 3. Facies with *Loripes lacteus*, *Tapes* spp.
- III. 2. 3. 5. Association with *Zostera noltii* on superficial muddy sands in sheltered waters
- III. 2. 3. 7. Facies of hydrothermal oozes with *Cyclope neritea* and nematodes

**III. 3. COARSE SANDS WITH MORE OR LESS MUD**

**III. 3. 1. Biocenosis of coarse sands and fine gravels mixed by the waves**

- III. 3. 1. 1. Association with rhodolithes

**III. 3. 2. Biocenosis of coarse sands and fine gravels under the influence of bottom currents (also found in the Circalittoral)**

- III. 3. 2. 1. Maërl facies (= Association with *Lithothamnion corallioides* and *Phymatolithon calcareum*)  
(can also be found as facies of the biocenosis of coastal detritic).

- III. 3. 2. 2. Association with rhodolithes

**III. 5. POSIDONIA OCEANICA MEADOWS**

- III. 5. 1. Posidonia oceanica meadows (= Association with *Posidonia oceanica*)

- III. 5. 1. 1. Ecomorphosis of striped meadows

- III. 5. 1. 2. Ecomorphosis of "barrier-reef" meadows

**Table 1 (cont.):** Reference List of Marine Habitat Types for the Selection of Sites to be included in the National Inventories of Natural Sites of Conservation Interest

<b>III. 6. HARD BEDS AND ROCKS</b>	
III. 6. 1. Biocenosis of infralittoral algae	
III. 6.1.2. Association with <i>Cystoseira amentacea</i>	
III. 6.1.3. Facies with Vermetids	
III. 6.1.10. Association with <i>Cystoseira tamariscifolia</i> and <i>Saccorhiza polyschides</i>	
III. 6. 1.14. Facies with <i>Cladocora caespitosa</i>	
III. 6. 1. 15. Association with <i>Cystoseira brachycarpa</i>	
III. 6. 1. 16. Association with <i>Cystoseira crinita</i>	
III. 6. 1. 17. Association with <i>Cystoseira crinitophylla</i>	
III. 6. 1. 18. Association with <i>Cystoseira sauvageauana</i>	
III. 6. 1. 19. Association with <i>Cystoseira spinosa</i>	
III. 6. 1. 20. Association with <i>Sargassum vulgare</i>	
III. 6. 1. 25. Association with <i>Cystoseira compressa</i>	
III. 6. 1. 35. Facies and Associations of Coralligenous biocenosis (in enclave)	
<b>IV. CIRCALITTORAL</b>	
<b>IV. 2. SANDS</b>	
IV. 2. 2. Biocenosis of the coastal detritic bottom	
IV. 2. 2. 7. Association with <i>Laminaria rodriguezii</i> on detritic	
IV. 2. 2. 10. Facies with large Bryozoa	
<b>IV. 3. HARD BEDS AND ROCKS</b>	
IV. 3. 1. Coralligenous biocenosis	
IV. 3. 1. 1. Association with <i>Cystoseira zosterooides</i>	
IV. 3. 1. 2. Association with <i>Cystoseira usneoides</i>	
IV. 3. 1. 3. Association with <i>Cystoseira dubia</i>	
IV. 3. 1. 4. Association with <i>Cystoseira corniculata</i>	
IV. 3. 1. 5. Association with <i>Sargassum</i> spp. (indigenous)	
IV. 3. 1. 8. Association with <i>Laminaria ochroleuca</i>	
IV. 3. 1. 9. Association with <i>Rodriguezella strafforelli</i>	
IV. 3. 1. 10. Facies with <i>Eunicella cavolinii</i>	
IV. 3. 1. 11. Facies with <i>Eunicella singularis</i>	
IV. 3. 1. 12. Facies with <i>Lophogorgia sarmentosa</i>	
IV. 3. 1. 13. Facies with <i>Paramuricea clavata</i>	
IV. 3. 1. 15. Coralligenous platforms	
IV.3. 2. Semi-dark caves (also in enclave in upper stages)	
IV. 3. 2. 2. Facies with <i>Corallium rubrum</i>	
<b>V. BATHYAL</b>	
<b>V. 1. MUDS</b>	
V. 1. 1. Biocenosis of bathyal muds	
V. 1. 1. 3. Facies of soft muds with <i>Funiculina quadrangularis</i> and <i>Apporhais seressianus</i>	
V. 1. 1. 4. Facies of compact muds with <i>Isidella elongata</i>	
<b>V. 3. HARD BEDS AND ROCKS</b>	
V. 3. 1. Biocenosis of deep sea corals	
V. 3. 2. Caves and ducts in total darkness	

Given data availability, logistical considerations, and the relative value of certain habitats for supporting wider biodiversity, the best possible single indicator of this type in the Mediterranean marine context is likely the coverage of *Posidonia* beds. In fact, many of the national reports prepared for SAPBIO do describe seagrass beds and other key habitats, although standardized and comprehensive mapping and quantitative assessment at the national level is non-existent (see UNEP, 2003). Spain has developed a habitat typology that might be used as indicators, and the Mediterranean-wide Natura 2000 sites could also be a starting point (SAP BIO Annexes, UNEP 2005). However, even rigorous mapping of seagrass meadows will not necessarily give an indication of either the health of the ecosystem, nor the numbers of genetically distinct populations, species, or species assemblages it supports (these being some of the true measures of biodiversity). Thus an extensive seagrass bed, and possibly one that is even growing, may experience a loss in biodiversity that would not be picked up through use of this indicator in isolation.

Given new advances in remote sensing and even aerial photography, as well as advances in photo interpretation and GIS, it would also be useful to supplement more detailed biocenotic or biotopic mapping of seagrass and other key habitats with more comprehensive mapping of shallow water environment. Aerial photographs combined with ground truthing is particularly useful for saltmarsh and sea grass (Froude, 1998).

The aerial photographs should be taken at low spring tides when there is no cloud cover and missions should be reflown every five to ten years to identify changes in habitat extent. For deeper habitats, side scan sonar has proven a useful tool to determine habitat coverage, although species identification and ecological condition cannot be determined. However, if samples are obtained via dredges and grabs, side scan sonar information can be adequately ground-truthed, at least in soft sediments. Similarly, remote operation vehicles with video can be used to confirm the nature of habitats in deeper waters, though cost is a major limiting factor with this technique (Froude, 1998).

When data are not available for measuring extent of a particular kind of habitat (i.e. one measure of the state of biomes, ecosystems, or habitats), it is possible to determine trends from monitoring of pressure indicators. For instance, amount of wetland loss could be estimated using agriculture- and aquaculture-related land conversion rates in the coastal zone. Similarly, benthic habitat loss can be looked at through the prism of bottom trawling rates. In order for such proxies to be effective, however, they should be supplemented with more localized data on habitat coverage and condition. Throughout, extrapolations must rest on scientifically-tenable assumptions about pressures and response, at the level of biodiversity itself and not merely in terms of environmental change.

## **B. Trends in abundance and distribution of selected species**

It is possible to look at species indicators in different ways. One could express biological diversity as the number of taxa within an area, as the number of different life-history stages, the different species interactions within an area, or simply the number of individuals of a selected species or group of species and the geographic distribution of those individuals. The different ecological roles that species may play at different stages in their lifecycles can be considered or the level of endemism and/of biodisparity (range of morphologies and reproductive styles in a community, determined by heterogeneity and unpredictability of habitats) (RAMSAR, 1999, in Derous, 2004). In fact, it is difficult to know where exactly the trends in habitat coverage end and the trends in species abundance and distribution begin, since these factors are so highly interconnected and many of the most advanced biodiversity assessments use a combination of indicators that measure some aspects of each. In some cases, certain species are used as indicators for condition of habitat (e.g. Bustos-Baez and Frid, 2003), while in other cases habitat coverage is used as a predictor for presence of certain species (e.g. Cendrero et al., 2005).

Costello et al. (2001) review the use of marine biodiversity measures, within species, between species, and of ecosystems, and describe in detail full measures of interspecific diversity (i.e. without the use of indicators), using single-species indices, graphical methods, multivariate methods, and taxonomic relatedness. They conclude that two measures prove to have special utility for marine environmental assessments: 1) taxonomic relatedness or distinctness, and 2) measures of beta diversity or spatial turnover, as the difference in species composition between samples or habitats. Even though these measures depend on data coming from individual species presence and abundance, it is clear that the indices tell as much about trends in coverage of habitats or biotopes as about trends in species abundance.

One of most common used indicator of trends in species abundance is species richness within a prescribed area. Usually these areas are chosen for their relatively high species richness, and/or endemism, as compared to other areas (a phenomenon described as a “hotspot”). However, simply monitoring the number of species in an area can be misleading, because it is generally the assemblage of organisms in an area that is important, rather than simply the number of species present (Deros, 2004). Monitoring sites selected on the basis of one level of diversity (such as species richness) will not adequately represent diversity at other levels. Thus though species richness is valued as the common currency of the diversity of life, the “face” of biodiversity –the problem with this emphasis on using species for quantifying biodiversity is that such a focus on species may provide patterns that mask many important patterns and trends for other properties of biodiversity beyond taxonomy (Norse, 1994). Given the complexity of biodiversity, species- or other taxon-based measures of biodiversity rarely reflect the real attributes of biodiversity that are the focus of conservation monitoring.

Where species lists are limited it is likely that surrogates of species richness will have to be found. The choice of site for looking at richness or other attributes of assemblages of selected species is key: the ability of higher taxonomic levels to reflect the distribution patterns of species is likely to depend on the scale of consideration (i.e. the size of the area being considered). At large scale, shifts in the species composition of assemblages are likely to be more substantial and there are also likely to be shifts in higher level taxa (Deros, 2004).

Thus a major problem exists with assessments of species richness, and that is what to monitor, where. Taxonomic bias toward vertebrates or, as mentioned above, species assemblages in certain biomes such as coral reefs, is common (Gray, 1997; Gray et al., 1997). The result can be indication of hot-spots with significant species richness only in certain taxa. Ferrier et al. (in press) describe an alternative method for terrestrial hot-spots determinations, in which they look at species richness and compositional turnover in invertebrates as well as vertebrates. This certainly could be applied to marine environment, although data limitations are such that broad-based species assessments are unlikely, even in well-studied areas like the Mediterranean.

The approach of Ray (1999) to use species richness of birds as a surrogate for biodiversity in general is based on the fact that birds have dispersed to and diversified in all regions of the world, but subsequent analyses have shown that hotspots of birds coincide poorly with those of other taxonomic groups (Deros, 2004). Ward et al. (1999) and Ward (2000) also investigated the use of surrogates for overall biodiversity and found that habitat types suited this function best. But no surrogate was able to cover all species, from which it can be concluded that the hotspot paradigm is difficult to apply. In addition, indication of increasing numbers of species in a single or many taxonomic groups (as for example data presented in Table 2, UNEP, 2003) is likely to mean better, more in-depth analyses, not an increase in biodiversity. While some Mediterranean countries do have species lists (see Ozturk, 2002 for marine species found in Turkey, for example), abundance and distribution data are generally lacking.

Whitfield and Elliott (2004) presented the benefit of using fishes as a taxonomic group to develop indices and stressed that there are differences between indicator systems that rely solely on the diversity of species, or the life history traits of the species, and those that rely on a combination of fish species/abundance/biomass/ and some environmental variables. The authors suggest that fish studies at a variety of levels from communities to individual physiology are sensitive indicators, are scientifically and biologically relevant, and more meaningful than other indicators in a policy, management and public perception manner, although they concede that fish studies should be one component in a holistic approach (Whitfield and Elliott, 2004).

The use of higher taxonomic levels as surrogates for species level data is therefore more likely to be successful when undertaken at a broad scale. However, selection at a broad scale might fail to identify areas that are locally significant in terms of species composition. When the sizes of the areas chosen are substantially smaller than biogeographic regions, this will result in a high degree of species replication and will fail to protect those species most at risk (rare species), which mostly have unique habitat preferences (Deraus, 2004).

A combined index of species richness, rarity and vulnerability was proposed by Rey Benayas & de la Montaña (2003) as a measure of the diversity of an area ( $\sum_{i=1}^S (1/n_{ri})V_{ji}$  with species richness implicit in  $\sum_{i=1}^S$ ). Applying this index had a much better performance to reveal the areas of high diversity than when the constituting criteria were used. It should be noted that this index was developed for terrestrial ecosystems and that it has not been tested on its applicability in the marine environment (Deraus, 2004).

Others have developed indices specifically for the marine context. The concept of 'benthic complexity' was introduced by Ardrón et al. (2002) as a surrogate for benthic species diversity. The authors assume that the bathymetric (topological) complexity of an area is a measure of benthic habitat complexity, which in turn would represent benthic species diversity. But the data to perform the spatial variance analyses needed to quantify 'benthic complexity' are usually lacking.

A different way to look at the trends in species abundance and distribution is by selecting species that are umbrella species whose status can be used as a proxy for other species in the ecological community. This concentration on a single species, or possibly a subset of only a few marine species, is a proxy to flag trends in biodiversity overall. Such is the focus of projects such as the Marine Species of Common Conservation Concern, undertaken by the North American Commission on Environmental Cooperation (CEC). In this initiative, the three countries of North America (Canada, United States and Mexico) identified 15 marine species of common conservation concern (MSCCC). MSCCC designations included cetaceans, marine turtles, and seabirds, and most species are considered umbrella species. The designation is meant to bring increased attention to shared, endangered marine species, and to serve as a catalyst for better cooperation in monitoring populations and addressing threats. Action plans were developed for these species, and a central part of prioritizing action was identifying indicators and proxies for assessing threats (see Table 4 as an example of threats indicators developed for the leatherback sea turtle). It was decided that focusing on pressure indicators rather than solely state indicators might be more effective at creating an early warning system for population declines, since most of these MSCCC are very difficult to census directly (Agardy, 2003).

**Table 4:** Typology of major threats to leatherback turtles (*Dermochelys coriacea*) and potential indicators and proxies for monitoring changes in population size (Agardy, 2003)

#### **Mortality Effects**

- Directed take of adults  
Indicators: Number of adults found dead in boats or in port by fishing authorities, market surveys (port sampling); necropsy results pointing to deliberate killing.
- Directed take of eggs/young  
Indicators: Number of poached nests determined by nesting beach surveys, interviews with coastal communities; number of eggs appearing on the market from market surveys.
- Light pollution causing disorientation of young  
Indicators: Nesting beach surveys. Proxy indicators include determining amount of light pollution on nesting beaches
- Incidental capture  
Indicators: Number of entangled or incidentally caught turtles from observer data, fishermen interviews, and necropsy results from strandings.
- Ingestion of plastics/entanglement  
Indicators: Data derived from necropsies on stranded animals; *ex situ* behavioral studies showing tendency to ingest certain plastics; data on plastic pollution volume and fate to do predictive modeling on the likelihood of ingestion.
- Predation by invasive species or due to human-induced natural predator imbalances  
Indicators: Number of adults, young and eggs killed by predators determined from eyewitness accounts and tracking studies. Proxy measures include censusing predator populations at nesting beaches.
- Catastrophic events  
Indicators: Anecdotal evidence from oil spill and beach monitoring programs
- Ship strikes  
Indicators: Data on ship strikes from observer programs, voluntary reporting.

#### **Morbidity or Reproductive Failure**

- Population fragmentation / inbreeding effects
- Destruction of breeding habitat
- Barriers to migration, loss of migratory stopover habitats and staging areas
- Toxics poisoning and bioaccumulation
- Reduced prey or forage availability  
(directed take of prey, increased competition, eutrophication impacts, barriers to foraging areas)
- Sex ratio changes  
(differential mortality and morbidity, hatchery practices, global warming impacts)
- Disturbance-related behavioral changes  
(ship noise, nesting beach visitation and ecotourism)
- Introduction of new and/or increased spread of pathogens (including HABs)
- Changes to salinity, UVA/UVB, ocean temperature

#### **Constraints to Recovery**

- Destruction of habitat that could be used for population expansion or reintroduction
- Lack of knowledge / ignorance
- Lack of funding for monitoring, management, or enforcements
- Logistical difficulties in monitoring and enforcement
- Socio-economic considerations and pre-existing agreements (tribal rights, etc.)
- Genetic bottlenecks
- Population below MVP and ex situ / captive breeding not possible

### **C. Coverage of protected areas**

The areal extent or spatial coverage of marine and coastal protected areas is an easily obtained metric. Marine protected areas (MPAs) are catalogued not only in national databases but also in international ones, such as the World Conservation Monitoring Center with its partner IUCN, the World Conservation Union, and UNEP itself. However, most databases on MPAs are not up-to-date, otherwise incomplete, or, in some cases, filled with error. Nonetheless, with better information such as that currently being compiled by University of British Colombia (J. Alder, pers. comm.), MPA coverage will likely serve as a useful indicator of limited aspects of biodiversity status.

The biomic bias that was described in the section on trends in habitat coverage is also a factor in the way MPAs tend to be directed at conserving only certain kinds of biodiversity. In the Mediterranean, MPAs tend to be established in shallow water, soft-bottom habitats, or rocky shorelines. This means that even in those areas or countries in which there are many MPAs, the full range of biodiversity might not enjoy the protections afforded by these spatial management measures.

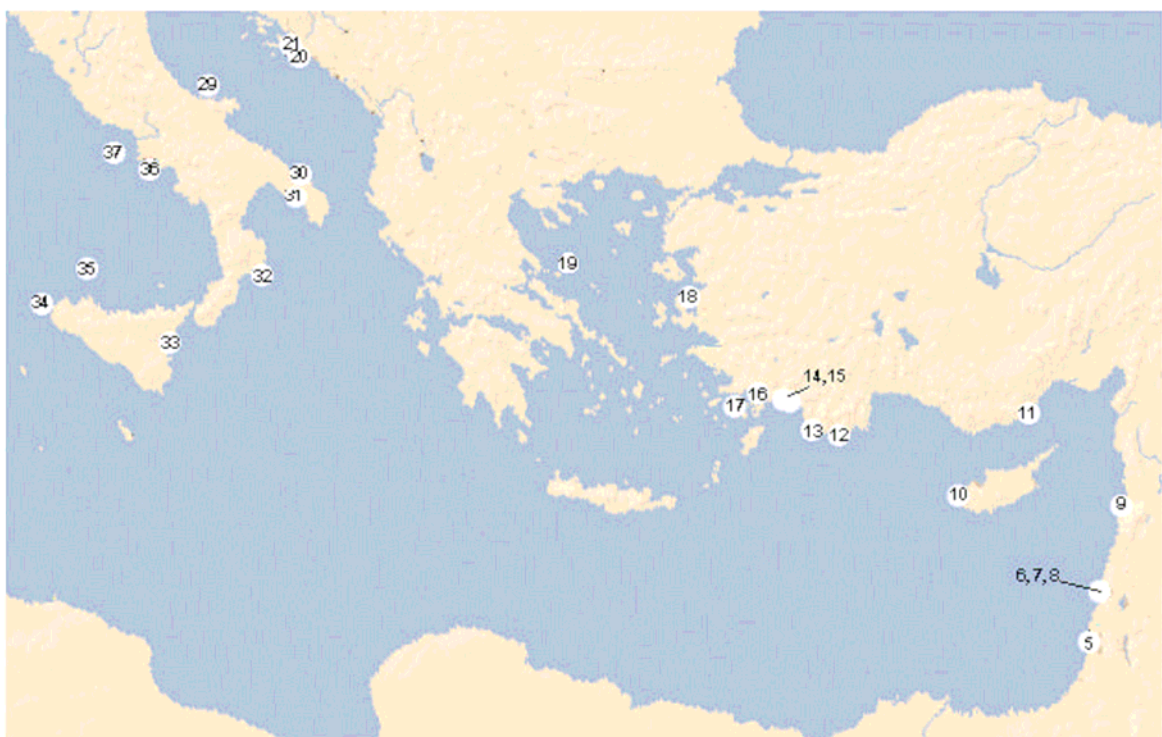
The 1982 Geneva Protocol on Mediterranean specially protected areas established a directory of Mediterranean marine and coastal protected areas. Protected areas are listed therein on the basis of notification made to RAC/SPA by the national focal points for specially protected areas (Rais, 2004). In all, 152 protected areas have now been declared in the Mediterranean coastal area in the different countries of the region, according to Rais (2004). Creating protected areas to protect particularly important natural sites is a practice, which goes back several dozen years in the Mediterranean. For the marine environment, according to data notified to RAC/SPA, the first Mediterranean protected area was set up in 1961: the Mljet National Park in the former Yugoslavia (now in Croatia), followed by the Port-Cros National Park, France (in 1963). Most of the protected areas in the coastal region are not marine but rather terrestrial. Of the 57 Mediterranean marine protected areas, 23 are exclusively marine, and the other 34 cover both terrestrial and marine environments (Table 4).

**Table 4:** List of Mediterranean MPAs (from Rais, 2004)

1 AL HOCEIMA MOROCCO	28 MIRAMARE ITALY
2 EL KALA ALGERIA	29 ISOLE TREMITI ITALY
3 GALITON TUNISIA	30 TORRE GUACETO ITALY
4 ZEMBRA AND ZEMBRETTE TUNISIA	31 PORTO CESAREO ITALY
5 SHARON CLIFFS ISRAEL	32 CAPO RIZZUTO ITALY
6 MA'AGAN MICHAEL ISLANDS ISRAEL	33 ISOLE CICLOPI ITALY
7 DOR-HABONIM ISRAEL	34 ISOLE EGADI ITALY
8 ROSH HANIKRA ISRAEL	35 USTICA ITALY
9 RABBIT ISLANDS LEBANON	36 PUNTA CAMPANELLA ITALY
10 LARA TOXEFTRA CYPRUS	37 VENTOTENE E S. STEFANO
ITALY	
11 GOKSU DELTASI TURKEY	38 CAPO CARBONARA ITALY
12 KEKOVA TURKEY	39 PENINSOLA DEL SINIS ITALY
13 PATARA TURKEY	40 ISOLA TAVOLARA ITALY
14 FETHIYE GOCEK TURKEY	41 ASINARA ITALY
15 KOYCEGIZ-DALYAN TURKEY	42 CINQUE TERRE ITALY
16 DATKA BOTZBURUN TURKEY	43 PORTOFINO ITALY
17 GOKOVA TURKEY	44 LARVOTTO MONACO
18 FOCA TURKEY	45 RED CORAL RESERVE MONACO
19 ALONNISSOS N. SPORADES GREECE	46 LAVEZZI ISLANDS FRANCE
20 MLJET CROATIA	47 SCANDOLA FRANCE
21 MALOSTONSKI ZALJEV CROATIA	48 FINOCCHIAROLA FRANCE
22 KORNATI ISLANDS CROATIA	49 PORT CROS FRANCE
23 BRIJUNI CROATIA	50 CERBERE-BANYULS FRANCE
24 LIMSKI ZALJEV CROATIA	51 MEDAS SPAIN
25 CAPE MADONA SLOVENIA	52 CABRERA SPAIN
26 DEBLI RTIC SLOVENIA	53 S'ARENAL REGANA SPAIN
27 STRUNJAN SLOVENIA	54 COLUMBRETES SPAIN
	55 CABO SAN ANTONIO SPAIN
	56 TABARCA SPAIN
	57 CAPO DE GATA SPAIN

The total surface area of the Mediterranean marine protected areas is 596,790 hectares, i.e. about 10,500 hectares on average (Rais, 2004). This average is, however, to be treated with caution because of the great range in size observed, from 1 hectare to over 200,000. It should be noted that 89% of Mediterranean marine protected areas are smaller than 20,000 hectares, and that only one is larger than 200,000 hectares (the Alonissos Marine National Park in Greece) (Rais, 2004).

The distribution of the main Mediterranean marine protected areas shows a sharp geographical imbalance (Figures 2 and 3). The southern shore has only 9 marine protected areas and these have a total surface area of less than 30,000 hectares, i.e. about 5% of the total marine area that is protected in the Mediterranean (Rais, 2004).



**Figures 2 and 3:** Geographic distribution of MPAs in the Mediterranean

In the Mediterranean, the names used to designate marine protected areas differ from country to country. Most of the countries have national laws that define the national categories of protected area and the terms of their creation. Generally speaking, except for a few cases, Mediterranean marine protected areas suffer from a lack of effective protection and suitable management. Also, in many cases the data on marine biotopes is either unavailable or is very old and needs updating. Statistics on the protected areas are far from

precise. Systems of classification differ from one country to another, making it difficult to inventory the protected areas as part of a common list (Rais, 2004).

An important new development in protecting Mediterranean biodiversity, but one that would not ordinarily be picked up by this indicator, is the protection of the Mediterranean benthos from trawling at depths of 1000 meters (MPA News, 2005). It is likely that other unconventional spatial management measures would be overlooked using MPAs as an indicator, even when their contribution to biodiversity protection is great.

In one sense this indicator can be taken as a kind of reverse pressure indicator, as long as the underlying assumption that designation of a marine protected area will stem or significantly reduce biodiversity loss. However, few databases anywhere in the world include size and extent of MPAs as well as their effectiveness in conserving biodiversity. In the Mediterranean region, many MPAs are established with no specific objectives, and very few Mediterranean MPAs have action plans with measurable benchmarks against which to monitor progress in conservation.

#### **D. Nitrogen deposition**

Nitrogen is a limiting resource in marine ecosystems and is therefore a useful indicator for monitoring potential changes in biodiversity. It should be noted that this indicator, unlike some of the others, does not measure a particular aspect of biodiversity but rather assesses one of the key pressure factors driving marine biodiversity loss. Most nitrogen coming into the marine system comes from land, and once entering coastal ecosystems can either settle with sediments or is taken up by dinoflagellates and other primary producers.

Some 77% of the pollutant load reaching the coastal ecosystems currently originates on land, and 44% of this comes from improperly treated wastes and run-off (Cicin-Sain *et al.*, 2002). Agricultural inputs often result in excessive nutrient loading which in turn causes large coastal areas to become eutrophied, hypoxic, or even anoxic (Boesch *et al.*, 2001; D'Avanzo *et al.*, 1996). An extreme example is the massive (to 15,000 km<sup>2</sup>) dead zone in the Gulf of Mexico (Turner and Rabalais, 1994). Eutrophication is pervasive close to most of the world's large estuaries and all centers of human population, and the resulting ecosystem changes are difficult (though perhaps not impossible) to reverse once algae take over benthic habitats or cause shifts in trophic structure. Numerous river basin and coastal zone studies (e.g., the Baltic region; Mississippi River / Gulf of Mexico; North Sea; Northern Adriatic; the Black Sea) have shown that elevated levels of nutrients, coastal eutrophication, toxic phytoplankton blooms, and bottom water hypoxia are a consequence of human settlement and industrialization. It has been estimated that fluvial fluxes of inorganic N and P to the world oceans have increased several-fold over the last 150 to 200 years. In certain regions, such as in Western Europe, the increase in N and P are 10 to 20 fold over pre-industrial levels (MA, 2005; Vörösmarty *et al.*, 2001).

Like many other parts of the world, the Mediterranean has experienced ever-increasing rates of eutrophication, in which the addition of large quantities of fertilizers, sewage, and other non-natural nutrients has acted to change the processes occurring in coastal ecosystems (NRC, 2000). All the country reports for SAP BIO list eutrophication as a threat, and many detail "hotspot" areas of high pollution. Fertilizer consumption in the Mediterranean countries increased several fold in the last 3 decades in all riparian countries, leading to significant increases in nutrient levels reaching coastal waters (EEA, 2000). Eutrophied conditions are evident in virtually all coastal waters near areas of human habitation, and is an acute problem in the shallower biogeographic subdivisions of the Mediterranean such as the

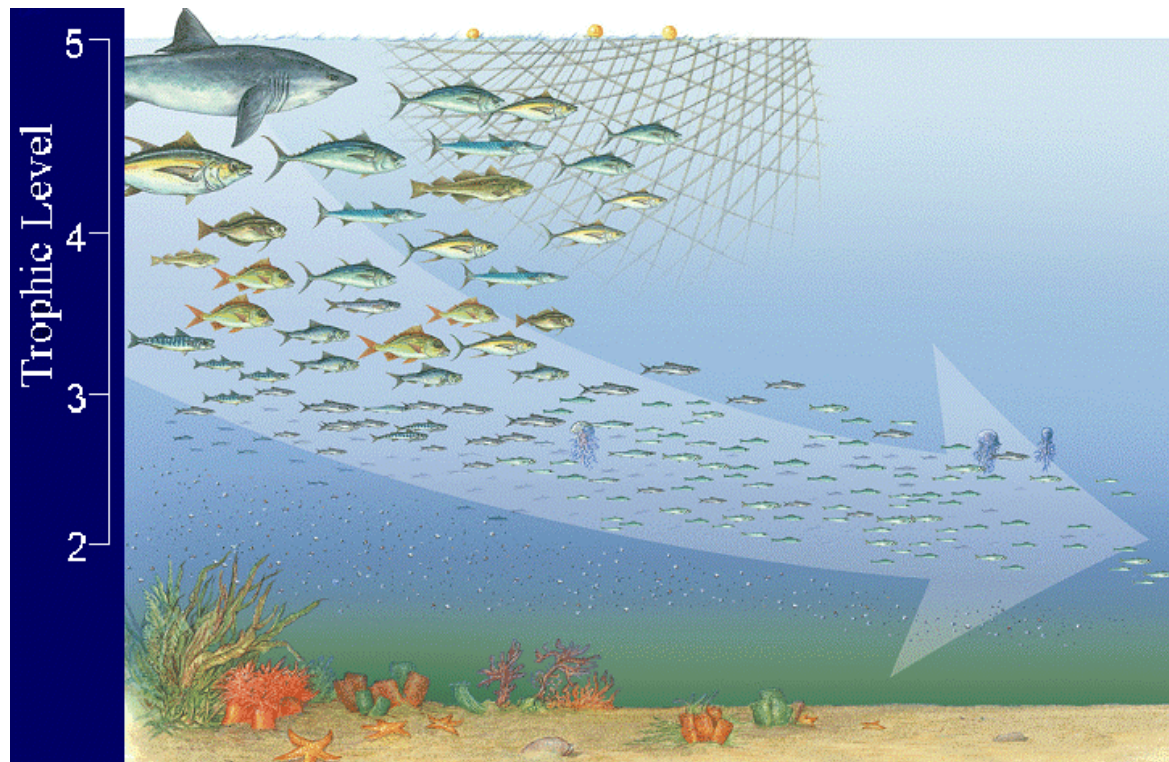
Adriatic Sea. In the Bosphorus region of Turkey, sewage pollution has been implicated in the decline of many fish species. Particularly vulnerable are parts of the sea that are naturally low in nutrients (Beman et al., 2005).

Measuring nitrogen is more complicated than one might think. Nitrogen exists in water both as inorganic and organic species, and in dissolved and particulate forms. Inorganic nitrogen is found both as oxidised compounds (e.g. nitrate ( $\text{NO}_3^-$ ) and nitrite ( $\text{NO}_2^-$ )) and reduced compounds (e.g. ammonia ( $\text{NH}_4^{++}\text{NH}_3$ ) and di-nitrogen gas ( $\text{N}_2$ )) (Logan and Longmore, 2005). The occurrence of different forms of ammonia depends on pH. At the pH of seawater, ~95% of ammonia is in the cationic form which is called ammonium ( $\text{NH}_4^+$ ). However, roughly half the ammonia will be in the (toxic) gaseous form ( $\text{NH}_3$ ), if the pH rises to ~9.5 in seawater. Total dissolved nitrogen (TDN) consists of dissolved inorganic nitrogen (DIN) and dissolved organic nitrogen (DON), and is readily available for plant uptake. DIN comprises  $\text{NO}_2^- + \text{NO}_3^- + \text{NH}_4^+$ . Dissolved organic nitrogen is found in a wide range of complex chemical forms such as amino acids, proteins, urea and humic acids. Ammonium is the form of nitrogen taken up most readily by phytoplankton because nitrate must first be reduced to ammonia before it is assimilated into amino acids in organisms. The particulate nitrogen pool consists of plants and animals, and their remains, as well as ammonia adsorbed onto mineral particles; it can be found in suspension or in the sediment (Logan and Longmore, 2005). Total nitrogen is a measure of all forms of dissolved and particulate nitrogen present in a water sample, and is usually what is measured for this particular indicator.

In some parts of the Mediterranean, nitrogen levels are measured directly (see Sistema Afrodite of Italy, Section V., for example). However, this may not be as easy as first appears (Giovanardi & Tromellini, 1992a; Zurlini, 1996). Giovanardi and Vollenweider (2004) state that though the principles governing coastal marine eutrophication are the same as for the lakes, it is difficult to evaluate the variations in concentrations of nutrients and calculate the residence time of water in the estuarine zone. At some sites nitrogen loading is determined by proxy such as amount of biotic encrustation on blades of seagrass. A key consideration for national or regional monitoring of biodiversity using nitrogen as an indicator is where to sample: in the most highly polluted areas (pollution hotspots), in biogeographically representative areas, or in according to an evenly-spaced grid across entire regions. In geographically larger scale studies, nitrogen deposition may be better extrapolated from chlorophyll concentration data determined by remote sensing.

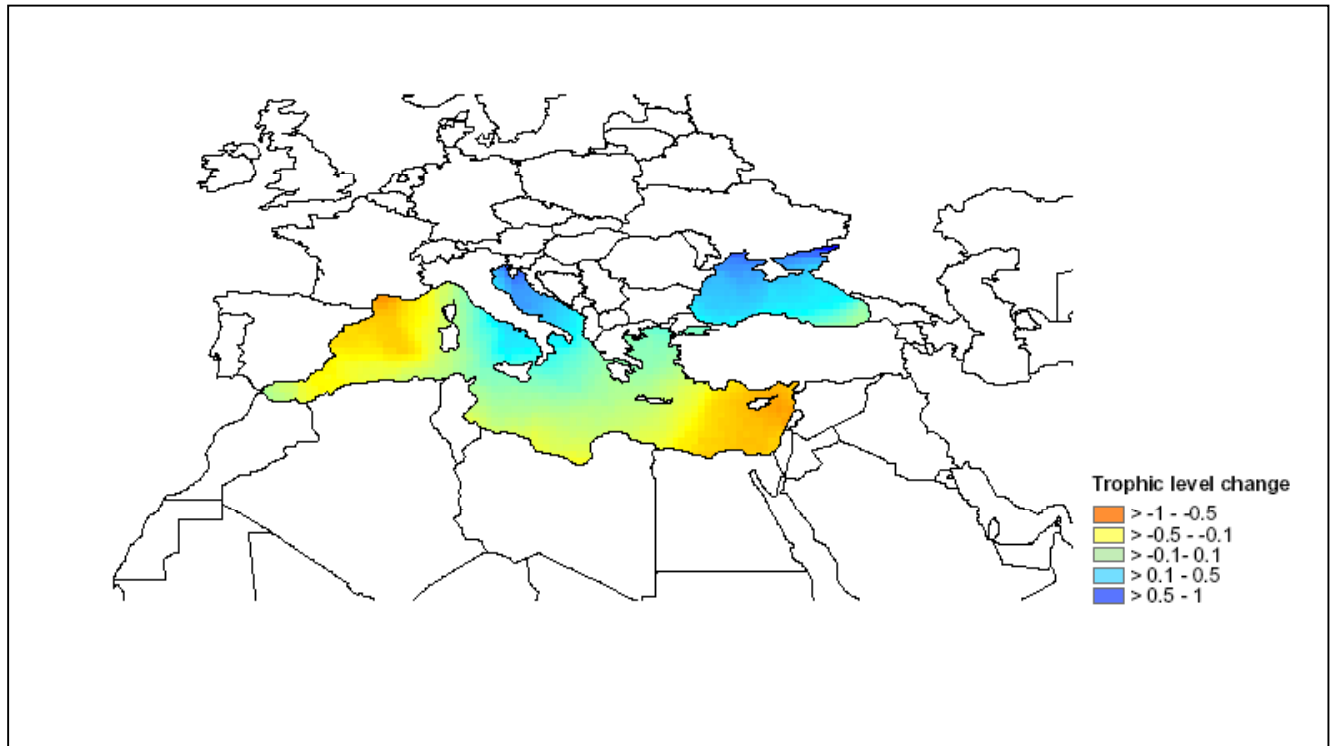
## **E. Marine trophic index**

The marine trophic index measures the change in mean trophic level of fisheries landings; calculations at the global and regional scales have showed that commercial fishing pressures on organisms high on the food web have created shifts in the web itself, with fewer large, high-value, predatory species and more low-value species from lower trophic levels (Chrstitensen et al., 2003; Pauly et al., 1998) (Figure 4). The intensification of fishing beyond sustainable levels has lead to ecological extinction of some species in some regions, and major imbalances in the entire marine food web (Myers and Worm, 2003; Watson et al., 2003).



**Figure 4:** A schematic of the fishing down the food web phenomenon, from Pauly at <http://www3.telus.net/rwatson/FDMFW2.html> (also cover image)

Marine trophic index is presented as a weighted average of trophic level change since fishing started; catch per trophic level is presented in tonnes (1000kg) by each exclusive economic zone. For national level marine trophic indices (see Case Studies), the distortion of range increases in fisheries are minimized by calculating marine trophic level change as a trophic slice – by using data from just those species with a trophic level of 3.35 and under. This excludes the large pelagic fisheries commonly caught offshore. A positive change is an increase in trophic level (up the food web toward the large fish-eating fishes) – a move usually associated with an expansion of the fishery and greater targeting of large pelagic species like tunas. A negative change, which more typical in many places, is usually associated with the failure of fisheries on species at higher trophic levels. This is fishing down the marine food web, with its attendant implications for ecosystems and biodiversity. A regional map of changes in the Mediterranean Marine Trophic Index is shown in Figure 5.



**Figure 5:** Mean Trophic Level change in the Mediterranean region.  
(Prepared by The Sea Around Us Project, UBC)

It is due to these imbalances and the resulting impacts on biodiversity that the marine trophic index is thought to be a useful indicator for marine biodiversity loss. However, it should be made clear that fisheries impacts do not always translate to measurable losses in biodiversity, and conversely, fishing can cause major changes to biodiversity, especially at the genetic level, with no perceptible change in the marine trophic index. Therefore this indicator should be used in conjunction with other indicators that more directly reflect biodiversity in all its manifestations.

Giovanardi and Vollenweider (2004) calculated a different sort of trophic level index, called TRIX, to gauge eutrophication. Their TRIX point values assign an immediate measurement to the trophic level of coastal waters. In Italian seas, values exceeding 6 TRIX units are typical of highly productive coastal waters, where the effects of eutrophication determine frequent episodes of anoxia in bottom waters. Values lower than 4 TRIX units are instead associated to scarcely productive coastal waters, while values lower than 3 are usually found in the open sea.

Other approaches use different measures of fish community dynamics (Nicholson and Fryer 2002; Nicholson and Jennings, 2004). Some scientists view these marine trophic indices as pressure indicators rather than state indicators (Rice, 2000). Regardless, when marine trophic index is used in combination with other biodiversity indices such as coastal and marine habitat coverage and trends in species richness and abundances of key species, changes in marine biodiversity can indeed be adequately monitored.

## **F. Water quality in aquatic ecosystems**

This indicator is related to nitrogen deposition, but goes well beyond it. In addition to nitrogen, other nutrients pollute coastal waters, including phosphorus, and toxins also contribute to reduced water quality. Loadings of biotically-active elements, metals, hormones, antibiotics, and pesticides are known to have increased several-fold since the beginning of the Industrial Era, and levels of these toxins in coastal areas are expected to continue to increase (MA, 2005).

The Mediterranean is typical of semi-enclosed seas: they tend to be highly productive (primarily due to exogenous inputs from lands nearby), often have high species diversity and endemism, are heavily used by the countries and communities that border them, and are often at high risk from pollution. Semi-enclosed seas are highly degraded, due to demands placed upon them and their physical configuration. Freshwater inflows to semi-enclosed seas have been severely curtailed in most areas, robbing them of recharging waters and nutrients. At the same time, water reaching these basins is often of poor water quality due to land-based sources of pollution such as agricultural and industrial waste (GESAMP, 2001). Such degradation is highly prevalent in semi-enclosed seas with major river drainages, such as the Mediterranean Sea (Cognetti *et al.*, 2000). The limited flushing and long recharge times in semi-enclosed seas means that pollutants are not as quickly diluted as in the open sea, and eutrophication and toxics loading is often the result.

There are many different approaches to assessing water quality, but it is always accomplished through a variety of indicators used simultaneously. UNESCO (2003) suggests that indicators be scientific (i.e. rigorous, replicable), functional (i.e. policy relevant, compelling) and pragmatic (i.e. feasible, timely and democratic). Since the range of parameters to be measured under water quality indicators is so large, it is also useful if the set of measurements pertains as well as possible to the particular region, and the changes occurring in that region.

Measurements of water quality can include directly assessing the quantities of particular toxins such as hydrocarbons, PCBs, copper and tin butyltin, mercury, pathogens (especially *E. coli*) and sediments. – or it can be assessed by proxy through measures such as turbidity, BOD (biological oxygen demand) and SOD (sediment oxygen demand), etc. Even less direct are measures of ecological change brought about by changes in water quality, such as meiofaunal encrustations on soft-bottom habitats (Mirto and Danovaro, 2004).

Most Mediterranean countries already monitor at least some of these parameters to meet the obligations of the Barcelona Convention, though a standardized research protocol is needed. An additional problem with the use of this indicator is the complex relationship between water quality and biodiversity: while a generally inverse relationship exists between water quality and the diversity of aquatic life, species turnover with more environmentally tolerant species replacing sensitive ones can lead to major ecological changes that might not necessarily be picked up by species diversity indices.

### **G. Status and trends of linguistic diversity and numbers of speakers of indigenous languages**

This indicator was developed for the portfolio of biodiversity indicators to underscore the fact that humans are a *bona fide* part of ecosystems, and that biodiversity extends to the human species as well. The use of this indicator for measuring marine biodiversity is limited at best, however, since human communities do not exist in the sea. One could imagine an assessment of linguistic diversity in coastal regions might be possible, however it presumes agreement on what constitutes the narrow strip of land that is defined as the coastal area.

A better measure of human diversity in coastal environments might be the existence of traditional marine-based livelihoods, such as the tonnara fishery in Sicily, for example. Additionally, one could develop a metric for diversity within the portfolio of livelihood options in coastal communities, with the assumption that at least some traditional ways of life would be preserved in places where the diversity of job opportunities is high.

### **H. Official development assistance provided**

Much like MPA coverage, the amount of development assistance provided to countries to conserve their biodiversity is a kind of negative pressure indicator, presuming that some correlation exists between funds allocated and actual biodiversity conservation. The problem with this measure, however, is that the relationship between financial resource allocation and threat abatement is not as clear as one would like. Much conservation funding is directed at interventions that are inefficient, or do not address the drivers behind biodiversity loss.

Data do exist on development assistance for marine biodiversity (eg Tables 2, 3, and 4 of SAPBIO in UNEP, 2003), although an accurate assessment would have to include funding from different multilateral institutions, government funding in developed countries, and environmental NGO and private sector investments.

### **I. Other indicators to be tested**

#### **1. Change in status of threatened species**

This indicator focuses on those species that are either at greatest risk of extinction (and as such, can be used as an early warning system for biodiversity loss), and uses change in

status as a trend indicator. Species status determinations such as those made by IUCN for the Red List (see Appendix 1) can be used to select indicator species. Using these status determinations to evaluate trends in abundance and determine geographic distributions can help determine regional trends in species extinctions, regional trends in the numbers of species categorized as threatened or endangered and protected under legislation, and regional trends in the number of at-risk species (Flather et al., 2003). However, there are limitations in using this indicator, including that benchmark conditions are rarely known, data on species (especially marine species) are often limited, species status determinations can vary year-to-year or by methodologies used to determine status, and geographic distribution information is usually inconsistent throughout a region.

In any case, the use of this criterion is limited since most rare, threatened and endangered marine species remain poorly surveyed, if at all and many of the charismatic rare megafauna have untenably large ranges that make assessment throughout their range difficult (Ardrón et al., 2002). Unlike in the terrestrial environment, very little marine species are included in Red Data Books or receive statutory protection. This is due to a lack of systematic assessment and study of marine species (Sanderson, 1996a; 1996b). Since information on the population sizes of the vast majority of marine benthic species is unavailable, it is impossible to make rarity assertions based on a species' comparative population size; the sites of occurrence of species are better known hence a rarity assessment based on units of the area of occupancy might be a more practical solution (Sanderson, 1996b).

## 2. Trends in genetic diversity of domesticated animals, cultivated plants, and fish species of major socio-economic importance

In the marine context, this indicator tries to capture changes in genetic diversity of wild populations brought about primarily by fishing, and changes in the genetic diversity of species raised in aquaculture/ mariculture operations. The latter has some connection to the former as well, since genetically alien cultured species have been known to escape farming operations and genetically contaminate wild stock.

For fisheries managers, tracking the changes in genetic composition of stocks is critically important, as it is for aquaculture operators (Costello et al., 2001). Genetic composition defines population assemblages, and is one of the key factors in determining stock delineations for management. There is longstanding evidence that heavy fishing pressure reduces the genetic diversity of stocks, possibly to the point where homogeneity puts organisms at a genetic disadvantage (Smith, 1994). In addition, intense fishing pressure can select for ever-smaller sizes of individuals (Dayton et al., 2002).

The major problem with the use of this indicator is cost and time for sampling and performing genetic analyses. Such investments can be significantly reduced if the choice of species for genetic analysis is a good indicator for wider genetic changes, and if sampling can be built into the fishery. In aquaculture operations, cost is less of a factor since the investment is regularly made by the private sector underwriting the farming operation.

## 3. Area of forest, agricultural, and aquaculture ecosystems under sustainable management

This is a potentially useful pressure indicator that could be especially helpful when taken in combination with coverage of marine protected areas and with more direct state indicators like trends in species and habitats. The immediate difficulty with this indicator is determining what constitutes "sustainable management". Presumably, some index of sustainability akin to the aquaculture certification programs being talked about (to follow the example set by the

Marine Stewardship Council's eco-labelling of fish products as sustainably harvested) might be used, but such certification is not yet in place. Then there is the added complication of how one would weight this indicator against trends in habitat, such that when wetlands or other coastal habitats are converted for use in agriculture, plantation forestry, or aquaculture, the effects on biodiversity could be assessed even in the case that the area then becomes under sustainable management.

#### 4. Numbers and costs of alien invasions

Alien species invasions and the ecological imbalances they wreak are a large problem for coastal and marine biodiversity. The ecological consequences of the invasions include: habitat loss and alteration; altered water flow and food webs; the creation of novel and unnatural habitats subsequently colonized by other exotic species; abnormally effective filtration of the water column; hybridization with native species; highly destructive predators; and introductions of pathogens and disease (MA, 2005).

Estuarine systems are among the most invaded ecosystems in the world, with exotic introduced species causing major ecological changes. Often introduced organisms change the structure of coastal habitat by physically displacing native vegetation (Grosholz, 2002; Harris and Tyrrell, 2001). For example, San Francisco Bay (US) has over 210 invasive species, with one new species established every 14 weeks between 1961 and 1995 (Cohen and Carlton, 1998). Most of these bioinvaders were borne by ballast water of large ships or occur as a result of fishing activities (Carlton, 2001). Altering soft bottom habitat to hard bottom in the process often affects estuaries indirectly by creating conditions for new assemblages of species, and facilitating range expansions of invasive species (MA, 2005). The resulting ecosystems may have losses in some ecosystem services and biodiversity. In New Zealand invasive species have displaced commercially important mussel beds, resulting in significant economic losses for many mussel farmers.

In the Mediterranean, the number of introduced species has risen dramatically in the last century (Zenetos et al., 2005). Some of these non-indigenous species have arrived via the Straits of Gibraltar, the Straits of Suez, and the Dardanelle. Much attention has been paid to *Caulerpa taxifolia* and related species, and other invasive introduced species. Such plants are a good biodiversity indicator because their spread is easy to monitor, the data can be mapped, and predictions on future extent can be made given understanding of currents, temperature regimes, etc. Other species that are indigenous but spread to new areas can be monitored in the same way. In the Mediterranean Basin, the northwards spread of thermophilic species represents a sort of alien species invasion, made possible by the warming of the sea's waters (Chevaldonne and Lejesne, 2003; Gomez and Claustre, 2003).

The introduction, establishment, and spread of alien species is relatively easily monitored, and provide a good pressure indicator for assessing trends in marine biodiversity. However, the costs of such invasions are difficult to measure, and it is not clear how cost factors into assessing biodiversity. The reason for this is that costs vary according to value of the resources in a particular place. Hypothetically, if a *Posidonia* bed is invaded by *Caulerpa* and suffers a loss of fisheries productivity, the cost of that impact is measures in fisheries losses. If the fishery is profitable, i.e. it supplies markets in Europe, than the cost is greater than if the fishery is supporting artisanal markets in North Africa, for example. Rather than measure economic costs of invasions, it would make sense to determine ecological costs, though this is a complex undertaking in and of itself.

## 5. Connectivity and fragmentation of ecosystems

In theory this is an important indicator to assess changes in biodiversity, but habitat or ecosystem fragmentation is more difficult to determine in marine systems than in terrestrial. One reason for this is the more three-dimensional nature of marine systems, such that connectivity between ecosystems can be sustained in the water column even where connectivity is impaired between different areas of the benthos. Perhaps a better indicator to get at this aspect of ecosystem functioning and biodiversity would be to use an indicator that measures physical barriers to the flow of nutrients or movements of organisms. Included in such an indicator could be dams, seawalls, groins, impenetrable causeways, etc.

## **III. Case Studies**

The case studies that follow are a preliminary attempt to gather information on the eight primary indicators selected by the CBD. Five countries were chosen for these case studies: Spain, Italy, Turkey, Tunisia, and Morocco, because these five countries cover the seven biogeographic subdivisions of the Mediterranean described in section IC. (Table 1). For most of the biodiversity indicators, determining their utility in identifying trends is impossible since time-series data are unavailable; in cases where there are indeed three data points (the minimum needed to assess a trend), the data were collected at different times, so comparative analysis is equally impossible. It is clear from the case studies that no country is currently equipped to use the 8 preferred indicators to monitor changes in biodiversity, and that a standardized protocol for doing so is badly needed.

### **A. Spain**

Spain has a long coastline and a wide diversity of habitat types, including Atlantic coastal ecosystems not included in the purview of this report. The coast has been a huge draw for tourism, and coastal development has been rampant. In addition, fisheries are a very important part of Spain's economy, and a central part of its culture. These two pressures, along with urban, industrial and agricultural pollution, have compromised the marine biodiversity of the country. Recognizing this, Spain adopted the objective of establishing a protected area every 30 kilometers along the coast to ensure the preservation of ecosystems and the maintenance of marine and terrestrial fauna and flora (Batisse and Jeudy de Grissac, 1995).

After a period of adaptation to the transfer of competence dealing with conservation of environment from the national to the regional authority, the regions showed a great interest in marine conservation and are very active in the field of MPAs. In the period between 1982 and 1994, Spain created some 25 protected areas on the Mediterranean coastline at the national or regional level. Six of these areas include a marine component. The following MPAs were recorded in 1995:

- Cabo de Gata Nature Park and Marine Reserve (26,000 hectares including 13,000 hectares terrestrial)
- Archipelago de Cabrera National Park (Balearic Islands) (10,000 hectares including 1,836 hectares terrestrial)
- Columbretes Nature Park and Marine Reserve (5,766 hectares including 43 hectares terrestrial)
- Medas Islands Marine Reserve (40 hectares including 20 hectares terrestrial)
- S'Arenal Regional Protected Landscape (400 hectares)
- Tabarca Marine Reserve (1,463 hectares).

In addition, the Island of Menorca has been declared a Biosphere Reserve, with protection of the sea adjacent to the protected core areas (Batisse and Jeudy de Grissac, 1995).

Several more MPAs have been designated since the 1995 Global Representative System of Marine Protected survey was undertaken, including a marine reserve on north Menorca and the Reserva Marna de los Freus de Ibiza y Formentera. The SAPBIO synthesis of national reports (UNEP, 2003) cites 63 marine protected areas all told, covering 136,335 hectares, including 7 SPAMI sites: Alboran Island, Seabed of the Levant of Almeria, Cape Gata-Nijar Natural Park, Mar Menor and the East coast of Murcia, Cape Creus Natural Park, Medes Island and Columbretes Islands.

Cabrera national Park and Maro-Cerro Gordo Cliffs were presented and adopted as SPAMIs during the SPA National Focal Points (Marseille, June 2003) and the Contracting Parties to the Barcelona Convention meetings (Catane, November 2003)

Spain has very active and well-developed fisheries. The Marine Trophic Index for Spain's Mediterranean waters was calculated at 2.99 E -02, suggesting a slightly expanding fishery and some movement up trophic levels to target higher value fishes (The Sea Around Us Project, 2005). Little is known about trends in species, habitat or water quality.

## **B .Italy**

With over 8,000 kilometers of coastline, Italy has the second longest coast on the Mediterranean. The marine biodiversity of Italy is especially significant, since it borders four Mediterranean biogeographic subdivisions (Algerian Basin, Tyrrhenian Sea, Ionic Basin, and Adriatic Sea). As it is the case for many other countries of the Mediterranean, the coast and seas are a critical component of both the economy and the culture of the country.

Italy has made significant investments in developing species lists of both marine plants and animals, however these lists may be incomplete and abundances are not nationally monitored (Boero, 2005). However, in 2001 Italy did establish its Sistema Afrodite, through which Italian researchers undertake standardized research in 16 MPAs simultaneously (Greco et al., 2004). In addition to the main goal of helping create a national MPA network, the objectives of Sistema Afrodite include the creation of a shared, standardized base of knowledge, the promotion of a higher level of co-operation among scientists in Italy and in the Mediterranean countries, and the provision of a potential model for the creation of a regional Mediterranean network of MPAs. Through Sistema Afrodite, Italy uses, *inter alia*, many of the indicators for assessing changes in biodiversity, including trends in habitat determined through biocenotic mapping and benthic sampling, trends in selected species via fish visual census, water quality through direct measures and biomarkers for certain pollutants, and spread of invasive species.

The results from the first years of monitoring under Sistema Afrodite are to be published soon.

Marine protected areas are established by presidential decree. In recent years Italy established 16 marine protected areas, and nearly twenty more MPAs are currently in the pipeline (Agardy, 2002; UNEP, 2003). Italian MPAs are multiple use protected areas, typically including one or more core zones (i.e., no-entry, no-take reserves), also known as "A" zones, buffer zones ("B" zones) where limited human activities are permitted, and "general reserve" zones ("C" zones) having a lesser degree of protection (Villa et al., 2001). At present, each MPA functions as a separate entity, and as such it faces institutional, administrative, socio-economic and management challenges that must be resolved in an ad hoc, piecemeal fashion (Agardy, 2002).

While protected areas used to be mainly concentrated along the Tyrrhenian Sea, recent and important MPAs have been declared in Sicily, in the Adriatic and Ionian Seas.

The following MPAs were recorded in 1995 (Batisse and Jeudy de Grissac, 1995):

- Archipelago Toscano National Park (57,500 hectares with islands)
- Castellabate Fishery Reserve (4,400 hectares)
- Ciclopi Marine Reserve (Sicily, 35 hectares)
- Miramare Marine Reserve and Biosphere Reserve (Trieste, 30 hectares including 3 hectares terrestrial)
- Portoferraio Fishery Reserve (160 hectares)
- Tremiti Marine Reserve (Adriatic, 1550 hectares)
- Ustica Marine Reserve (7500 hectares)
- Egadi Islands Marine Reserve (Sicily, 50,000 hectares)
- Torres Guaceto Marine Reserve (Adriatic 15,500 hectares)
- Capo Rizzuto Marine Reserve (Ionian Sea, 11,000 hectares)

Like other countries of the region, Italy has well-developed fisheries off all of its coasts, at artisanal and commercial levels. The Mean Trophic Index for Italy is 0.116, indicating a slight positive change with either an expanding set of fisheries, or new fisheries targeting high trophic level species (The Sea Around Us Project. 2005).

### **C. Turkey**

The Turkish Mediterranean coast is of high importance for the protection of the monk seal and of sea turtles as well as for biodiversity in general. In comparison with Spain and Italy, Turkey's coasts are relatively underdeveloped. Coastal tourism, however, has expanded in recent years. Fortunately, Turkey has developed a strong policy for conservation of nature with the implementation of a 1989 law allowing the creation of Specially Protected Areas (in reference to the Barcelona Convention Specially Protected Areas Protocol). From 1990 to 1995, 385,000 hectares and 774 kilometers of coastline were protected and management was begun (Batisse and Jeudy de Grissac, 1995). By 1994, 25 percent of the total coast length (1,332 kilometers) had been declared under protection.

The following MPAs were recorded in 1995:

- Datcha Botzburum Specially Protected Area (147,400 hectares including 116,900 hectares terrestrial)
- Fethiye Gocek Specially Protected Area (61,300 hectares including 30,000 hectares terrestrial)
- Foca Specially Protected Area (2,750 hectares including 1,550 hectares terrestrial)
- Gokova Specially Protected Area (52,100 hectares including 24,500 hectares terrestrial)
- Goksu Delta Specially Protected Area (23,600 hectares including 17,800 hectares terrestrial and wetland)
- Kekova Specially Protected Area (26,000 hectares including 14,500 hectares terrestrial)
- Koycegiz Dalyan Specially Protected Area (38,500 hectares including 28,300 hectares terrestrial)
- Patara Specially Protected Area (19,000 hectares including 14,800 hectares terrestrial)

In addition, Turkey has seventeen protected sea turtle nesting beaches and seven wetland areas with differing levels of protection (UNEP, 2003).

Turkey's fishing in the Mediterranean is well developed, and supports mostly national consumption. The Mean Trophic Level is 0.1519, suggesting a slight movement towards fisheries resources at higher trophic levels. Besides fishing, other pressures that are growing are agricultural and industrial pollution that continue to threaten important biodiversity, especially on the Cilician coast.

Little else is known about Turkey's overall biodiversity, with no species inventories and no regular monitoring of water quality, nitrogen deposition, or habitat coverage. Nonetheless, Turkey's investment in conservation has been substantial, with \$3.6 million estimated investment in its National Action Plan, which includes projects to conserve marine turtles, develop MPAs, reduce the negative impacts of destructive fishing, and conserve cetacean species (UNEP, 2005).

#### **D. Tunisia**

Tunisia has significant marine biodiversity, despite its relatively small size. The National Report cites 2135 marine and wetland species – more than a third of the country's total known biodiversity (Romdhane and Missaoui, 2002). Tunisia reinforced its institutions with the creation of a Ministry for the Environment in 1991 (Batisse and Jeady de Grissac, 1995). One of the Ministry's priorities is effective management of the protected areas along the coast. However, the Tunisia's wetlands are threatened by pollution, unplanned development, and agriculture. It is estimated that Tunisia has lost 28 percent of its wetlands in a little over a century, mainly as a result of drainage, while urbanization accounts for the loss of over 3,300 hectares of wetlands each year. In addition, 27 percent of Tunisia's lakes and marshes, and 21 percent of its rivers are polluted.

Tunisia has made strong commitments to conservation. Its seven National Action Plans for biodiversity conservation are an estimated investment of \$2.85 million (US) elaborated within the SAP BIO Project (UNEP, 2005).

The following MPAs were recorded in 1995:

- Galiton Marine Reserve with 450 hectares
- Zembra and Zembretta National Park and Biosphere Reserve (Large d'EL Haouaria) with 4,700 hectares, including 391 hectares terrestrial.

An additional four protected areas have been established in the intervening decade, some of which have a marine component (Romdhane and Missaoui, 2002):

- Ile de la Galite ( Nord de Tunisie)
- Parc national de l'Ichkeul ( Tunisie Septentrionale)
- Réserve naturelle de l'Ile de Chekly ( dans le lac de Tunis)
- Réserve Naturelle des Iles Kneiss ( Large de Sfax).

In 2004 the government of Tunisia officially announced its commitment to designate at least 15 locations as wetlands of international importance under the Ramsar Convention. The designations mean that these North African wetlands, covering a total area of over 750,000 hectares (2,895 square miles), will be protected from development. The wetlands to be

protected include salt lakes, swamps, peat bogs, dunes, karstic caves, oases, and lagoons, and harbor some 85 aquatic plant species. Tunisia's wetland environment attracts up to a half a million birds each year, and supports 350,000 individuals of 33 species of sandpipers alone. During the migratory season, Tunisian wetlands host 250,000 ducks, which make up 58 percent of the Maghreb's total population. In addition, 25,000 flamingos that form a third of the Mediterranean population migrate to Tunisia. Tunisian wetlands provide thousands of families with income from fishing and shellfish collection. Lake Ichkeul, the only Ramsar site now found in the country, provides 150 to 200 tons of fish a year, while the Ghar el Melah lagoon provides 80 tons. Thousands of tourists visit the Korba lagoon, Lake Ichkeul, and the salt lakes of Thyna and Monastir each year.

The Mean Trophic Index for Tunisia is 5.47 E-02, again suggesting an expanding fishery and movement towards higher trophic level species (The Sea Around Us Project. 2005). The Gulf of Gabes is an area of particular concern with regard to fisheries impacts and threat of industrial pollution. The Gulf is important for loggerhead sea turtles, sea birds, and supports the most extensive *Posidonia* meadow in the region. Cetaceans also reside in Tunisian waters (Ktair Chakroun, 1980).

## **E. Morocco**

Within the framework of the national policy for the environment, Morocco has begun to create a network of protected areas covering all the ecosystems and habitats in the country. On the Mediterranean shore, Al Hoceima National Park was established to cover marine as well as terrestrial areas with an important buffer zone of 42,900 hectares (Batisse and Jeudy de Grissac, 1995). In addition, Morocco has seven natural reserves (UNEP, 2003):

- Moutpeice of the Moulouya
- Jbel Moussa
- Sebkha Bou Areg
- Trois Fourches Cape
- Irque of El Jabha
- Coast of Rhomara
- Koudiet Taifour
- Lagoon of Smir.

The Alboran Sea hosts one of the richest species assemblages in the Mediterranean, the result of mixing of Atlantic flora and fauna with Mediterranean species. Large predatory fish like swordfish and tunas, whales, dolphins and sea turtles all use this corridor to pass between water bodies. Seabirds, including the yellow-legged gull (*Larus cachinnans*) and Audouin's gull (*Larus audouinii*) are present, as are colonies of deepwater red coral (*Corallium rubrum*). Humans use this area, too – the Marine Trophic Index is at -9.90E-02, suggesting that heavy fishing pressure has caused a local fishing down the food web phenomenon (The Sea Around Us Project, 2005). However, Morocco presents a problem in investigating the Marine Trophic Index and other indicators, as do other countries that have coasts outside the Mediterranean, in that data are not disaggregated for Mediterranean and other coasts (in this case the Atlantic, which takes up a much greater proportion of the total coastline).

Little else is known about biodiversity on Morocco's Mediterranean coast. There has been recent speculation that the rocky shores and caves of the coast may harbor Mediterranean monk seals, but surveys have not yielded sightings (E. Salvati, pers.comm). Morocco has 6

National Action Plans for biodiversity, for an estimated investment of approximately \$1.05 million US elaborated within the SAP BIO Project (UNEP, 2005).

#### IV. Additional Indicators

Froude (1998) investigated a range of additional marine biodiversity indicators for monitoring marine biodiversity and environmental condition in New Zealand, presented in Table 5. Some of these are state indicators while others are pressure indicators. Many are related to indicators discussed in this paper.

**Table 5:** Marine indicators considered in New Zealand (from Froude, 1998)

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|--|
| MB 1. Percentage And Number of Endemic Species Per Habitat Type  |
| MB 2. The Number of Taxa In Different IUCN Status Categories   |
| MB 3. Estimated Population Sizes For Selected Marine Mammal Species  |
| MB 4. Estimated Population Sizes For Selected Species of Seabirds  |
| MB 5. Relative Breeding Success For Selected Species of Seabirds in Selected Breeding Sites                                      |
| MB 6. Relative Breeding Success For Selected Marine Mammal Species At Selected Breeding Areas                                    |
| MB 7. The Spatial Extent Of Selected Habitats In Relation To Historic And Current Baselines                                      |
| MB 8. The Extent And Condition of Benthic Communities Sensitive To Sedimentation   |
| MB 9. The Distribution And Density Of Marsh Bird Species Found In The Natural Vegetation Along The Margins of Selected Estuaries |
| MB 10. Extent of Coastline With An Appropriately Vegetated Buffer  |
| MB 11. The Spatial Distribution And Relative Abundance of Selected Alien Species   |
| MB 12. The Number of Alien Species Established In the Marine Environment And The Rate of Establishment                           |
| MB 13. The Extent And Location of Extractive Industries By Type  |
| MB 14. The Number of Marine Species Protected By Legislation By Taxonomic Group  |
| MB 15. The Percentage And Area of Different Habitat Types/Ecosystems That Are Legally Protected                                  |
| MB 16. The Number, Location And Type Of Marine Species And Habitat Restoration/Enhancement Programmes                            |
| MB 17. The Condition Of Selected Marine Ecosystem Types  |

Some of these indicators could be adapted for use in the Mediterranean, including distribution and abundance of selected marine endemic species.

There are several initiatives underway in Europe that could lead to additional indicators for marine biodiversity. These include the Phytoplankton Trophic Index currently being developed by Dave Mills, Michelle Devlin and others (Cefas, 2005), the indicators applications project of MARBEF, the proposed North Sea Integrated Assessment ([www.ices.dk/iceswork/wgdetailacfm.asp?wg=REGNS](http://www.ices.dk/iceswork/wgdetailacfm.asp?wg=REGNS)), the successor to Rapid Assessment of Marine Biodiversity Linked to Environmental Remediation Studies (AMBLE) being developed by PML, the development of an application scheme for the marine benthic component by the Environment Agency, and scoring criteria developed by the IOC benthic

indicators group (Cefas, 2005). In addition, the UK government is heavily invested in identifying and applying biodiversity indicators (Defra, 2002, 2003, 2005; Rogers and Dufuy, 2005), while Europe-wide analyses continue (EPBRS, 2003; Feral et al., 2003), and lessons from these experiences could well guide Mediterranean efforts.

## **V. Recommendations on Future Biodiversity Monitoring Research**

It is quite apparent that there are almost as many ways to measure changes in biodiversity, as there are aspects to biodiversity. It is also clear that even well-studied areas like the Mediterranean region could benefit from a standardized approach to choosing and applying indicators to determine biodiversity loss. Thus it is recommended that parties to the Barcelona Convention review the pros and cons of each indicator, determine the best possible suite of both pressure and state indicators that optimally assess all aspects of biodiversity changes, and develop a mechanism to establish a standardized research protocol for the region. The standard data entry forms developed under the Specially Protected Areas protocol are a significant step in this direction, and manuals such as those developed by Italy for identifying habitats and species for national inventories will help the process. The region will also have to think collectively about how to best set up the monitoring regime so that sites provide the best samples for monitoring change throughout the Mediterranean. To this end, it is recommended that a research program that uses MPAs as sampling sites, much like Italy's Sistema Afrodite, be developed for the entire region, at least as a starting point for comprehensively assessing biodiversity.

Marine protected areas that exist within a national system or a regional network provide the basis for an ideal, cost-effective sampling regime to determine what exists in the marine environment – not just in the MPA sites themselves but also in the larger marine ecosystem(s) as a whole (Agardy, 2002). Species checklists, species abundance measures, studies tracking changes in distribution all provide critical inventory information. Physical attributes such as water quality, temperature, salinity, presence of pollutants, etc. provide information on the status of marine and coastal habitats. The health of marine ecosystems, as measured through stability and resilience, as well as indices of biotic integrity and marine trophic structure, can also be studied as baselines for detecting trends and being able to predict future conditions.

Standardized one-off inventories can provide a snapshot in time of what lives in the marine environment and in what condition. However, periodic inventorying can allow us to detect trends in living resource abundance, environmental quality, and biodiversity overall. Biodiversity indices, for instance, can provide the necessary information to establish trends in biodiversity, and allow models to be developed to predict future biodiversity loss/gain, especially if as many indicators are used as possible (Baan and van Burren, 2003). Trends in environmental quality, such as changes in turbidity, BOD, pollutant levels, etc. will also allow development of models that will allow predictions of future condition to be made. Finally, periodic sampling will allow researchers and managers to track invasives such as introduced and thermophilic species. Thus both state and pressure indicators can and should be used to determine changes to biodiversity. Using GIS, modelers can then use such information to describe how the marine environment will likely look in the future if no management interventions to control invasives is taken.

Taking stock of information on trends is critically important – not only to determine the current and expected condition of coastal areas, including those within protected areas, but also for

gauging the current and future condition of large marine ecosystems and water bodies such as semi-enclosed seas like the Mediterranean. MPA networks can thus provide a sampling regime for tracking changes in systems generally too large to study intensively with a standard basic research agenda.

## **VI. Conclusions**

There are serious constraints to using all of the CBD listed indicators for measuring changes to marine biodiversity in the Mediterranean region. One major problem is whether the choice of proxy will adequately reflect what is actually happening to biodiversity. Many of the indicators chosen by the CBD and those chosen by other multilaterals or national agencies would be better used to monitor environmental change overall, as opposed to the consequence of that environmental change as measured by biodiversity loss (Magni et al., 2004).

An important consideration in reviewing the utility of any marine indicator is how well it complements other indicators. Many of these marine biodiversity indicators have little value when taken alone, but as a set prove to contribute to robust analyses of trends in biodiversity.

Another problem, and one that is outside the purview of this analysis, is the question of what to do with indicator data when they are gathered, and whether any of these indicators can serve as an early warning system to catalyze major shifts in management before biodiversity is lost. This is a problem that is not unique to either marine systems or the Mediterranean, but it may constrain meaningful application of monitoring using these indicators.

The complexity and non-linear characteristics of ecosystems make ecological impacts difficult to monitor and predict, resulting in rapid ecosystem transitions and/or considerable inertia and time lags in management responses. A mismatch exists between the dynamics of natural systems and human responses to those changes, with inertia and lag times in both natural systems and social systems complicating the ability of humans to anticipate and develop adaptation strategies to cope with change. Anticipating major changes is complicated by lags in ecological responses, complex feedbacks between socio-economic and ecological systems, and the difficulty of predicting thresholds prior to such thresholds being overshot. Multiple impacts (especially the addition of climate change to the mix of forcing functions) can cause thresholds to change, and monitoring methods are often inadequate due to poor choice of indicators, inappropriate periodicity of monitoring, and infrequent analysis of results.

In the marine environment, managers and conservationists are still confounded by very significant data limitations. Marine research is logistically difficult, expensive, and hindered by the relative newness of the science. In addition, certain biomes remain wholly understudied, leading to a biased view of marine biodiversity more generally.

Nevertheless, a strategically designed, standardized research protocol using a subset of indicators will lead to more effective and efficient monitoring of trends in biodiversity. For a region as large and as ecologically- and sociologically diverse as the Mediterranean, a complex yet coordinated approach is needed. With a large universe of parameters that could be studied, the choice of indicators, the periodicity with which parameters are monitored, and the choice of sampling sites will have to be given critical consideration. A coordinated monitoring program using a small subset of indicators and standardized methodologies could

provide both an early warning for biodiversity loss and a way to improve marine management techniques.

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## Appendix 1: IUCN Criteria for Species Status Categories

### Extinct

A taxon is extinct where there is no reasonable doubt that the last individual has died.

### Extinct in the wild

A taxon is extinct in the wild when it is known only to survive in cultivation, in captivity or as a naturalised population well outside its past range. A taxon is presumed to be extinct in the wild when exhaustive surveys in known and/or expected habitats, at appropriate times, throughout its historical range, have failed to record an individual.

### Critically endangered

A taxon is critically endangered when it is facing an extremely high risk of extinction in the wild in the immediate future as defined by any of the following criteria:

1. An observed, estimated, inferred or suspected population reduction of at least 80% over the last ten years or three generations, whatever is the longer.
2. A population reduction of at least 80%, projected or suspected to be met within the next ten years or three generations, whichever is the larger.
3. The extent of occurrence is estimated to be less than 100km<sup>2</sup> or the area of occupancy is estimated to be less than 10km<sup>2</sup> and estimates indicate any two of the following apply:
  - there are severely fragmented populations or it exists at only one location;
  - there is continuing decline in the extent of occurrence, area of occupancy, extent and/or quality of habitat, number of locations, and number of mature individuals;
  - there are extreme fluctuations in any of the extent of occurrence, the area of occupancy, the number of locations, and the number of mature individuals.
4. The population is estimated to number less than 250 individuals and there is a continuing decline of at least 25% within three years or one generation, whatever is longer: or there is a continuing decline with severely fragmented sub populations; or all individuals are in one single sub population.
5. The population is estimated to number less than 50 mature individuals.
6. Quantitative analysis shows that the probability of extinction in the wild is at least 50% within ten years or three generations, whichever is the longer.

### Endangered

A taxon is endangered when it is not “critically endangered” but is facing a very high-risk of extinction in the wild in the near future as defined by any of following criteria:

1. An observed, estimated, inferred or suspected population reduction of a least 50 percent over the last ten years or three generations, whichever is the longer.
2. A population reduction of at least 50 percent, projected or suspected to be met within the next ten years or three generations, whichever is the longer.
3. The extent of occurrence is estimated to be less than 5 000km<sup>2</sup> or area of occupancy estimated to be less than 500km<sup>2</sup> and estimates indicate any two of the following apply:
  - severely fragmented populations or populations at less than 5 locations;
  - continuing decline in the extent of occurrence, the area of occupancy, the area and/or quality of habitat, the number of sub populations, and the number of mature individuals;
  - extreme fluctuations in any of extent of the occurrence, the area of occupancy, the number of sub populations, and the number of mature

**individuals.**

4. The population is estimated to number less than 2500 individuals and there is either:
  - a continuing decline of at least 20% within five years or two generations, whichever is longer
  - or there is a continuing decline in the numbers of mature individuals in the population because of severe fragmentation or all individuals are from one sub-population.
5. The population is estimated to number less than 250 mature individuals.
6. Quantitative analysis shows that the probability of extinction in the wild is at least 20 percent within 20 years, or five generations, whichever is the longer.

**Vulnerable**

A taxon is vulnerable when it is not critically endangered or endangered but is facing a high-risk of extinction in the wild in the medium-term future as defined by any of the following criteria:

1. There is a population reduction of the least 20 percent over the last ten years or three generations, whichever is the longer.
2. There is a projected population reduction of the least 20 percent for the next ten years, or three generations, whichever is the longer.
3. The extent of occurrence is estimated to be less than 20 000km<sup>2</sup> or the area of occupancy is estimated to be less than 2000km<sup>2</sup> and estimates indicate any two of the following apply:
  - there are severely fragmented populations or populations are less than ten locations;
  - there is a continuing decline of any of the extent of occurrence; the area of occupancy; the area and/or quality of habitat; the number of sub populations and the numbers of mature individuals;
  - extreme fluctuations in any of area of occurrence, the area of occupancy, the number of sub-populations, and the number of mature individuals.
4. The population is estimated to number less than 10 000 mature individuals and there is either;
  - a continuing decline of least ten percent within ten years of three generations, whichever is longer
  - or there is a continuing decline in the numbers of mature individuals and the population structure.
5. The population is estimated to number less than 1000 mature individuals.
6. The population is characterized by an acute restriction in its area of occupancy(usually less than 100 km<sup>2</sup>) or the number of locations (typically less than 5).
7. Quantitative analysis shows that the probability of extinction in the wild is at least ten percent within 100 years.

**Lower risk**

A taxon is lower risk where does not satisfy the criteria for critically endangered, endangered or vulnerable taxa. There are three sub categories of lower risk taxa:

1. A conservation dependent taxon is the focus of continuing taxon specific or habitat specific conservation programmes. The cessation of these programmes would result in the taxon qualifying for one of the threatened categories within five years
2. A near threatened taxon does not qualify as being conservation dependent but it is close to being of vulnerable status.
3. A least concern taxon does not qualify as being conservation dependent or near threatened.

**Data deficient**

This is when there is insufficient information to assess the risk of extinction for a particular taxon.