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Biodiversity in the Marine Environment

éditions Quæ
Éditions Cirad, Ifremer, Inra, Irstea
www.quae.com

Springer

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Photo 1. Shrimp fisheries in French Guiana. (© Ifremer, Chaloupe Project, Fabian Blanchard)

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Biodiversity in the Marine Environment

Translated by Janet Heard-Carnot

Springer

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ISBN 978-94-017-8565-5 ISBN 978-94-017-8566-2 (eBook)
DOI 10.1007/978-94-017-8566-2
Springer Cham Heidelberg New York Dordrecht London

Library of Congress Control Number: 2013958064

Printed on acid-free paper

Springer is part of Springer Science+Business Media (www.springer.com)

Foreword

Oceans and seas cover more than 70 % of the Earth and hold extraordinarily rich biodiversity, right down to great depths where abundant life forms thrive near ocean ridges. But marine biodiversity remains poorly known and faces numerous threats. Endangered by ever-increasing pressures from human activities, it is also sensitive to climate-based disturbances, in particular their consequences on ocean acidification.

Therefore, we must learn more about marine biodiversity and protect it. It is truly essential in ecosystem function and provides people with a vast number of resources and services. Maintaining marine biodiversity has now become a global priority clearly identified in several international treaties and agreements, like the Convention on Biological Diversity, and is correlatively part of European policies and national strategies (e.g. the national strategy for biodiversity and the *Grenelle* environmental and marine stakeholder consultation and legislative processes in France).

Indeed, France has special responsibility in this domain. With nearly 11 million km^2, the French exclusive economic zone (EEZ) is the second largest in the world, sheltering a great part of global biodiversity, especially in its overseas maritime area, with coral reefs, mangroves, etc.

Ifremer is one of the marine research bodies with the broadest range of expertise, spanning fisheries and aquaculture, coastal environment, biotechnologies, geosciences, mineral and energy resources, operational oceanography, underwater technologies and operation of offshore and inshore research fleets. Thanks to this extensive multidisciplinarity and the integrated approach it enables, our Institute is a natural partner in numerous projects and actions related to biodiversity. Indeed, one of the ten key objectives set out in the Ifremer strategic plan is "*learn about and characterise marine biodiversity to better protect it*".

As a true scientific challenge, an appropriate research strategy must be defined for this biodiversity. That is why I wanted a collective expert review to be conducted by a group of recognised French and foreign specialists and researchers, to answer the following question: what should Ifremer's priorities be for marine biodiversity research?

Chaired by Gilles Boeuf, who is a professor at Pierre & Marie Curie University and president of the MNHN national museum of natural history, the group of fourteen intentional experts formed for this purpose analysed existing literature and compared the results of their analysis with Ifremer's specificities. This detailed report examining the state of knowledge for marine biodiversity, drawn up during the first half of 2010, is the direct outcome of this expert review. It defines five high-priority orientations for marine biodiversity research and proposes that a partnership-based research programme be implemented. Its recommendations will enable a coherent programme to be developed, offering a framework for Ifremer, working with our partners, to further strengthen our ability to provide advice and expert assessments, in contact with the decision and policy makers in charge of managing and protecting biodiversity.

This expert panel review was supported by the Ministry of Ecology, Sustainable development, Transport and Housing (MEDDTL). Of course, it also falls under the scientific foresight work on French research on biodiversity, drawn up upon request from the Ministry of higher education and research on behalf of the national strategy for research and innovation (SNRI), by the scientific council of the Foundation for research on biodiversity, of which Ifremer is a founding member.

Jean-Yves Perrot
Chief Executive Officer of Ifremer

Introduction

The term “biodiversity” was first used in 1985 by the American ecologist W.G. Rosen and then broadly disseminated by the American entomologist E.O. Wilson. What is meant by biodiversity? Entire chapters have been devoted to presenting and explaining the concept. *Simply put, biodiversity designates the variety, amount and distribution of life on earth. It is the living part of Nature.* Much more than a simple inventory of species inhabiting ecosystems, it highlights the relationships established between these species and their environment. It is the outcome of ecological and evolutionary processes modified by human and environmental impacts. Biodiversity is intricately linked to ecosystem functions and the provision of ecosystem services (i.e. the products and processes supplied by the environment) that people benefit from. Efforts to ensure the sustainable use and conservation of biodiversity are driven by social, economic and ethical concerns and informed by scientific expertise. Numerous international commitments exist for the sustainable use of biodiversity, recognising its fundamental importance to human well-being and setting targets to halt the loss of biodiversity (MA 2005; Barbault 2006; CSPNB 2007, 2008).

The scientific requirements for knowledge needed to describe the variety of life and provide a rational basis for its management can be put into five categories:

- Cataloguing biodiversity where is it found (the variety, quantity and distribution of genes, individuals, populations, communities and ecosystems) and developing the tools and metrics needed to describe it.
- Understanding the ecological and evolutionary processes that account for the variety, quantity and distribution of genes, individuals, populations, communities and ecosystems over space and time, (i.e. how has Nature engendered more than 1.5 billion species in less than 4 billion years?) and assessing how biodiversity responds to environmental and human drivers based on analysis of the past and present, and scenarios for the future.
- Appraising how patterns of biodiversity influence the functioning of populations, communities and ecosystems in providing ecosystem services, including large-scale biogeochemical cycles and all relationships with the non-living world, as well as assessing the resulting social and economic benefits.

Fig. 1 Tuamotu (French Polynesia) land and seascape, an atoll. (© Ifremer, Olivier Dugornay)

- Understanding the factors of change in human use of marine biodiversity at various scales, including economic, social, cultural, institutional and political dimensions, as well as the ability of individuals and societies to adapt to changes in the state of marine biodiversity.
- Implementing management systems to meet objectives for biodiversity conservation, based on designing innovative approaches and tools to aid decision-makers. This involves models and indicators of changes in biodiversity and management tool performance assessments. They are informed by the first four points above, and backed up by understanding, on various scales, of the social-economic consequences of management approaches.

Future trends in human and environmental impacts on biodiversity remain uncertain and yet, it is essential that current planning and management take account of changes that may occur. Scenarios are widely used, an approach which is probabilistic by nature and takes account of the range of uncertainties related to current scientific knowledge. A key avenue for progress in this field lies in finding better ways to integrate scientific knowledge in decision-making processes, including innovation and development of adaptive learning in processes to regulate activities impacting marine biodiversity.

This document aims to explain why marine biodiversity research holds highly strategic interest for society and the scientific community.

Fig. 2 Illustrations of bivalve molluscs. (Taken from Tryon 1879, *Manual of conchology, structural systematics*, Vol. III, plate 131)

For society, research on marine biodiversity will offer new insights into marine life and could provide the necessary evidence to justify conservation priorities, while helping to prepare alternate management actions for the future. For scientists, strategic refocusing on biodiversity research will lead to shared vision and, by spotlighting the subject, help attract scientists from a range of fields and stimulate new knowledge being brought to the fore. Such a strategy will foster an interdisciplinary approach and better coordination between scientists, especially by bringing together various strands of research, as the ecosystem-based approach becomes the standard choice in marine resource management. This shift in perspective will meet the vital need to grow our capacity to provide scientific advice to policy makers in charge of managing and protecting biodiversity, as shown by the development of the IPBES Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.[1]

[1] http://www.ipbes.net/.

Acknowledgements

It is with great pleasure that the authors extend their thanks to Jean-Yves Perrot, the President-Managing Director of Ifremer, who was the initiator of this expert panel review on research needs in the social and environmental sciences in the field of marine biodiversity. He supported this work and made it materialise in Ifremer's scientific strategy, working jointly with Ifremer's Scientific Director Marie-Hélène Tusseau-Vuillemin and Associate Managing Director Patrick Vincent.

This review is the result of rich and fruitful exchanges between scientists at various French and international research institutes and the Foundation for scientific cooperation for Research on Biodiversity (FRB). Several experts have compared and cross-checked the issues raised in their respective fields of study (exact and natural sciences, human and social sciences) to identify the priorities for marine biodiversity research. Our warmest thanks go to them, and most particularly to Christophe Béné, Gary Carvalho, Philippe Cury, Bruno David, Daniel Desbruyères, Luc Doyen, Susan Hanna, Simon Jennings, Harold Levrel and Olivier Thébaud.

The outstanding old plates enhancing the chapters in this book come from the documentary collection of the Ifremer (Nantes) Atlantique centre's library. We are grateful to Marielle Bouildé and Valérie Thomé for their time and assistance in the search and discovery of books from the eighteenth and nineteenth centuries!

The photographs in this volume were kindly offered by our colleagues and fellow researchers with a keen interest in the subject. Our very sincere thanks go to Marc Taquet, Nicolas Chomérat, Lionel Loubersac, Lenaick Menot, Hugues Lemonnier, Sophie Arnaud, Fabian Blanchard, Stéphane Robert, Daniel Desbruyères and Olivier Dugornay. Several photos come from Ifremer's well-stocked photo library. We express our gratitude to Ifremer's Communications and Institutional Relations Management, especially Aurélie Desaint, Danièle Lemercier and Pascale Pessey-Martineau.

The making of this book was based on choices shared with the Quae publishing house. Great thanks go to Nelly Courtay for her patience and invaluable advice, as well as to Clarisse Robert for the page and illustration layouts of this volume.

And finally, it would not have been possible to draw up this review without the confidence and support of the French Ministry of Ecology, Sustainable Development and Energy.

Contents

Abbreviations and Acronyms

International agreements and organisations

EEA	European Environment Agency (http://www.eea.europa.eu/).
CBD	Convention on Biological Diversity (http://www.cbd.int/).
ICES	International Council for the Exploration of the Sea (www.ices.dk).
RAMSAR	Convention on Wetlands (http://www.ramsar.org/).
	United Nations Convention (http://www.un.org/Depts/los/index.htm).
COP	Conference of Parties (Convention on Biodiversity Diversity).
DIVERSITAS	International Programme of Biodiversity Science, under the institutional auspices of international organisations such as UNESCO, SCOPE, IUBS, ICSU and IUMS (http://www.diversitas-international.org/).
EC	European Commission (http://ec.europa.eu/index_en.htm).
FAO	Food and Agriculture Organization of the United Nations (www.fao.org).
IPCC	Intergovernmental Panel on Climate Change (http://www.ipcc.ch/).
IPBES	Intergovernmental Platform on Biodiversity and Ecosystem Services (http://ipbes.net).
OECD	Organisation for Economic Co-operation and Development (http://www.oecd.org/).
IMO	International Maritime Organization (www.imo.org/).
OSPAR	Oslo-Paris Convention (http://www.ospar.org/).
SBSTTA	Subsidiary Body on Scientific, Technical and Technological Advice (http://www.cbd.int/sbstta/).
IUCN	International Union for the Conservation of Nature (http://www.iucn.org/).
UNCED	United Nations Conference on Environment and Development (http://www.un.org/esa/sustdev/documents/agenda21/english/Agenda21.pdf).

UNCLOS United Nations Convention on the Law of the Sea (http://en.wikipedia.org/wiki/United_Nations_Convention_on_the_Law_of_the_Sea).
WorldFish Center (http://www.worldfishcenter.org).

International policies

MFSD Marine Framework Strategy Directive (http://ec.europa.eu/environment/marine/index_en.htm).
WFD Water Framework Directive (http://ec.europa.eu/environment/water/water-framework/index_en.html).
EUNIS European Nature Information System (http://eunis.eea.europa.eu/).
CFP Common Fisheries Policy (http://ec.europa.eu/fisheries/cfp_en.htm).
EEZ European Union's Exclusive Economic Zone (http://en.wikipedia.org/wiki/Exclusive_Economic_Zone).

Agencies, research institutes and foundations

AAMP French Agency for marine protected areas (http://www.aires-marines.fr/).
CSIRO Commonwealth Scientific & Industrial Research Organisation, Australia (http://www.csiro.au/).
DEFRA Department for Environment, Food & Rural Affairs, United Kingdom (http://www.defra.gov.uk/).
EPA Environmental Protection Agency, United States (http://www.epa.gov/).
FWS U.S. Fish and Wildlife Service (http://www.fws.gov/).
Ifremer French Research Institute for Exploitation of the Sea (http://www.ifremer.fr/).
MacArthur Foundation (http://www.macfound.org/).
MEDDTL The French Ministry of Ecology, sustainable development, transport and housing, which became the Ministry of Ecology, sustainable development and energy (MEDDE) in May 2012 (http://www.developpement-durable.gouv.fr/).
MMS Minerals Management Service (http://www.mms.gov/).
MNHN National museum of natural history (http://www.mnhn.fr/museum/foffice/transverse/transverse/accueil.xsp).

NCBI	National Center for Biotechnology Information (http://www.ncbi.nlm.nih.gov/).
NOAA	National Oceanic and Atmospheric Administration (http://www.noaa.gov/).
Sloan Foundation	(http://www.sloan.org/).
VLIZ	Flanders Marine Institute (http://www.vliz.be/EN/INTRO).
World Resources Institute	Earth trends (http://www.earthtrends.wri.org/).
WWF	World Wildlife Fund (http://www.wwf.fr/).

National (France) and international Programmes

CHALOUPE	ANR project http://www.univ-brest.fr/gdr-amure/projet-chaloupe/.
COML	Census of Marine Life (www.coml.org).
CBOL	Consortium for the Barcode of Life (http://www.barcoding.si.edu/).
CORONA Project	Coordinated Research on North Atlantic NSF-DEB-0130275/ Biogeographic Study on North Atlantic.
CPR	Continuous Plankton Recorder Project (http://www.sahfos.ac.uk/).
EDMONET	European Marine Observation and Data Network http://208.254.39.65/coastmapnews/e_article001208695.cfm.
EUR-OCEANS	Climate Change & Marine Ecosystems (http://www.eur-oceans.info/EN/home/index.php).
FISH-BOL	(www.fishbol.org).
GEOBON	(http://www.earthobservations.org/geobon.shtml).
GISP	Global Invasive Species Program (http://www.gisp.org/).
GOOS	Global Ocean Observation System (http://www.ioc-goos.org/).
HMAP	History of Marine Animal Populations (http://www.hmap-coml.org/).
IMBER	Integrated Marine Biogeochemisty and Ecosystem Research (http://www.imber.info/).
INSDC	International Nucleotide Sequence Database Collaboration (http://www.ebi.ac.uk/embl/Contact/collaboration.html).
IUBS/DIVERSITAS	(http://www.diversitas-international.org/).
MarBEF	Marine Biodiversity and Ecosystem Functioning (http://www.marbef.org/).
MA	Millennium Ecosystem Assessment (http://www.millenniumassessment.org).

MESH European	(http://www.searchmesh.net/Default.aspx?page=578).
MOREST	Summer mortality of Pacific oysters project http://www.ifremer.fr/morest-gigas/.
NOEP	National Ocean Economics Program (http://www.oceaneconomics.org/).
REPER	Environmental research observatory (ORE)-Pertuis Charentais region observatory.
RSL	Lagoon monitoring network (http://rsl.cepralmar.com/).
SAUP	Sea Around Us Project-Fisheries Ecosystem & Biodiversity (http://www.seaaroundus.org/).
SEBI	Streamlining European Biodiversity Indicators http://biodiversity.europa.eu/topics/sebi-indicators.

Technical acronyms and abbreviations

AM	Adaptive Management (http://en.wikipedia.org/wiki/Adaptive_management).
DPSIR	Drivers, Pressure, State, Impact, Response (http://www.springerlink.com/content/v447r90jl1wh0430).
EAF	Ecosystem-based Approach for Fisheries Management (http://www.jncc.gov.uk/).
EBFM	Ecosystem-based Fisheries Management Approach.
ENSO	El Niño and Southern Oscillation Climate Pattern.
HABs	Harmful Algal Blooms (http://en.wikipedia.org/wiki/Algal_bloom).
IAS	Invasive Alien Species (http://www.cbd.int/invasive/).
ITQ	Individual Transferable Quota (Fishery Management).
IUU	Illegal, Unreported, Unregulated Fishing.
MEY	Maximum Economic Yield (http://stats.oecd.org/glossary/detail.asp?ID=6504).
MPA	Marine Protected Area.
MSVPA	Multi-Species Virtual Population Analysis.
MSY	Maximum Sustainable Yield (http://en.wikipedia.org/wiki/Maximum_sustainable_yield).
NIS	Non-Indigenous Species (http://fl.biology.usgs.gov/Nonindigenous_Species/nonindigenous_species.html).
PSR	Pressure-State-Response (http://www.ens.gu.edu.au/AES1161/Topic1/Topic1R1.htm).
PVA	Population Viability Analysis (http://en.wikipedia.org/wiki/Population_viability_analysis).

SST	Sea Surface Temperature (http://en.wikipedia.org/wiki/Sea_surface_temperature).
TEV	Total Economic Value (http://en.wikipedia.org/wiki/Total_Economic_Value)

Chapter 1
The Importance of Marine Biodiversity

The study of marine biodiversity is timely and fundamental for a number of reasons (CBD, Global Biodiversity Outlook 3, 2010). Marine biodiversity plays a key role through ecosystem services (provisioning and regulation, amongst others). They provide economic wealth and resources that range from active ingredients for pharmaceuticals and medicine to products from fisheries and aquaculture, as well as contributing to cultural well-being and supplying relevant "biological models" for both basic and applied research. The role and dynamics of biodiversity are central themes when addressing climate change, earth and universe sciences or sustainable use of natural resources. Thus the issues of application involve policy, regulations and ways to globally manage energy and food security.

We now have access to a breadth of diverse tools and sensitive indicators to explore marine biodiversity, in realms which have been limited to terrestrial habitats until now, and have been difficult to apply. They range from molecular barcoding approaches that can explore entire communities, to the use of real time marine sensors incorporating innovative stimulus and photo-responsive materials and Lab-on-a-Chip (LOAC) technologies. In addition, satellite data and petaFLOP (10^{15} *FLoating-point Operations Per Second*) computing power to analyse extensive data sets are available.

The marine environment is highly sensitive to various climatic and other environmental perturbations, such as thermohaline or overturning circulation in the North Atlantic, changes in polar ice cover and greater stratification in surface waters and their acidification; resulting in already observed changes in species' phenology and ranges of distribution. Today, the ability to robustly and quantitatively assess the implications of climate scenarios on marine ecosystems and their associated services, and appraise the scope, nature and projected effectiveness of management actions in a changing context, is of prime importance.

This has led to a growing need to understand overall marine ecosystem responses, particularly to large-scale offshore developments. These include renewable energy structures (e.g. farms exploiting offshore wind and marine currents), ever-deeper drilling for oil and the associated changes in habitats, and growing demand

P. Goulletquer et al., *Biodiversity in the Marine Environment,*
DOI 10.1007/978-94-017-8566-2_1, © Éditions Quæ, 2014

for marine resources (living resources and mining), in a context of policy objectives aiming to implement holistic integrative approaches to marine management based on the principles of an ecosystem-based approach.

The human population reached 7 billion individuals in 2011, and is forecast to reach 8 billion in 2024 (Palumbi et al. 2009; UNPD 2011) and 9.3 billion in 2050 (more precisely, between 8.1 and 10.6 billion), along with population movements towards urban developed coastal areas and consequently, increased pressure on marine ecosystem services. It is currently estimated that 60 % of the global population lives within 100 km of the coast, relying on marine habitats, resources and space for food, housing, food production, recreation and waste disposal. The majority of big mega-cities with more than 15 million inhabitants are and will continue to be located near coasts. Much of the remaining non-coastal population is concentrated along rivers and other waterways and generates indirect effects on marine biodiversity (Kay and Alder 2005).

Assessing the global footprint and impact on biodiversity that these changes will entail for the topology of human society is a major question. Synergies between human drivers, the timescales and locations of thresholds, the trajectory and speed of biological adaptation to climate change, and the resistance and resilience of marine biodiversity to anthropogenic disturbances are only partially understood. They are key priorities in the quest to maintain ecosystem services. Likewise, better understanding and anticipation of the consequences that changes in biodiversity will have on individuals and human societies, particularly in their ability to adapt to them, are urgently needed.

Drawing up methods to protect and sustainably utilise marine biodiversity represents a complex issue of collective choices to be made; requiring consideration of geographic (land-sea interfaces), political (conservation, exploitation) and economic (fisheries, tourism, intellectual property, etc.) aspects. It is thus becoming increasingly important to clarify, quantify and communicate across social, academic and industrial sectors, these stakes, values, priorities and conflicting demands (Fig. 1.1).

Key Features

There are several salient features of marine biodiversity, i.e. the exceptional biodiversity in our oceans, its importance in ecosystem functioning and the fast-growing series of threats to which marine taxa are exposed. Oceans encompass approximately 72 % of the planet's surface and more than 90 % of habitats occupied by life forms. The diverse habitats there support 31 phyla of animals, 12 of them endemic to the marine realm. In comparison, there are 19 phyla from terrestrial habitats (Angel 1992; Boeuf 2007, 2010a, 2011; Boeuf and Kornprobst 2009).

High species and phylogenetic diversity is commensurate with a plethora of lifestyles, from floaters and swimmers, to those which can withstand partial aerial ex-

Fig. 1.1 *Pisces Scandinavae*: *Clupea harengus* (**a**), *Clupea Alosa* (**b**). (Taken from *Pisces Scandinavae*, Tab XI III,1895)

posure in intertidal zones or inhabit deep-sea hydrothermal vents at >2,800 m. Marine species diversity is lower than on land, estimated today at fewer than 240,000 species, the equivalent of 13 % of total species known today (1.9 million) (Census of Marine Life 2010; Boeuf 2008).

We know that life originated in the seas, so marine taxa have been evolving for more than 3 billion years longer than their terrestrial counterparts. This means that the marine environment is inhabited by archaic groups which can provide interesting and useful biological models to support basic research and for use for pharmaceutical purposes (Boeuf 2007, 2011).

Almost all extant phyla have marine representatives, compared to slightly less than two-thirds having terrestrial representatives (Ray 1991). As advanced taxonomic methods become available (Savolainen 2005) and new technologies enable previously inaccessible habitats to be explored, many new marine species are discovered on a regular basis (e.g. Santelli et al. 2008). These include both microscopic and microbial taxa (Venter et al. 2004; Goméz et al. 2007) as well as more familiar larger organisms such as fish, crustaceans, corals and molluscs (Bouchet and Cayré 2005). An example of this is the marine bryozoan *Celleporella hyalina*, thought to be a single cosmopolitan species. But DNA barcoding and mating tests revealed that geographic isolates comprised >20 numerous deep, mostly allopatric genetic lineages (Gómez et al. 2007). Moreover, these reproductively isolated lineages share very similar morphology, indicating rampant cryptic speciation.

The extent of this hidden diversity is exemplified by recent discoveries in Australian seawaters where over 270 new species of fish, ancient corals, molluscs, crustaceans and sponges have been discovered on seamounts and in canyons off Tasmania[1]. During the Lifou (Loyalty Islands) expedition in 2002, more than 4,000 species were found in an area of slightly over 300 ha (Bouchet and Cayré 2005). Unexpected microbiodiversity, invertebrates and four new species of groupers were discovered around the small island of Clipperton (Pacific Ocean) in 2007.

The phenomenon has also been observed in marine transition zones between biogeographical provinces (e.g. between the Lusitanian and boreal provinces, Maggs et al. 2008). It is estimated that new species are currently being discovered and described at a rate of 16,000–18,000 per year, including 1,600 marine species (Bouchet 2006). All but one of the cosmopolitan diatom species investigated to date are composed of multiple cryptic species (see review in Medlin 2007). Even in especially well-studied taxonomic groups, our overall understanding of the state of biodiversity is poor. For example, about 60 % of known fish species live permanently in the sea and 11,300 of them are found in coastal waters down to depths reaching 200 m (Nelson 1993). However, Reynolds et al. (2005) showed that information about conservation status was available for less than 5 % of the world's marine fish species.

This makes it difficult to formulate advice for the protection of biodiversity. It is estimated that the broodstocks of 98 North Atlantic and North-East Pacific populations of marine fishes have declined by an average of 65 % from known historic levels; and 28 populations have dropped by more than 80 %.

Most of those declines would be sufficient to warrant "threatened with extinction" status under international agreement criteria.

In addition, despite the high levels of extant species diversity, marine systems are exposed to excessive and accelerating threats from environmental change

[1] http://www.csiro.au/science/SeamountBiodiversity.html.

Table 1.1 Main human and natural pressures interacting with one another, and their combined effects on pan-European marine and coastal ecosystems. (© EEA 2007)

Pressures	Main impacts
Climate change	Increased/changed risk of floods and erosion, sea-level rise, increased sea surface temperature, acidification, altered species composition and distribution, biodiversity loss
Agriculture and forestry	Eutrophication, pollution, biodiversity/habitat loss, subsidence, salinisation of coastal land, altered sediment balance, increased water demand
Development of industries and infrastructures	Coastal squeeze, eutrophication, pollution, habitat loss/fragmentation, subsidence, erosion, altered sediment balance, turbidity, altered hydrology, increased water demand and flood-risk, seabed disturbance, thermal pollution
Urbanisation and tourism	Coastal squeeze, highly variable impacts by season and location, artificial beach regeneration and management, habitat disruption, biodiversity loss, eutrophication, pollution, increased water demand, altered sediment transport, litter, microbes
Fisheries	Overexploitation of fish stocks and other organisms, by-catch of non-target species, destructions of bottom habitats, large-scale changes in ecosystem composition
Aquaculture	Overfishing of wild species for fish feed, alien species invasions, genetic alterations, diseases and parasite spread to wild fish, pollution, eutrophication
Shipping	Operational oil discharges and accidental spills, alien species invasions, pollution, litter, noise
Energy and raw material exploration, exploitation and distribution	Habitat alteration, changed landscapes, subsidence, contamination, risk of accidents, noise/light disturbance, barriers to birds, noise, waste, altered sediment balance, seabed disturbance

and human activity (Table 1.1; OSPAR 2010). Threats such as pollution, overexploitation, eutrophication, biological invasions and climate change bring about changes in distribution and abundance of marine species (Jackson et al. 2001; Pauly et al. 2009; Worm et al. 2006; Cury et al. 2008) as well as localised extinctions. It is important to understand the mechanisms of such changes and infer what their consequences will be, as well as to encourage opportunities for recovery, resilience and reversibility of the disturbances in question (Palumbi et al. 2008).

And thirdly, marine biodiversity underpins the scope and dynamics of ecosystem functioning. Marine biota play a key role, for example, in global nutrient recycling, and supply people with a multitude of resources and ecosystem services (products and processes provided by the natural environment), including carbon storage, atmospheric gas regulation, waste processing and provision of food and raw materials (MA 2005).

Current estimates suggest that marine microalgae contribute to 40 % of global photosynthesis. For instance, coccolithophorids play a vital role in ocean exchanges, sedimentology and, generally speaking, in climate processes (Tyrrell and Merico 2004). The same holds true for deep-sea ecosystems. In a widely debated publication by Costanza et al. 1997, the economic value of all 17 biosphere services (major nutrient cycle, regulation of environmental disturbances, exploitable biological production and recreational activities amongst others) was estimated at between $ 16 to 54 trillion—mostly likely reaching US$ 33 trillion. Two-thirds of these services can be attributed to marine ecosystems (US$ 21 trillion per year, with 12.6 for coastal and continental shelf ecosystems and 8.4 for the open and deep sea). This further confirms that conservation of marine biodiversity is a priority to secure sustainable functioning of the world's oceans.

Hierarchical Components

Biodiversity is an all inclusive term to describe the variety of living organisms and their environments. It comprises four main components: (1) *genetic diversity*, referring to 'within-species' genetic variation, a crucial determinant of the ability of populations and species to withstand and recover from environmental perturbations; (2) *species diversity*, which describes the variety of species or other taxonomic groups within an ecosystem and represents the key identifiable units that determine the complexity and resilience of habitats; (3) *ecosystem diversity*, i.e. the range of biological communities and the dynamics and nature of their interdependence and interactions with the environment. Ecosystem diversity is distinct from (1) and (2) in that it comprises both a living (biotic) and non-living (abiotic) component; and (4) *functional diversity*, which includes the array of biological processes, functions or characteristics of a specific ecosystem. Some argue that functional diversity may well be the most meaningful way of assessing biodiversity because it does not necessitate the cataloguing of all species within a given ecosystem, and may thereby provide a relevant way of understanding marine natural systems for the purposes of achievable sustainable use. Although such an approach has been well documented in genomic approaches, its application is constrained by the challenge of relating diversity to function at different spatial scales (Bulling et al. 2006; Naeem 2006), and the fact that many species and their function have not yet been described. In recent times, significantly greater research efforts have been devoted to components (1) and (4), but better understanding of the linkages between the various components of biodiversity is still needed. The complexity of units and scale makes it difficult to measure biodiversity. Obviously, no single measurement can suffice. Although most studies focus on species richness, this is not necessarily the most suitable proxy for the structure or function of ecosystems. Today, the question is clear (Boeuf 2010b): seeing the extinction rate, how can biodiversity be estimated using meta-approaches, without systematically describing and knowing all the species inhabiting an ecosystem? Different methods have been proposed to address this issue (Purvis and Hector 2000; Boeuf 2010b).

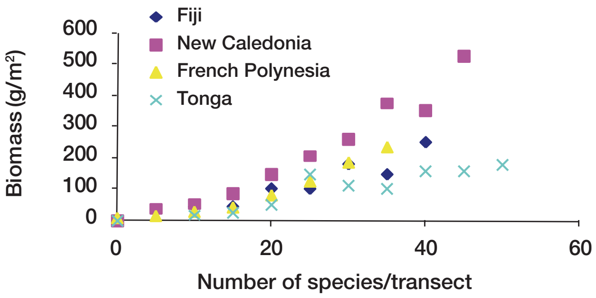

Fig. 1.2 Empirical correlation between specific biodiversity and the level of productivity for several archipelagos in the Pacific Ocean. (Courtesy of M. Kulbicki)

The Functional Significance of Biodiversity

There is growing and compelling evidence that the sustainability of ecosystem services depends upon diversified biotopes (reviewed by Palumbi et al. 2008). For example, using several independent indicators of ecosystem functioning and efficiency, a global-scale case study from 116 deep-sea sites showed that ecosystem functioning was exponentially related to deep-sea biodiversity (Danovaro et al. 2008, Fig. 1.3). This relationship, and those shown in related studies (Palumbi et al. 2008), indicate that greater biodiversity can support higher rates of ecosystem processes like organic matter production and biogeochemical cycling (Fig. 1.2). A loss of biodiversity, at least in these cases, is likely therefore to bring about a marked decline in ecosystem function.

Several studies have now demonstrated that high biodiversity—including within-species diversity—also supports either higher productivity, greater resilience or both, for example, for sessile invertebrates, large seaweeds and marine plants (Stachowicz et al. 2002; Allison 2004, Hughes and Stachowicz 2004; Reusch et al. 2005), grazing crustaceans (Byrnes et al. 2006), salmon populations (Hilborn et al. 2003), and oceanic cyanobacteria (Coleman et al. 2006). Moreover, some processes which are key to ecosystem resilience, such as recovery, resistance and reversibility, are enhanced by natural levels of biodiversity (Palumbi 2001; Palumbi et al. 2008, 2009). These studies indicate a strong positive relationship between biodiversity and ecosystem processes and services (Fig. 1.4). The ecological mechanisms that generate such correlations are well established (Bruno et al. 2003) and include complementary resource use, positive interactions among species and the increased likelihood of keystone species being present when species richness is high. For instance, complementarity, i.e. functionally similar species which occupy different niches and play slightly different roles, is certainly widespread in the marine environment. Facilitation, whereby one species may improve the environmental conditions of another, is common in marine systems, (e.g. coral reefs, wetlands and kelp forests) (Knowlton 1999). Species richness provides a repository of biological options that help promote ecosystem response to perturbation and reduces the risk of major failure.

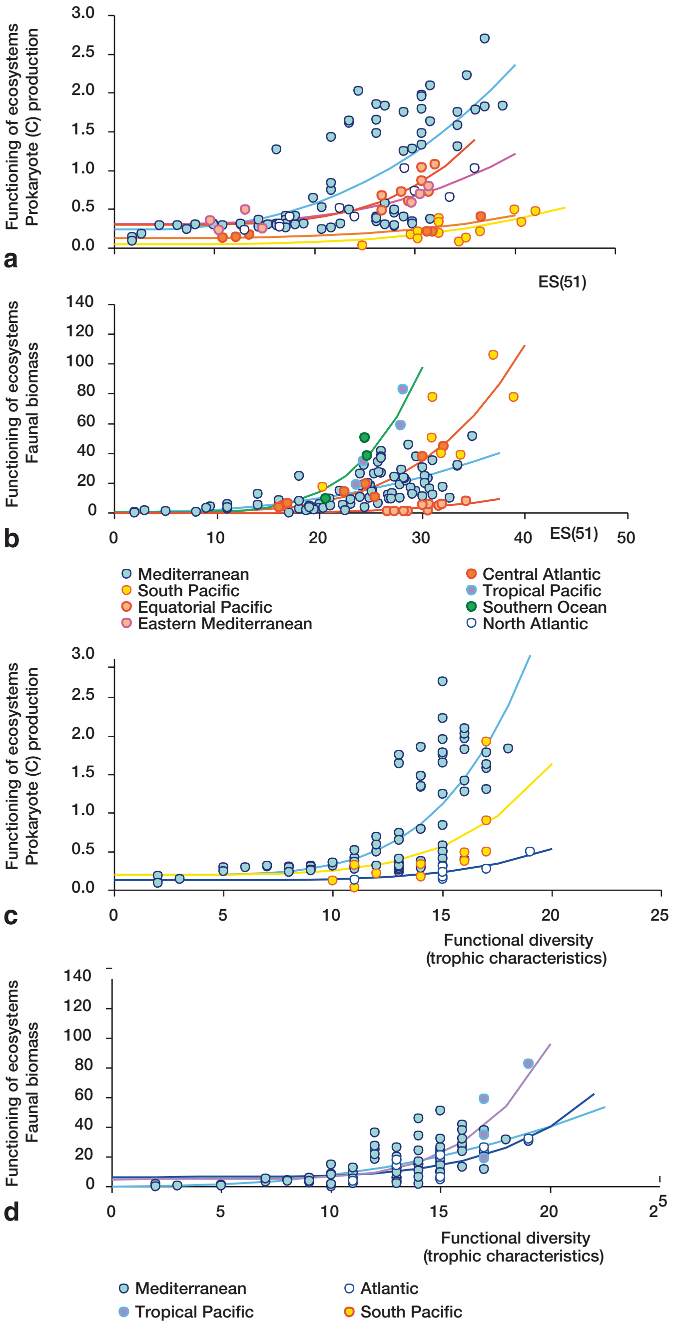

Fig. 1.3 Relationship between biodiversity and ecosystem function (From Danovaro et al. 2008). Data show a correlation between increased biodiversity of benthic meiofauna estimated geographically, (**a**, **b**), depending on trophic traits (**c**, **d**) and ecosystem function indicators (e.g. prokaryote carbon production (**a**, **c**), and the faunal biomass (**b**, **d**)

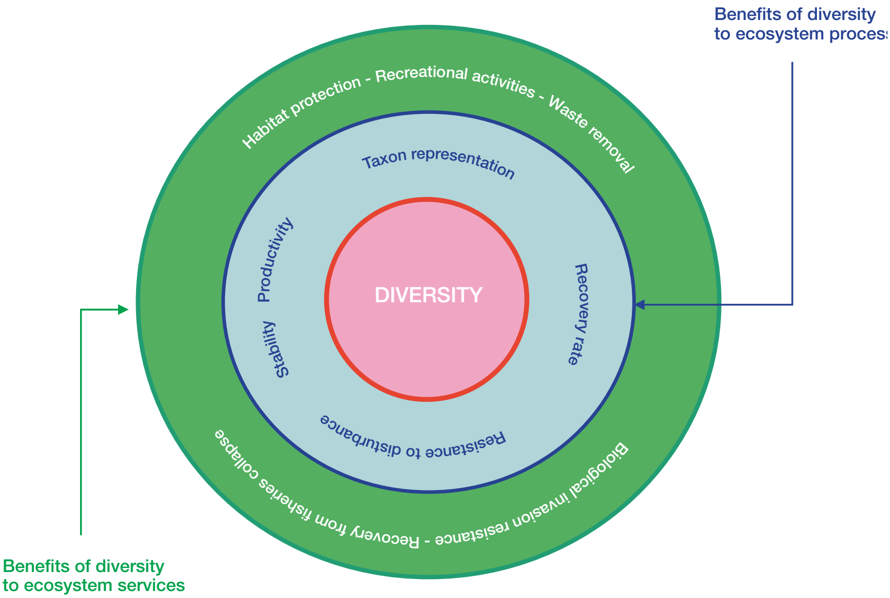

Fig. 1.4 Schematic representation of the ecosystem benefits of marine biodiversity (from Palumbi et al. 2009). Biodiversity (*pink* portion) at various biological levels (genetic, species, ecosystem and functional) enhances a variety of ecological processes (*blue* portion), which themselves accelerate the services that ecosystems provide in terms of recovery (resilience), resistance, protection, recycling, etc. (*green* portion)

Marine Biodiversity and Ecosystem Services

The benefits that society derives from ecosystems are generally called "ecosystem services" (Figs. 1.5 and 1.6). They can be put into four broad categories (Millennium Ecosystem Assessment 2005; Levin and Lubchenco 2008), which are: (1) *provisioning services*, such as food, fresh water, resources like wood; (2) *regulating services*, for instance, regulation of coastal erosion, climate regulation, diseases and water quality; (3) *supporting services*, including primary production, soil formation, detoxification and sequestering of contaminants, nutrient cycling; (4) *cultural services*, such as aesthetic ones, generally intangible and related to recreation, education and spiritual experiences. It is important to view such resources from an ecosystem-based perspective, recognising their interdependence. For example, mangrove ecosystems provide a nursery habitat for a variety of taxa, as well as trapping sediment, and contribute to recycling nutrients, regulating diseases, sheltering coastal areas from erosion, detoxifying and sequestering contaminants. They also provide food, fibre and fuel, and yield various recreational and other cultural benefits. Thus, human welfare depends on the interactions among plants, animals, microbes and their physical environment, which in turn means that alterations or degradation at one level can have cascading effects on others. Unintended

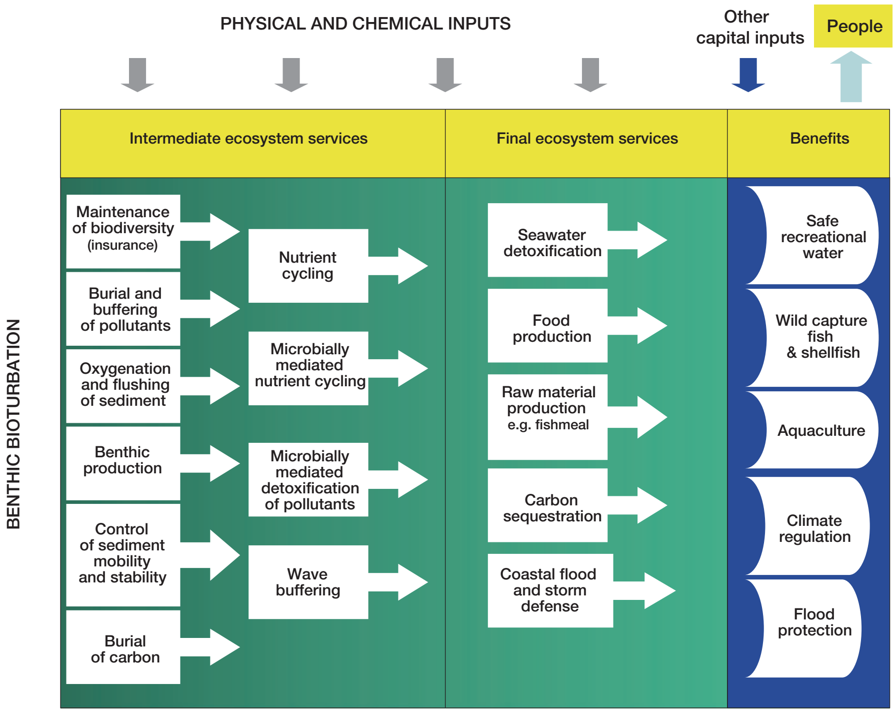

Fig. 1.5 Conceptual framework identifying intermediate and final ecosystem services as well as benefits for people based on the example of organisms involved in bioturbation (Indicative values; adapted from Major Issues in Marine Biodiversity and Ecosystem Change: Oceans 2025/NERC theme actions). Framework adapted from Fisher et al. (2008) with the help of G. Mace (2009), Austen et al. (2010) and P. Williamson (2010)[2]

modifications arising from human activities, like production and recreation, can increase vulnerability to natural phenomena like storms or to pest outbreaks.

The ecosystem-based approach's challenge lies in going beyond the way stakeholders modify ecological system function, to implement holistic management of human activities which can sustain the provision of services in the long-term and in the face of environmental change. Since complex services are supported by various animal, plant and microbial communities, several aspects of the ecosystem must be considered when adopting this approach. For instance, the breakdown of contaminants depends on the detoxification of several pollutants, and the subsequent range of metabolic processes requires a diverse microbial community (Nystrom and Folke 2001).

Many other complex ecosystem services such as fisheries also depend upon a wide range of complex ecological interactions. For example, Worm et al. (2006) provided an empirical demonstration that highly diverse marine ecosystems gen-

[2] www.oceans2025.org

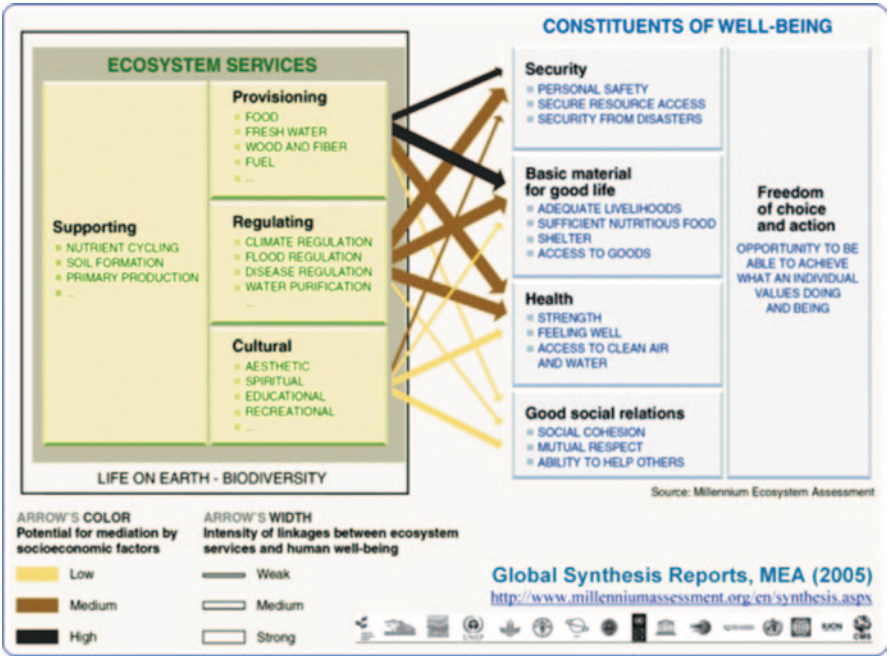

Fig. 1.6 Typology of ecosystem services taken from the Millennium Ecosystem Assessment. The *arrows*' width indicates the intensity of linkages between ecosystem services and human well-being. The *arrow's* colour indicates the extent (low, medium or high) to which socio-economic factors may mediate the linkage (e.g. there is a high potential for mediation when it is possible to purchase a substitute for a degraded ecosystem service). (From: MA, Ecosystems and Human Well-being 2005)

erally showed slower rates of fished stocks collapse and higher rates of recovery than less diverse marine ecosystems. It is also worth pointing out that the linkages between services and biodiversity cross ecosystem boundaries (Palumbi et al. 2009)

The combination of global interdependence of biodiversity, energy flow and nutrient cycling with the high and apparently dynamic species diversity in oceans provides a compelling case for ramping up our efforts to identify new taxa. The DNA barcoding approach and the use of metagenetics can provide detailed records of species richness, including estimates of species loss due to anthropogenic disturbances—however, establishing links with functional diversity remains a challenge (Fig. 1.7; Creer et al. 2010). Although it is possible to assess function by examining genes (gene expression) and metabolites, this approach does not work well when trying to characterise ecological function in eukaryotes. These functions are mediated by multiple, trophic and habitat-related aspects that cannot be predicted by a few genes. Such information is important when predicting the impact of species loss on ecosystem function and services (Fig. 1.8).

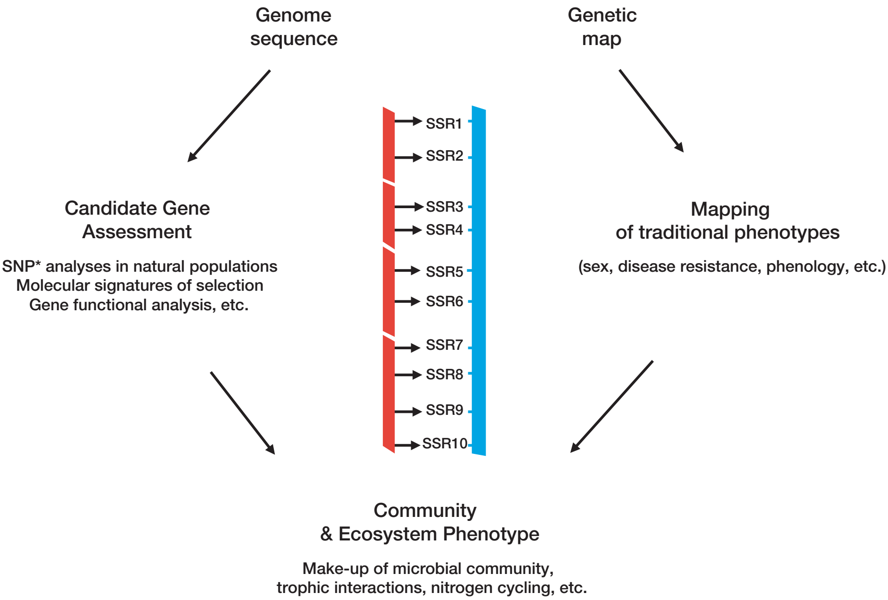

Fig. 1.7 Exploring linkages between genomic information and ecosystem function. *SNP* Single Nucleotide Polymorphism. (SSR: Simple Sequence Repeat, from Witham et al. 2008)

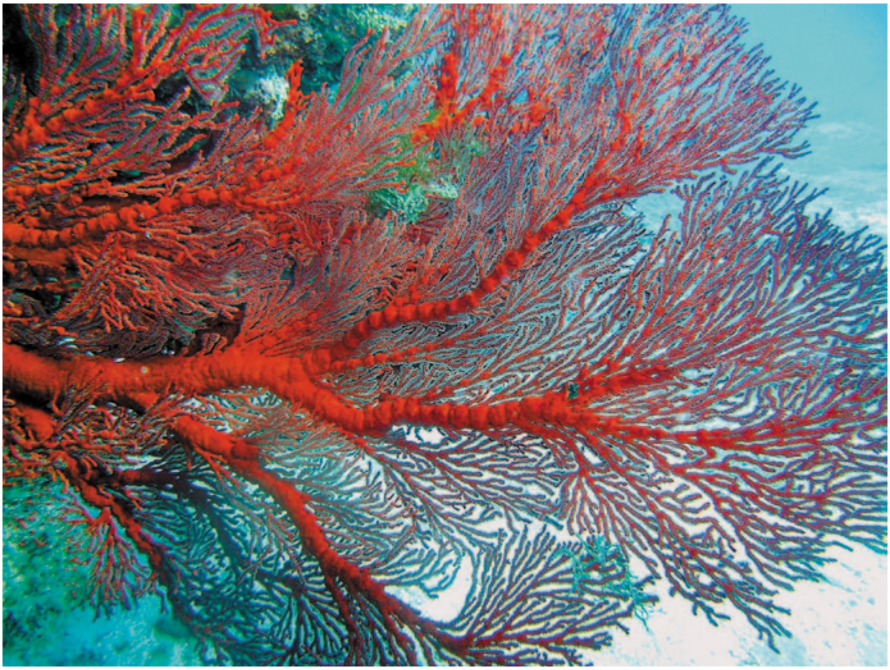

Fig. 1.8 Red gorgonian, outer reef of ilot Mato in the great South Lagoon of New Caledonia. (© Ifremer, Lionel Loubersac)

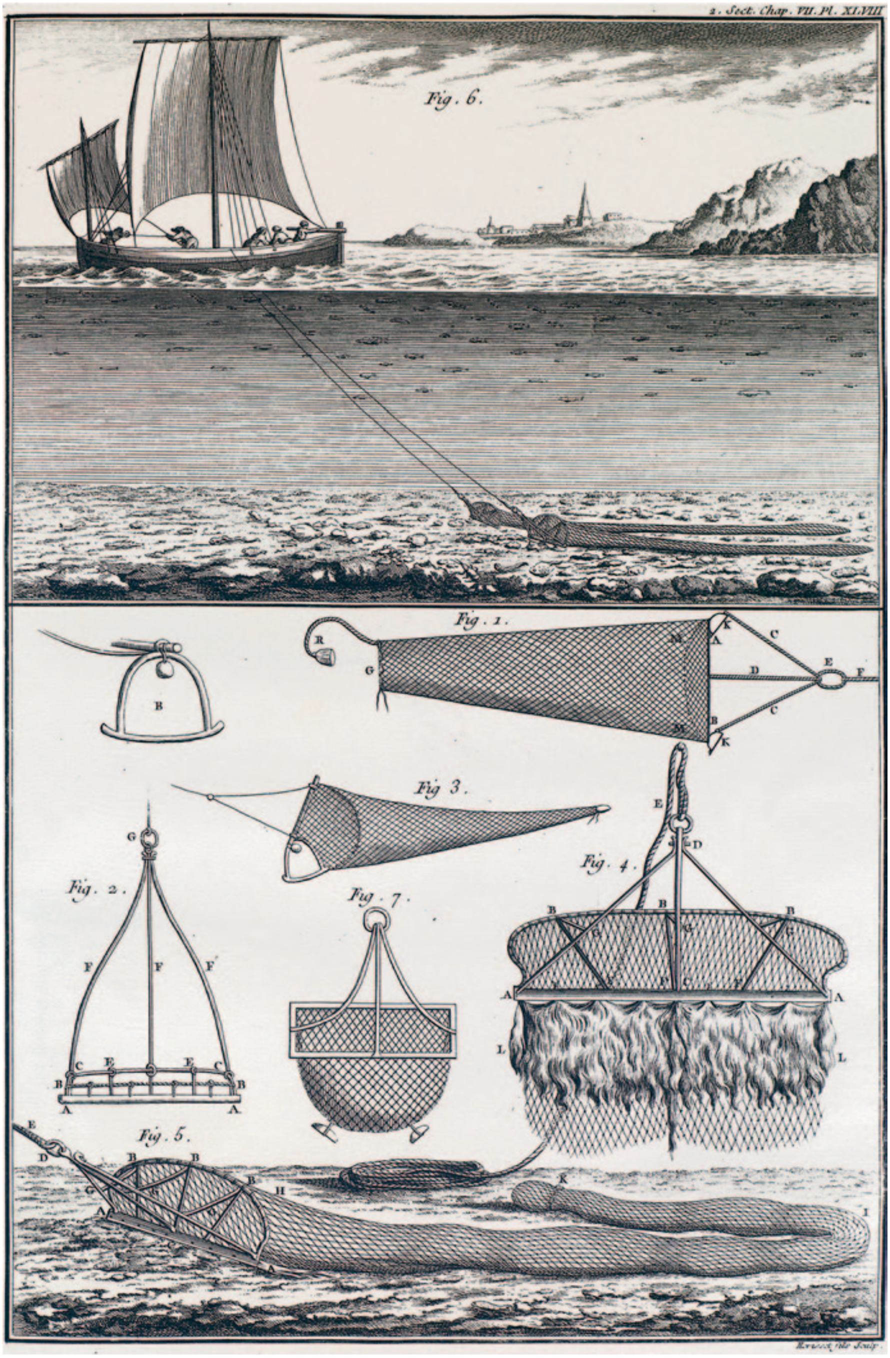

Fig. 1.9 Taken from Duhamel du Monceau and Delamarre 1769, *Traité général des pêches*, Sect. 2, chapter VII, plate XLVIII

Chapter 2
The Impacts of Human Activities on Marine Biodiversity

Human impacts have been shown to profoundly modify genetic and species diversity (Palumbi 2001) (Fig. 1.9). The main direct impacts are caused by overexploitation and habitat loss, while indirect effects may result from cascading interactions in the food web (e.g. removing competitors and predators from the system) and the effects of environmental change. Dulvy et al. (2003) reviewing local, regional and global marine extinction, identified "exploitation" and "habitat loss" as being respectively responsible for 55 and 37 % of 133 reported extinctions.

Fishing is the main cause of mortality for numerous fish and invertebrate species. Since growth and reproduction are both size-related and to some extent heritable, size-selective fishing gear puts selection pressure on populations. De facto, the exploited populations will evolve in response to harvesting pressure.

Lower local species richness will not necessarily entail a drop in fisheries productivity. However, if the targeted species are functionally "redundant", the ecological function values may change. This issue has led several authors to call for more research on functional similarities (Collins and Benning 1996). Although species richness does not appear to play a vital role in this case in maintaining ecosystem functions, we should remember than the species which could fulfil new roles when environmental conditions change must already be present. The keystone species concept also applies in the marine environment (Mills et al. 1993).

The evolutionary effects of exploitation by fisheries can be investigated with quantitative genetics. Using a model of population dynamics incorporating quantitative genetics, Law and Rowell (1993) conducted the first assessment of the effects of exploitation on body length in North Sea cod and suggested a small selection response after 40 years of exploitation. Subsequent work on fisheries-induced adaptive change has been extensive, showing continuous shifts towards maturation at earlier ages and at smaller sizes (Heino and Dieckmann 2004). These trends correspond to the outcomes predicted by the theory. Fisheries managers should be aware of this evolutionary change, because it will be hard to reverse and, if properly controlled, could bring about an evolutionary gain in yield (Law 2000).

Human-induced genetic impacts on wild populations can also result from interactions with their domesticated counterparts. Factors that may influence the magnitude, rate and reversibility of genetic responses, shifts in reaction norms and

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DOI 10.1007/978-94-017-8566-2_2,

reduced plasticity, loss of genetic variability, outbreeding depression and their demographic consequences for wild fishes have been shown in many fish populations (Hutchings and Fraser 2007).

The direct effects of fishing can also influence species diversity at two levels. First, by removing components of populations that may show some genetic differentiation and second, by depleting species that are most vulnerable. Large slow-growing and late-maturing species suffer greater population declines for a given fishing mortality rate, because these attributes are associated with intrinsically lower rates of population growth. An example of collapse in the abundance of intensively fished vulnerable species is that of the common skate *Dipturus batis*, a large ray found in the North-East Atlantic. The case is particularly striking in that overfishing was further exacerbated by the confusion between two taxa. Iglesias et al. (2010) showed recently that the so-called *D. batis* "species" actually corresponds to two distinct species, one of them (*Dipturus flossada*) reaching maturity at about 120 cm in size and the other (*D. intermedia*) at 200 cm. This discovery answers the questions raised by the apparent—and surprising—ability of the skate, a species showing low resilience, to withstand fisheries pressure brought to bear on it (Brander 1981). In fact, the depletion of *D. intermedia*, one of the largest rays in the world, was masked by ongoing catches of *D. flossada,* a smaller and very likely more resilient species, also overfished. In this case, the lack of basic genetic and biological information made it impossible to draw up protection strategies and adequate management measures (Iglesias et al. 2010).

Other important impacts on biodiversity are the effects of species transfer and introduction, which may result in biological invasions (see EU project DAISIE[1]). A prime example is represented by the Mediterranean Sea (Walther et al. 2009; Blondel et al. 2010). To be successful, an invasive species must have ecological, physiological, genetic and morphological characteristics that promote long-distance dispersal of offspring and propagules, rapid colonization rates and high competitive ability (Lambdon et al. 2008). Other human-induced factors like the deballasting of water and sediment by merchant vessels contribute to these invasions and to deteriorating biodiversity.

The impacts of climate change have also been well documented in the marine environment (Walther et al. 2009; Crain et al. 2009; Lejeusne et al. 2009). Those aspects will be discussed in Chap. 3.

Finally, if human pressures lead to sharp drops in the abundance of some species and changes in biological diversity, we can ask what the effect will be on ecosystem stability. While links between diversity and ecosystem stability are an active field of research for terrestrial ecologists, they are little studied in the marine environment (Korobeinikov and Petrovskii 2008; Fig. 2.1).

[1] http://www.europe-aliens.org/.

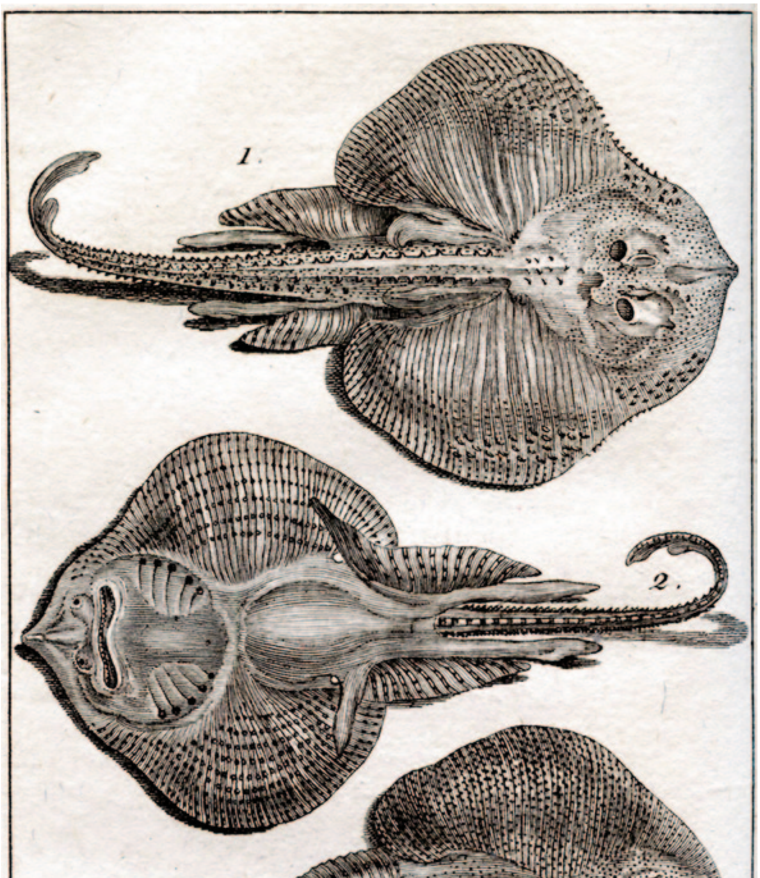

Fig. 2.1 Male rough ray (*1 and 2*), female rough ray (*3*). (Taken from Lacépède 1798, *Histoire naturelle des poissons*, vol. I, plate 5)

The Strategic Value of Research

The main reasons put forward for biodiversity conservation and research typically fall into three categories: (1) *conserving life in the oceans* is a moral and ethical responsibility, seeing the pleasure, wealth and welfare some species secure for people; (2) *species yet to be identified* are potential sources for new drugs, medical treatments and pharmaceuticals (over 15,000 to date), which is especially due to the original and archaic nature of marine biodiversity, providing a rich reservoir of food or genes and models for research; and (3) *organisms contribute to supplying ecosystem services* (Kunin and Lawton 1996; Boeuf 2007), including biological productivity, as well as controlling—or even preventing—the arrival and establishment of invasive species. There are physical, chemical, biological and physiological links between the ocean and public health. A few marine species serving as "biological models" have contributed to major progress being made in the field of life sciences, leading to several Nobel prizes, ranging from the discovery of phagocytosis to anaphylactic shock, the transmission of nerve influxes, molecular bases of memory, discovery of cyclins, organisation of the eye, neurotransmitter membrane receptors for neurotransmission and the bases of the specific immune system. These marine models are quite useful in understanding the origin and function of the mechanisms of human life and sometimes give rise to effective treatments and applications. Thus, studying and protecting marine diversity is crucial for the future of mankind.

Amongst the arguments put forward, moral and ethical aspects and the enrichment of human lives have led some sectors of society to campaign effectively for improved management of some emblematic species. However, these arguments often do little to ensure that millions of lesser known and lower profile species are also sustainably managed. Here, the main scientific inputs to the process are to identify the species concerned, assess trends in their abundance or range of distribution and establish the key factors underpinning these trends, particularly the role of human activities and their drivers. Research must also evaluate how they would respond to alternate management methods.

Assessments of the direct value of genes, species or communities to society, through the value of services they render, often provide economic arguments for the conservation of biodiversity. It is widely considered that such arguments will better influence new policies, since the costs and benefits of management actions can be directly compared (Balmford et al. 2002; CAS, 2009). Moreover, a growing field of research is focusing on the theoretical and practical (economic, social and ecological) implications of using economic incentives in support of biodiversity conservation policy. Conversely, economic incentive measures that are harmful to biodiversity need to be accurately assessed (CDB 2010), as was done on a nationwide scale in 2011 (CAS 2011).

When drawing up scientific advice on uses of biodiversity, advances made by research in understanding its role are especially important. This includes understanding the relationship between biodiversity and the provision of services, as well as (1) *how this relationship is affected* by human impacts and the environment,

Fig. 2.2 Illustrations of gastropods. (Taken from Tryon 1879, *Manual of conchology, structural systematics*, vol. III, plate 62)

(2) *the drivers of human activities* which depend on and impact marine biodiversity, and assessing (3) *the effects of alternate management actions* for biodiversity on the associated services and society.

The scientific inputs needed to describe and manage biodiversity will require collective efforts by scientists who are not necessarily used to working in an interdisciplinary approach. Taxonomists, geneticists and statisticians form the mainstay of contributors to cataloguing biodiversity (and where it is located) and developing tools and methods needed to describe it. Their work will need to be supported by building technical capacity in marine sciences, including the sampling of pelagic and deep water environments. Ecologists will work with geneticists to disentangle the ecological and evolutionary processes accounting for the distribution of biodiversity over space and time. The types and dynamics of links between biodiversity and ecosystem services will be of prime interest to both applied and theoretical ecologists and social scientists and economists. This too must be supported by innovative technological developments. Assessing the links between biodiversity and human and environmental drivers, including historical analysis such as scenarios and their social and economic impacts, will involve physical and ecological sciences as well as social sciences and economics. Likewise, diverse groups of scientists will be needed to support the development of management systems to meet objectives for biodiversity conservation, based on the above-mentioned research (Fig. 2.2).

Chapter 3
Status and Trends

How Many Marine Species are There?

Our limited knowledge of the world's biodiversity, coupled with the limitations of the current approaches to cataloguing biodiversity, are the main driving forces behind new approaches to species identification. Estimates of the total number of existing eukaryotic species range from the most conservative of 3.6 million to over 100 million, with a figure of 10 million favoured by most analysts as the nearest order of magnitude. To date, some 1.9 million species have been deposited in museums.

Approximately 1.5–1.8 million species have been described, 15 % of which are marine species. They belong to the 31 animal phyla on earth (12 of them exclusively marine), compared to 19 in the continental domain (only one of these being of terrestrial origin) (Boeuf 2010a, 2011). As of 18 August 2011, there were 213, 215 marine species listed in the World Register of Marine Species, or WoRMS[1], with 186,393 (87 %) of them validated (Table 3.1). Fish species are among the best documented and represent more than half of all living vertebrates, i.e. 48,000 species altogether. These are mainly marine or fresh water species, respectively 58 and 41 %, with just 1 % of them occupying both environments. Among the 32,000 fish species described in the international database called Fishbase[2], over two-thirds live in shallow waters such as coral reefs, and only a small percentage of them are pelagic species (sardines, anchovies, tunas).

According to Bouchet (2006), there are two notorious grey areas in evaluating the number of valid species. The first is the number of unicellular eukaryotes, in particular foraminifera and radiolarians. They have accumulated over geological periods to constitute a large fraction of marine sediments and were first studied by micropaleontologists. Since recent and fossil species were not counted separately, total estimates have varied by an order of magnitude (e.g. 4,000 species in Groombridge and Jenkins (2000) *vs.* 40,000 for Brusca and Brusca (2003).

The second grey area is due to synonymy: different authors may have unknowingly described the same species under different names in different parts of the

[1] http://www.marinespecies.org/.

[2] http://www.fishbase.org/.

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Table 3.1 Different estimations of the number of marine species currently existing (excluding microbes)

Winston 1992	250,000
Van der Land 1994	150,000
Reaka-Kudla 1997	274,000
Gordon 2000	160,000
Groombridge and Jenkins 2000	250,000
Bouchet 2006	230,000
Allemand 2008	275,000

world, or described variants (ecological, geographical, ontogenic and sexually dimorphic) as different species (Miranda et al. 2010). The problem of synonymy is most frequent in the groups of spectacular organisms (e.g. molluscs) that appeal to amateurs and collectors.

For these reasons, no consensus has currently been reached on the numbers of species living in the sea, except that the number of extant species is grossly underestimated in current records. Past and present estimates are based on experts' opinions, extrapolations from samples by habitat or area, inventories of known fauna (e.g. ERMS, Fishbase) or the rate at which species are discovered. Since they only take account of the best-known components of marine biodiversity, they exclude microbes, viruses and parasites. The estimates range from 500,000 (May, 1994) to over 10 million species (Grassle and Macioleck 1992; Poore and Wilson 1993). Based on extrapolation of well inventoried European fauna, Bouchet (op.cit.) proposed 1.4–1.6 million species of multicellular marine organisms, whereas Costello et al. (2006) suggest 1.15 million species based on similar information. At the current rate at which new species descriptions are validated, it would take 1,000 years to complete the inventory of marine biodiversity.

There are also several taxonomic black boxes making up huge reservoirs of unknown species (e.g. nematodes, parasites, symbionts, microbes and viruses; Creer et al. 2010). Although the description of new microbial strains still relies on our ability to isolate and cultivate them, molecular techniques that do not require cultures are now routinely employed as the primary approach in exploring microbial biodiversity. Not surprisingly, new molecular approaches are leading to far-reaching re-evaluation of microbial diversity in natural ecosystems (Venter et al. 2004; Carvahlo et al. 2010; Creer et al. 2010).

Symbiosis and other forms of sustainable interaction, like commensalism and parasitism, contribute significantly to total biodiversity, but the organisms involved are undersampled and their numbers remain underestimated. In his essay (cited by Bouchet, op.cit.) entitled "How many copepods?" Arthur Humes (1994) noted that of the copepods associated with benthic invertebrates in Madagascar, New Caledonia and the Moluccas, 95 % were new species. Viruses, the most common biological entities in the marine environment, which in all likelihood infect all organisms and influence many ecological processes (e.g. nutrient cycling, respiration systems, bacterial and algal biodiversity and genetic transfers) are virtually unknown. Marine viral diversity is high, probably reaching a few hundred thousand virus species according to Angly et al. (2006), with a mean abundance of 10 million viruses per millilitre of surface sea water.

Taxonomic Records

Taxonomic censuses can mainly be accessed via reference books, like guides and handbooks, and on line via interconnected international databases.

Guidebooks for the identification of marine species are far from comprehensive, both in geographical and thematic terms. The most common, large and or ecologically significant species are generally well covered in guides and handbooks. In contrast, in spite of their potentially important role in ecosystem functions, many of the smaller, rarer or taxonomically difficult-to-identify species are left out of these reference books. A look at the bibliography of 842 identification guides shows that few exist for the seas of Southern Europe, although the latter hold more species than Northern European waters. Guidebooks suitable for the scale of Europe only exist for fish. Therefore, new guides are needed for areas with high biodiversity and for smaller-sized groups like polychaete, oligochaete and tube-building worms and harpacticoid copepods. A database listing over 600 (self-identified) experts and a sub-sample of experts recognised as scientists by their peers has been set up. In spite of a larger number of experts for groups containing numerous species, there is no apparent correlation between the number of experts and the number of species per taxon. Some taxa containing thousands of species are studied by a small number of taxonomic experts. Additional funding is necessary to fill these gaps and produce new identification guides (Costello et al. 2006).

A wide range of information about biodiversity can be found on line, from international databases connected to international programmes, to local databases used for scientific projects (Figs. 3.1 and 3.2, Databases 1). Only a few of these databases can be considered as biodiversity and taxonomic references. Figure 3.1 indicates the type and flow of information within a set of international databases. Obviously, interconnection and interoperability are essential aspects in furthering knowledge about marine biodiversity.

Significant funding is provided by national and EU agencies for databasing of existing sets of data and metadata. However, this priority is built on a perception tending to overestimate the richness of existing data, whereas in fact, the greater part of information is still lacking on marine species and habitats. What should be prioritised is the systematic acquisition of basic data, with a built-in quality assurance approach, but this acquisition can be expensive. For example, the cost of mapping of the EU continental shelf for the implementation of the Marine Strategy Framework Directive (MFSD) was estimated at 900 million €.

In France, the national SINP nature and landscapes information system, mainly focused on biodiversity with patrimonial value, is based upon the MNHN’s national

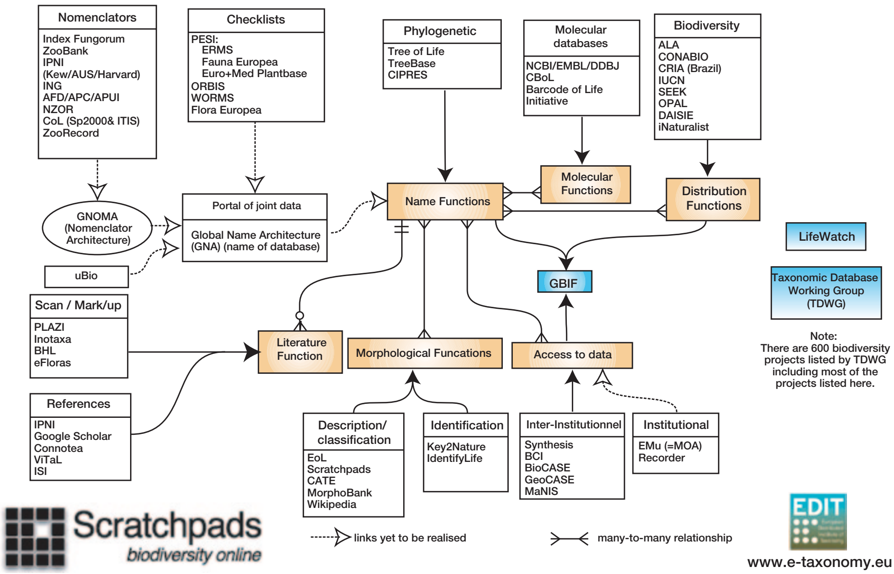

Fig. 3.1 Data flow and information transfer among biodiversity international programs and databases (Courtesy of Roberts)

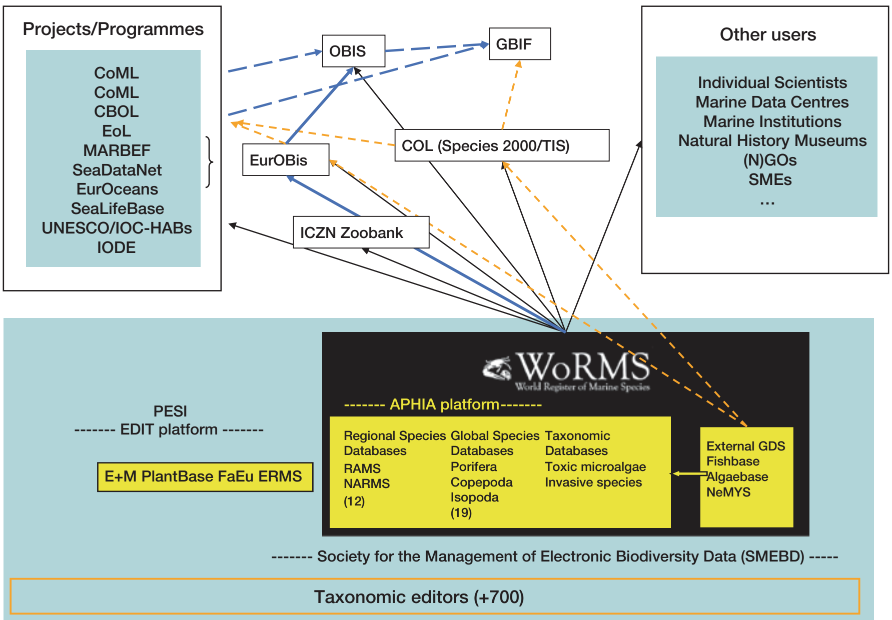

Fig. 3.2 Description of interactions between WoRMS and other databases. (From WoRMS, http://www.marinespecies.org/, April 2010)

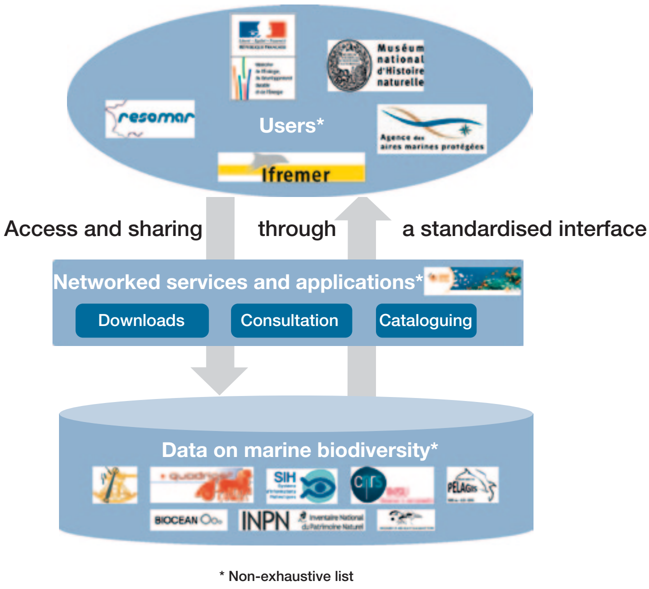

Fig. 3.3 Development of the SINP-Mer information system which is interoperable with numerous databases. (Courtesy of A. Huguet)

natural heritage inventory (INPN) database, although up to now, it has contained mostly terrestrial data. Collaborative effort is currently underway to develop an open system for marine data acquisition (**SINP-Mer**) (http://www.naturefrance.fr/sinp). This information system is based on an IT platform which provides interoperability with existing databases, including INPN and Ifremer's Quadrige, and mapping via the **Sextant** (http://w3z.ifremer.fr/sextant/) geoportal (Figs. 3.3 and 3.4). The SIH fisheries information system (https://www.ifremer.fr/isih/) databases information about maritime fisheries and their ecosystems.

Cryptic Species

Cryptic speciation in the marine realm is widespread, and it is quite likely that large numbers of hitherto "hidden" species exist. As advanced taxonomic methods become available (Savolainen 2005), and as new technologies enable exploration of previously inaccessible habitats, many new marine species are being discovered. They include both microscopic and microbial organisms (Venter et al. 2004; Gomez et al. 2007) and more familiar groups such as fish, crustaceans, corals and molluscs (Bouchet 2006).

Until recently, the marine bryozoan *Celleporella hyalina* was thought to be a single, cosmopolitan species (Fig. 3.5). DNA barcoding and mating tests revealed

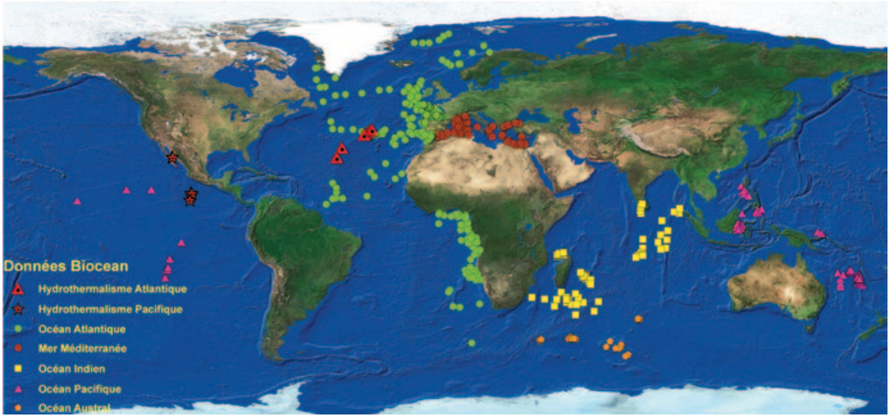

Fig. 3.4 Global distribution map of deep-sea benthic faunal data stored in the BIOCEAN core database and available through OBIS portal. (From Fabri et al. 2006)

that geographic isolates comprised >20 deep, mostly allopatric, genetic lineages (Gómez et al. 2007). Moreover, although reproductively isolated, these lineages share very similar morphology, indicating rampant cryptic speciation (Gómez et al. 2007).

The DNA Barcode

Identifying the large number of marine species not yet described (e.g. fishes, algae, bryozoans, microbial taxa) represents an enormous task. It creates a bottleneck in the research process, whose steps go from sampling to data analysis. And yet this identification is the critical starting point of any research in marine biology. Conventional identification approaches based on phenotype characters may be apparently straightforward. However, there are numerous situations in which they may fail or have limited efficiency, such as for cryptic species, inherently difficult taxonomic groups, or taxonomically ambiguous eggs and larvae.

The discovery of new marine habitats and communities and the growing threats to marine species from environmental change and disturbances, make it increasingly important to develop rapid and robust ways of describing and cataloguing marine biodiversity. Hebert et al. (2003) introduced the concept of a DNA barcode, and proposed a new approach to species identification which could overcome many of the limitations mentioned above. The method is based on the premise that the sequence analysis of a short fragment (658 nucleotide base-pairs) of a single gene (cytochrome c oxidase subunit 1, commonly called COI), enables unequivocal identification of almost all animal species. The main reason for selecting COI is its typical pattern of variation, i.e. with no overlap between intraspecific genetic distances on the one hand and interspecific ones on the other (Bucklin et al. 2011). However,

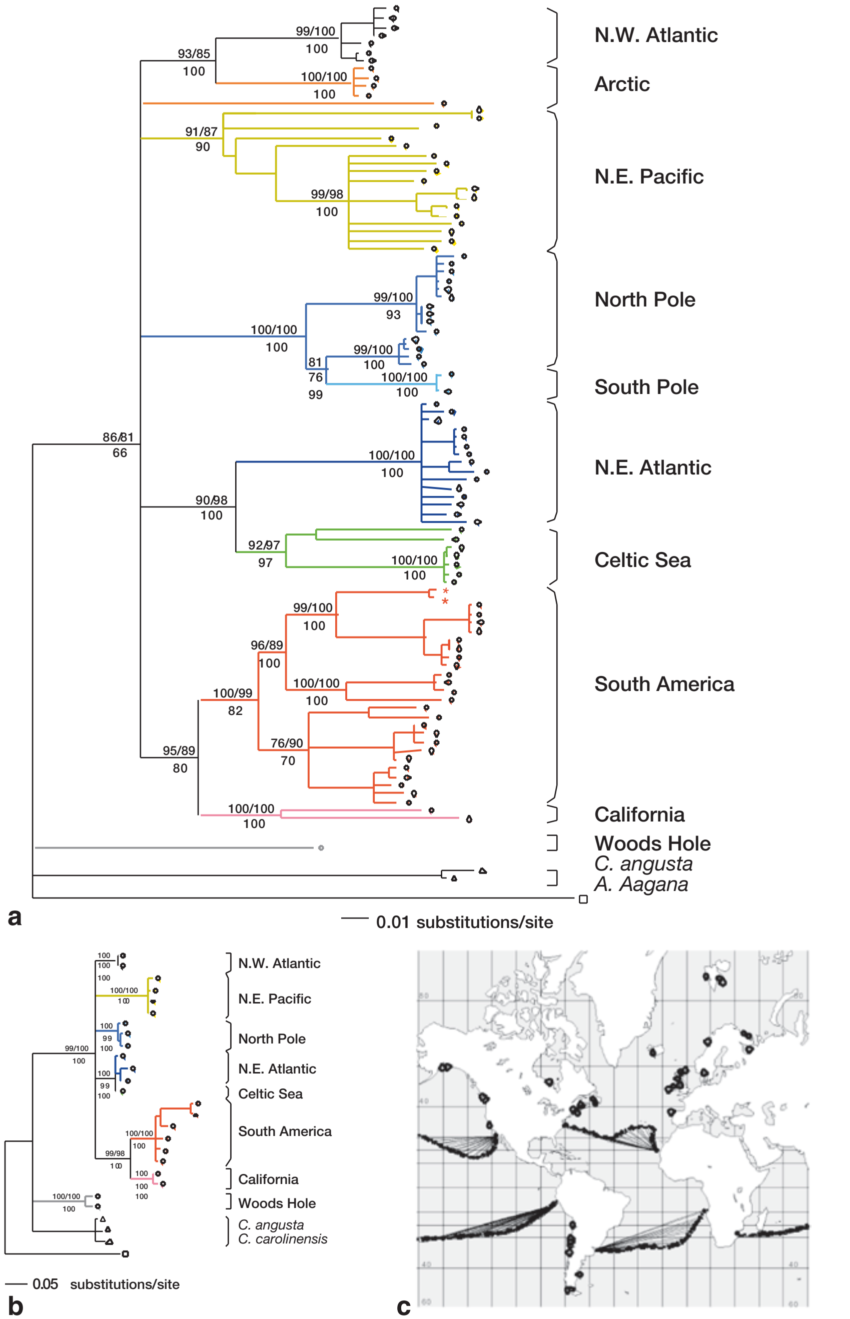

Fig. 3.5 Cryptic speciation of the marine bryozoan, *Celleporalla hyalina*. (From Gómez et al. 2007). **a** Maximum-likelihood tree of haplotype data from barcoding gene, COI. **b** Maximum-likelihood tree of the nuclear gene, elongation factor, EF-1a haplotype data. Traditionally described individuals like *C. hyalina* marked with *coloured circles* for geographical regions listed on *right*. **c** Map of sample locations included in genetic analysis. *Circles* indicate the major lineage according to the phylogenetic analysis. *Dotted lines* indicate temperate ocean limits (20 °C isotherm)

this resolving power can prove insufficient to discriminate between species in some groups whose mitochondrial evolution compared to other metazoans is lower by a factor of 10–20, i.e. phyla of *Porifera* and *Ctenophora* and cnidarians in the class of *Anthozoa*.

Like the barcodes used for commercial products, the DNA barcode provides a standardised tool for fast, simple, robust and accurate species identification. This genomic barcode fragment must have evolved fast enough to discriminate between species, while remaining roughly identical between members of the same species.

Finally, it would have to be flanked by stable DNA so that the polymerase chain reaction (PCR) method of targeted gene replication can be applied. Such molecular tools of universal implementation (DNA barcodes, "*a rigorously standardized sequence of a minimum length and quality from an agreed-upon gene, deposited in a major sequence database, and attached to a voucher specimen whose origins and current status are recorded*") can counter conventional limitations, providing a simple, yet robust system to unambiguously identify not only whole individuals, but eggs, larvae and body fragments. International efforts are now coordinated by the Consortium for the Barcode of Life (CBOL)[3], founded in 2004 and now including over 200 member organisations from 50 countries. The CBOL is closely allied to the Census of Marine Life (CoML), aiming to assess the past, present, and future biodiversity of the oceans.

The Drive to Identify New Species

Due to a false perception that most species are already known, efforts by institutions to promote cataloguing of flora and fauna remain limited. And yet it is hard to evaluate the number of species described to date without centralised reference libraries. Bouchet's (*op. cit.*) analysis of data from 2002–2003 indicates that approximately 1,600 new species are described each year (Fig. 3.6). The rate at which new species descriptions increase reflects both the size of the *phylum* they belong to and the number of taxonomists from the scientific community studying them (Fig. 3.7). Assuming that the proportion of synonyms is about 10–20 %, this means that from 1,300 to 1,500 species are added to the marine inventory annually. Apart from molluscs, the contribution from knowledgeable amateurs is small for marine species descriptions, compared to the contributions of amateurs or citizen scientists in terrestrial environments which account for 46 % of new descriptions in Europe. In European waters, there has been a linear increase in the cumulative frequency of species discovered over time (for instance, 20 % of gastropod species were scientifically named in the past 25 years (Fig. 3.6).

[3] http://www.barcoding.si.edu/.

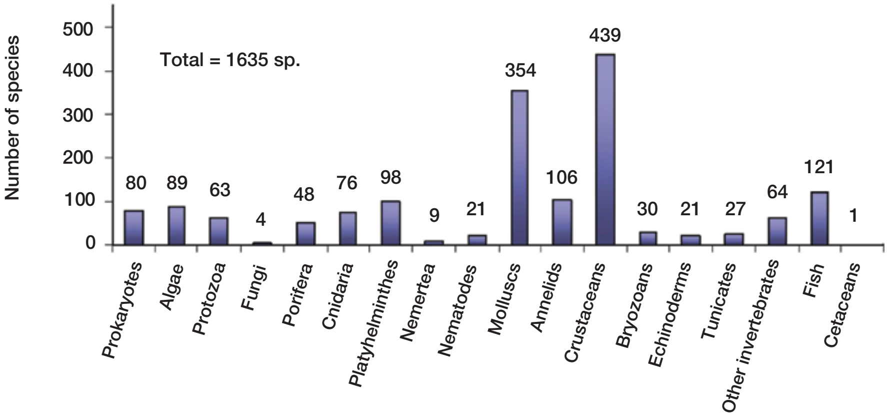

Fig. 3.6 Average number of marine species described in 2002–2003 by taxonomic group. (Bouchet 2006)

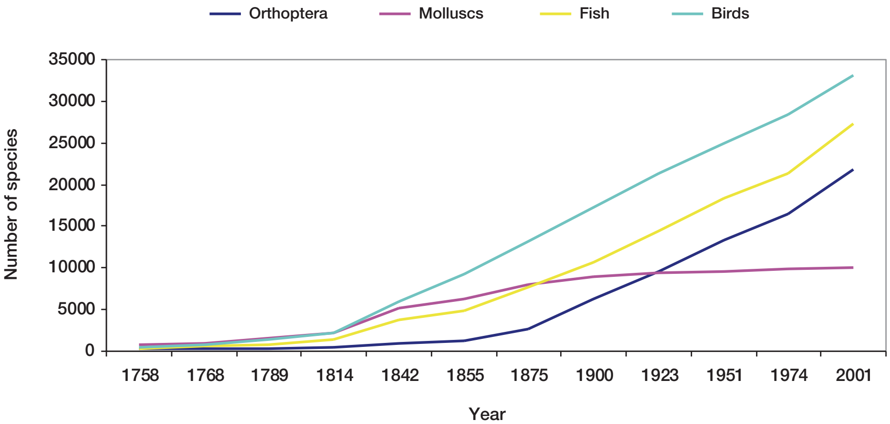

Fig. 3.7 Trend over time of species descriptions since 1750. (Courtesy of P. Bouchet)

The "Taxonomic Impediment"

Taxonomy is the science which identifies, describes, classifies and names living beings. As a science, it is becoming crucial to biodiversity management, conservation, public health, agriculture, and many other aspects of life and society.

It can be said that abundant and inexpensive molecular data have revolutionised phylogenetics and identification techniques, but they have not diminished the importance of traditional taxonomic work. Morphology, which links living and fossil species, inspires the search for causal explanations and democratizes science. Also, visual morphological knowledge is ideally suited for communication and outreach. The need for morphological research has been masked by the fact that molecular researchers could draw on centuries of banked morphology knowledge. That knowledge, however, is limited to a fraction of Earth's species and will very soon be exhausted. The reality is that, for all but a few taxa, much of the data is outdated or unreliable. Many specimens represent undescribed or misidentified species.

Global biodiversity is being lost at an unprecedented rate as a result of human activities, and several decisions have been taken to reverse these trends (CBD[4] 2010–2020 Targets). The Conference of the Parties to the Convention on Biodiversity (COP) has requested a report from its technical committee (SBSTTA[5]) on ways to offset the shortage of taxonomists available to inventory and characterise the world's biodiversity This scarcity has been documented in many reports around the world (e.g. House of Lords Report, UK 2008). IUBS/Diversitas dubbed it the "taxonomic impediment", because a lack of taxonomic expertise prevents other biodiversity research from being carried forward. The taxonomic impediment to progress in the study of biodiversity is linked to a worldwide shortage of taxonomists who can be called upon to identify species, describe species that are new to science, determine their taxonomic relationships and make predictions about their properties. The situation is expected to worsen, due to an ageing taxonomic workforce, combined with a decline in the number of students majoring or being trained in taxonomy. This will ultimately be a major issue.

Species Under Pressure [1][6]

What is the true volume of fisheries catches worldwide? [1]

Today fisheries are the only activity capturing wild species to be used for human food (and in part for animal feed) which is developed worldwide and whose practices range from small-scale to industrial. Fisheries are subjected to special governance and regulations at regional, national and international levels. Their footprint extends from inshore areas to the open sea, in practically every latitude of the earth, though the majority of catches come from the continental shelf (depths less than 200 m) and areas of upwelling (where deep, nutrient-rich water flows upward). During the 2004–2010 period, the global volume of officially declared catches was 10 million t (Mt) on average per year in inland waters (mostly Asia and Africa) and 81 Mt per year in marine waters. These statistics are based on data sent by countries to the FAO, and underestimate the true amount of biomass extracted, since they do not include two significant causes of bias, i.e. discards and Illegal, Unreported and Unregulated (IUU) fishing.

[4] http://www.cbd.int/.

[5] Subsidiary Body on Scientific, Technical and Technological Advice.

[6] Numbers in brackets refer to text boxes.

By “discards” is meant organisms which are caught and brought on board, then put back into the sea for a variety of reasons (non-target by-catch species, exceeded quotas, undersized fish, not of commercial interest, etc.). In 1994, FAO alerted scientists and managers of the sheer scale of waste in fisheries, estimating the total world discards for the period from 1980–1992 at between 18 Mt year and 40 Mt year (Alverson et al. 1994). Although as of 1996, FAO corrected these Figs. (bringing the estimation down to about 20 Mt year), the initial assessment had made such an impact that for a long time it remained the main reference cited. Ten years later, FAO published an update for the decade 1992–2001, where for 84 Mt year of declared catches there were approximately 7–8 Mt year of discards (strictly marine fisheries, not including IUU and recreational fisheries; Kelleher 2005).

All countries are affected by IUU (Illegal, Unreported and Unregulated) fishing, which is rife everywhere from the high sea to Exclusive Economic Zones. Agnew et al. (2009) suggested that the overall value of yields from illegal and unreported fisheries would be between US$ 10 and 24 billion yearly (i.e. a volume of 11–26 Mt year), not including unregulated catches in small-scale fisheries or discards. The high profitability (much higher than potential penalties incurred), intensification of world fish trade, chronic overcapacity of fisheries fleets and weak governance are motivators of this scourge, which can only be effectively combated by measures based on multilateral cooperation (High Seas Task Force 2006). The FAO adopted the first legally binding international treaty in this context, designed to counteract illegal fisheries by putting the onus of responsibility on the Port State to deny access of IUU catches and prevent them being traded worldwide (FAO 2009). The same holds true for the recent strengthening of fisheries control in the EU (European Council regulation on IUU fisheries, see O.J. L 286/1), which entered into force in January 2010.

On the basis of marine fisheries alone, by adding discards and IUU catches to the 80 Mt officially landed, while taking account of the lack of available information for small-scale fisheries in many countries, it can be surmised that at least 110–130 Mt year are actually caught in seas worldwide. Over half of all catches are made in just 20 % of the ocean’s surface area, and several authors have estimated that in order to adjust the extraction to the biological productivity of ecosystems, the fishing effort must be significantly decreased (Jennings et al. 2008; Libralato et al. 2008; Coll et al. 2008; Chassot et al. 2010).

Whether their extinction is recent or long-standing, marine and estuarine invertebrate species are rarely mentioned in scientific literature. However, three observations can be made, which suggest that they have generally been underestimated. Hundreds of taxa have not been listed since the eighteenth and nineteenth centuries.

Taxonomists have varyingly considered them as “unidentifiable”, “rare” or “synonymous” with other species. Some species may have disappeared before ever being described. And finally, the significant drop in the number of scientists studying systemics, biogeography and natural history in the late twentieth century has had a negative impact on the analysis of marine extinctions (Carlton et al. 1999).

In another sphere, the use of fishing gear like dredges and bottom trawls strongly alters biodiversity. There are several classic reviews available on the subject, such as Jennings and Kaiser (1998, pp. 208–236), Hall (1999, Chaps. 3 and 4), Jennings et al. (2001, Chap. 14), and articles 54–72 quoted in Hilborn et al. (2003). Morgan and Chuenpagdee (2003) classified the main types of gear used in three categories according to their environmental impact scores, especially in terms of habitat degradation and incidental catches. These are high-impact (bottom trawls and dredges and midwater or bottom-set nets), medium-impact (both pelagic and bottom-set longlines, traps and pots) and low-impact (hook and line, ring net seines, and midwater trawls). The authors proposed several solutions to minimise these impacts. Marine ecosystems are subjected to pressures generated by fisheries (and by aquaculture, with the introduction of species, transfer of pathogens, genetic pollution, releases of pharmaceutical substances, dependence on industrial fodder fisheries to feed carnivorous farmed species, and so on). More generally, altered marine habitat quality is firstly due to the development of activities other than fisheries and aquaculture, which mostly take place in catchments. They create forms of point-source or non-point-source pollution from farming, industrial or residential sources which are carried by air or water and can be chronic or accidental (see Table 1.1). These impacts become particularly acute in coastal areas where they are amplified by growing urbanisation of the coastal fringe (developments in estuaries, regression of wetlands, destruction of many biotopes, etc.) and contribute to the erosion of biodiversity.

Extinct Species

Extinction rates for marine species cannot be compared with those of terrestrial species. There are only a few dozen proven cases of extinction (i.e. species that have totally disappeared) in the marine environment (Carlson et al. 1999): which is very low in comparison to the known scope of extinctions on land. It could be assumed from this that human impact on the oceans is lesser. However, it is the risk of extinctions going undetected that should be envisaged, since only a small part of marine invertebrate species—even in shallow shelf waters—have been described. This could be even truer in less-explored regions of the world. In functional terms, extinction is the final outcome of a decline in population size, which is the situation for many marine species, especially those subjected to overfishing.

Carlson et al. (1999) considered that only a few marine species have gone extinct over the past 300 years. 12 species are considered to be truly extinct (3 mammals, 5 birds and 4 molluscs). Dulvy et al. (2003) more recently reviewed earlier esti-

Fig. 3.8 Silky shark accompanied by rainbow runners off the Seychelles (Indian Ocean), 2005 © Ifremer, Marc Taquet

mates, assessing the number of extinct species at local, regional or global levels. They compiled 133 examples of extinct animal and plant species (mammals, birds, fishes, chondrichthyans, echinoderms, molluscs, annelids, coelenterates and algae). This figure may be underestimated, seeing the methodology employed (backcasting analyses, fisheries statistics, interviews with fishermen). In fact, the mean time interval between the observed disappearance of a species and the moment it is declared extinct is 53 years. Amongst the numerous factors contributing to the extinction of a species, Jones and Reynolds (1997) identified overexploitation (55 %) as the main one, followed by habitat loss (37 %) and disturbances due to invasive species. Synergies also exist with pollution, which may disrupt reproductive physiology, sexual maturity and life history features of organisms. Pollution was involved in one of the best documented global extinctions, i.e. that of the macrophytic Bennett's seaweed *Vanvoorstia bennettiana* (Millar 2002).

Extirpations concern species such as dugongs, sea otters, skates, sharks and organism which build coral reefs (including deep "cold water" corals) which have gone locally extinct. This is mainly due to overfishing and/or gear impact. The North-East Pacific abalone is an example of a locally extinct species (Fig. 3.8).

Del Monte-Luna et al. (2007) suggested that the number of extinctions reported by Dulvy et al. (2003) were sometimes overestimated by almost 50 % for some groups. Several species reported as extinct were actually still alive, with a few individuals spotted recently, such as sea otters in the North-East Pacific or dugongs along the Chinese coasts. The Common or flapper skate (*Dipturus batis*), which is one of the largest rajids, has disappeared from most of its range of distribution.

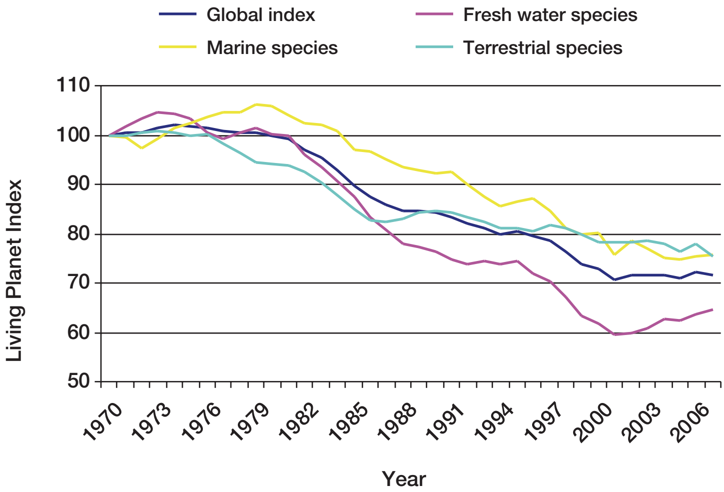

Fig. 3.9 Main trends of the Living Planet Index, for the period 1970–2010. (Courtesy of WWF)

Furthermore, as described in Chap. 2, the true status of this "species" (Critically Endangered on the IUCN Red List) was long hidden by the confusion with another. Ecological analyses to quantify marine extinction are laborious and difficult to implement and often provide uncertain conclusions. The abundance of several species has dropped so sharply that they may already be extinct. However, the main biodiversity concern is not so much the number of cases of extinctions but rather the number of depleted species which are leading to drastic and durable changes in the productivity of marine ecosystems and the ecosystem services they provide. This point is stressed by the Living Planet Index, which is based on a group of 3,000 vertebrate populations representing more than 1,100 terrestrial, marine and fresh-water species (Loh et al. 2005; Fig. 3.9). The index shows that population abundance has dropped more significantly for marine species. Moreover, it must be emphasised that obtaining maximum sustainable yield (MSY) from a fish population—a classic fisheries management target—will typically reduce its biomass by approximately 50 % of the virgin biomass (i.e. before fishing).

Endangered Species

On the regional scale, OSPAR (2010) has published a list of threatened species in the North-East Atlantic which includes: carbonate mounds, coral reefs and gardens, deep-sea sponge communities, *Cymodocea* mats, intertidal *Mytilus edulis* mussel beds on mixed and sandy sediments, littoral chalk communities, *Lophelia pertusa* reefs, maerl banks, *Modiolus modiolus* beds, ocean ridges with hydrothermal vent fields, *Ostrea edulis* native oyster beds, *Sabellaria spinulosa* reefs, seamounts, burrowing megafauna, zostera seagrass beds and the sea-pens.

More globally, the IUCN[7] has established lists of species which are extinct, endangered or vulnerable, for several years now. However accurate these lists may be, their categories are now broadly acknowledged and accepted, and they are a useful management tool for monitoring changes in biodiversity. That said, they have not been applied much to marine biodiversity, with the exception of iconic groups like sharks, turtles, corals, marine mammals or seabirds. Although the IUCN regularly engages its efforts in this direction, they remain limited, seeing the sheer scope of marine biodiversity. Thus it is hard to diagnose “good” or “poor” status for the vast majority of marine life-forms. In other words, this proves the need to develop research on these species, giving priority to those which are subjected to the greatest pressures (*e. g.* overfishing, destruction of coastal habitats) [2]. To appropriately address the challenge, a drastic increase in resources allocated to this issue is required.

Putting an end to fishing of bluefin tuna Thunnus thynnus in the Mediterranean [2]

Scientists and conservationists are concerned about the future of the bluefin tuna, an iconic Mediterranean fish species of great ecological and economic importance. This top predator can exceed 3 m in length with a body mass of 800 kg or more. A series of studies published by the Tuna Research and Conservation Center showed how the use of modern tagging techniques (electronic implanted and pop-up archival tags and isotope analysis of otoliths) has highlighted trans-Atlantic migrations of bluefin tuna (Block et al. 2005), their diving behaviour and preferred residence areas (Walli et al. 2009), natal homing as they return to their place of birth in the Gulf of Mexico and the Mediterranean and shared feeding grounds of these populations located on either side of the Atlantic (Rooker et al. 2008). This species has been fished in the Mediterranean Sea for over 2,000 years and severely overfished for more than a decade now. (Fromentin 2009). The International Commission for the Conservation of Atlantic Tuna (ICCAT—http://www.iccat.int/en/) determines a yearly quota, which was around 30,000 t between 1998 and 2008, for the Eastern Atlantic Ocean and Mediterranean Sea. However, this quota was far from being enforced and annual landings were about 50,000 t between 1998 and 2007. Seeing this situation of overexploitation, the scientific committee of ICCAT recommended a Total Allowable Catch that should not exceed 15,000 t a year and stressed that stocks could collapse in the East Atlantic and the Mediterranean without it (ICCAT 2006). Since 2008, BFT fisheries are better managed: strict control measures have been implemented, the minimum size limit was set at 30 kg and annual quotas were substantially reduced, i.e. respectively 28,500, 22,000 and 13,500 t in 2008, 2009 and 2010. Inter-

[7] http://www.uicn.fr/.

national trade data statistics show that fishing quotas are still being exceeded today. The scientific committee of ICCAT will evaluate the effects of these measures and how they may evolve, but at the least, strict management measures will have to be maintained during a decade or so to rebuild this population. Tuna aquaculture, especially capture-based farming and fattening, is not considered to be a sustainable alternative to fisheries for several reasons. They include interactions with the fishery quota system, the fact that adult fish in captivity (fattened in floating cages) do not contribute to the reproduction of wild populations and the very low conversion rate (12–20 kg of feed is required for 1 kg of tuna production, which means it is 4–5 times less efficient than salmon farming, see Mylonas et al. 2010).

Ecosystems Under Pressure: The Deep Sea

As new technologies develop, industries such as gas and oil exploitation, deep-sea fisheries, bioprospecting or mining are making rapid inroads into deep-water areas. Injection and sequestering of CO_2 in deep water reservoirs are currently being studied. These human activities, as well as illegal dumping of toxic materials (despite the London Convention) can affect deep-seafloor ecosystems, before we even learn about their biodiversity and functioning (Benn et al. 2010). Anthropogenic disturbances are especially important in the deep sea because deep macrofauna species often exhibit long life-spans, slow growth and late breeding maturity, making recovery from disturbance a long process and even, in some cases, causing local extinction.

Mining of polymetallic sulphides provides a telling example. Currently 99 % of extracted minerals come from the 29 % of the world that is land. Reserves of many metals, such as copper, are being depleted at a greater rate than new reserves are being discovered and mining companies are now investigating the possibility of mining marine metal sulphide deposits, including chalcopyrite ($CuFeS_2$), which have been formed at hydrothermal vents. Marine technology has now improved to a stage at which engineers are confident that mining machines can be constructed to work at several thousand metres of water depth. There are some apparent environmental advantages to mining on the seabed; for example there will be no acid mine drainage. There may be cost advantages; a large mining ship or barge is mobile and could be moved from one ore deposit to another. This mobility is not a feature of most current onshore methods. In addition, the legal problems of tenure may be fewer and less complex than those on land. Despite these trends, it is difficult to predict when potential future exploitation will take place at vent sites. The timing will depend upon economic conditions favouring marine mineral development over mineral conservation, recycling, substitution, technological advances in onshore mining, exploration in other areas, and declining price trends in most metal markets. Nautilus Mineral Corporation has a license issued by the Papua New Guinea government for the exploration of polymetallic sulphide deposits at all hydrothermal sites in the East Manus Basin. Neptune Resources applied to the New Zealand government for an exploration license in the Havre Trough. Neptune has recently merged with Deep Sea Minerals, which has 3 further applications pending for exploration licences. Above and beyond these case studies, a consortium of French firms is aiming to assess mining potential around the islands of Wallis & Futuna in the Pacific Ocean. Moreover, several hydrothermal sites now fall under the National Protection regulations in Canada and Portugal.

Climate Change

Today, ocean dynamics are evolving under human-induced pressures which lead to rapid changes in physical, chemical and biological characteristics of marine ecosystems, similar in scope to major past geophysical events. Recent studies confirm that rapidly rising greenhouse gas emissions are driving ocean systems towards conditions not seen for millions of years, with an increased risk of sudden non-linear ecological transformations (Doney 2010; Hoegh-Guldberg and Bruno 2010). In order to understand the present and future effects of climate change on marine biodiversity, more knowledge about the interactions of marine life with the dynamics of its physical-chemical environment must be acquired.

Faster Changes in the Ocean's Physical-Chemical Properties

The mean global surface temperature has risen at a rate of 0.2 °C per decade over the past 30 years (Hansen et al. 2006). Most of this added energy is absorbed by the world's oceans. The ocean has stored more than 90 % of the additional heat received by the Earth during the second half of the twentieth century (Bindoff et al. 2007), i.e. much more than any other store of energy in the Earth's heat balance (continents, atmosphere, ice sheets and sea ice). Levitus et al. (2009) estimated that the heat content of the 0–700 m layer of the world ocean has increased by 16×10^{22} J since 1969, leading to a 0.17 °C rise in the average temperature of the upper layers over the past 40 years (Fig. 3.10, left). The geographical distribution of the linear trend is non-uniform in space, with the maximum increase (0.4 °C) occurring in the Northern Atlantic and in European waters and more particularly in the North Sea (Fig. 3.10, right). Overlaying the general trend of European surface water warming over the past 30 years are interannual to decadal variations, such as the Atlantic Multidecadal Oscillation or AMO. The amplitude of these variations is 2–3 times greater than the long-term change observed over the twentieth century (Fig. 3.11).

As in March, April and May 2010, the global average surface temperature recorded for June 2010 was the warmest in 131 years, 0.68 °C above the average of

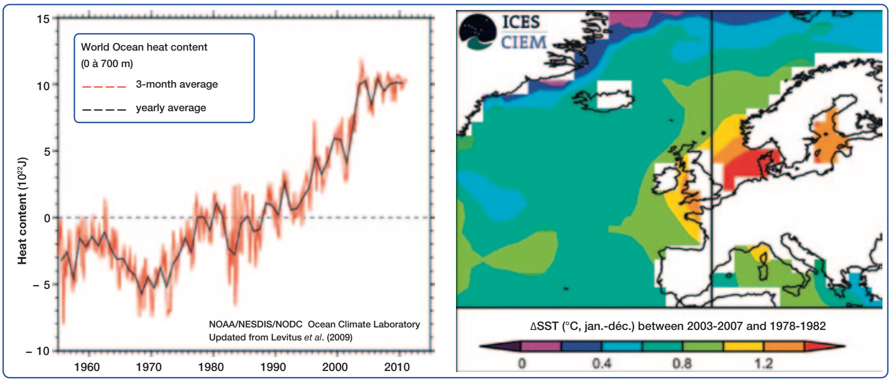

Fig. 3.10 Warming observed in surface sea water since 1955. *Left*: global scale, 0–700 m layer of the world ocean. Times series of yearly (*black curve*) and seasonal (*red curve*) mean heat content over the 1955–2006 period; one unit on the ordinate axis corresponds to 10^{22} J. (Levitus et al. 2009). *Right*: North-East Atlantic. Plot of the differences between the average sea surface temperatures (Δ SST, °C) of the 2003–2007 and 1978–1982 periods (Tasker 2008)

Fig. 3.11 Serpentine ocean research cruise on the Mid-Atlantic Ridge, February-April 2007. Logatchev hydrothermal site (depth 3,000 m): swarm of *Rimicaris* rift shrimps and Modiolus vent mussels © Ifremer, Victor 6000, Serpentine 2007

15.5 °C for the month of June in the twentieth century (NOAA-NCDC Global Analysis Report[8]). This warming can lead to sea-level rise (from thermal expansion and inflow of freshened water due to the melting of ice caps and glaciers (Kemp et al. 2011) or to increasingly intense storm phenomena (Knutson et al. 2010). Another of

[8] http://www.ncdc.noaa.gov/sotc/global/2010/6-June2011.

its effects is greater stratification of the ocean surface layers (because it is warmer, the less-dense surface water mixes less with deeper water, a phenomenon which tends to reduce vertical fluxes of dissolved or particulate compounds; Doney 2006). This leads to a drop in nutrient input in the (euphotic) surface layer, which limits primary production and promotes the expansion of biological deserts in the ocean. The extent of these deserts rose by 15 %, i.e. 6.6 million km^2, in the Pacific and Atlantic between 1998 and 2006 (Polovina et al. 2008). Moreover, there is a downward trend in oxygen concentrations in marine waters (Keeling et al. 2010) and this depletion can worsen the effects of eutrophication in coastal waters to the point of anoxia (Dead Zones, Diaz and Rosenberg 2008). Early warning signs of very abrupt changes are appearing in polar regions where water temperature (Levitus et al. 2009) and acidification (Fabry et al. 2009) are increasing twice as fast as in the rest of the world ocean. In the Arctic, this is the case for the not-entirely explained phenomenon of near-surface air temperature, called Arctic amplification, which may be mainly due to sea ice cover shrinking in summer (Screen and Simmonds 2010). From 1979 to 2010, the surface area of Arctic sea ice measured in September dropped by 12 % per decade (compared to its mean extent in 1979–2000). Several authors consider that the Arctic could be practically free of sea ice during summer in the next few decades (e.g. Wang and Overland 2009). The heterogeneity in warming distribution strongly influences water body circulation, leading to large-scale alterations to marine ecosystems, including changes in their dynamic regime (Andersen et al. 2008) and population swings in synchrony (Alheit and Bakun 2010). The ocean's multiscale variability has a far-ranging impact on many ecological processes, reaching the spatial-temporal patterns of species and population abundance, as highlighted in studies on the effects of the North Atlantic Oscillation (NAO); Fig. 3.13), which is the most energetic climate mode in the North Atlantic (Hurrell and Deser 2010; Deser et al. 2010). The NAO index shows the variability of the dipole formed from the low pressure zone over Iceland to the north and the zone of high pressures which extends into the mid North Atlantic between 35° N and 40° N, centred on the Azores high. Change in the position of the dipole has been observed since the mid-1970s (Dong et al. 2010) (Fig. 3.12).

In the North Atlantic, the alternating phases of the NAO influence the number, strength and track of storms and hence the processes which regulate atmospheric humidity and precipitation. Sea surface temperature fluctuations, and probably the depth of the mixed layer, also contribute to the "NAO signal". Atmospheric pressure field states depend on slow variations in the temperature field in surface sea waters, which determines the atmosphere's main oscillation modes. Only the ocean has the inertia needed for the decadal variability of an atmospheric oscillation like the NAO to appear. The NAO index has exhibited considerable variability over the past 110 years, with *inter alia* a downward trend from the early 1940s to the start of the 1970s (European wintertime temperatures were frequently lower than normal during negative phases), followed by a sharp increase in the index, culminating—with the notable exception of 1996—in the last two decades of the twentieth century (NAO+ , temperatures above normal in Europe). Many authors, when able to access long time series of observations for plankton or exploited fish populations, for

Fig. 3.12 Giant tube worms (*Riftia Pachyptila*) in their habitat and abyssal fauna at depths of 2,630 m on the East Pacific Rise during the Phare ocean research cruise © Ifremer, Victor 6000, Phare 2002

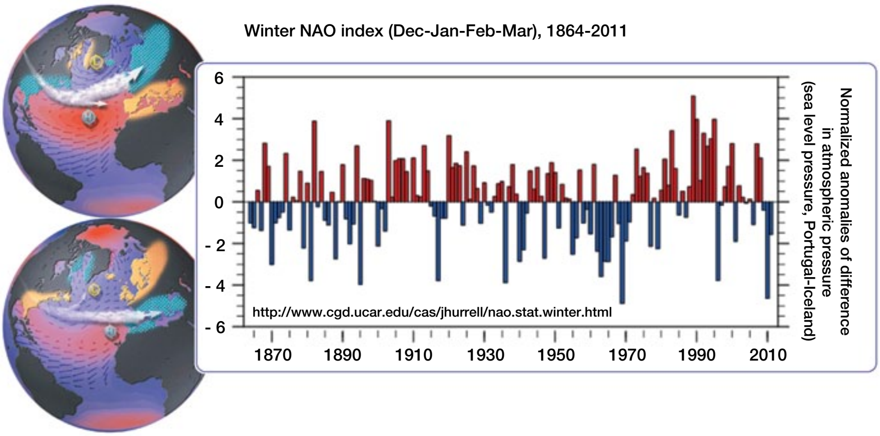

Fig. 3.13 Variations from 1864–2011 in the winter (November to March) NAO index. *Left* insert: diagram showing the influence of NAO^{+} and NAO^{-} states on the westerly wind regime Thanks to a greater number of measurement stations, an index taking account of long-term spatial variation of the NAO was computed using empirical orthogonal functions (EOF); for the period between 1899–2010, the correlation is 0.93 between the "stations" index (shown here) and the "EOF" index

instance, have highlighted the variety of responses from different marine organisms to alternating NAO phases (amongst them are Ottersen et al. 2001; Stige et al. 2006; also see remarks by Kimmel and Hameed 2008).

Contrary to the NAO, which is an atmospheric forcing (thus, with fast dynamics), the Atlantic Multidecadal Oscillation (AMO) is the slow variation in sea surface temperature (SST) occurring in the North Atlantic between the equator and 70° N. Monthly AMO index values are expressed in °C and can be calculated beginning in 1870. The AMO index equals, for a given month and year, the mean of the monthly SST anomaly in the North Atlantic, minus the global (i.e. world ocean) monthly average of the SST anomaly. During the period from 1870–2008, AMO phases alternated between cool (1900–1925, 1970–1990) and warm (1930–1960, 2000 to today) anomalies, with respect to the mean SST anomalies in the world ocean (Deser et al. 2010). The AMO comprises a component due to external anthropogenic (greenhouse gas and aerosol emissions) and natural (volcanic and solar) forcing and another component of internally generated variability which is part of the dynamics of the North Atlantic itself. By analysing various simulations of coupled ocean–atmosphere models of the IPCC[9], Ting et al. (2009) identified this internal component. The latter's trend suggests that the AMO may have entered a warm phase in the late twentieth century.

The OSPAR[10] Commission has published several documents on the regional characteristics of climate change and its effects on the marine environment in the North-East Atlantic. The main conclusions are summarised in Table 3.2 (Christophersen et al. 2009).

Changes in the Distribution Range and Abundance of Marine Species

Using a joint attribution method similar to that of Rosenzweig et al. 2008), ICES[11] performed a meta-analysis on ecological effects likely to occur in response to rising SST in the OSPAR maritime area. There are five regions in this maritime zone: (I) Arctic waters, ca. 40 % of the OSPAR maritime area—(II) English Channel, North Sea (incl. Shetland islands)—(III) Celtic Seas up to the 200 m depth contour to the west of the 6° W meridian—(IV) Bay of Biscay and Iberian Coast, from 36° N to 48° N and from 11° W to the coastline—and (V) the wider Atlantic, from 36° N to 62° N and from 42° W to 11° W off Iberia and France, and beyond the 200 m depth contour. Most observations were made in Region II (North Sea, English Channel) and there are none from Region V (wider North-East Atlantic). Due to the lack of sufficient data, phytoplankton and other organisms positioned low in the food chain or web were not taken into account. In European waters, it is likely *a priori* that warming has led to marine organisms moving northwards, with clearer variations in abundance seen near the "cold" or "warm" ends of their range of distribution. The results are summarised in Table 3.3, showing that shifts in distribution, abundance and other characteristics (e.g. phenology) are consistent with expected climate effects in 77 % of the 288 cases studied (83, 85, 100 and 20 cases for zooplankton,

[9] http://www.ipcc.ch/.

[10] http://www.ospar.org/.

[11] http://www.ices.dk/indexfla.asp.

Table 3.2 Projected climate change impacts on physical and chemical aspects of the marine environment of the North-East Atlantic and what has been observed (OSPAR, QSR 2010; Christophersen et al. 2009)

Impact	What might happen	What has been observed
Increased sea temperature	Warming in all OSPAR[a] areas but with strongest warming in Region I	Regions I–IV have warmed since 1994 at a greater rate than the global mean. Warming most evident in Region II
Reducing sea ice	Region I: sea ice may disappear in the summer in coming decades	Region I: extent of sea ice has decreased in recent decades
Increased freshwater input	Region I: 10–30 % increase in annual riverine input by 2,100 with additional inputs from the melting of landbased ice	Region I: the supply of freshwater to the Arctic appears to have increased between the 1960s and the 1990s
	Regional precipitation is difficult to project but Region IV and the southern part of Region V may experience decreases in precipitation	
Changed salinity	Regions I and V: The Atlantic ocean north of 60° might freshen during the twenty-first Century	Freshening in the deep waters of Regions I and V over the last 4 decades of the twentieth century
Slowed Atlantic overturning circulation	Slowdown of circulation in twenty-first Century is very likely	Monitoring is now in place that will be able to observe long term change in the Atlantic Overturning Circulation
Shelf sea stratification	Regions II and III: Shelf seas may thermally stratify for longer, and more strongly but in the same locations	Regions II and III: some evidence for earlier stratification in recent years and onset of the associated bloom
Increased storms	Projections of storms in future climate are of very low confidence	Regions I to V: severe winds and mean wave heights increased over the past 50 years, but similar strength winds were also present in earlier decades
Increased sea level	Between 0.18 and 0.59 m by 2,100 mostly through thermal expansion and noting high uncertainty at the upper range due to ice sheet processes. A rise of 2 m in a century cannot be discounted as a possibility based upon past change	Global sea level rose on average at 1.7 mm/yr through the twentieth Century. A faster rate of sealevel rise was evident in the 1990s

Table 3.2 (continued)

Impact	What might happen	What has been observed
Reduced uptake of CO_2	Dependent on water temperature, stratification and circulation	North Atlantic: reduced flux of CO_2 into surface waters in 2002–2005 compared with 1994–1995
Acidification	During the twenty-first Century ocean acidity could reach levels unprecedented in the last few million years with potentially severe effects on calcareous organisms	Global: average decrease in pH of 0.1 units since the start of the industrial revolution
Coastal erosion	Predictions are very uncertain and highly location specific	In many areas the combined effect of coastal erosion, infrastructure and sea defence development have led to a narrow coastal zone
Nutrient enrichment	Predictions are linked to impacts of various factors, such as rainfall patterns on freshwater input and run-off, storminess on turbidity, sea temperature on stratification	Regions I to IV: Drier summers may already be contributing to a decrease in nutrient inputs Higher nutrients inputs in wet years have caused harmful algal blooms

[a] OSPAR regions, see p. 55

Table 3.3 Detection of climate change effects on marine organisms in the North-Eastern Atlantic (OSPAR regions, 1st column). Colour coding of cells represents the percentages of changes that were in the direction expected as a result of climate impact (green: <50 %; orange: 50–75 %; red: >75 %). (Simplified from Tasker 2008)

Region OPSAR	Zooplankton			Benthos		Fish		Birds	Change in expected direction
	Distribution	Abundance	Other	Distribution	Abundance	Distribution	Abundance	Distribution and abundance	
I									74
II									77
III									83
IV									76

benthos, fish and seabirds respectively). (Simplified from Tasker 2008). Overall, the null hypothesis (H0: changes in either direction are of equal likelihood) can be rejected (Table 3.3).

Most of the available time series spanning several decades of plankton observations come from the Continuous Plankton Recorder (CPR) (Edwards et al. 2010). Analysis of these series has provided significant results, including (1) the remarkable trend (1948–1955) in the Phytoplankton Colour Index (PCI) proxy for chlo-

Fig. 3.14 Diatom (*Odontella mobiliensis*) © Ifremer, Nicolas Chomérat

rophyll concentrations (Reid et al. 1998); (2) the rapid northward biogeographical shift (10° latitude between 1960 and 1999) in the North-East Atlantic of warm to temperate calanoid copepods, Beaugrand et al. 2002, 2009; (3) the 1958–2002 increasing trends in PCI and dinoflagellate abundance versus a decrease in total diatoms, consistent with the hypothesis that more stratified surface waters create an environment favouring the growth of dinoflagellates over diatoms (Leterme et al. 2005); (4) the long-term (four decades) variability in harmful algal blooms (HABs) in the North-East Atlantic and North Sea (Edwards et al. 2006); (5) trends in cnidarian abundance in the North Sea (Haddock 2008, Attrill and Edwards, 2008) and in the North-East Atlantic (Licandro et al. 2010), and (6) co-variations (1958–2003) between phytoplankton and hydrodynamics in the North Sea (Leterme et al. 2008) (Figs. 3.14 and 3.15).

Looking beyond plankton, numerous studies have addressed the issue of links between variability in fish populations and climate fluctuations (Cushing 1982; Klyashtorin 2001; Finney et al. 2002; Lehodey 2004; Ravier and Fromentin 2004, amongst others). The subject has been given particular attention in European waters, where fish are the marine taxon for which the largest amount of data is available. Figure 3.16 illustrates changes occurring in three species in the North Sea over

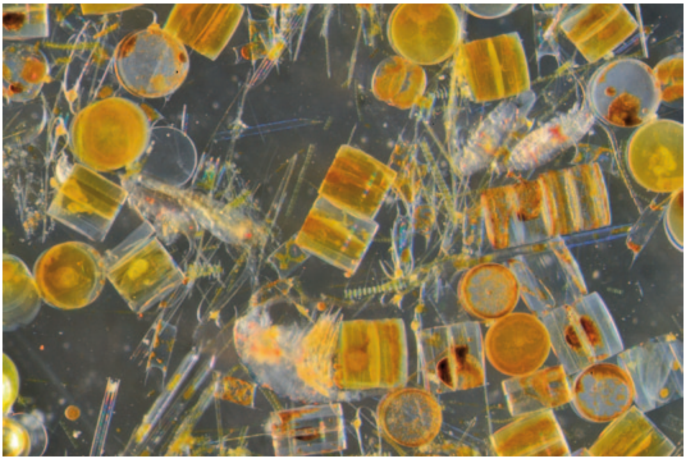

Fig. 3.15 Phytoplankton © Ifremer, Olivier Dugornay

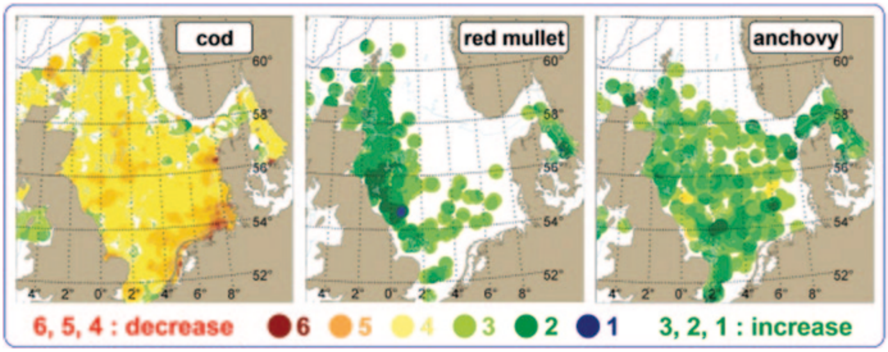

Fig. 3.16 North Sea. Changes in local abundance of three fish species between 1977–1989 and 2000–2005 mapped using 6 colour-coded categories. Warm colours (*red* to *yellow*) indicate areas where species abundance decreased and the cold colours (*green* to *blue*) where abundance increased. The dark colours (categories 1 and 6) indicate where the largest changes occurred. (From Tasker 2008)

more than 30 years. (1) The Atlantic cod *Gadus morhua* is a boreal species whose NE Atlantic southern distribution limits are around the British Isles and west of Brittany and northern limits in the Barents Sea and it is overfished in the North Sea. (2) The red mullet (*Mullus surmuletus*) is a Lusitanian species inhabiting coastal waters, whose distribution extends from Norway to North-West Africa, and covers the Mediterranean and Black Sea. In the North Sea, red mullet did not appear in surveys before 1989. (3) The anchovy (*Engraulis encrasicolus)* is a pelagic species with subtropical affinities, occasionally recorded in the period from 1977–1989, and now widespread over 80 % of the North Sea. Generally speaking, a trend can be observed of fish populations moving northward and/or deeper which respectively increases or reduces the northern and southern parts of their ranges of distribution. However, it was recently shown that this expansion of anchovies is due to higher productivity of populations in the North Sea and not, for example, to a northward shift in the distribution of southern conspecifics from the Bay of Biscay (Petitgas et al. 2012).

Similar phenomena have been described in the Bay of Biscay, where Hermant et al. (2010) analysed the responses of 20 flatfish species to warming (for the period from 1987–2006). The range of 6 species which are expanding in the Bay (including the imperial scaldfish, wedge sole and thickback sole) extends on average from 8° N to 46° N, and that of the 5 declining species (particularly dab, plaice and flounder) lies between 47° N and 58° N. The decline of plaice and dab is not due to these fish moving northwards, but rather to locally degraded conditions for their development.

Phenology and Desynchronisation of Biological Interactions

Phenology can provide sensitive indicators of climate change. It studies variability in the triggering of periodical life cycle events in response to climate fluctuations, generally on a seasonal scale. Using long-term CPR data (1958–2002) for 66 plankton taxa in the North Sea, Edwards and Richardson (2004) highlighted the patterns of phenological change in 5 functional groups (diatoms, dinoflagellates, copepods, non-copepod holozooplankton and meroplankton) found at three trophic levels (primary, secondary and tertiary producers). The entire pelagic community responded to climate change (surface water temperatures in the centre of the North Sea rose by 0.9 °C during the 1958–2002 period), but the level of response varied significantly across the functional groups and trophic levels. Organisms that are temperature-dependent for their physiological development and larval release have moved forward in their seasonal cycle (by more than 40 days for meroplankton stages in echinoderms). The majority of dinoflagellates have seasonal cycles which are earlier than before (over 3 weeks for the *Ceratium*, *Protoperidium* and *Dinophysis* genera) while the dates for spring blooms—and thus the beginning of the pelagic seasonal cycle—and autumn blooms of the diatoms which are more light-dependent (both in intensity and photoperiod) remain the same on average—but with high variability. These results were part of a meta-assessment of all phenological changes observed from 1976 to 2005 in UK terrestrial, freshwater and marine ecosystems (Thackeray et al. 2010). In primary producers, as in primary and secondary consumers, seasonal biological events have advanced on average by a dozen days over a period of 30 years. All three types of environment show the same trend, but there is high variability between taxonomic groups. The authors emphasised the fact that in marine and continental areas, there is less of an advance in secondary consumers compared to primary producers and consumers, concluding that all types of ecosystems are facing the risk of desynchronised biological interactions.

It is hard to infer the global community response, not only due to the variety of responses from the organisms making up these communities, but also because its determinism cannot be attributed to climate change alone, particularly when fishing pressure is a confounding factor. A typical example is that of Atlantic cod, whose commercial catches have dropped over most of its distribution range over the past few decades, with the notable exception of the Barents Sea where the adult cod biomass exceeded 1.8 million t (Mt) in 2011 and will reach 2 Mtin 2012. In the North Sea, cod's decline since the 1970s is the outcome of combined effects of overfishing and ecosystem modifications due to climate change (see the long-term 1958–2007 data analysis by Beaugrand and Kirby 2010). Beaugrand et al. (2003) gave a classic example of food web disturbance, in the mismatch (Cushing 1982) between a predator (cod larvae) and a prey (calanoid copepod) in the North Sea. Analysis of changes in plankton communities (using 1958–1999 CPR data) showed that the so-called boom period for gadoids between1964–1983 was followed by environmental conditions which were unfavourable for these fish. Prey for cod larvae became scarcer: the size of calanoid copepods decreased from the early 1980s on, *Calanus finmarchicus* was replaced by *C. helgolandicus* and the *Calanus* began appearing in late autumn rather than spring. The rise in mortality of larvae and juveniles and overfishing jointly brought on the collapse of cod stocks in the North Sea. In the 1960s, adult cod represented some 30% of the stock's total biomass and one third of that total biomass was caught each year. By the end of the 1990s, adults made up only a fourth of the total biomass, approximately half of which was fished each year. Over the past decade, in spite of the decreasing trend of fisheries mortality and the implementation of a management plan, cod stocks have not recovered and the total biomass in 2011 was less than one fifth of its 1970 level[12] (Mieszkowska et al. 2009).

Ecosystem Shifts

There is, for the set of (physiological, ecological, etc.) signals specific to each species, a corresponding potential to recompose communities, a process which can disturb biological, e.g. trophic interactions (see above, as well as Lindley and Kirby 2010; Schlüter et al. 2010). This phenomenon has occurred in several marine ecosystems, leading to sudden change in their state (the abruptness of the shift is relative to the generation time of high-level predators affected by the event). In English language publications, this has been called the Ecosystem Shift, or Ecosystem Regime Shift. The pragmatic definition proposed by De Young et al. (2008) is that of a far-ranging restructuring of the ecosystem, lasting long enough for a quasi-equilibrium state to be observed. The driver of the shift is external to the biological component of the ecosystem. Generally, it is physical (i.e. climate and oceanic phenomena) and can be combined with anthropogenic pressures, like overfishing. However, the underlying processes remain poorly understood and most regime shifts have only been identified several years after they have actually occurred.

One example of this is the change in the status of the North Sea ecosystem in the 1980s (Beaugrand et al. 2009; Llope et al. 2009), which was well borne out

[12] http://www.ices.dk/advice/icesadvice.asp ICES Advice 2011, Book 6, North Sea. Cod in Subarea IV (North Sea), Divisions VIId (Eastern Channel) and IIIa (Skagerrak).

by analysing time-series for several biological descriptors and by identifying the related modifications to the physical environment. In keeping with this, Beaugrand et al. (2009) suggested that the shift in 9–10 °C SST isotherms, concomitant with the warming of the North Atlantic, would indicate a threshold which once crossed, would entail major ecological changes.

Future Impacts on Ecosystem Services

As understood by the Millennium Ecosystem Assessment (MA 2005), marine primary production is the "supporting service" which provides approximately half of primary production in the biosphere and is the source of 99 % of organic matter used in marine food webs. In the next few decades, surface water warming will subject phytoplankton to antagonistic effects. Increased thermal stratification effectively tends to keep phytoplankton cells in surface waters, where they benefit from maximal solar energy. However, it also reduces the input of nutrients in the euphotic layer by reducing vertical mixing. The response seems to vary depending on the biogeographical provinces, leading to possibly lower productivity in the tropics and mid-latitudes and respectively to possibly higher productivity at high latitudes (Doney 2006). This hypothesis has been confirmed overall by several observations and projections from models. For instance, Behrenfeld et al. (2006) analysed synoptic satellite ocean colour recordings made over a decade (1997–2006) of 74 % of the ocean's surface (SeaWiFS) and established an inverse relationship between the variation in net primary production and that in surface temperatures. This result reaches the same conclusions as recent simulations by Steinacher et al. (2010) who suggested there would be a 2–20 % decrease in primary production from phytoplankton overall by 2,100 relative to pre-industrial conditions. Moreover, the increase in the recycled part of production would reduce the fluxes of particulate organic matter exported to food webs.

In warmer, more stratified surface sea water, fewer nutrient inputs could lead to reconstruction of the plankton community. Small, single-celled organisms (flagellates, cyanobacteria) would predominate over diatoms, microzooplankton would develop and there would be a greater number of trophic levels. Less efficient transfers between the latter, combined with greater losses from respiration due to warming, would reduce fluxes of organic matter available to other food web compartments (e.g. for fish this would range from plankton-eaters up to high-level predators). *In fine*, the pelagic ecosystem would shift to a less productive alternative state which would particularly affect the "provisioning service" of living resources used for human consumption (fish, molluscs, crustaceans). This means that less organic matter would be exported towards the abyssal zone, entailing alterations in deep benthic populations (Smith et al. 2008). Finally, it has been suggested that synergies between climate change, human pressures—like overfishing of plankton-feeders (herring, sprat, sardine, etc.)—and bottom-up transfers in food webs, could favour top predators being replaced by jellyfish, not just offshore, but also in eutrophicated coastal zones (Pauly et al. 2009; Richardson et al. 2009; Condon et al. 2011, also see reservations expressed by Condon et al. 2012).

Acidification, a "Chemical Mirror" of Ocean Warming

Marine surface waters are slightly alkaline and today their mean pH is close to 8.1. Ocean acidification is the name given to the approximately 0.1 unit decrease in mean pH since 1800, due to the ocean absorbing from 25 to 30 % of the CO_2 emitted by human activities. At the end of the pre-industrial period, the partial pressure of CO_2 (pCO_2) was 280 ppm, reaching 380 ppm in the 1970s (Sabine et al. 2004) and currently slightly exceeding 390 ppm (see Trends in Atmospheric Carbon Dioxyde[13]). This leads to lower pH levels in sea surface waters, dropping at a rate of 0.017–0.020 units per decade. At the turn of the century, it was estimated that the ocean had accumulated some 135 billion t of anthropogenic carbon (Sabine and Tanhua 2010). The fate of CO_2 absorbed by surface sea waters depends on several interacting non-linear processes. In addition to hydrodynamic transport and mixing, vertical gradients of dissolved inorganic carbon content, which are -under current conditions—approximately 90 % of HCO_3 bicarbonate ions, 9 % of CO_3 carbonate ions and 1 % of $CO_2 + H_2CO_3$ are governed by two mechanisms, i.e. the solubility pump and the biological pump. The former is the result of increased CO_2 solubility when temperatures drop, combined with the formation of deep water (downwelling of cooled surface waters at high latitudes). The biological pump encompasses two fluxes directed from the euphotic layer towards the deep sea (Denman et al. 2007 in Solomon et al. (eds) 2007), i.e. transport of photosynthesised organic matter and sinking of carbonate particles produced by "calicified plankton", e.g. coccolithophorids (single-cell microalgae) and pteropods (small pelagic molluscs).

Our understanding of how marine biodiversity responds to ocean acidification remains rudimentary. Early studies were primarily focused on calcifying organisms (pluri- and unicellular) as many of them precipitate calcium carbonate or $CaCO_3$ either in the form of calcite (e.g. coccolithophorids, foraminifera, echinoderms, various crustaceans) or aragonite (e.g. tropical and cold-water corals), both forms being present in mollusc shells. High-Mg calcite is found in crustose coralline algae, benthic foraminifera, bryozoans and echinoderms Most of the results come from *in vitro* experiments which followed various protocols. Several of these studies, whose aim was neither to study the effect of this phenomenon, nor to study the long-term adaptive responses of communities, tested much higher pCO_2 levels than those foreseen in the most pessimistic IPCC scenarios, moreover without reproducing the progressiveness of ocean acidification. The growing interest in this issue is shown by the recent publication of several reviews (reports by the Royal Society 2005, the ICES report edited by Fernand and Brewer 2008, that of the Secretariat of the Convention on Biological Diversity 2009 and the report by the European Science Foundation 2009) as well as special sections in journals (*Marine Ecology Progress Series*, Vézina and Hoegh-Guldberg 2008; *Biogeosciences*, Gattuso et al. 2008; and *Oceanography*, Kappel 2009). This is also a general orientation of several research

[13] http://www.esrl.noaa.gov/gmd/ccgg/trends/.

programmes like the integrated FP7 EPOCA[14] project (European Project on Ocean Acidification). The recent book published by Gattuso and Hannson (2011) provides a summary of the current key knowledge available.

Future Changes in the Physical-Chemical Properties of Marine Waters

All of the scenarios envisaged by IPCC predict that the partial pressure of atmospheric CO_2 will continue to rise in coming decades and that it could reach from700 ppm (scenario A1B) to over 900 ppm (scenario A1FI) by 2100. If emissions continue at the current rate, the mean pH in surface sea water would reach 7.7 towards the end of the century (the concentration of hydrogen ions would increase 2.5-fold). It appears that unprecedented levels could be reached by the end of the twenty-first century with respect to the past 40 million years. Analysis of paleo-recordings (δ11B paleo-pH-metre, B/Ca ratio in fossil foraminifera) has revealed that the future decrease in pH would be three times greater than the disturbances recorded during glacial and interglacial cycles over the past 2 million years (Fig. 3.17) and above all, would occur over a much shorter time span (Pelejero et al. 2010).

The largest pH changes would occur in the Arctic, in the vicinity of European marine ecosystems, during this century, from 0.23 (scenario B1) to 0.45 units (scenario A2) (Steinacher et al. 2009). As oceans turn more acidic, the concentration of CO_3 carbonate ions becomes lower. It is said that the saturation state Ω of calcium carbonate $CaCO_3$ decreases. It should be noted that precipitation of $CaCO_3$ is thermodynamically favourable where $\Omega > 1$ (super-saturation) and unfavourable where $\Omega < 1$ (under-saturation). By definition $\Omega = [Ca^{2+}][CO^{2-}]/K^*$, where the equilibrium constant K* is the solubility product. Today, the $[Ca^{2+}]$ concentration, which essentially depends on salinity, is more or less constant in the ocean. The variations in Ω currently observed mostly express rapid changes in $[CO_3]$. Nowadays, surface waters are over-saturated in carbonates ($\Omega > 1$) and this condition promotes the precipitation of calcium carbonate in the form of crystal structures whose solubility increases with higher pressures and lower temperatures. They are, from most to least thermodynamically stable: calcite, aragonite and high-magnesium calcite. Acidification leads to a rise in the “carbonate saturation horizon”, i.e. the depth at which $\Omega = 1$, below which seawater becomes corrosive for $CaCO_3$ (Fig. 3.18). In coming decades, aragonite’s saturation horizon will reach the ocean surface (Hoegh-Guldberg et al. 2007). By the end of the twenty-first century, the surface waters of the Southern Ocean (lat. > 60° S) and of some areas of the North Pacific (lat. > 50° N) will be under-saturated ($\Omega arag < 1$).

[14] http://www.epoca-project.eu/.

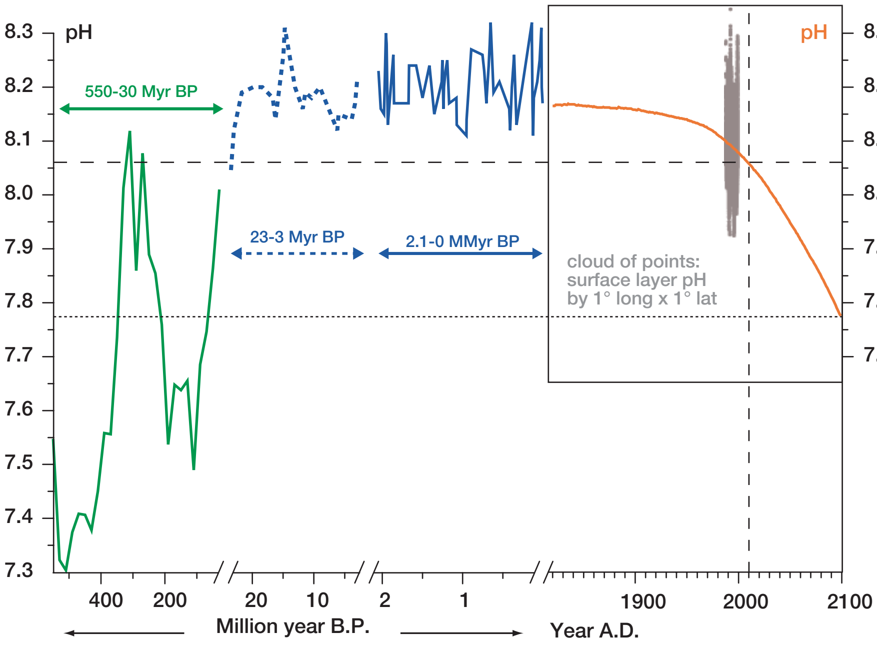

Fig.3.17 Past trends and future changes in ocean pH. By the end of the twenty-first century, ocean pH values would be lower than those estimated for the past 40 million years. *Green curve*: simulated paleo-pH values for the period between 550 Ma and 30 Ma, output from an atmosphere-ocean-sediment carbon cycling model. *Dashed blue curve* and *solid blue line*: reconstructions from two boron isotopes in for aminifera for the period between 23 Ma and 3 Ma and for the last 2.1 Ma respectively. Inset top *right*, *red curve*: mean surface ocean pH from historical data (1800–2000) and simulated up to 2100 using IPCC A2 emission scenario. The *grey* cloud of points shows all 1 × 1 degree mixed surface layer (upper 50 m) pH values in the oceans. (In: Pelejero et al. 2010)

Biological and Ecological Effects

Although the magnitude of ocean acidification can be predicted reasonably well, predicting the ways marine biodiversity may respond to changes in the physical-chemical ocean environment remains a considerable scientific challenge. Indeed, advances in forecasting the impacts of ocean acidification on marine biodiversity require that the main factors of global change simultaneously in play in the real world must be taken into account.

This means integrating the synergies between rise in temperature, CO_2 concentrations, the drop in oxygen concentration, not to mention interactions with other pressures, especially anthropogenic ones, exerted on marine ecosystems (Turley et al. 2010).

Let us first consider the effects that increased pCO_2 will have on marine phytoplankton, which are the “motor” driving the biological pump. Many microalgae use carbon- concentrating mechanisms to compensate for the low affinity of ribulose-1,5-biphosphate carboxylase-oxygenase (RuBisCO) for this substrate (Rein-

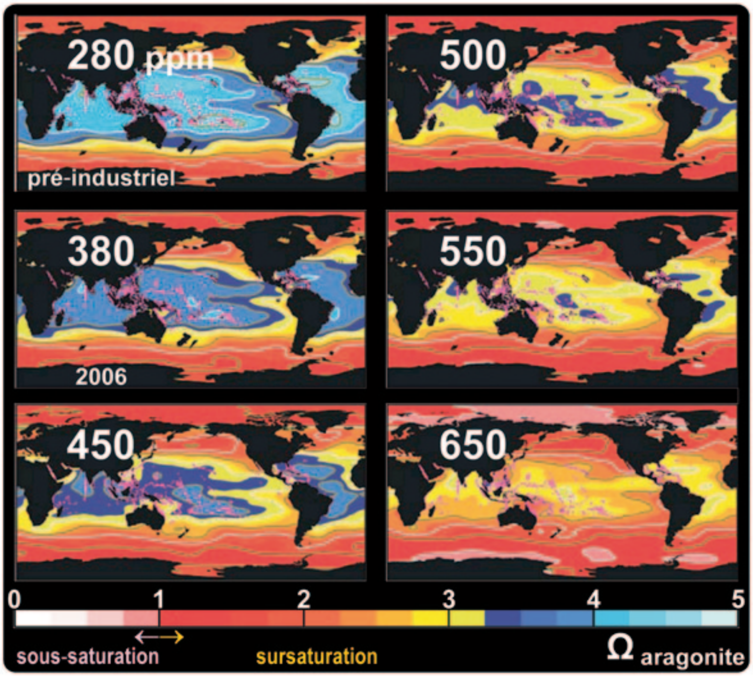

Fig. 3.18 Changes in the aragonite saturation state Ωarag plotted over the world ocean surface for different levels of atmospheric CO_2 concentrations. Maps highlight the dramatic reduction of overall area of warm waters, shown in *blue* (Ωarag > 3.25), where nearly all shallow-water coral reef communities (locations shown as *pink dots*) develop properly today. (From Hoegh-Guldberg et al. 2007)

felder 2011). This is the first enzyme catalysing carboxylation reactions in the process of photosynthesis, an ancient enzyme which evolved under high pCO_2 and low O_2 conditions. Its half-saturation constant, between 20 and 185 µmol/l, limits carbon assimilation in waters where its concentration is low (5–25 µmol/l). Hence, the impact of increasing CO_2 concentrations on photosynthesis in marine diatoms would most likely be limited, or even favourable for the productivity of nitrogen-fixing cyanobacteria (like those in the genus *Trichodesmium*). Of the few coccolithophores which have been studied, some may occasionally benefit from a rise in atmospheric CO_2 (Rost et al. 2008), but the variety of responses observed in a small number of species (*Emiliana huxlei* and *Gephyrocapsa oceanica* in particular) do not seem to be consistent enough as of now for a general pattern to be identified (Barcelos e Ramos et al. 2010; Jiang et al. 2010). Beaufort et al. (2011) attributed this complexity to the assemblage of physiologically differentiated morphotypes whose distribution in the ocean depends on local carbonate chemistry characteristics.

As for metazoans, many studies on marine invertebrates have highlighted the sensitivity of biocalcification to ocean acidification (Doney et al. 2009). Calcification plays various roles, like stabilising body form and function, protecting against predators or—in the case of corals—building reefs and specific habitats. Sea water's degree of saturation in carbonates Ω is often cited as the factor determining the rate of calcification, however bio-mineralisation rarely occurs in contact with seawater. Indeed, contrary to phytoplankton cells, those of marine ectothermic metazoans are surrounded by an extracellular fluid (blood, hemolymph or cœlomic fluid) which is isolated from seawater by various epithelia (gill, gut, nephridia). This body fluid irrigates the organism's cells and ensures convective transport of substances including dissolved gases (O_2, CO_2) associated with respiration. The ionocytes of the epithelial cells hold an ion-exchange system (driven by Na^+/K^+ ATP-ase activity, which maintains—within limits—the acid-base balance of extracellular fluid by regulating the effects of pH variations in the external environment. The Na^+/K^+ pump drives ion exchanges in marine ectotherms. It is responsible for the respectively high and low concentrations of Na^+ and K^+ ions inside branchial epithelium cells. The Na^+ gradient enables H^+ protons to be released into the seawater. These H^+ protons are produced in the gill epithelium when the CO_2 hydration reaction is catalysed by carbonic anhydrase. The reaction produces HCO_3^- ions which are exported to the extracellular fluid. Electroneutrality is maintained by additional exchanges of ions. It should be noted that the way Na^+/K^+ ATP-ase activity is controlled is complex and has not been fully elucidated (Fig. 3.19).

In functional terms, the epithelia which regulate the pH of extracellular fluid in marine ectotherms (fish and invertebrates) react similarly overall to acidification. However, sensitivity to CO_2 varies greatly from one organism to another. CO_2 sensitivity thus may be higher in the lower marine invertebrates (e.g. echinoderms, bryozoans, cnidarians and bivalve molluscs), not because they are calcifiers but because they are sessile, hypometabolic organisms with poor control of their extracellular pH. In contrast, the high-metabolic physiotypes (e.g. teleost fish, cephalopods and brachyuran crustaceans) which need efficient blood O_2 and CO_2 transport mechanisms to support their active behaviour are less sensitive. These organisms must excrete the CO_2 produced by their respiration, above all by diffusion to seawater, and this means that the partial pressure pCO_2 in their blood or hemolymph must be much higher than that of the water. The hypothesis of Rosa and Seibel (2008) can nevertheless be mentioned here: that because of future shrinking of its habitat (acidification, warming, vertical migration in oxygen minimum zone), the jumbo or Humboldt squid (*Dosidicus gigas*, pelagic top predator) may have to reduce the range of its nycthemeral migrations in order to preserve its functional performance. As regards the early development stages (gametes, eggs and larvae) of many marine ectotherms, lack of knowledge about their tolerance to hypercapnea has become a subject of great concern (Melzner et al. 2009; Talmage and Gobler 2010). Overall, fitness differences between competitive species may lead to changes in biodiversity components, and ultimately to restructuring of marine ecosystems. To what extent such changes could affect ecosystem services is presently unknown.

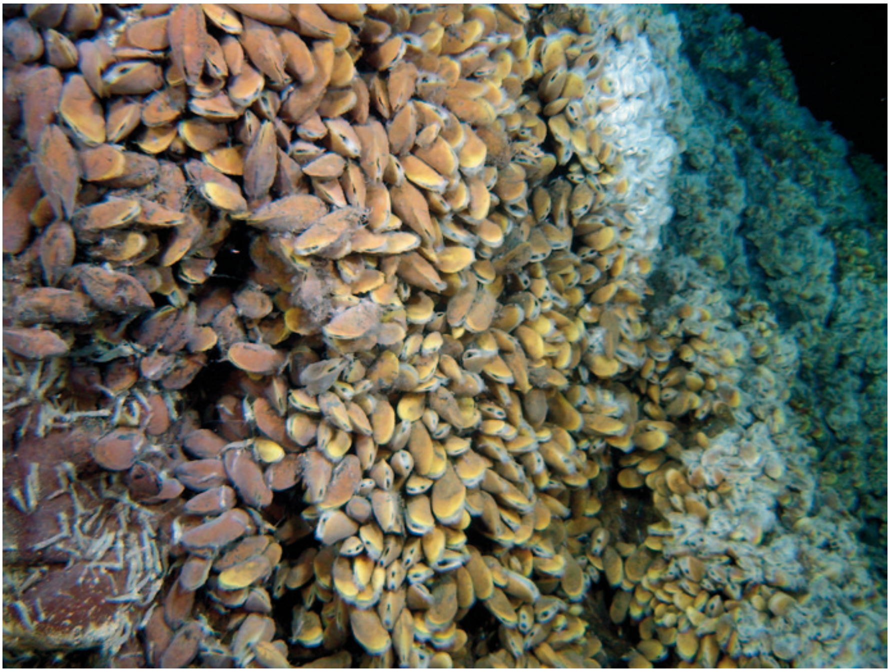

Fig. 3.19 Modiolus, mytilid bivalves similar to mussels. (Photo taken by Victor 6000 during Exomar research cruise on the Mid-Atlantic Ridge. © Ifremer, Victor 6000, Exomar 2005)

Feedbacks—Threshold Phenomena and Tipping Points

Several feedbacks contribute to the complex dynamics of combined warming and acidification of the world ocean. They are the result of physical-chemical processes, generally associated with biological processes (Riebesell et al. 2009; Hofmann 2010).

- Warmer sea surface temperatures (SST) create feedback on atmospheric pCO_2 by weakening the solubility pump; since higher temperature reduces CO_2 uptake, but above all because warming at high latitudes in the Northern hemisphere could slow down the formation of deep waters. Global oceanic carbon uptake could decrease by several dozen gigatonnes by the end of this century, although uncertainty about the Southern Ocean's role as a carbon sink weighs on this estimate.
- The increase in atmospheric pCO_2 lowers the concentration of carbonate CO_3 ions in surface waters. The resulting feedback is lower capacity of excess CO_2 in the ocean. By the end of the twenty-first century the average CO_2 uptake capacity could drop to less than a third of its pre-industrial value, in 1750.
- This means that a tipping point could be reached in oceanic regions where surface waters will become corrosive for carbonates. Model forecasts indicate that in two or three decades from now, the saturation horizon will reach the surface at high latitudes (in the Arctic and Southern Ocean during boreal and austral winters, respectively). This change, amongst others, may impact a small

pelagic mollusc (thecosome pteropod from the *Limacina* genus) whose larval development takes place in part during winter. *Limacina* produce calcium carbonate shells and form dense groups of individuals. They are a key species in marine food webs, particularly in the Southern Ocean, which is one of the reasons for investigating the viability of their populations in acidified waters. The other is their contribution (~ 10 %) to fluxes of $CaCO_3$ exported to the deep ocean (McNeil and Matear 2008; Comeau et al. 2009).

- In tropical environments, the future of coral reefs is a major subject of concern. The shallow habitats built by corals only cover a small part of the ocean's surface area, but they shelter very rich biodiversity and over 450 million citizens in 109 countries live in proximity to coral reefs. Coral reefs are limited to regions surrounded by warm ocean surface waters whose aragonite saturation (Ωarag) is greater than about 3.3. Decreasing Ωarag has led several authors to consider that there will be a general degradation of coral ecosystems when atmospheric pCO_2 reaches 500 ppm (Hoegh-Guldberg et al. 2007). Reef-building organisms have lived—and survived crises—for 540 million years, under higher conditions of temperature and pCO_2, however Ωarag is decreasing today at a rate faster than any previously known. Palumbi et al. (2009) consider however, that projections of global collapse of tropical corals are too simplistic, since there is no general theory integrating the variability of responses due to (1) species' different sensitivity to combinations of factors influencing calcification (Ωarag, temperature, light and nutrients), (2) environmental heterogeneity (kinetics of dissolving reefs, drop in pelagic productivity due to sea surface water stratification), (3) anthropogenic pressures (fisheries, coastal urbanisation and effluents from catchments). In this respect, to ensure that the resilience of reef ecosystems is not altered, Palumbi et al. (2009) recommend that the uses of the numerous services reefs supply be better managed (also see Burke et al. 2011).

Spatial Patterning of Characteristics

Marine biodiversity is unequally distributed in the oceans, being greater in the benthic than the pelagic domains and in the coastal areas than in the open sea; with however a few notable exceptions such as the fauna associated with seamounts and coral reefs. The total surface area of coral reefs covers 600,000 km^2 or 0.2 % of the ocean's surface area, and they provide habitats for some 93,000 species (Bouchet and Cayré, in Cury and Morand 2005).

Seamounts are active or extinct undersea volcanoes with heights exceeding a hundred metres, usually located on oceanic crust. Satellite altimetry which detects those over 1.5 km high has made it possible to inventory about 13,000 seamounts. By using frequency distribution of seamount size, by extrapolation there are an estimated still unmapped 100,000 of them over 1 km high and it can be speculated that there are 25 million which exceed 100 m (Wessel et al. 2010). Only a few thousand of them have been visited and they represent the last major frontier in geographic, geological, and ecological exploration on our planet. Seamounts form a complex and little known system of habitats which can potentially link various populations (or isolate them from each other), stimulate genetic differentiation and structure biodiversity. Early biological prospecting suggested that they were highly endemic, a hypothesis which was difficult to confirm due to insufficient sampling (seamounts can host diversified benthic communities whose species composition is often similar to those on the continental shelf). Seeing the perspectives for the exploitation of mineral and biological resources on seamounts, it is essential to secure more knowledge about the role they play in marine population connectivity (Shank 2010). In addition, seamounts are topographical accidents which interact—in varying ways depending on their own characteristics and those of the local ocean circulation—with the dynamics of hydrodynamic and biological processes (Lavelle and Mohn 2010). Thus they create perturbations which are propagated in the water column and contribute to spatially structuring the pelagic ecosystem (Morato et al. 2010). In order to protect biodiversity, the Josephine Seamount located in the North-East Atlantic off Portugal was added to the international on-line repository that scientifically described "Ecologically or Biologically Significant Marine Areas" in 2012.

Large-Scale Patterns

What large-scale patterns can be seen in the spatial distribution of marine life? This is not a new question, having been addressed as of the mid-nineteenth century by numerous scientists, including Charles R. Darwin (1809–1882), Alfred R. Wallace (1823–1913) and Augustin Pyrame de Candolle (1778–1841). We know that life is neither evenly, nor randomly distributed on Earth, but rather along latitudinal and topographic gradients, with hotspots of biodiversity and endemicity, etc. We also know that fauna in the South Atlantic differs from that in the South Pacific. On a smaller scale (see section on habitats), ecological studies have linked the success of species or communities to physical, chemical or biological drivers. In these cases, the aim is to identify the main processes driving the local distribution of life. But above and beyond local factors, spatial distribution of life also results from the history of clades and that of the Earth's geology. These parameters, combined with local factors, give rise to biogeographical patterns on different nested scales, where, the greater the scale, the stronger the influence of historical contingency will be (Ricklefs 2004). In this context, macroecology entails more than simply updating biogeography (Brown and Maurer 1989; Gaston and Blackburn 2000). Its objective is to model large-scale biogeographic patterns related to biological entities such as populations, species, communities and clades, as well as abundance, life history traits and relations between size and environmental and historical parameters.

Macroecology often relies on statistical tools and geographic information systems to detect these relationships (Chao and Shen 2005; Chao et al. 2005). Because it seeks to identify general mechanisms, it is an appropriate approach to understanding how climate change may affect marine biodiversity. In more ambitious terms, the predictive power of macroecology is needed to tackle the question of what shapes marine biodiversity over time and space. In the years from 2000–2010, the Web of Science returned only 485 references for the key-word "macroecology", and less than 20 % of them pertained to marine areas. This suggests that scientists working in oceanology are unfamiliar with the macroecological approach. For instance, we can ask why the incidence of endemism is stronger in the Southern Ocean than in the Arctic Ocean. Why does biodiversity in the Antarctic seem so much more diverse than that in the Arctic? At this point, we are generally unable to tell if the large-scale distribution of a given marine clade or community is controlled by environmental and physical factors such as water temperature, nature of sediment, depth, or whether it is best explained by historical biogeography. Against the backdrop of global change, it is hard to answer that question, whether for shallow areas, strongly structured by their relationships with the continents, or the theoretically more homogeneous open and deep-sea areas (assumed to be homogeneous, but this is generally due to a lack of knowledge).

Local Patterns (Habitats)

Numerous definitions of the term "habitat" are available in scientific literature. Charles Darwin's (1859) referred to the environment in which a single species lives. However, today is is a broader concept, designating a place where multiple species occur together under similar environmental conditions, meaning that a habitat is characterised by both its faunal and floral species composition and its physical environmental features (e.g. type of seabed, hydrodynamic conditions and the physical-chemical properties of water—temperature, salinity, dissolved oxygen, etc.). Although coastal and shelf habitats are relatively well known (albeit incompletely mapped), much less knowledge is available about deep-sea and pelagic habitats. Achieving progress in this field still depends on large infrastructures to observe the dynamics of marine habitats and communities, i.e. vessels and vehicles (*in situ* measurements, ROV, AUV, gliders, sonars, multibeam echosounders, and so on), satellites (sea colour/biogeochemistry, surface CO_2 fluxes, surface temperature and salinity, wave and current fields), free-drifting floats and instrumented moored buoys, and sea floor observatories.

Habitat Classification

A common first step in marine resource management and protecting biodiversity is to identify and classify habitat types, and this has led to a proliferation of habitat classification systems. Creating a classification system is a difficult challenge due to the complex and shifting array of habitats across various spatial and temporal scales. To meet this challenge, several countries have developed, or are developing, national classification systems and mapping protocols for marine habitats. To be effectively applied by scientists and managers, it is essential that classification systems be comprehensive and that they incorporate the relevant physical, geological, biological characteristics, as well as anthropogenic stressors and pressures. Two main marine habitat hierarchical classifications are currently available:

- The Coastal and Marine Ecological Classification Standard (CMECS), developed in partnership principally by NOAA, NatureServe, the U.S. Environmental Protection Agency and the U.S. Geological Survey. CMECS[15] is a nested, hierarchical framework that applies a uniform set of rules and terminology across multiple habitat scales. It uses a combination of oceanographic (e.g. salinity, temperature), physiographic (e.g. depth, substratum) and biological (e.g. community type) criteria.
- The EUNIS[16] (European University Information System) habitat type classification is a pan-European system to facilitate the collection and harmonised description of data based on various criteria for habitat identification. It covers all types of habitats from natural to artificial, from terrestrial to freshwater and marine. A significant effort has been carried out by Ifremer to adapt the EUNIS classification at local level to specify habitat typology in Brittany (western France) (Bajjouk 2010).

Recently Guarinello et al. (2010) published a critical paper on these top-down classifications and proposed a new multiscale hierarchical classification of habitats.

Habitat Mapping

Although European Seas cover a large surface area (9 million km^2 for continental Europe, including 2 million km^2 for the shelf), as well as the French Exclusive Economic Zone (11 million km^2), knowledge about them remains patchy. Several mapping projects are underway in European countries like Ireland, Italy and Denmark, and others are being prepared (UK and France). The EU EMODNET[17] (European Marine Observation and Data Network) project has set out the basic design principles for habitat mapping: (1) collect data once and share it many times (2) develop standards across disciplines as well as within them (3) process and validate data at different levels (4) provide sustainable financing (5) build on existing efforts where data communities have already organised themselves and (6) accompany data with statements on ownership, accuracy and precision.

[15] http://www.csc.noaa.gov/benthic/cmecs/.

[16] http://www.eunis.org/.

[17] http://www.emodnet-chemistry.eu/portal/portal/.

- The EU MESH[18] project has compiled the first models and seabed habitat maps for North-West Europe within the EUNIS classification system. It has established a framework for mapping marine habitats by developing standard Data Exchange formats and guidelines for habitat mapping, together with a bespoke web-based GIS application to integrate mapping data at an international level. The increased importance of marine habitat mapping is reflected in new EU policy mechanisms, such as the Marine Strategy Framework Directive (MSFD) and the proposed Atlas of the Oceans for maritime strategy.
- The Sextant (Ifremer server) portal aims to collect and distribute a repository of georeferenced data sets on the marine environment. Sextant covers biodiversity, integrated coastal management, fisheries, coastal and deep-sea environments and exploitation of the seabed. The general public can access it via internet (with restricted access for some data) and the system gathers (vectorized or meshed) data produced by both Ifremer and its partners. A data management system creates the data and metadata. Sextant is compliant with the international OGC and ISOTC211 standards.
- SINP-Mer is part of the Nature and landscapes information system to manage biodiversity supported by the French Ministry of ecology, sustainable development, transport and housing (MEDDTL). Ifremer, the Agency for marine protected areas and the National museum of natural history MNHN are working together to develop its marine strand. Its objective is to identify and improve biodiversity inventories and mapping, list endangered species and habitats and provide monitoring methods as decision-making support for the management and protection of biodiversity. As an IT platform, SINP-Mer relies on the interoperability of various information systems (INPN, Sextant, SIH, Quadrige) and provides a shared portal for different users (Fig. 3.3, Databases).

This list is neither complete, nor closed. There will be additions to it, like the new system called the national biodiversity observatory (ONB) set up by the French ministry MEDDTL.

Marine Protected Areas

In the face of increasing pressure from human activities, Marine Protected Areas are now acknowledged as important management tools to conserve and restore natural and cultural resources and ecosystems, in both inshore and offshore areas. For the Convention on Biological Diversity (CBD 2010), protected areas are the cornerstone of any strategy for conserving marine zones (Fig. 3.20 and 3.21).

Networks of Marine Protected Areas are vital in maintaining Good Environmental Status (GES) (MSFD 2011). MPAs make it possible to compare the physical connectivity of marine waters with greater knowledge about genetically isolated species. An extensive, appropriate and representative system of marine protected areas can contribute to protecting all components of biodiversity—including the

[18] http://www.searchmesh.net/.

Fig. 3.20 Crinoids on cold-water coral skeletons. (© Ifremer, Caracole, Daniel Desbruyères)

habitats and key species in some ecosystems—on an eco-regional, i.e. intra- or inter-regional, scale. The areas to be protected are selected according to their natural heritage interest, the importance of the ecological functions they perform (e.g. nurseries, highly productive areas, migratory routes) and the type of human activities occurring there (such as fisheries, aquaculture, tourism and mineral extraction). The overarching aim, i.e. to conserve the functions and resilience of ecosystems, is generally expressed through objectives for the sustainable use of biodiversity. A functional network of marine protected areas, eliminating destructive fishing practices and implementing ecosystem-based management can also significantly contribute to reaching the objective of maintaining or restoring fish stocks, by 2015, at levels enabling their maximum sustainable exploitation. Traditional conservation measures initially designed to be applied on land cannot be directly transposed to the marine environment. That is why the first national strategy for biodiversity (SNB) was accompanied by a "marine action plan" which set out the priorities for the marine environment. MPAs are specifically indicated in this plan, in order to:

- extend the tangible application of the Habitats Directive at sea by implementing the ecological network of special protected areas in the framework of the Natura 2000 Directive;
- establish a new legal instrument in the form of marine nature parks (PNM) to align measures to maintain ecological functions, use made of resources and

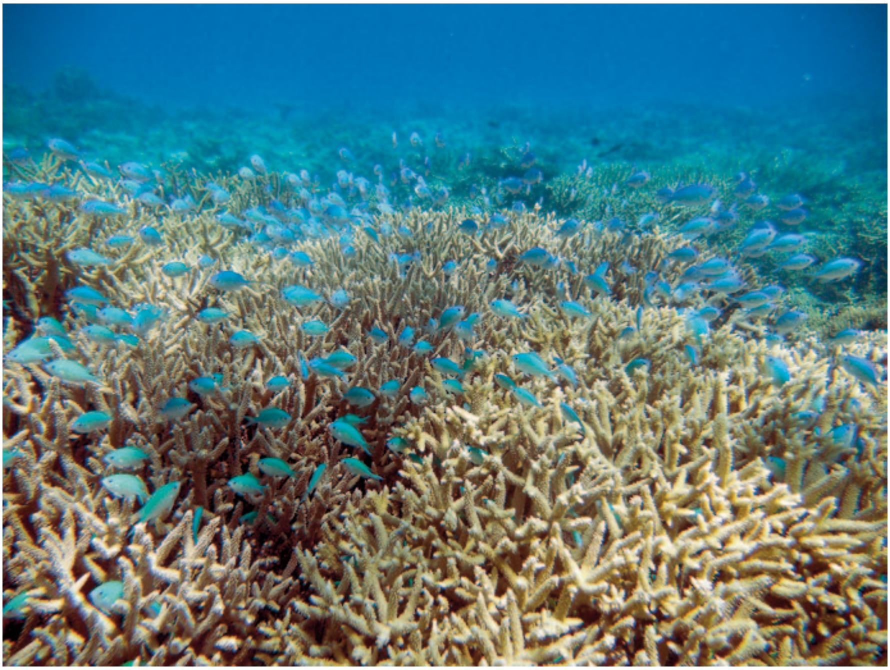

Fig. 3.21 Blue damselfish in branched coral, Ilot Golfield reef, great South Lagoon in New Caledonia. (© Ifremer, Lionel Loubersac)

means of governance. The objective is to create 10 marine parks by the end of 2012, including the four which already exist: Iroise Sea, Mayotte, Gulf of Lion, and Glorieuses islands created in 2007, 2010, 2011 and 2012 respectively. Several other projects are currently being finalised, such as the marine nature park in three estuaries in Picardy (Somme, Authie, Canche), the Pertuis Charentais—Gironde estuary, or in the initial phases of investigation (e.g. Normandy-Brittany Gulf).

The required legislation has been passed. A law on marine nature parks was enacted in April 2006, and then the law on water for Marine Natura 2000 Directive compliance came into force in January 2007. The French law of 14 April 2006 created the national agency for marine protected areas (AAMP[19]), a public institution governed by a board of directors whose members represent Ministries, local authorities and the main partner stakeholders. Its aim is to promote policy developments in the realm of MPAs, both in creating and managing them; managing the financial and human resources devoted to marine nature parks and to supply administrative and technical support to MPA managers.

The national strategy for biodiversity (SNB) was updated in 2011, in particular to meet the targets set at the Nagoya Conference (CDB 2010).

[19] http://www.aires-marines.fr.

That for marine protected areas was revised in November 2011 for the waters of metropolitan and overseas France, through the *Grenelle de la mer* (marine environmental consultation) operational committee framework. The objective was to protect 20 % of French EEZ waters, half of them as fishery reserves. Moreover, the definition of a fishery reserve, as identified in the *Grenelle* multi-stakeholder discussions has also been finalised.

The European maritime strategy also requires that Member States draw up a network of MPAs to help achieve good environmental status (GES) by 2020 at the latest, as specified in the Marine Strategy Framework Directive.

Population Structure and Connectivity

The classical notion that marine environments tend to be demographically "open", and that many species have either high mobility or potential for dispersal during the egg and larval stages coincided with many early genetic studies that typically indicated a lack of genetic differentiation across often even wide geographical scales (Ward et al. 1994). The implication was that most marine fish populations have vast population sizes that would not be subjected to either rapid or stochastic genetic change (Fig. 3.22).

Selection and gene flow were considered to be the predominant evolutionary forces affecting marine species, resulting in expectations that large-sized populations with vast ranges of distribution would most likely exhibit low rates of change. In other words, opportunities for local adaptation would be hindered by intensive migration and extensive habitats. However, several recent studies have challenged this view by demonstrating population subdivisions on scales ranging from tens of kilometres to a few hundred kilometres (Cimmatura et al. 2005; Olsen and Moland 2011). These results are especially useful for determining the boundaries of marine protected areas and for assessing population connectivity, status and dynamics. Furthermore, these patterns can be temporally stable (Cimmaruta et al. 2008). Several hypotheses have been proposed to explain population structuring, including local retention of juvenile stages (Pogson et al. 2001), life history characteristics (Bekkevold et al. 2005), habitat preference, e.g. benthic versus pelagic spawning (Hemmer-Hansen et al. 2007) and environmental gradients or obstacles (Bekkevold et al. 2005; Waples 2002). It is becoming increasingly evident, for example, that despite the potential for high gene flow based on egg and larval dispersal, a combination of oceanographic processes, behaviour and high mortality (Cowen et al. 2006; Gawarkiewicz et al. 2008; Jones et al. 2008) promote retention or loss of life history stages and a finer scale of genetic structuring than once appreciated.

Additionally, recent data now indicate that genetically effective population sizes (Ne) in marine fishes, especially those characterized by high fecundity and high larval mortality, are typically 2–6 orders of magnitude smaller than census sizes (N) in the same populations. (The effective population size Ne designates the number

Fig. 3.22 Yellowfin tuna fisheries. (© Ifremer, Marc Taquet)

of spawners in the so-called "ideal population", i.e. where balance of genotype and allele frequency is maintained, theoretically possessing the same genetic diversity as the census size population *N*). Such discrepancies have profound implications for estimating both quantitative change in population size relative to recruitment and harvesting, and qualitative change, in terms of the nature and speed of genetic change in marine populations. A low Ne/N ratio suggests vulnerability to changes in genetic diversity, patterns of genetic differentiation and responses to environmental change (selection pressures) even in apparently large, commercially exploited populations.

Compared to what we know about terrestrial organisms, our knowledge is less advanced on marine organisms' biology and how they interact with the physical-chemical environment. There are inherent challenges in studying the biology of marine populations. Few marine organisms can be observed *in situ.* Most of the standard biological descriptors like population size, migrations and individual behaviour, generally speaking, are obtained using indirect methods. Despite the abundant literature devoted to connectivity among marine populations (see Selkoe et al. 2008) and recognition that connectivity is variable in seascapes, available information is too patchy to quantitatively estimate demographic exchanges which maintain this connectivity year after year.

Estimations of dispersal and connectivity are generally indirect, and produce contradictory results, e.g. some suggest that dispersal over long distances is common, whereas more recently emerging results suggest the opposite. Moreover, it is hard to distinctly identify the effects of various types of factors which influence connectivity (e.g. oceanographic context and local hydrodynamic features, life history variability, biological phenomena like behaviour and predation, population size), and this situation creates great uncertainty in management scenarios which take demographic and evolving processes into account. In addition to the need for data on the scale of demographic and evolutionary exchanges, the level of connectivity also plays a major role in determining the capacity for local adaptation

For many taxa, it has not been possible to incorporate the scale of population-level and genetically based adaptive variation into estimates of response to environmental change because studies to date have been primarily descriptive (Hauser and Carvalho 2008). Furthermore, even when they exist, investigations have mostly focused on genetic differentiation revealed by neutral markers. This approach provides a framework to assess and analyse marine taxa's ability to adapt to environmental change. However, it will usually not supply empirical information about the type, causes and consequences of local adaptation. Concomitant differences in ecologically important traits now indicate extensive adaptive differentiation and biocomplexity, potentially increasing the resilience to exploitation and disturbance. Two recent examples of connectivity studied through the analysis of complex systems are presented in Figs. 3.23 and 3.24.

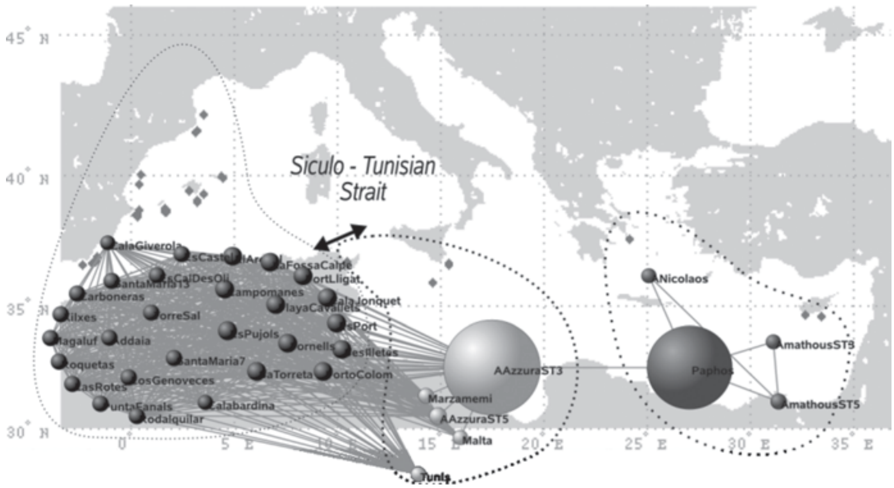

Fig. 3.23 Network of *Posidonia oceanica* populations in the Mediterranean Sea, analysed at the percolation threshold. Node size indicates the level of betweenness-centrality, an indicator used to denote the proportion of shortest paths passing through the represented item—in this case, the gene flow—and maintaining the system's connectivity. (Rozenfeld et al. 2008, © National Academy of Sciences (USA) 2008)

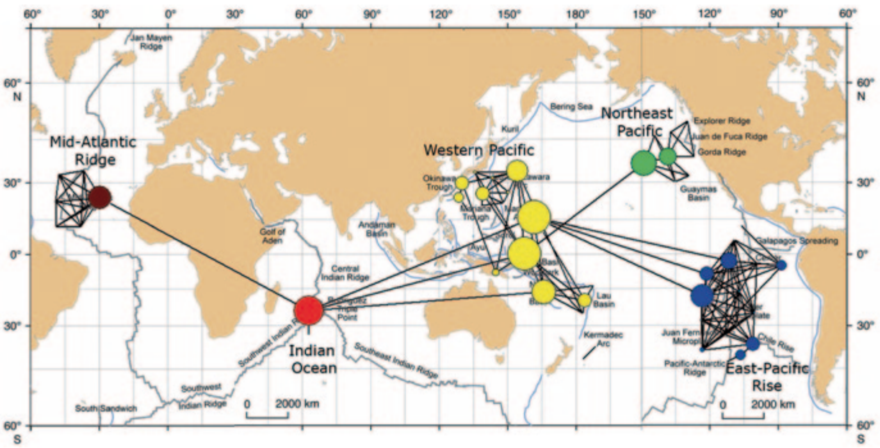

Fig. 3.24 Global biodiversity network of hydrothermal vents created using complex system analysis. Each node in the network represents a hydrothermal vent field. The connectivity of nodes indicates the level of proximity in terms of genera shared by related communities. The clustering index defines the 5 biogeographical regions identified, highlighting a high rate of endemism (colour-coding). The dot size indicates the betweenness-centrality index (see Fig. 3.23) showing how information is relayed by populations. In this case, the past gene flow has led to unique genera and species which are specific to each area. (Moalic et al. in press)

Biological Invasions

Non-Indigenous Species or NIS (also called non-native, exotic, alien, or allochthonous) are species, sub-species or lower taxa introduced outside of their natural (past or present) distributional range or their potential dispersal area.

Invasive Alien Species or IAS are non-indigenous species which have have spread, are spreading or have demonstrated their potential to spread elsewhere. When indigenous species exhibit these invasive characteristics, they are called "invasive" (e.g. green algae). In the regions they invade, these species have negative effects on biodiversity, ecosystem functioning and on human societies through socio-economic and/or public health impacts. Species of unknown origin which cannot be ascribed as being native or alien are termed "cryptogenic species". They also may demonstrate invasive characteristics and should be included in IAS assessments (see Olenin et al. 2009, European Commission—Marine Strategy Framework Directive report, Descriptor n°2). The United Nations Convention on the Law of the Sea (UNCLOS 1982) stipulates in Article 196 that "*States shall take all measures necessary to prevent, reduce and control pollution of the marine environment resulting from [...] the intentional or accidental introduction of species, alien or new, to a particular part of the marine environment, which may cause significant and harmful changes thereto*".

On 22 May 2006, the European Commission published a Communication on biodiversity [COM(2006) 216 final] aimed at "*halting the loss of biodiversity by 2010 and beyond*". One of the objectives is to substantially reduce the impacts of invasive alien species and alien genotypes. Four key supporting measures were proposed for this purpose, the most important being to encourage Member States to draw up national strategies in line with the Action Plan which "*addresses both Community and Member States, specifies the roles of each in relation to each action, and provides a comprehensive plan of priority actions towards specified, time-bound targets. Success will depend on dialogue and partnership between the Commission and Member States and common implementation.*" This EU Action Plan has been supported by international agreements such as the 1979 legally binding Bern Convention on the Conservation of European Wildlife and Native Habitats, which created a Group of Experts on Invasive Alien Species in 1992. Likewise, the 1971 intergovernmental treaty on wetlands of international importance called the Ramsar Convention and the 1979 Bonn Convention on the conservation of migratory species also include resolutions on IAS. And finally, combating invasive species is one of the six key objectives of the new EU 2020 Biodiversity strategy presented by the Commission in May 2011.

Invasive species have a strong impact on marine areas, as reported in the European FP6 DAISIE R&D[20] project (Delivering Alien Invasive Species Inventories for Europe), mostly due to marine environmental characteristics. Compared to the properties of air and soil, those of seawater, like its viscosity and thermal capac-

[20] http://www.europe-aliens.org/.

ity, can promote dispersal over large distances. Marine water dynamics on various scales (ocean circulation, mesoscale eddies or "meddies" and tidal currents) have an influence on this dispersal, especially for pelagic species which can actively follow the displacements of water bodies or be transported by them. Also, benthic species usually have a pelagic phase in their life cycle. Marine species can also be artificially disseminated, e.g. by shipping, in the ballast water of vessels (Leppäkoski et al. 2002). Demographic strategies of species, human interventions and the properties of the marine environment all contribute to making biological invasions more successful.

Marine invasive species can be ranked in three categories according to the way they became invasive: (1) some take advantage of man-made infrastructures to move by themselves through previously separate regions. So-called "Lessepsian" species using the Suez Canal [3] illustrates this. (2) Others are voluntarily introduced for specific purposes. Aquaculture provides a classic example. The majority of species used in mariculture are farmed outside of their natural geographical range of distribution. Amongst numerous examples can be mentioned the Pacific whiteleg shrimp *Litopenaeus vannamei* farmed in the Atlantic, the Japanese *Crassostrea gigas* oyster introduced to Europe where it has become an invasive species and Atlantic salmon *Salmo salar* reared in farms on the Pacific coasts of North and South America. This practice has become generalised, carrying the risk of "biological pollution", due not just to these voluntarily introduced allochthonous species, but also to their diseases and parasites. Biological pollution comprises a genetic component (risk that individuals escaping from farms will breed with wild conspecific populations). (3) However, most introductions are unintentional, as those from shipping (species attached to ship hulls, or most often in ballast waters of ocean-going cargo vessels). The latter represents the most significant vector, mediating marine invaders from a wide range of taxa, especially molluscs and crustaceans (Carlton and Geller 1993).

Often the reality is more complex, since the above-mentioned forms of invasion can be cumulative or successive. A good example of an introduction sequence is supplied by the gastropod mollusc *Crepidula fornicata*, which has invaded French coastal waters in the Channel and the Atlantic. The crepidula, or slipper limpet, first arrived in European waters in the nineteenth century via oyster transfers from North America (mode 2); then in June 1944, they were accidentally transported to beaches in Normandy with the Allies' D-day fleet of landing boats, (mode 3). In the 1970s, the species was re-introduced when American *Crassostrea gigas* cupped oysters were imported to save French oyster farms from collapse (mode 2) and finally, the crepidula's dissemination was further reinforced by massive transfers made regularly between various oyster farming areas along the French coastline (modes 3 and 2).

The numerous invasive species in the sea have as many impacts on marine ecosystems and coastal economies. At the turn of this century, more than 104 non indigenous species established along the French Atlantic coastline were counted, with a dozen of them inducing significant economic side effects (Goulletquer et al. 2002). We know that numerous species with a pelagic larval phase are potentially invasive and that even a slight change in water temperature can open new pathways for biological invasions.

Lessepsian species [3]

Mediterranean biodiversity is characterised by the large—and still growing—number of non-native species. There are over 600 of them, if we take only metazoans and around 1,000 if unicellular organisms and foraminifera are included (Coll et al. 2010). Streftaris et al. (2005) estimated that in recent years a new species has been introduced every 4 or 5 weeks. The opening of the Suez Canal in 1869 and its later enlargements have allowed numerous exchanges of species between the Red Sea and the Mediterranean. The "traffic" is not symmetrical, and the flow of species northward is greater from the former, warmer and more saline (22–34 °C, 42 psu), to the latter (13–31 °C, 39 psu). The so-called Lessepsian species, after de Lesseps who developed the Suez Canal, (Por 1978) have either freely migrated via the canal (mode 1–which does not include the "Herculean" species which have arrived naturally via the Strait of Gilbraltar), or were transported by ships coming through the canal (mode 3). Since 1869, several hundreds of species have became established in the Eastern Mediterranean Sea, where a new "Lessepsian" biogeographic province was defined (nearly three-quarters of fish observed along the coast of Lebanon are Siganus rivulatus rabbitfish which originally come from the Indian Ocean; Bariche et al. 2004). Several polychaetes, molluscs (the first cephalopod species being detected in 2002; Lefkaditou et al. 2009), types of plankton (Belmonte and Potenza 2001) and non-native fish (Mavruk and Avsar 2008) now occupy this new biogeographic province. The immigration process is still continuing, notably with the arrival of rabbitfishes on the Mediterranean coasts of France. Taking just the 664 fish species recorded in the Mediterranean (nearly 80 of them endemic), that means that 127 alien species have established their presence there since the beginning of the twentieth century, 65 having arrived by way of the Suez Canal and 62 via the Strait of Gibraltar (Ben Rais Lasram and Mouillot 2009). Although the endemic fishes' range of distribution has remained stable, that of most non-native species has spread northward by 300 km on average since the 1980s, a trend which suggests that rising sea temperature in the Mediterranean should continue to attract warm-affinity species. Ben Rais Lasram et al. (2010) indicated that the endemic fish assemblages in the Mediterranean will be entirely different from those in the 1980s by the end of the century.

In spite of numerous research projects (see DAISIE mentioned above), no comprehensive list is yet available, and the side-effects and economic impacts of displaced species most likely remain underestimated. Impacts are usually assessed after the fact, revealing the lack of anticipation and risk analysis (see example of *Caulerpa taxifolia* introduced in the Mediterranean) (Fig. 3.25). However, it should also be emphasised that the first sighting of a species may be made long before it becomes invasive following

Fig. 3.25 Illustration of *Caulerpa racemosa* in the Barbados islands, now an invasive species in the Mediterranean Sea. (Taken from Vickers A. 1908, *Phicologia Barbadensis*, plate XLV)

environmental changes (e.g. *Crassostrea gigas* cupped oysters and the oyster drill *Ocinebrellus inornatus* along the Atlantic coast) (Martel et al. 2004).

Thus far, prevention and action plans have not been effective in the marine environment, in spite of attempts towards ballast water regulations (e.g. the Ballast Water Decision Support System in Australia; or the IMO's Ballast Water Management Convention which was adopted in 2004 but still not implemented as of early 2010). Most methods proposed are based on eradicating or/and managing populations, involving operations which are much more costly and less effective than a pro-active approach based upon prevention. Indeed, very few examples of successful eradication of marine invasive species have been reported.

Experiences with the North Pacific sea star *A. amurensis* have shown that once a population is established there is little chance for successful eradication (Parry et al. 2000). *Asterias amurensis* invaded the coasts of Tasmania and Southern Australia. In 2 year's time, 12 million individuals had colonised Port Phillip Bay, near Melbourne—where this predator attacks both the local fauna and farmed shellfish stocks (mussels, king scallops and clams). Physically removing tens of thousands of specimens in attempts to regulate the *A. amurensis* population had no effect (Johnson 1994). In the final analysis, the only option is to adapt to the presence of the invasive species, and devote research to assessing the ecological processes of the impact, evaluating its consequences and sometimes, mitigation. This was the case for *A. amurensis*. Knowledge about this sea star's life cycle in the South Australian environment made it possible to define and choose "ballast windows" for ballasting and deballasting in order to reduce or even eliminate the risk of transporting the species (Byrne 1996). Biological control through parasitic castration was also envisaged to reduce *Asterias* populations.

Temporal Patterns

Temporal characteristics of biodiversity can be described over several scales, i.e. geological, historical and those depending on current anthropogenic pressures.

Geological Scale

Paleobiology records provide information about long-term cycles in the history of marine life, especially previous extinction processes. Long-term historical factors have participated in shaping biodiversity patterns on large scales, i.e. biomes, biota, latitudinal belts and gradients. Without going too far back in time, simply taking the major geodynamic events of the last 34 million years (i.e. Post Oligocene) can shed light on current biodiversity patterns. In terms of the climate, this was the starting point of the icehouse period, with the development of glaciers and polar ice caps and stronger latitudinal thermal gradients (Deconinck 2006).

During these 34 million years (Ma), the events most significantly affecting the marine biosphere were: (1) the closure of the Tethys (30 million years ago, or Myr) related to the Alpine orogeny (event triggering vicariance between the Atlantic and Indian Oceans); (2) concomitant formation of the Mediterranean Sea and consecutive evolution of Paratethys remnants (Aral, Caspian, and Black Seas); (3) separation between Australia and Antarctica (20 Myr), opening of the Drake passage between Tierra del Fuego and Antarctica (16 Myr), and formation of the Antarctic circumpolar current and the south polar front, prefiguring today's thermohaline circulation; (4) mid-Miocene cooling, corresponding to a drop in sea surface temperature by 7 °C; (5) the Messinian salinity crisis (6 Myr), closing of the Strait of Gibraltar and formation of powerful evaporite deposits in the Mediterranean), the uplift and closure (3 Myr) of the isthmus of Panama, entailing local vicariance between the Caribbean Sea and the Eastern central Pacific (Lessios 2008); and (6) Pleistocene glaciations causing eustatic regressions (closure of the Bering strait, connection of Australia with South-East Asia, etc.) and expansion of sea ice.

These paleogeographic reconstructions help explain the current distribution ranges of marine species. For instance, in Euphausiaceans, the genus *Nyctiphanes* comprises only four neritic species which are often predominant in neritic zooplankton communities in the Atlantic (*Nyctiphanes couchii*, *N. capensis*) and Pacific (*N. simplex*, *N. australis*) Oceans. Two of them are exclusively found in the northern hemisphere (*N. couchii* and *N. simplex*) and the other two in the southern hemisphere (*N. capensis* and *N. australis*). D'Amato et al. (2008) used molecular techniques to identify their likely phylogenetic relationships and suggested a chronological series of steps which may have led to their contemporary distribution range. The separation between the lineage that Euphausiacea in the *Meganyctiphanes*, *Nematoscelis* and *Thysanoessa* genera descend from, on the one hand and the lineage of the *Nyctiphanes* genus on the other, would have occurred 20–35 Myr. This would have been about the time of the onset of the Antarctic convergence which led to favourable conditions for the neritic productivity zones in the South Pacific where the *Nyctiphanes* genus may have originated. The hypothesis is in agreement with the fact that *N. simplex* is the oldest species (−4 to −13 Ma) in the *Nyctiphanes* genus. During the Pliocene-Pleistocene transition (2.6 Myr), *N. simplex* would have dispersed in the North-East Pacific, just like a number of other taxa. The *N. australis* lineage would have appeared between −3 and −9 Ma, after closure of the Thetys was completed during the Miocene and the uplift of the Indonesian archipelago, events modifying ocean circulation and regional climates. This lineage would then have diversified in the Indo-Pacific and western Indian Ocean, *N. australis* thus giving rise (from 1 to 5 Myr) to the sister species *N. capensis* (currently endemic to the Benguela current, which came into being about 10 Myr) and *N. couchii*. The latter was most likely dispersed in the North Atlantic due to a glacial episode.

The legacy of this history is that today, even for many taxa with relatively high dispersal capacity, regional species pools are broadly split between the main ocean basins in the northern and southern hemispheres with the warm equatorial waters forming a natural barrier to the movement of temperate and cold-water species from northern to southern hemispheres and land masses limiting dispersal to the east and west. Besides physical barriers, the main natural drivers that contributed to shaping marine biodiversity are depth (resulting from tectonic and climatic processes) and temperature (in covariance with energy input to the system and O_2 concentration).

Historical Scale

The international Census of Marine Life programme (CoML[21], 2000–2010) developed a project named "History of Marine Animal Populations" (HMAP) whose approach used the study of historic records (mostly dealing with human activities exploiting living resources of the ocean) to better understand past and present interactions between people and marine life.

Specific focus was put on the global ecological impacts of fisheries (long-term quantitative and qualitative changes in fisheries stocks), the ecological impact of large-scale harvesting by humans, and the historic role of marine resource utilisation in the development of human societies (Schrope 2006). The HMAP project brought together a multidisciplinary team of ecologists, marine biologists, historians, anthropologists, archaeologists, paleoecologists and paleo-oceanographers, who analysed data from a variety of unique sources, such as colonial fisheries and monastic records, modern fisheries statistics, ship's logs, tax documents, sediment cores and other environmental records. The objective is to reconstruct the events which have contributed to changes in specific marine populations throughout history.

Seven case studies focused on a few species of commercial importance and biodiversity changes:

- Gulf of Maine, Newfoundland-Grand Banks, Greenland cod fisheries;
- South-East Australian Shelf and Slope fisheries, New Zealand Shelf fisheries;
- Russian and Norwegian herring, salmon and cod fisheries, and Atlantic walrus hunting;
- South-West African clupeid fisheries (Agulhas and Benguela currents);
- multinational cod, herring and plaice fisheries in Norwegian, North and Baltic Seas;
- worldwide whaling
- impact of the removal of large predators in the Caribbean Sea.

The list of HMAP publications (books, special issue of *Fisheries Research* journal) can be consulted on-line[22].

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[21] http://www.coml.org/.

[22] http://hmapcoml.org/publications/.

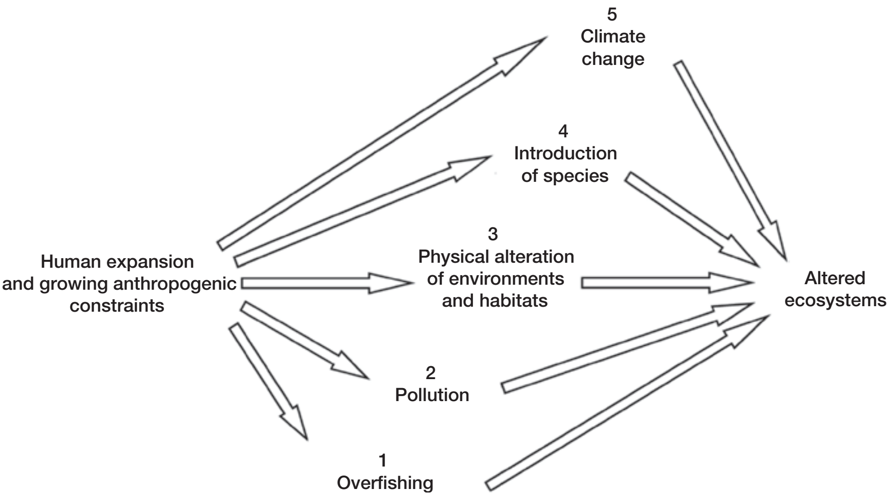

Fig. 3.26 Sequence of human disturbances affecting four main types of coastal ecosystems (*Macrocystis* kelp forests, coral reefs, tropical and subtropical marine phanerogam meadows, eutrophicated coastal environments). Subsequent steps 2 through 5 have not been observed in every example and may vary in order but fishing always preceded other human disturbance in all cases examined. (From Jackson et al. 2001)

Cascading Effects

Historically speaking, fisheries were the first anthropogenic influence to alter coastal marine ecosystems. They came before pollution, physical deterioration and destruction of habitats, introduction of alien species and climate change (Fig. 3.26). That is the conclusion drawn by Jackson and 17 co-authors in a retrospective analysis of trends for four main types of ecosystems (*Macrocystis* kelp forests, coral reefs, tropical and subtropical phanerogam meadows and eutrophic coastal environments) published in 2001. Modifications to the structure and functions of ecosystems occurred as early as the aboriginal and early colonial period, although the process has since grown faster and more diverse and is of greater magnitude. Early pressures increased the sensitivity of coastal marine ecosystems to subsequent disturbance. They prepared the ground for the abrupt changes we are witnessing today (population collapse, habitat loss) as well as a general deterioration of marine ecosystem services. Referring to the five mass extinctions which have occurred on Earth in the past 540 million years, Jackson (2008) concluded that without global measures to regulate the uses of biodiversity, these synergistic effects could lead to a sixth, comparably great, “Anthropocene” mass extinction.

Overall, the abundance of fished resources is dropping and their recovery potential decreases exponentially with declining diversity (Worm et al. 2006). Confronted with the natural and anthropogenic disturbances which accompany global change, ecosystems are becoming more vulnerable. Using paleoecological, archaeological and historical data, Jackson et al. (2001) explored changes in the structure and functioning of coral reefs, estuarine and coastal ecosystems over the past few centuries. Several species of marine mammals (e.g. whales, seals, dugongs, manatees and sea otters) or large reptiles and fish (crocodiles, marine turtles, sailfish and sharks) are now functionally extinct or have become rare in many areas, which also contributes to the degradation of ecosystems.

In the late 1990s, Pauly et al. (1998) observed that in most large ocean regions, the mean trophic level of fisheries catches had dropped regularly since the 1950s, a trend which was interpreted as revealing changes in the structure of marine food webs. According to these authors, the world development of fisheries has tended to first eliminate individuals from large, slow-growing and generally late-maturing species. This phenomenon is called "fishing down marine food webs". It is typical in the North Atlantic and has led to a decrease in the mean trophic level of fished communities. The analysis largely contributed to disseminate acknowledgement that fisheries disturb the structure of communities, and to the adoption of the Mean Trophic Index based on the mean trophic level of catches (Catch MTL) as an indicator of marine food web integrity and diversity by the Convention on Biological Diversity, amongst others. The indicator is calculated on the basis of declared landings, for which abundant data and global geographical coverage is available. Of course, the reality is much more complicated, if nothing else, due to the difference between reported catches and actual catches (see box [1]). Moreover, fishing down the food web is not the only reason for the drop in the mean trophic level of catches, which is also the result of the spatial expansion of fisheries targeting low trophic level species, like small pelagic fishes which feed on plankton (*Fishing through marine food webs*, Essington et al. 2006). In re-examining the way the Catch MTL indicator has been interpreted, using various approaches (simulations, new data—some of them not fisheries-related) Branch et al. (2010) concluded rather that fisheries are currently exploiting all trophic levels, leading to general scarcity, more pronounced nonetheless for populations of high trophic level fish species.

The decline in large predators (Myers and Worm 2003) loosens the top-down control that these predators exert on food webs, and can thus trigger "trophic cascades" (Casini et al. 2009), like that described by Myers et al. (2007). On the Atlantic coast of the United States, 35 years of observations have shown that populations of great sharks have collapsed and that the abundance of their prey (dogfish, skates and ray—especially the cownose ray *Rhinoptera bonasus*), which have decimated a population of bay scallops that had been fished for a century in North Carolina, has increased by an order of magnitude. Figure 3.27 gives a simplified view of trophic level response to less predation by top predators (Cury et al. 2003). The decreasing size of the top predator populations leads to reduced predation on prey, which in turn leads to an increase in abundance of the prey fish (particularly plankton-feeders). This is followed by greater predation on zooplankton, thus creating favourable conditions for phytoplankton to develop.

Amongst the consequences of a trophic cascade is increased abundance of species which are small in size, on a low trophic level and have a high turnover rate

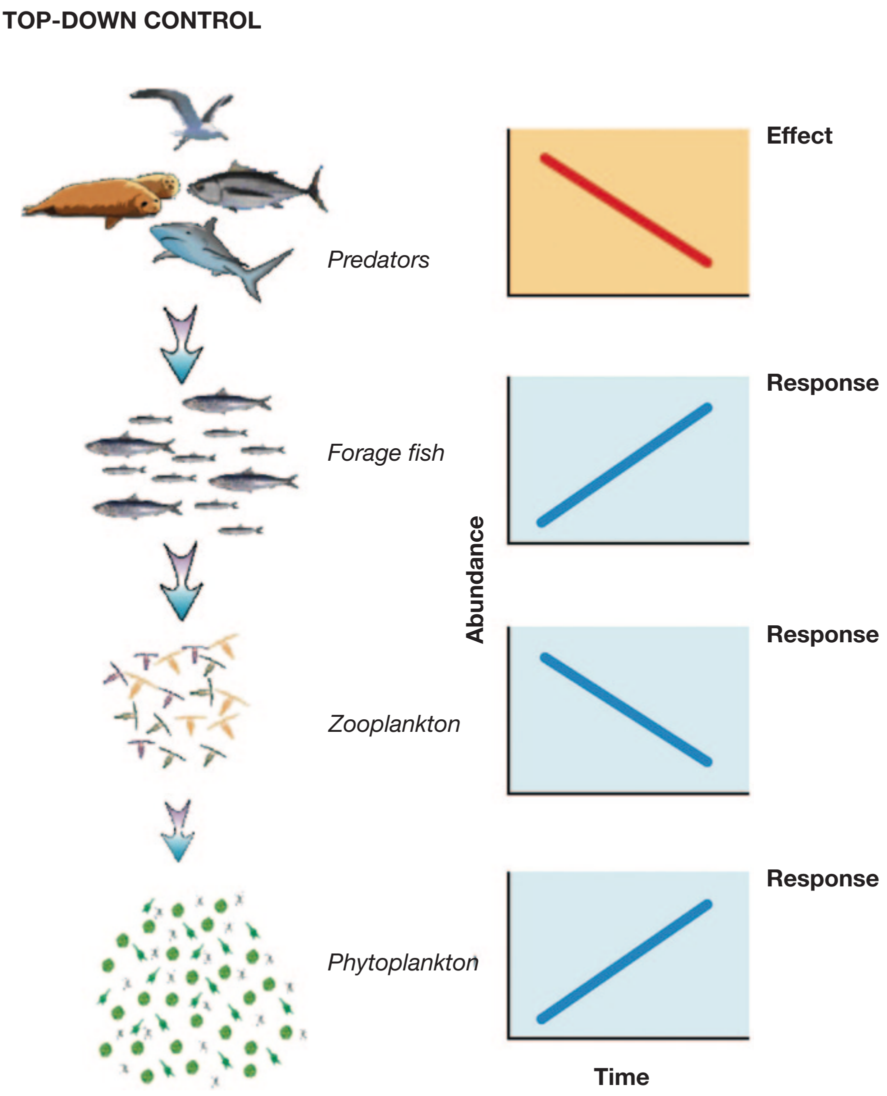

Fig. 3.27 Example of top-down control (effect of decreasing abundance of predators, *red curve*). Simplified presentation of responses—*blue curves*—of 3 levels of a marine ecosystem. (From Cury et al. 2003)

(e.g. small pelagic fish, shrimp and octopus). In Western Africa, the population boom of octopus has been ascribed to groupers being overfished. In genetic terms, these fisheries-induced disturbances combine with other factors, particularly climate influences. On the Atlantic coast of Canada, the collapse of cod stocks, which led to the fishing moratorium in 1993, has enabled herring, capelin and prawn populations to grow. These changes in communities, combined with changes in meteorological and ocean conditions, have been investigated in numerous studies which are still furthering our understanding of how the trophic cascade set off by the collapse of cod, particularly on the coasts of Nova Scotia, has evolved (see, *inter alia*, Bundy and Fanning 2005; Greene and Pershing 2007; Fisher et al. 2008; Strong and Frank 2010). After the moratorium, there was a boom in the abundance of small pelagics and macro-invertebrates, which was followed by a fresh increase in abundance of benthic fish populations from the years 2005–2006 on (Frank et al. 2011).

These sorts of modifications in marine community structure also have socio-economic repercussions. For instance, the Canadian moratorium on cod fishing led to the loss of 20,000 jobs in Newfoundland, without the possibility of switching to crustacean fisheries because of restricted access. Likewise in the North-East Atlantic, the rise in the proportion of low commercial value species landed has contributed to a lasting drop in turnover for French fishing fleets (Steinmetz et al. 2008).

For a long time, the Black Sea was considered to have "good environmental status", with various marine predators topping the food webs. At the end of the twentieth century, it was subjected to anthropogenic impacts due to overfishing, eutrophication and biological invasions. The drop in abundance of top predators combined with the massive blooms of the comb jelly *Mnemiopsis leidyi* (invasive pelagic ctenophore) caused a trophic cascade and depletion of the ecosystem's productivity (Daskalov et al. 2007). Above and beyond the damage to fisheries, this has more generally resulted in the degradation of ecosystem services supplied by marine biodiversity (not just provisioning and regulating services but cultural ones as well).

Thus biodiversity functions are vital in stabilising marine food webs. The rapid changes currently observed are a reminder of how important protection measures are, particularly those to protect top predators for the stabilising role they play in the overall productivity of marine ecosystems.

Fisheries Trends—Other Uses of Marine Ecosystems

In 2009, 57 % of world fisheries stocks were fully exploited, 30 % were overexploited (*vs.* 10 % in 1974) and 13 % were not fully exploited (*vs.* 40 % in 1974) (FAO 2011). Moreover, the proportion of large demersal fishes in catches has decreased from 23 to 10 % of total catch since 1950, meaning that lower-priced species have become more predominant in landings (Steintmetz et al. 2008). The increase in the proportion of exploited fish stocks over more than three decades is indicative of the pressures exerted on ecosystems. Along with many others, this indicator supported the recent observation that the Target set by governments from all over the world in 2002, to "*achieve by 2010 a significant reduction of the current rate of biodiversity loss at global, regional and national level*" has not been reached (Convention on Biological Diversity 2009). Today, 63 % of assessed fish stocks worldwide still

Table 3.4 Pressures that can influence biodiversity. (Based on Table 3.1, EC 2008)

Physical loss	Smothering and sealing
Physical damage	Siltation, abrasion and extraction
Physical disturbance	Underwater noise, marine litter
Interference with hydrological processes	Changes in thermal and salinity regimes
Contamination by hazardous substances	Synthetic compounds, non-synthetic compounds, radioactivity
Nutrient and organic matter enrichment	Eutrophication, hypoxia, etc.
Biological disturbance	Pathogens, introductions of non-indigenous species, extraction

require rebuilding (Worm et al. 2009). Despite the crises and conflicts associated with these evolutions and the efforts made to regulate the sector's activity, today's fishing capacities considerably exceed the resources' potential (Gros 2010); the capacity of the world fisheries fleet was tripled in two decades' time (1980–2000), whereas official landings for the same period were only multiplied by 1.3 (see Garcia and de Leiva Moreno 2001). This is also true in Europe where the EU fleet's capacity is two to three times the sustainable level of available stocks, in spite of reductions in the number of vessels and jobs in the sector since the 1940s (European Commission 2009, 2011). This raises the question of how viable the exploitation systems are, both in terms of resources and the human communities which depend on them and of the capacity of marine ecosystems to withstand the current level of fishing effort (Beddington et al. 2007; World Bank and FAO 2008); also see text box [1].

These issues are all the more urgent in that there are significant social stakes for fisheries and aquaculture (part of the latter's production being dependent on capture fisheries). Today, for 3 billion humans, at least 15 % of their average animal protein intake comes from fish, and this percentage can reach or sometimes exceed 50 % in several countries in the Far East or in Western Africa. FAO statistics indicate that in 2010 some 128 million t (Mt) of fish products were utilised as food for people, with 68 Mt from capture fisheries and 60 Mt from aquaculture. The global consumption of fish products has doubled since the beginning of the 1970s because of major trends like population growth, rising incomes and developing urban centres (Delgado et al. 2003). In addition, according to FAO, 170 million jobs in the fisheries-aquaculture sector—including ancillary activities in value chains—support the livelihoods of 660–820 million people, or about 10–12 % of the world's population in 2010).

Along with capture or farmed fish production systems, many other activities (industries exploiting marine energy and mineral resources, shipping, coastal urbanisation, etc., as well as waste and discharges from industry, farming and other activities on land) put additional pressure on marine ecosystems (as shown in Table 3.4). These stressors have both direct and indirect impacts on marine biodiversity and habitats, and especially on water quality (e.g. chemical and microbiological contamination).

Overall, there is a recognised lack of basic monitoring and statistics for human uses of marine ecosystem services, and the associated benefits derived by society. This limits the analytical scope of changes in uses and their consequences on biodiversity status and human well-being. In some cases, no observations, apart from purely anecdotal evidence, have been collected. And when data is available, via national statistics for instance, the sampling scale used is often not appropriate for the issues at hand. Hence it remains quite difficult to determine the links between changes in uses, modifications to ecosystem attributes and resulting changes in well-being. Other stumbling blocks involve the sector-based aspect of information (e.g. by theme or geographic area) or lack of access to existing data (Fig. 3.28).

Dedicated Time Series

Societal concerns over the potential impacts of climate change have given rise to renewed interest in long-term monitoring of large marine ecosystems. Ecologists are working hard to either maintain or create monitoring facilities and systems to secure the data needed to validate predictive models.

Marine ecosystem dynamics are influenced by numerous processes and physical-chemical properties of the ocean-atmosphere system (sea surface temperature or SST, horizontal and vertical transport, stratification, storm intensity, etc.) which fluctuate over several temporal and spatial scales. Space and time form convenient dimensions upon which to classify natural phenomena, ranging from the diurnal migration of zooplankton to the seasonal impact of hurricanes, the climatic shifts of the Quaternary to the mass extinctions recorded in geological records (Edwards et al. 2010). Yet surprisingly few long-term marine ecology time series are available. The best known are those from the continuous plankton recorder (CPR[23]) surveys running since 1931 in the North Atlantic and performed since 1949 in the East Pacific by the California Cooperative Oceanic Fisheries Investigations (CalCOFI[24]) programme. Today the CPR is operated by the Sir Alister Hardy Foundation for Ocean Science (SAHFOS) and has expanded into the North Pacific and the Southern Ocean, to the waters of Australia and South Africa. CalCOFI is operated by a partnership of state-wide (California Department of Fish and Game) and federal (NOAA-NMFS) agencies and by the research institute Scripps Institution of Oceanography. It is of interest to note that these two monitoring systems, which systematically sample phyto- and zooplankton in surface waters, were originally set up to help elucidate the causes of variability in the abundance of fish populations exploited by fisheries, whereas today their primary purpose is to monitor the health status of large marine ecosystems. All

[23] http://www.sahfos.ac.uk/.

[24] http://calcofi.org/.

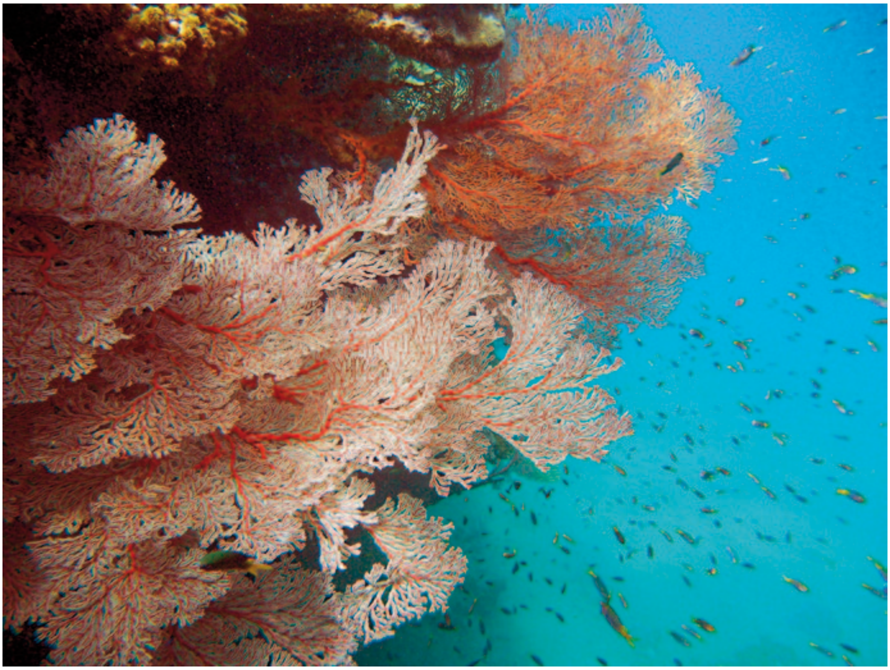

Fig. 3.28 Gorgonians, drop-off of fringing reef in Port-Bouquet. Borindi area on the east coast of New Caledonia. (© Lionel Loubersac)

the sampling and analytical methods are designed to maintain the consistency of the time series. By the end of the last century, more than 900 papers using CPR data had been published (Reid et al. 2003). The CPR system has now become an integral component of the Global Ocean Observation System (GOOS). A final example is that of the REPHY network monitoring phytoplankton and phycotoxins, run by Ifremer since 1984 in French inshore waters (370 stations) (Gailhard et al. 2002)[25] (Fig. 3.29).

As mentioned above, variations in abundance of fish stocks used in capture fisheries were often the reason that long-systematic monitoring was set up to observe fluctuations in marine communities and their environment (Fig. 3.30). In Europe, great impetus was given by the pioneering fisheries scientist Johan Hjort (1869–1946), who promoted the idea that recruitment in natural populations of bony fish mainly depended on stochastic processes (Hjort 1914). Along with the fact that this vision has been borne out by a very large number of scientific studies, it has also contributed to the later guidance for fisheries management oriented towards the ecosystem approach to fisheries or EAF adopted in by FAO in 2001 (Reykjavik Declaration) and included in the Common Fisheries Policy (CFP) reform in 2002. This means that fisheries management is no longer limited to the stock/fleet pair

[25] http://envlit.ifremer.fr/surveillance/phytoplancton_phycotoxines/presentation.

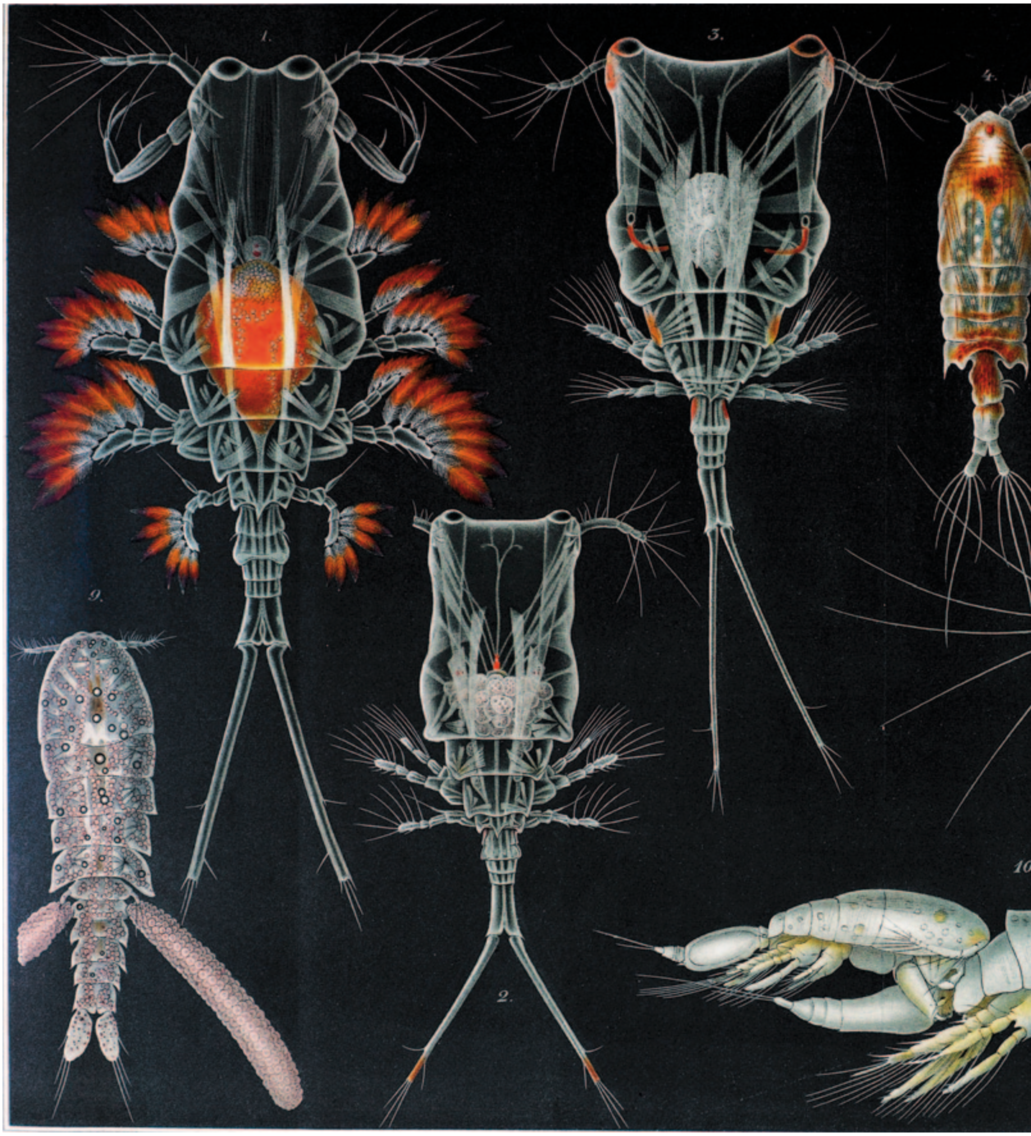

Fig. 3.29 Illustration of copepods. (Taken from Giesbrecht W., *Systematik und faunistik der pelagischen copepoden des golfes von Neapel*, 1892, vol. 2)

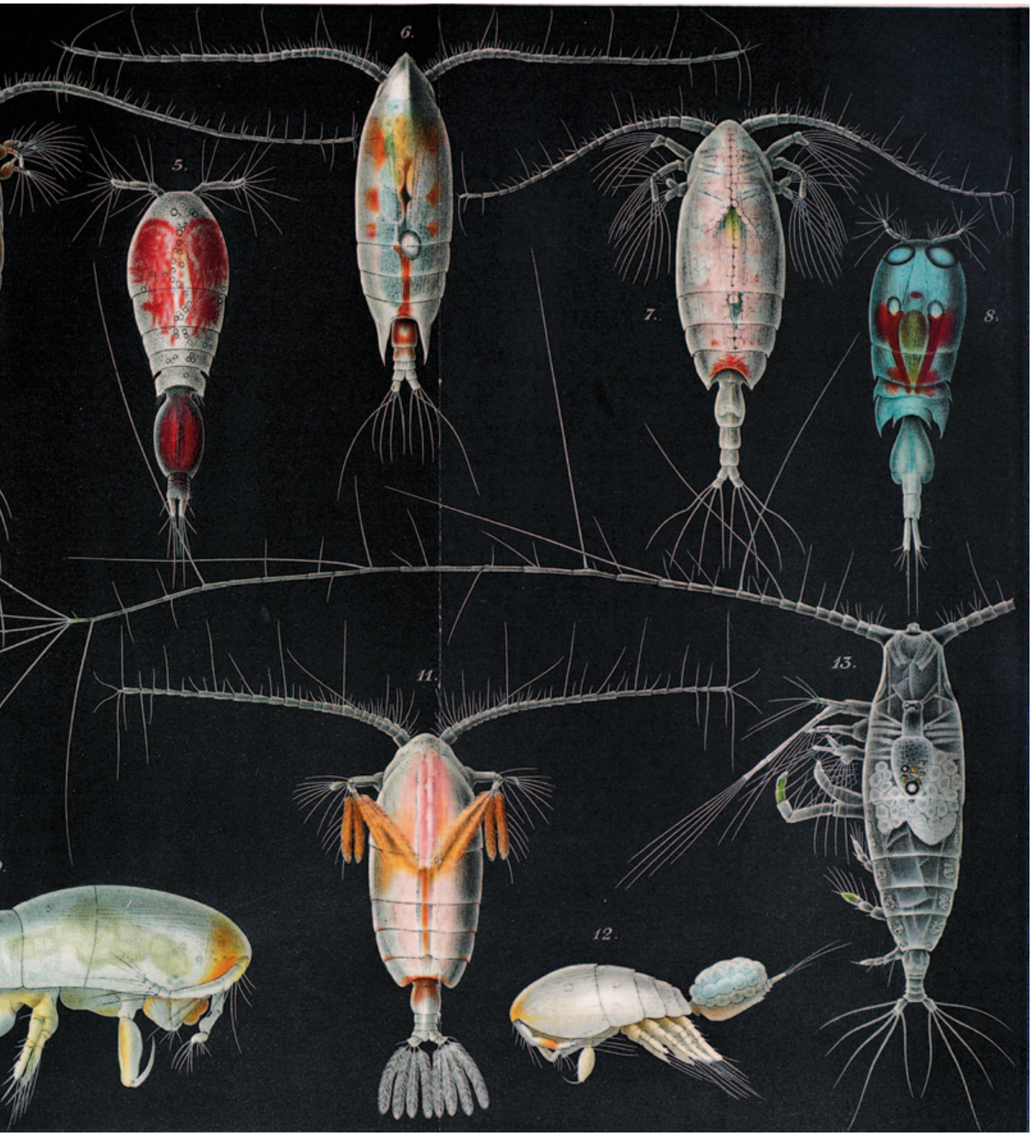
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Fig. 3.30 Taken from Duhamel du Monceau and La Mare 1769, *Traité général des pêches*, Sect. 2, Chap. III, plate XIV

alone (although abundant catch data is available for the main ones). It must now incorporate ecosystem-related information (including the Catch MTL indicator already mentioned), as well as taking account of the future effects of climate change, as has been done on a European scale by ICES (Tasker 2008; Rijnsdorp et al. 2010; Drinkwater et al. 2010) and globally by the FAO Fisheries and Aquaculture Department (Cochrane et al. 2009).

For non-pelagic ecosystems, knowledge is patchy on scales spanning a few decades. The areas sampled are much smaller and benthic time-series usually cover local or regional scope. In Europe, two "benthic" series of over 30 years can be mentioned.

The first is in the North Sea off the Northumberland coast (Frid et al. 2009). The second is called the Pierre Noire site, located in the bay of Morlaix (Western English Channel), where a shallow water, silty fine-sand benthic community has been sampled since 1977 (Ibañez and Dauvin 1988). In France, the REBENT[26] monitoring network is focusing on several biocenoses (e.g. intertidal seagrass meadows, maerl beds, honeycomb worm reefs and rocky habitats) to detect and characterise qualitative and quantitative changes in communities and to map habitats. The network is in the pilot stage on the coasts of Brittany, with the aim of developing nationwide.

Historically, the deep ocean was considered a relatively stable environment which was better protected from climate and other influences than terrestrial and marine coastal ecosystems. It is only recently that long-term trends have been detected at a number of deep-sea sites (Billett et al. 2001; Smith et al. 2006). Muddy, deep-sea sediments represent the most widespread habitat on the Earth's solid surface, occupying approximately 96 % of the ocean floor (Glover and Smith 2003). With an average ocean depth of 3,800 m, they are also one of the least accessible. For example, the Porcupine Abyssal Plain (PAP) site lies at a depth of 4,850 m in the North-East Atlantic and has been sampled in various ways and varying intervals over the past 15 years. The most complete datasets are for invertebrate megafauna (1989–2005) and fish (1977–1989 and 1997–2002) collected using bottom-trawls. From 1989 to 2002, there was a three-fold increase in megafaunal abundance along with major changes in species composition. These time series therefore suggest that decadal-scale changes have occurred among shallow-infaunal foraminifera at the PAP site, more or less coincident with megafauna changes, and that indications of shorter-term events are related to seasonally-pulsed phytodetrital inputs.

Although the deep-sea chemosynthetic ecosystems of hydrothermal vents and seeps are driven by quite different physical and chemical processes from those in "classic" sedimentary habitats, their fauna share a relatively recent evolutionary origin (Little and Vrijenhoek 2003) mostly likely due to the similar physiological constraints organisms have to cope with there. We are starting to better understand how geology dynamically influences ecology and biology over decadal scales (Sarrazin et al. 1997). Time-series have often been launched on ocean research

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[26] http://www.rebent.org/.

cruises performed following major eruptions (e.g. the famous 1991 eruption at 9° N) on the East Pacific Ridge; Shank et al. 1998) or by merging data from a number of separate cruises. Twelve hydrothermal vent sites were identified as having useful long-term data (to date, the best documented hydrothermal fields are 9° N/EPR and Lucky Strike/MAR). (See Figs. 3.4 and 3.24). For cold seeps, repeated visits to the same sites are scarce (mainly focused on Haakon Mosby Mud Volcano in the Arctic sector of the Scandinavian Margin), or have not yet been published. Moreover, there is a paucity of information about cold seep zones, which are targets for future mining prospection. For chemosynthetic ecosystems of biogenic origin (e.g. whale-falls), time-series data are available for sites in the North-East Pacific and fjords on the Swedish coast.

Chapter 4
Conceptualising Biodiversity

Conceptual Frameworks for Relationships Between Biodiversity and Human Societies

Human and environmental pressures modify biodiversity features, leading managers to develop new ways of controlling or compensating for these pressures to meet objectives for sustainable use and conservation.

Management methods are limited to timescales of years to decades to modify human pressures and management decisions must be informed, as much as possible, on environmental stressors and their main interactions with anthropogenic stressors. One conceptual model that can be used to characterise these relationships, incorporating human pressures, the state of biodiversity and environmental response to management, is the PSR framework originally proposed by the OECD (1997)

The pressure-state-response (PSR) framework is based on the recognition that human activities exert pressures on the environment that result in a change in state. If the change in state is inconsistent with the objectives of society, then society makes a response using environmental, economic or social policies and management interventions that are intended to prevent, reduce or mitigate the pressures and achieve desirable states. Implementing the PSR framework requires a knowledge base to draw correlations between pressure and state, so that the state can be monitored with respect to management objectives and to predict how a management response will modify pressure and hence state (Figs. 4.1 and 4.2).

The "core set of indicators" based on PSR were developed by the Organisation for Economic Cooperation and Development in the 1990s (OECD 1994; Lehtonen 2002). In the field of interaction indicators, the PSR indicators play a fundamental role. They have given rise to the European Environment Agency's DPSIR indicators (EEA 2003), the Commission for Sustainable Development's DSR indicators (CSD 2001) and the Convention on Biological Diversity's use of Pressure-State-Use-Response-Capacity (UNEP 2003). These indicators provide a frame of reference to illustrate the interactions between nature and human societies. PSR indicators effectively provide a simple tool to empirically frame the environmental issues—anthropogenic pressures on the environment and policy responses to implement solutions (Fig. 4.1). However, since the OECD's first report devoted to PSR was

P. Goulletquer et al., *Biodiversity in the Marine Environment*,
DOI 10.1007/978-94-017-8566-2_4, © Éditions Quæ, 2014

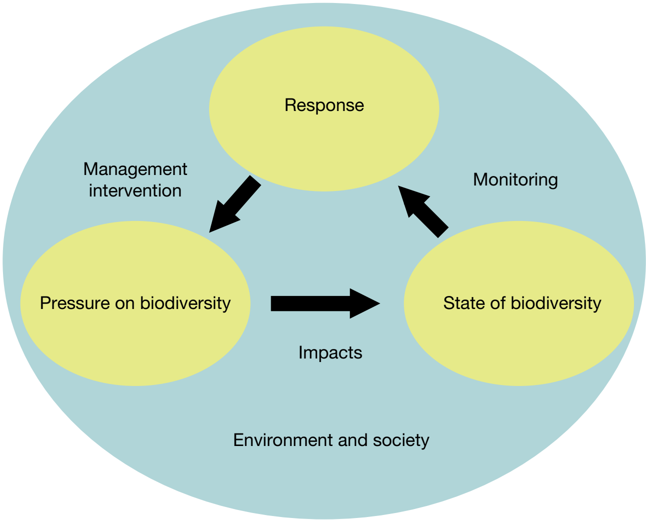

Fig. 4.1 The relationships between pressure, state and esponse. The state of biodiversity is a consequence of ecological and evolutionary processes and human and environmental pressures. By monitoring biodiversity status, progress towards the management management objectives can be assessed. The response typically involves management intervention, chosen to modify the pressures and ultimately, the state

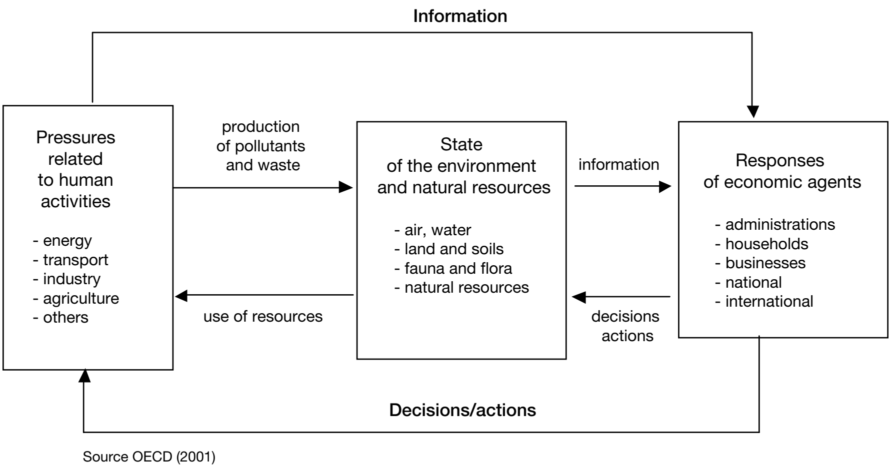

Fig. 4.2 Structure of PSR model and related data. (OECD 2004; from Levrel et al. 2009)

published, their theoretical limits have been highlighted in several critical analyses (OECD 1994; Hukkinen 2003; Wolfslehner and Vacik 2008), particularly those related to identifying links of causality between the three categories of Pressure, State and Response, and especially complex interactions between ecology and societies.

The relationships between PSR are not necessarily consistent over time, owing to changes in the environment and economic and social drivers underlying the pressures (Levrel et al. 2009). Although the PSR framework is mentioned here, more complex frameworks have been developed, such as DPSIR (Drivers, Pressures, State, Impact, Response). In the DPSIR framework, human driving forces (D), such as the demand for food driven by human population growth, exert pressure (P) on the environment through more intensive fishing for instance, thereby changing the state of biodiversity (S) in the ecosystems that are exploited. These changes in state have an impact (1) on society, such as failure to meet targets set at the World Summit on Sustainable Development in 2002. Society responds ® by trying to control the driving forces or pressure, e.g. the Common Fisheries Policy Reform in 2012. This integrated approach was adopted by the European Environment Agency[1] in its State of the Environment Reports (EEA 2006, 2010). The basic principles of the DPSIR approach are not fundamentally different, however, from those of the PSR model.

Its effectiveness depends on the ability to measure pressure, state and response and thus help understand how they are interrelated. Only by understanding how human pressures modify state and how the management response modifies pressure can managers modify biodiversity in order to meet societal objectives. Analysis of state will focus on cataloguing existing biodiversity and its location and developing the tools and metrics needed to describe it. Once again, this emphasises the need to understand the ecological and evolutionary processes that have shaped biodiversity over time and space, and for research to assess how patterns of biodiversity influence function and the provision of ecosystem services.

Based on the DPSIR model, most ecology models provide information about "states" and "impacts". In practice, these models require "forcing" by results from pressure models and produce simulations which can be used in response models. Classification of models, based on their degree of ecological integration and spatial resolution, was proposed by Klok et al. (2009, see Fig. 4.3).

A wide range of human pressures have the potential to modify biodiversity (Table 3.4). These can act antagonistically, additively or synergistically to modify biodiversity. Although the PSR conceptual framework is both simple and widely used, the links between pressure and state are usually complex and thus multiple responses may be needed to modify the state. Likewise, one pressure can affect several aspects of the state of biodiversity (Fig. 4.2).

There are several categories of management options to deal with pressure in the marine environment and they can be complementary: (1) *input controls*: management measures that influence the amount of a human activity that is permitted, (2) *output controls*: management measures that influence the degree of perturbation that is permitted, (3) *spatial and temporal distribution controls:* management measures that influence where and when an activity is allowed to occur, (4) *management coordination measures*, (5) *economic incentives*: management measures which make it in the economic interest of those using the marine ecosystems to modify pressure

[1] http://www.eea.europa.eu

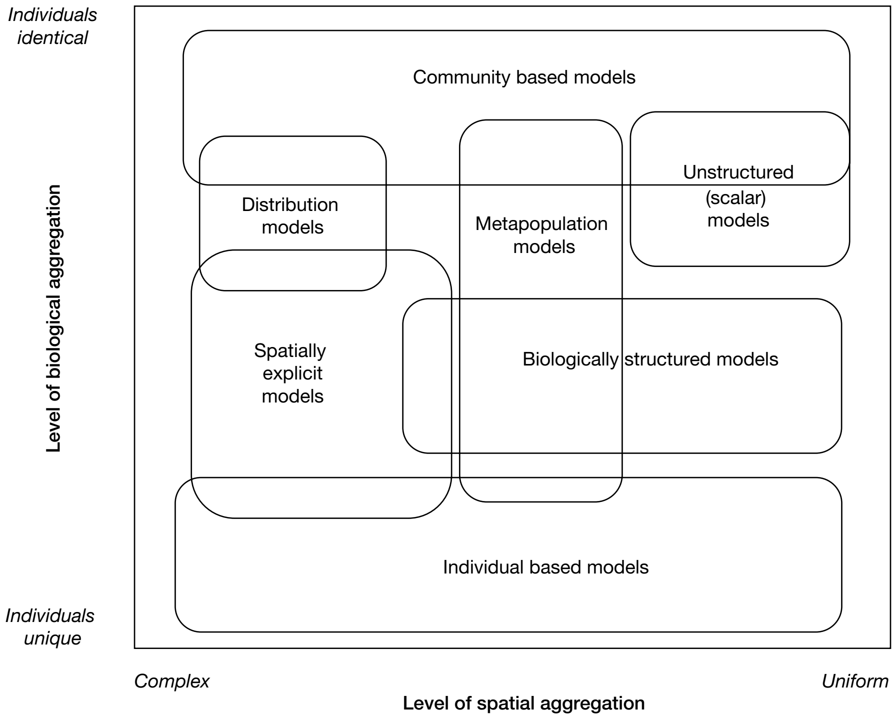

Fig. 4.3 Ecological model classification and specifications as proposed by Klok et al. (2009). (Adapted from Munns et al. 2007)

to meet objectives, (6) *mitigation and remediation tools* and (7) *communication*, stakeholder involvement and raising public awareness (ICES 2005; EC 2008).

In order to improve understanding of the interdependencies between human development and conservation objectives, the then United Nations Secretary-General Kofi Annan launched the Millennium Ecosystem Assessment or MA in June 2001. This 4-year core process, involving 1,360 scientists from 95 countries, was supervised by an 80-member independent board commissioned to validate the research programme output. The MA informed governments, non-governmental organisations, scientists and the general public about changes in ecosystems and their consequences for human well-being (MA 2005). This is the first programme to have incorporated—on a global scale—the economic, ecological and social components of biodiversity protection.

The MA is novel through its multiscale, multidisciplinary and integrated vision which can highlight the interdependent linkages between socio-economic and ecological dynamics (Fig. 4.5). The assessment takes account of relations between conservation and development as well as linking global and local changes. Biodiversity is considered as the keystone of ecosystem services (i.e. supporting, provisioning, regulating and cultural services; see below). The benefit of this approach is to highlight possible trade-offs between different types of services provided by it. The

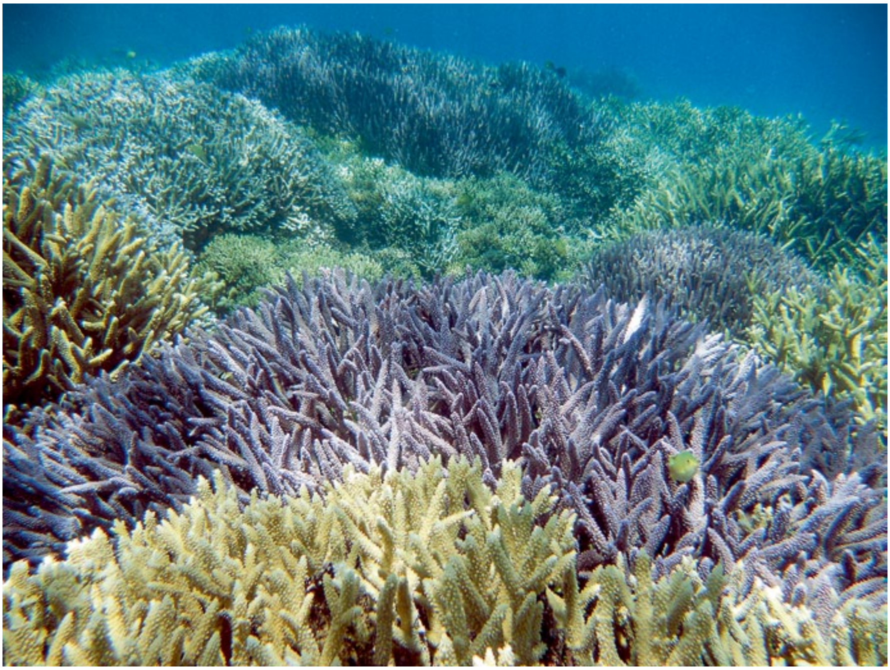

Fig. 4.4 Multicoloured branching corals, Great Aboré outer reef flat, on the south-western coast of New Caledonia. (© Lionel Loubersac)

rationale behind the MA's architecture is based on four "compartments" which are linked by interactions. Ecosystem services are put into four categories: supporting services, which are the basic biogeochemical processes which enable the development of life on Earth, provisioning services for mineral (water), fossil and living resources (e.g. fisheries resources), regulating services, particularly the major cycles which are the basis for ecosystem function and finally, cultural services, in the social realm (Fig. 4.4). When speaking of human development, the UNDP defines "well-being" as the "*opportunity to be able to achieve what an individual values doing and being*". This is quite a different concept of individual development from the World Bank's utilitarian, income-based vision. Direct drivers of change here explicitly designate the components of global change which influence ecosystem processes (especially, global warming, habitat destruction, invasive species, overexploitation of resources and pollution). Indirect forcings indicate the various factors which influence the direct driving forces of change. In the MA framework, these are economic, demographic, cultural, social-political, scientific and technical factors.

The PSR model's intuitive structure has facilitated its dissemination amongst economists and ecology scientists who find it an effective tool for outreach and education. However, the OECD has acknowledged that this analytical framework is flawed in that it suggests that relations between human activities and the state of biodiversity are linear, and thus understates the complexity of interactions. "*While*

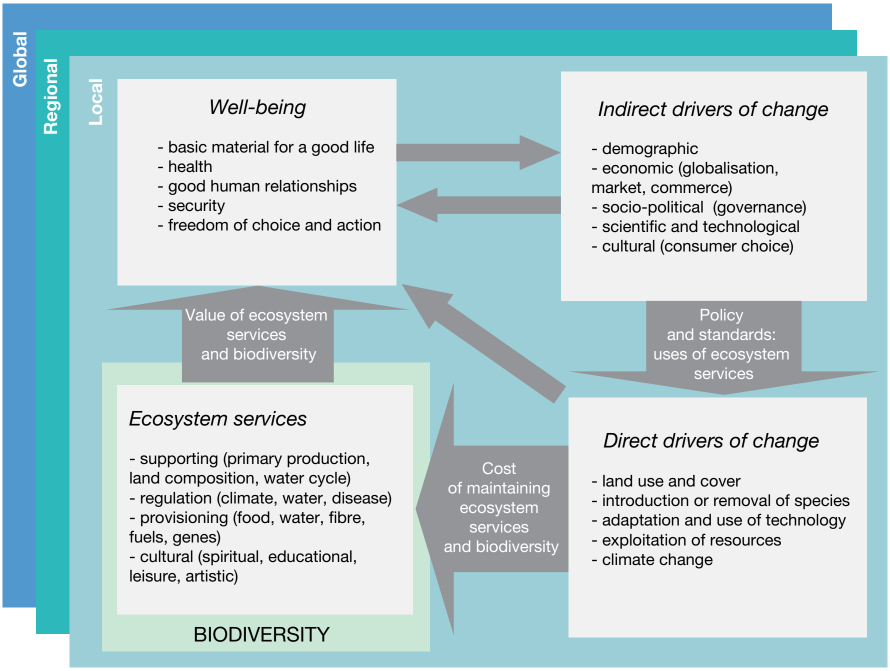

Fig. 4.5 Millennium Ecosystem Assessment model. (Source: MA 2005, pp. 13–14)

the PSR framework has the advantage of highlighting these links—pressures and responses—, it tends to suggest linear relationships in the human activity-environment interactions. This should not obstruct the view of more complex relationships in ecosystems and environment-economy interactions. " (OECD 1994, p. 10).

That is why a more realistic structure was suggested, with additional boxes (Fig. 4.6). Firstly, as in the OECD model, biodiversity is characterised by its "state" and societies as a source of change through their "human activities". A box for "ecosystem services" is still useful in order to highlight positive interactions between actions to conserve biodiversity and human societal development. "Response" indicators focus on measures society can take to slow biodiversity erosion, but leave out the "response capacity" of stakeholders. In this respect, response indicators raise a fundamental issue. If they are in keeping with the experts' responses based on the best available scientific knowledge, then this creates a normative instrument which could effectively substitute specialists' opinion for citizens' preferences. However, the societal response indicators proposed by conservation organisations have never been put to the test of public discussion, and when local people concerned by an issue are asked, it turns out that the choices they recommend to counter the erosion of biodiversity vary considerably (Levrel and Bouamrane 2008). This diversity of responses highlights the eminently political nature of this category of indicators. To be of use to managers, they must be employed in conjunction with other indicators

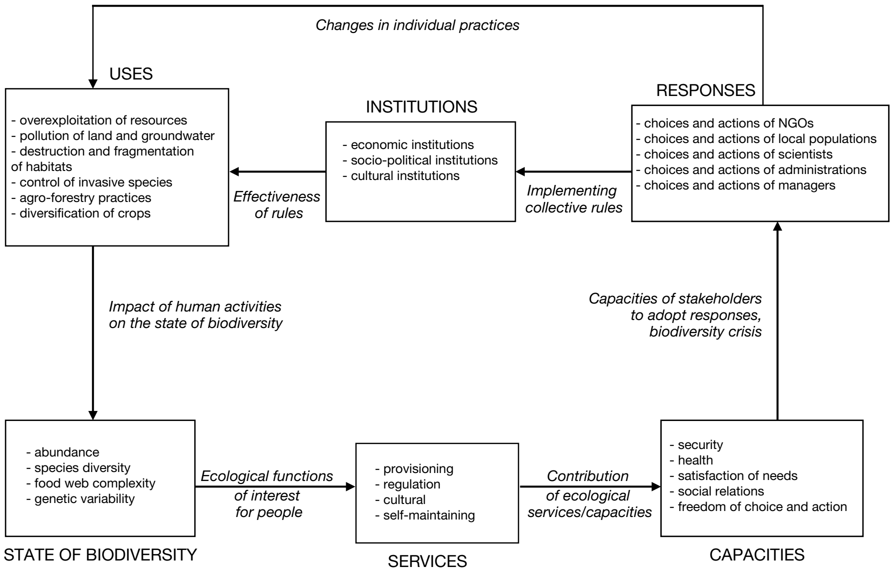

Fig. 4.6 An alternative model combining MA and PSR approaches. (Source: Levrel and Bouamrane 2008)

for both individual and collective capacities for response, and for the relevance of these responses. Individual response capacities are significantly related to how dependent stakeholders are on the resources they use. Collective response capacities include the "institutions" which enable local stakeholders to share responsibility in managing the resources they depend on.

Ultimately, the effectiveness of responses will largely depend on the legitimacy of the process used to adopt measures. On the basis of the points above, Levrel and Bouamrane (2008) proposed a new approach to identify the indicators of interactions which takes into account both the PSR and MA frameworks.

Choice of Model Framework

The PSR framework adopted in this report proposes to explicitly address human interactions with the state of biodiversity. It recognises that managing biodiversity ultimately means managing the pressures biodiversity is subjected to. However, PSR relationships must also be considered in the context of the ecological and evolutionary processes underpinning the state of biodiversity that is impacted by human stressors. Furthermore, social pressures and responses will depend on the environment which itself affects and influences society (Fig. 4.1). Seeing these mutual feedbacks and the avenues of research they could open, we propose to generalise the PSR model by putting it firmly within the context defined by society

Fig. 4.7 Hermit crab—seafloor landscape in Brittany. (© Ifremer, Olivier Dugornay)

Fig. 4.8 Nudibranch (Aeolidia papillosa). (© Ifremer, Olivier Dugornay)

and environment. This would make the relationships between pressures, state and responses explicitly "context-dependent" and some research priorities would focus on the processes by which the environment and society modify the interactions between pressures, states and responses (Figs. 4.6, 4.7, 4.8 and 4.9).

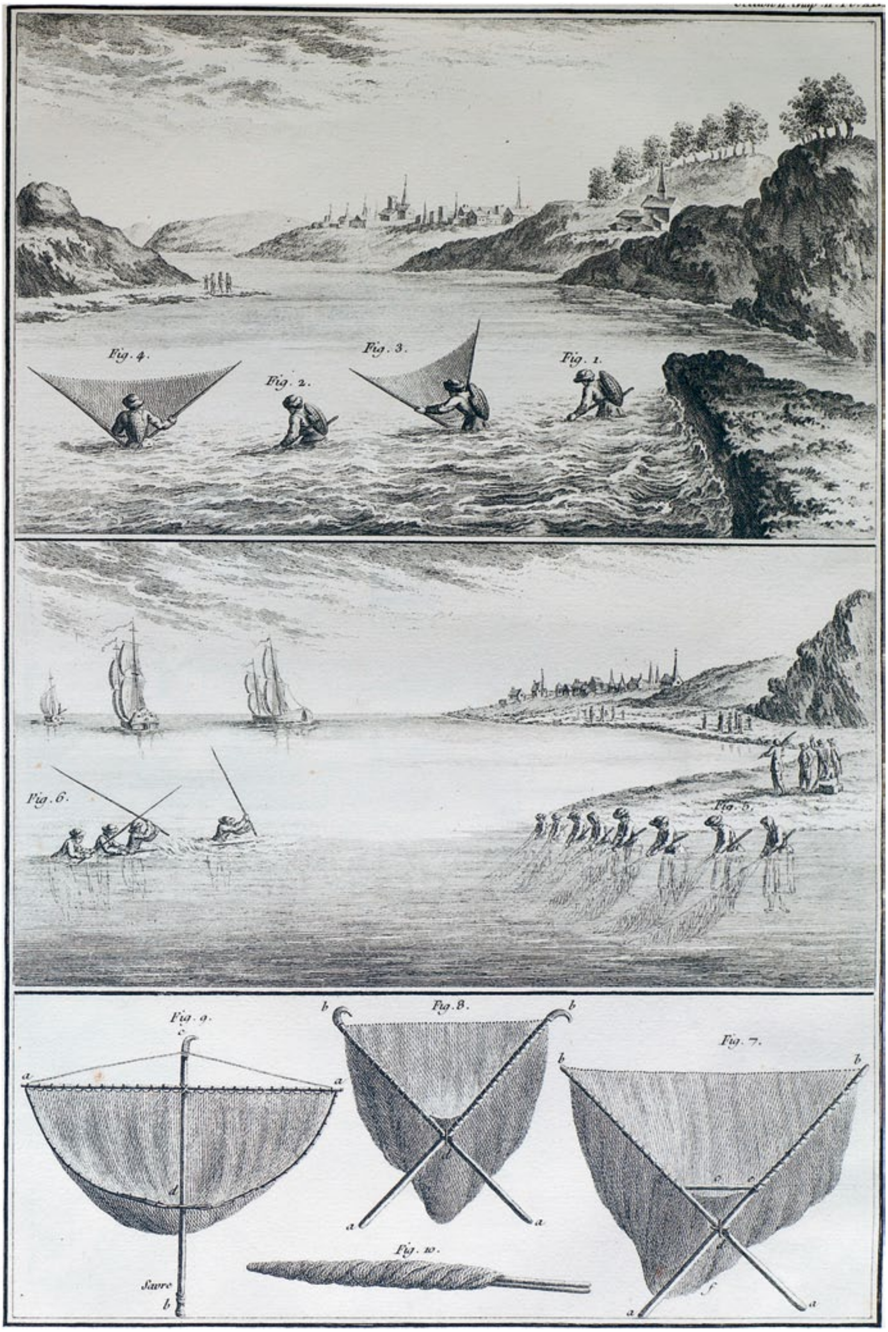

Fig. 4.9 Taken from Duhamel du Monceau and La Mare, 1769, Traité général des pêches, Sect. 2, Chap. II, plate XII

This opinion, in keeping with that stated by Pereira et al. (2010)—emphasises the need to develop scenarios so that the dynamics of PSR relationships can be studied within plausible contexts of future changes in society and in the environment. By providing different visions of the future, scenarios aim to better assess and understand the dependence of PSR relationships with respect to uncertain future societal developments.

Chapter 5
Measuring Biodiversity

Biodiversity as a Macroscopic Descriptor in the European Union Marine Strategy Framework Directive (MSFD)

In compliance with the requirements of the Marine Strategy Framework Directive (MSFD) adopted in 2008, in 2010 the European Commission produced criteria and methodological standards allowing for a consistent approach aiming to restore Good Environmental Status (GES) in EU marine waters by 2020. An ICES-JRC group was tasked with proposing criteria and methodological standards for Descriptor n°1 "Biological diversity. Target: Biological diversity is maintained. The quality and occurrence of habitats and the distribution and abundance of species are in line with prevailing physiographic, geographic and climatic conditions." One of the group's first tasks was to define the terms of reference for the descriptor. The definition of the Convention on Biological Diversity (CBD 1992) was used for the term "biological diversity". The term "maintained" was equated to no further loss of the diversity within species, between species and of habitats/communities. Ecosystems—at ecologically relevant scales—must also achieve a state in which biological diversity is restored and protected for its sustainable use. Consequently GES is assumed to be achieved given no further loss of the diversity of genes, species and habitats/communities at ecologically relevant scales and when deteriorated components, where intrinsic environmental conditions allow, are restored to target levels.

The task group recommended that biodiversity assessments be based on the states of: (1) species (including intra-specific variation, where appropriate), (2) habitats/communities (3) landscapes and (4) ecosystems. They further recommended that biological diversity should be considered in the following areas and groups identified in the Annexes to the original Directive: (1) the predominant seabed and water column types, (2) special habitat types (under Community legislation or international conventions), (3) habitats in particular areas (e.g. in pressured or protected areas), (4) Biological communities associated with the predominant seabed and water column habitats, (5) Fish, marine mammals, reptiles, birds, (6) other species (under Community legislation or international conventions) and (7) non-indigenous, exotic species and genetically distinct forms of native species.

Following these recommendations, the European Commission drew up a formal agreement in mid-2010 on the criteria and methods to apply in order to achieve GES. In this respect, levels and targets have not yet been set. The needs for research

P. Goulletquer et al., *Biodiversity in the Marine Environment,*
DOI 10.1007/978-94-017-8566-2_5, © Éditions Quæ, 2014

in developing operational indicators and methods to assess socio-economic consequences of moving towards GES targets can already be anticipated.

Moreover, the European Commission has recently initiated a dialogue on appropriate biodiversity targets after 2010 (EC 2010). Four options were suggested to achieve GES targets in the European Union by 2020: (1) significantly reduce the rate of loss of biodiversity and ecosystem services in the EU by 2020, (2) halt the loss of biodiversity and ecosystem services in the EU by 2020 (the same as the existing 2010 target but allowing more time to achieve it), (3) halt the loss of biodiversity and ecosystem services in the EU by 2020 and restore them insofar as possible, and (4) halt the loss of biodiversity and ecosystem services in the EU by 2020 and restore them insofar as possible, and step up the EU's contribution to averting global biodiversity loss. Whatever the final option selected in 2012, it will obviously require that appropriate indicators be developed to track trends in biodiversity and ecosystem services.

Measuring Genetic Diversity

Charles Darwin (1896) was the first to recognise the importance of genetics in maintaining natural populations. Conservation genetics aim to preserve both wild and farmed populations' potential to evolve and their maintenance. The discipline has grown considerably since Darwin's time. However, the need to empirically prove the relationship between genetic diversity and characteristics of fitness such as viability and fertility still remains. Initial classical ways of inferring genetic diversity from phenotypic variation (Carvalho 1998), have been progressively replaced since the 1960s by contemporary approaches employing a range of molecular genetic markers (Beebee and Rowe 2008). Irrespective of the technical method adopted, the essence of such approaches is to employ heritable, discrete and stable markers to identify genotypes that characterise individuals, populations or species. Importantly, there are two primary components of measuring genetic diversity in wild populations: first, the levels of genetic diversity found within populations, typically described as number of alleles per locus, heterozygosity or level of genetic polymorphism [4], and secondly, the level of genetic differentiation among populations, expressed by measures such as Fst, the fixation index, which is a measure of population differentiation, with genetic distance based on genetic polymorphism data such as single-nucleotide polymorphisms (SNPs) or microsatellites. It is a special case of F-statistics, the concept developed in the 1920s by Sewall Wright. FST is simply the correlation of randomly chosen alleles within the same sub-population relative to that found in the entire population. It is often expressed as the proportion of genetic diversity due to allele frequency differences among populations. This comparison of genetic variability within and between populations is frequently used in the field of population genetics. Several definitions of Fst have been used, all measuring different but related quantities. A common definition given is:

$$F_{ST} = \frac{\Pi_{\text{Between}} - \Pi_{\text{Within}}}{\Pi_{\text{Between}}}$$

where Π Between and Π Within represent the average number of pairwise differences between two individuals sampled from different (Π Between) or the same (Π Within) population. The average pairwise difference within a population can be calculated as the sum of the pairwise differences divided by the number of pairs. Note that when using this definition of FST, the value Π Within should be computed for each population and then averaged. Otherwise, random sampling of pairs within populations put all the weight on the population with the largest sample size. It is important to estimate the levels of genetic differentiation among populations because a component of such divergence will typically represent locally adaptive variation: features that enhance performance and persistence of populations exposed to environmental variation.

Estimating genetic diversity [1]

There are three primary measures of genetic diversity commonly employed.

- *Proportion of polymorphic loci* (or polymorphism): the proportion or percentage of the loci studied that reveal more than one allele in the population. Because sampling consistency is especially important for this measure (the more individuals sampled, the higher the probability of eventually finding an alternate allele), sample size bias is reduced by using P99 or P95. P95 is more stringent because the commonest frequency must not be greater than 95 % (meaning that minor alleles must account for at least 5 % of the total).
- *Allelic richness*: which is the mean number of alleles per locus (i.e. simply the total number of alleles divided by the number of loci).
- *Heterozygosity*: which includes both the observed heterozygosity (Ho), the mean proportion of individuals heterozygous across loci, or the mean proportion of loci for which an individual is heterozygous, and the expected heterozygosity (He), which is the proportion expected from allele frequencies in the sample assuming the population is in Hardy-Weinberg equilibrium.

For haplotypic data (such as mitochondrial DNA, mtDNA Restriction Fragment Length Polymorphisms, a simple estimate of diversity (D) can be calculated from $D = 1 - \Sigma xi2$, where xi is the frequency of the ith allele. For full sequence comparisons, nucleotide diversity (π) can be estimated using $\pi = \Sigma p/nc$, where p is the proportion of different nucleotides between DNA sequences and nc is the total number of comparisons, given by $nc = 0.5(n - 1)$, with n as the number of individuals sequenced.

Although early molecular markers focused on the use of protein variation or allozymes, contemporary approaches typically employ one of several classes of nucleic acid sequences such as microsatellites or SNPs. Whatever the method used, the underlying principle is the same, i.e. to detect genetic variation in nucleic acid sequences and to use the data to compare relationships, whether it is a comparison of allele frequencies among populations, or nucleic acid divergence among species [4]). The main advantage of these ubiquitous markers is the potential for high comparability across different taxonomic levels, thus enabling investigation of the mechanisms that serve to shape the extent and dynamics of genetic diversity in natural populations.

Measuring Species Diversity

Diversity has been measured in many ways (Magurran 1988). Estimation methods range from the most exhaustive possible census of species observed (species richness) to indicators of the distribution of individuals within the species richness (equitability indices). An equitability index can make a community where no one species is truly dominant appear to be more diversified (and a community where most individuals belong to only a small number of species seem less diversified). Table 5.1 gives examples of diversity indexes and others are described in Magurran (1988).

Diversity indices have been widely used to describe spatial patterns in the diversity of marine fauna and to assess changes in diversity over space or time in response to human impacts (Clarke and Warwick 1994). However, mechanistic understanding of the ways in which these indices respond to human impacts is still limited, in spite of attempts to model the underlying processes (e.g. Greenstreet et al. 1998; Hall et al. 2006).

Diversity indices offer statistical convenience, but they carry little information on the overall distribution of individuals among species. One alternative that has been proposed is to assess relationships between species and abundance using graphics. Amongst the most widely used techniques are abundance (or dominance), and cumulative abundance (or k dominance) curves, plotted against rank (or log-rank) in increasing order of abundance. The least elevated curves show the highest species diversity (and abundances can also be modified, e.g. log transformed (see Clarke and Warwick 2001).

The aforementioned methods allocate the same weighting to species, regardless of their taxonomic or phylogenetic proximity (or distance). As such, these above-mentioned diversity indices suggest that a community of four distantly related species of given abundance has the same diversity as a community of four closely related species with the same abundance. Taxonomy-based diversity indices (e.g. Warwick and Clarke 1998; Clarke and Warwick 2001) were introduced in an attempt to take account of taxonomic or phylogenetic distance, though in practice the indices have only been applied to taxonomies. Taxonomic diversity (Δ) is based on

Table 5.1 Some traditionally used indices of species diversity (From Jennings and Reynolds 2000)

Index	Measurements	Formula	Notes
Species richness (S)	Number of species	S	
Margalef (D)	Number of species for given number of individuals	$(S-1)/\ln N$	1
Menhinick's (D)	Number of species for given number of individuals	$S/\sqrt{N}$	
Shannon Wiener (H')	Richness and equitability	$-\sum p_i \ln p_i$	2
Eveness (for H')	Evenness	H'/H_{max} or $H'/\ln S$	
Brillouin (HB)	Richness and equitability	$(\ln N! - \sum \ln n_i)/N$	3
Pielou Evenness (for HB)	Evenness	HB/HB_{max}	
Simpsons (D)	Dominance	$\sum(n_i(n_i-1)/N(N-1))$	4
Hill N0	Number of species	S	5
Hill Nj	Number of "abundant" species	$\exp H'$	
Hill N2	Number of "very abundant" species	$1/D$	6
Taxonomic diversity (Δ)	Species diversity with taxonomic separation	$[\sum\sum_{i<j} \omega_{ij} x_i x_j]/[n(n-1)/2]$	7
Taxonomic distinctness	Species diversity without taxonomic separation	$[\sum\sum_{i<j} \omega_{ij} x_i x_j]/[\sum\sum_{i<j} x_i x$	7
Taxonomic distinctness (Δ+)	Taxonomic distinctness for presence/absence of data	$[\sum\sum_{i<j} \omega_{ij}]/[s(s-1)/2]$	7

Notes: (1) S = number of species and N = number of individuals. (2) Where Pi is the proportion of individuals of the ith species. The Shannon-Wiener index assumes random sampling from an infinitely large population and that all species present are represented in the sample. In reality the true value of pi is unknown and is estimated from pi=ni/N. (3) Used to describe a known collection of N individuals; tends towards Shannon-Wiener Index when N becomes very large. (4) Gives probability that two individuals drawn at random from an infinitely large community belong to the same species; an equivalent to this exists in the Gini-Simpson Index which estimates the probability that two individuals taken at random will be from different species. (5) Hill indices: family of diversity measures ranging from those that emphasize uncommon species to those that emphasize dominance; N_0, N_1 and N_2 are usually reported together. (6) Where D is Simpson's Index. (7) x_i is abundance of s species observed, $n=\sum x_i$ is total number of individuals in a sample and ω_{ij} weights the path length linking species i and j through the taxonomy

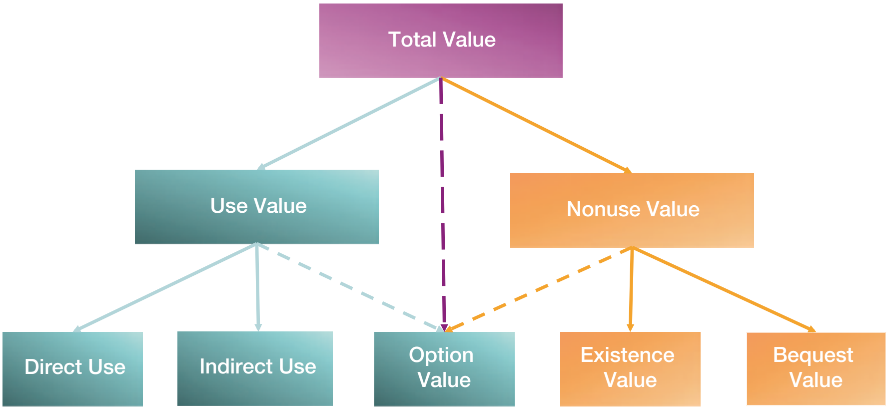

Fig. 5.1 Components of total economic value. (Adapted from Dziegielewska et al. 2007)

Simpson diversity with an additional component of taxonomic separation. It can be described as reflecting the mean path length along a taxonomic hierarchy between any two randomly chosen individuals. Δ^* is similar, but with a reduced role of species abundance, such that it measures the mean path length between any two randomly chosen individuals, providing that they come from two different species.

Assessing the Value of Marine Biodiversity

Marine biodiversity provides a wide range of values to people. Economic value is measured by the amount people are willing to pay for a good or service, or the amount they are willing to accept as compensation for not using the good or service. The concept of total economic value (TEV) encompasses different values generated by marine biodiversity, which fall into two main categories of "use" and "non-use." Use values specifically involve direct human interaction with the resource and include value produced through both direct and indirect use. This is not the case for non-use values, which are current or future potential values based on continued existence of a resource. Non-use values are further divided into existence and bequest values (Fig. 5.1; FAO 2003; Dziegielewska et al. 2007; Hanna and Sampson 2009).

Use value can be generated through direct use, indirect use, or option for future use.

- *Direct use value*: value from actual use of an ecosystem good or service, such as catching fish or kayaking.
- *Indirect use value:* value related to special functions, such as the habitat utilised by fish in a marine ecosystem or the knowledge generated through using a marine reserve (MR) as a research site.

- *Option value:* value of the ecosystem goods or services which will be available in the future. Several economists consider option value as both a use and non-use component, depending on the context. For instance, an option value could represent the future production of fish (use) or of marine biodiversity (non-use).

Non-use values, sometimes called "passive use" values, include option (see above), existence and bequest values.

- *Option:* as described above.
- *Existence value*: value from knowing that a certain good or service exists, e.g. the protection of endangered species against extinction, regardless of whether they are ever seen.
- *Bequest value*: value from ensuring that certain goods or services will be preserved for future generations.

For goods and services exchanged on a market, value is determined by market transactions. The market best reveals direct-use value through consumers' and producers' willingness to pay for a benefit or accept compensation for a cost. Indirect-use values are more difficult to quantify and are often ignored in resource-management decisions. However, markets are now emerging based on non-use values such as water temperature, endangered species habitat and carbon sequestration (e.g. Leslie et al. 2010).

The division of economic value into "use" and "non-use" categories is one way of characterising the trade-offs associated with marine biodiversity. Empirical assessment of these trade-offs is an emerging area of analysis to support decision making (Fisher et al. 2009; Whitmarsh and Palmieri 2011).

The following section on analytical methods is borrowed from a similar section in Hanna and Sampson (2009).

Analytical Methods Relevant to the Human Dimensions of Marine Biodiversity

Current emphasis on the ecosystem approach to fisheries management is already generating a need for assessments of the value of biodiversity through its ecosystem goods and services. There are five major avenues through which social science can contribute to the evaluation and management of marine biodiversity: assessment, feedback, prediction, mitigation, and acceptance (NOAA 2005).

- *Assessment*: baseline information provides information on existing uses. Incorporating economics into assessments can identify affected groups and potential areas of conflict resulting from protective actions. Early economic assessment can help predict potentially avoidable problems.
- *Feedback*: ongoing economic monitoring can help in evaluating the effectiveness of biodiversity management over time. Research can identify the economic

components of effectiveness and provide the public an opportunity to suggest management changes.
- *Prediction*: a range of economics methods can be used to predict the outcomes of biodiversity management actions, thereby helping to identify potential problems before they develop.
- *Mitigation*: understanding the economic positions and motivations of user groups and coastal communities may help reduce, or even avoid, conflicts associated with marine biodiversity protections.
- *Acceptance*: economic analysis can be used to understand public concerns, particularly with regard to the distribution of impacts from biodiversity protection. Concerns can be addressed through targeted outreach and education programs, which may in turn lead to better design of protection measures and increased public support.

Methods of Social Science Analysis

There is a range of social science research approaches appropriate to the analysis of marine biodiversity (especially Marine Protected Areas) (NOAA 2005). Economic modelling of MPAs was described in a special issue of the journal *Natural Resource Modeling*, edited by Sumaila and Charles (2002). These methods can be grouped into categories reflecting their primary, but not exclusive, application. The categories are to enhance understanding of economic context, human interactions, costs and benefits, and economic impacts.

Understanding the Human Context

- *Case study research*: an in-depth investigation of economic attributes and impacts associated with specific issues and locations (Fig. 5.2)
- *Content analysis*: a review of information sources such as newspapers, books, manuscripts.
- *Web sites,* etc.: to identify key words or phrases to help identify patterns and trends in discussions about marine biodiversity and to understand the context for economic impacts and values.
- *Demographic analysis*: a study of the characteristics of human populations, such as size, growth, density, and distribution in coastal communities.
- *Rapid rural appraisal*: a broad-level evaluation, usually through consultation with experts and stakeholders, that provides a general overview of the economic relationship between people and marine resources and identifies areas of concern about marine biodiversity as a precursor to planning.

Fig. 5.2 Scallop farming in Saint-Pierre-et-Miquelon (© Ifremer, Stéphane Robert)

Understanding Human Interactions

- *Focus group*: a group interview about a specific topic, for example fishery operating costs. Focus groups can also be used to identify economic motivations, styles of interaction, or perceptions of risk.
- *Observation*: personal observation and recording of patterns of resource use, interaction, and behavioural response.
- *Surveys*: primary economic data collection (by telephone, mail, or in person) through scientific sampling methods.

- *Predictive modelling*: simulation of real-world situations to predict future conditions; for example, the long-term impacts of areas set aside for biodiversity protections.
- *Bioeconomic modelling*: the integration of biophysical information and ecological processes with economic decision behaviour to analyse the possible effects of policies such as marine reserves on economic and resource welfare (see Anderson 2002).
- *Spatially explicit bioeconomic modelling*: addresses questions of economic and biological interactive effects with spatial effects explicitly taken into account, for example a spatial bioeconomic model to examine how various marine reserve

options affect fishermen participating in limited-entry fisheries (Sanchirico and Wilen 2002).
- *Game theory*: modelling of strategic interactions among agents based on economic motivations, for example, a model of distributional and efficiency effects of marine biodiversity protection areas to understand the effect of cooperative behaviour in management (Sumaila and Armstrong 2006).
- *Econometric analyses*: the application of statistical methods and empirical data to the testing of economic theories, for example, the testing of hypotheses about the economic response of fishermen to marine reserve implementation.
- *Secondary data analysis*: use of existing data and information (e.g. census data, fishery data, survey data) to identify characteristics of a group or analyse a particular issue.

Understanding Costs and Benefits of Biodiversity Protections

- *Cost-benefit analysis*: a tool for comparing the benefits of proposed marine biodiversity projects (e.g. alternative marine reserve sizes or sites) with the costs to identify the alternative with the maximum net benefit (benefits minus costs).
- *Non-market valuation*: methods to estimate indirectly an economic value that is not usually quantified in the typical markets where goods and services are exchanged for money, such as the value of ecosystem services (National Ocean Economics Program 2008).

Various methods have been used for conducting nonmarket valuations (Barbier 2009): (1) *contingent valuation*, determining willingness to pay (or to be compensated for loss) of a specified ecosystem good or service, through analysis of responses to structured questionnaires); (2) *travel cost*, estimating the value of marine biodiversity-supported recreational activity by analysing the relationship between participation and costs of travel to the biodiversity site. For example, the expenditure of people who travel to coastal locations to enjoy activities such as rock-pooling and bird-watching (Rees et al. 2010); (3) *avoided cost*, estimating the economic value of benefits that marine biodiversity provides via the cost of providing those benefits through some other action, for example, rebuilding overexploited fish stocks through reduced fish catch or through artificial propagation; (4) *benefit transfers*, estimating economic values by transferring existing benefit estimates from another location. The advantage is the avoided cost of a new study, but the disadvantage is the limited extent to which marine biodiversity in two locations is alike in the benefits it produces; (5) *choice experiments*, which consist in estimating economic values for ecosystem services by asking people to make trade-offs among sets of ecosystem or environmental services or characteristics. Willingness to pay is inferred from trade-offs people are willing to make among costly alternatives; and (6) *hedonic pricing,* assessing the value of an environmental feature by examining actual markets where the feature contributes to the price of a marketed good, for example, the monetary contribution of ocean views to home prices.

Understanding Impacts of Actions to Protect Marine Biodiversity

- *Economic impact assessment*: the identification of how user groups and coastal people and communities could react to a protected area, and the prediction of its probable impacts on regional income and employment and distributional effects among segments of the community.
- *Input-output analysis:* a representation of a regional economy through a description of linkages among industries. Changes on one economic component are traced throughout the economy, for example, a decline in fishery revenue or an increase in tourism revenue in a coastal community.
- *Comparative research*: comparison of different analyses over attributes, characteristics, or particular treatments across two or more biodiversity protection sites or within a single site over time to learn what contributes to different outcomes. Rees et al. (2010) consider a direct use valuation of the marine leisure and recreation industry to enable comparison with other sectors (e.g. fishing) that make use of the natural resource.
- *Multi-attribute utility analysis*: a tool for addressing a decision that has multiple criteria, e.g. quantifying trade-offs among the many ecological, economic, and social criteria accompanying biodiversity protection decisions. Proposed protections can be compared and scored using both quantitative and qualitative criteria (see Kiker et al. 2005).
- *Biodiversity portfolio analysis*: a tool for integrated coastal zone management using landscape level analysis balancing risks and returns within a portfolio of ecosystem values.
- *Institutional analysis*: the analysis of how organisations and people make economic and managerial decisions, for example, the structure and process of stakeholder involvement in decision making about marine biodiversity protection, or the governance of coastal ecosystems (Queffelec et al. 2009). Management strategy evaluation of protective actions also falls under this category (Tallis et al. 2010).

Marine and Coastal Biodiversity Indicators (SINP-Mer Jointly Operated by Ifremer, MNHN and AAMP)

In France, the Information system on nature and landscapes (SINP) is designed to promote synergies between managers and the people producing knowledge and data. One specificity of the SINP's marine strand is that indicators for marine and coastal biodiversity were developed from the very outset of the system's implementation. These indicators should clarify the linkages between marine biodiversity protection in France and various European regulations (e.g. Marine Natura 2000 Directive and the Marine Strategy Framework Directive) and international conven-

Table 5.2 List of institutional frameworks providing marine and coastal biodiversity indicators

Level	Institutional framework
International	Convention on Biological Diversity (CDB 1992)
	Ramsar Convention (Ramsar 1971)
	Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES 1973)
	Convention on migratory species (CMS 1979)
European[a]	Streamlining European 2010 Biodiversity Indicators, European Environment Agency (programme SEBI 2004, 2010)
	Habitats Directive 92/43/EEC (1992)
	Birds Directive 79/409/EEC (1979)
	Water Framework Directive 2000/60/EC (2000)
Regional	OSPAR Convention on the protection of the marine environment of the North-East Atlantic (1992)
	Barcelona convention for the protection of the Mediterranean Sea against pollution (1976)
	Agreement on the Conservation of Small Cetaceans of the Baltic and North Seas (ASCOBANS 1991)
	Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and contiguous Atlantic area (ACCOBAMS 1996)
National	French National Strategy for Biodiversity (SNB 2004)—Marine Action Plan

[a] Marine Strategy Framework Directive (MSFD, Directive 2008/56/EC) indicators are not given here, since they are still being developed

tions and agreements ratified by France (e.g. Convention on Biological Diversity, OSPAR, GBIF; see Table 5.2).

Ifremer's SINP-Mer project produced a comprehensive review which enabled the categories of marine ecosystem-based services to be identified (Table 5.3) as well as the 82 institutional indicators of marine and coastal biodiversity broken down into five fields of application (Table 5.4; Levrel et al. 2010).

- *Status and trends for components of biodiversity*: some of the institutional indicators assess the status and trends of ecosystems, species and populations of marine and coastal areas. These indicators provide elements for monitoring based on estimating variations in abundance of species and areal extents of habitats. However, they supply little information about the state of biodiversity at the genetic level.
- *Functioning and integrity of ecosystems*: this theme covers nearly half of the indicators identified, i.e. 40 of them. They are used to assess ecosystem health. In this context, the biodiversity indicator concept is often reduced to a simple "bio-indicator". They are used to assess trends in population subjected to major anthropogenic pressures such as eutrophication and pollution (e.g. the proportion of guillemots (*Uria aalge*) found dead or dying on beaches which are oiled birds).
- *Ecosystem conservation*: several indicators are proposed for monitoring the effect of measures implemented to protect biodiversity. They include the number and size of Marine Protected Areas (e.g. surface area of Natura 2000 sites at sea).

Table 5.3 Examples of ecosystem-based supporting services amongst 74 services provided by marine and coastal ecosystems (7 supporting, 20 provisioning, 26 regulating and 21 cultural; from Levrel et al. 2010)

Supporting services 7	List of services	Service's production source (structure or function)
1	Bioturbation (sediment mixing)	Biodiversity of benthic marine invertebrate species and some group of fishes (through spawning, feeding, hiding, resting)
2	Primary productivity	Photosynthesis, genetic and specific biodiversity of herbaceous community through several effects: redundancy, complementarity and selection
3	Secondary productivity	Food web dynamics, genetic and specific biodiversity of animal community through several effects: redundancy, complementarity and selection
4	Nutrient cycling and mineralization	Biodiversity of nitrogen fixing plants and of species along the food web through the production of waste
5	Water cycling	Biogeochemical cycles, oceans as a support of water cycling
6	Production of habitats for animals and plants (soil formation)	Biodiversity of soil invertebrate species, soil micro-organisms
7	Oxygen and carbon cycling	Biogeochemical cycles, oceans as a medium of carbon and oxygen cycling

- *Biodiversity uses*: this theme focuses on indicators assessing the variation in abundance of populations used by humans for food as well as indicators regarding direct human uses of biodiversity (e.g. fisheries' ecological footprint, annual aquaculture yields in Europe).
- *Other pressures on biodiversity*: four subsets are devoted to pressures on biodiversity: invasive species, climate change, pollution and eutrophication. Indicators provide direct information on the intensity of pressure (e.g. cumulative number of alien species in Europe since 1900, mapping of atmospheric deposition of reactive nitrogen).

Only a third of marine and coastal biodiversity indicators aim to assess the status and trends of components of diversity (Table 5.3). Indicators characterising species and populations (variations in species abundance, habitat distribution ranges) are over-represented, compared to those for genetic diversity. Furthermore, these "classic" indicators mainly focus on "remarkable" biodiversity, i.e. rare or vulnerable species, or those with special protection status, rather than so-called common, or ordinary, biodiversity.

Twenty-one percent of indicators (17 indicators) presented in institutional texts have not yet been estimated or implemented in practice. Most of them are still being developed (e.g. genetic diversity indicators). Eight indicators proposed in the National Strategy for Biodiversity (SNB) are still not in use in annual activity reports. Since two-thirds of those indicators are linked to databases, it is obvious that developing both databases and monitoring networks for marine and coastal biodiversity

Table 5.4 List of indicators of marine and coastal biodiversity identified in international frameworks (From Fossat et al. 2009; Levrel et al. 2010)

No.	Indicators
I. Status and trends of components of biodiversity	
I.1. Evolution and surface of biomes, ecosystems and habitats	
1	Evolution of the coverage of mangroves by countries or global (ha)
2	Evolution of wetlands in France (ha. and %)
3	Total area of mangrove in France and overseas (in km^2)
4	Status and trends of coral reefs worldwide and by region (km^2 and %)
5	Evolution of the surface of coral reefs and the rate of living coral cover (France)
6	Global distribution of seagrasses
7	Change in coverage of major habitat classes (CORINE Land Cover, CLC)
I.2. Evolution of species, populations and genetic diversity	
8	Marine Living Planet Index
9	Trends in abundance of marine fish caught
10	Trends in abundance of common birds in France
11	Trends in abundance of protected birds in France
12	Changes in the numbers and the abundance of spawning of sea turtles
13	Evolution of the abundance of cetaceans
14	Evolution of populations of porpoises in European waters
15	Evolution of populations of dolphins in European waters
16	Changes in the number of cetacean groundings (and by species) on the French coast
17	Evolution of the number of pinniped grounding (and by species) on the French coast
18	Genetic diversity of fish resources
I.3. Status changes of threatened and/or protected species and habitats	
19	Red list index (Status of marine species in the world)
20	Number of species on the French Red List of IUCN
21	Change in status of habitats of European interest (community)
22	Change in status of species of European interest (community)
23	State of conservation of protected habitats under the OSPAR Convention
24	State of conservation of protected species under the OSPAR Convention
25	State of conservation of protected habitats under the Barcelona Convention
26	State of conservation of protected species under the Barcelona Convention
II. Functioning and integrity of the ecosystem	
27	Marine trophic index
28	Proportion of transitional and marine water bodies in good ecological state
29	90 percentile chlorophyll a (mg/L)
30	Phytoplankton blooms
31	M-AMBI
32	Global recovery of intertidal macroalgae (hard substrate)
33	Number of representative species of intertidal macroalgae belts (hard substrate)
34	Covering of opportunistic species in intertidal macroalgae belts (hard substrate)
35	Limits of depth extension of the different subtidal algal belts (hard substrate)
36	Composition and density of species defining the layering (macroalgae subtidal hard substrate)
37	Species composition in subtidal macroalgae (hard substrate)
38	Total species richness of subtidal macroalgae (hard substrate)
39	Density and species composition of Zostera seagrass
40	Area and extents of Zostera beds

Table 5.4 (continued)

No.	Indicators
41	Fish indicator for Channel/Atlantic estuaries
42	Density of Posidonia (number of feet/m^2) to 15 m
43	Leaf area per foot of Posidonia (cm^2/foot) to 15 m
44	Epibiont loading on Posidonia leaves (dry weight of epibionts/leaf dry weight) to 15 m
45	Lower limit of Posidonia meadow
46	CARLIT Index
47	Richness of angiosperms and macroalgae in transitional waters in the Mediterranean
48	Covering index of reference species for angiosperms and macroalgae in transitional waters in the Mediterranean
49	Fish indicator in the Mediterranean
50	State of commercial fish stocks
51	Changes in population size of common seals (%)
52	Evolution of populations of young grey seals
53	Percentage of annual catches of harbour porpoises (%)
54	Proportion of guillemots (Uria aalge) oiled among those who were found dead or dying on beaches
55	Mercury levels found in eggs of seabirds
56	Organohalogen levels in eggs of seabirds
57	Plastic particles found in the stomachs of seabirds (fulmars, Fulmarus glacialis)
58	Reproductive success of kittiwakes (Rissa tridactyla)
59	Population trends of seabirds as an indicator of the health of the community of seabirds
60	Fish Index
61	Average level imposex in female purple (Nucella lapillus) or other gastropods selected, sensitive to tributyltin (TBT)
62	Density of sensitive species
63	Change/Mortality in zoobenthos in relation to eutrophication
64	Chlorophyll a content in phytoplankton (mg/L)
65	Phytoplankton eutrophication indicator (number of cells per litre; species composition)
III. Protection measures	
66	Protection level of marine areas (in %)
67	Density of MPAs in the high seas (areas existing and proposed sites)
68	Number and surface area (km^2) of Marine Protected Areas in France (mainland and overseas)
69	Area of Natura 2000 Sea France (ha or km^2)
70	Proportion and area of protected areas in France with plans for management or professional development, and rate of implementation of actions
71	Surface of coastal and marine protected areas (km^2)
IV. Uses of biodiversity	
72	Aquaculture ecosystems under sustainable management
73	Annual production of aquaculture in Europe and in European countries (tonnes)
74	Ecological footprint of fishing (Fishing Grounds Footprint)
75	Wild Commodities Index (marine)
V.1. Invasive species	
76	Cumulative number of alien species in Europe since 1900
77	List of worst invasive alien species threatening biodiversity in Europe

Table 5.4 (continued)

No.	Indicators
V.2. Climate change	
78	Evolution of sea level and coastal water temperature, frequency and intensity of hurricanes
V.3. Pollution	
79	Mapping of atmospheric deposition of reactive nitrogen (NOy and NHx)
V.4. Eutrophication	
80	Evolution of winter concentrations of nutrients (nitrogen (N) and phosphorus (P) in the transition waters)
81	Oxygen content (mg/L, oxygen saturation percentage)

Fig. 5.3 Yellowfin tuna seine fisheries, Seychelles, 2006 (© Ifremer, Marc Taquet)

on various scales is a critical objective for the SNB's implementation. Lastly, the concept of ecosystem services is increasingly coming to the fore as a reference in institutional frameworks. Half of all indicators are indirectly linked to it.

So it must be emphasised that indicators of biodiversity have mainly been developed for exploited species, the highest trophic levels and temperate areas. In contrast, biodiversity indicators for unexploited species, lowest trophic levels and the overseas areas are lacking. Likewise, the indicators developed by scientists are often not used by policy makers who prefer to use those developed by NGOs, and there is a recurrent lack of indicators for use over large spatial scales, due to gaps in information.

Fig. 5.4 Taken from Duhamel du Monceau and La Mare, 1769, *Traité général des pêches* (Sect. 1, plate VI)

Consequently, further development of biodiversity indicators must ensure that the following aspects are taken into account: social demand;—unexploited biodiversity—in the lowest trophic levels and living in the tropical areas; functional biodiversity to establish relationships with ecosystem services, as developed by the Millennium Ecosystem Assessment (2005); implementation from standardised databases over different spatial scales, and hence, networking of partners operating monitoring networks and/or biodiversity observatories (Figs. 5.3 and 5.4).

Chapter 6
Drivers of Changes in Biodiversity and its Uses

In describing the triggering element of a phenomenon, the terms "drivers" and "driving forces" will be used interchangeably here.

Environmental Drivers: A Working Framework

Interpreting the changes in the nature and patterns of biodiversity over time requires an overall vision, spanning several timescales. **Geological timescales** include orogeny, the formation and disappearance of large ocean basins, the fragmentation and collisions of continents, altering of the chemical composition of ocean and atmosphere, climate change and global events taking place over periods from a few tens to several 100 million years. **Evolutionary timescales**, without going all the way back to when life appeared on Earth some 3.8 billion years ago, are typically periods of several million years, marked by geophysical convulsions, crises and catastrophes, like the five great mass extinctions (over 75 % of species lost in less than 2 million years, or even much less than that) which helped shape biodiversity up until the most recent periods, including those since the last glaciation (20,000 years). Evolutionary time also encompasses short timescales of a few decades in extreme cases of rapid environmental change. **Ecological timescales** are more complex to define since they often relate to the biology of the taxa in question. They can differ by several orders of magnitude between rapidly proliferating microbial species and large marine mammals, for example. Understanding these nested ecological timescales provides the knowledge needed to examine more contemporary processes, i.e. those occurring over the past thousands to hundreds of years. Integrating a continuum of timescales to frame and test theories about biodiversity's origin and deployment stems from the nature of life itself: diverse events across their life history will shape the distribution and abundance of living organisms, dependent on features such as the spatial scale (global, regional or local), rate of change (across evolutionary time or in recent decades as anthropogenic pressures have become more intense) and biological level (e.g. individual, population or species).

P. Goulletquer et al., *Biodiversity in the Marine Environment*,
DOI 10.1007/978-94-017-8566-2_6,

One objective in biodiversity science is to identify the driving forces responsible. A few examples of the nature and impact of timescales on the evolution of biodiversity are given below. Environmental variations can also be seen on the world ocean scale, e.g. in meridional and zonal temperature gradients in surface water, or large biogeographic provinces[1] ("Longhurst Biogeographical Provinces"). The diversity of habitats, and numerous ecological processes such as dispersal capacity and connectivity, competition, predation, trophic interaction, disease and parasites, disturbance and facilitation (the conditioning of the environment by one species, allowing another species to become established there) further shape the large-scale variations observed.

Evolutionary Timescales

A species' "ecological time" is that of its populations, that is to say of individual life cycles. These vary depending on the species (for metazoans they range from a few weeks to a few decades). The times characterising ecosystem dynamics can also vary greatly. For example, carbon flux (photosynthesis/respiration, production of biomass, storage in soil) turnover times were estimated for five boreal terrestrial ecosystems using a modelling approach, showing that they varied from less than a year to more than a 100 years (Karlberg et al. 2006). These ecological time periods seem extremely short when compared to evolutionary time. Indeed, the time required to achieve speciation, taking estimations based on well-documented cases (islands or lakes) varies, from 1,000 to 100,000 years for animals such as insects or fishes. A few examples suggest that under very specific circumstances, the beginning of speciation can be observed within decades (Rolshausen et al. 2009). However, as a general rule, and particularly for marine mega- or macro-fauna, speciation remains a very long process. The time scale is even longer when the durations of species are considered. It is generally acknowledged that a species remains more or less unchanged over one to several million years (e.g. Kier 1974, for echinoids). As a consequence, the recovery time after a mass extinction covers several million years. In other words, ecological time is from 10,000 to a million-fold shorter than evolutionary time. This means the time needed by biological systems to respond to major ecosystem disturbances due to human activities may seem infinitely long (on the scale perceived by human societies).

Ecological Timescales

On ecological time scales, local diversity is influenced by the number of individuals in the community, diversity and abundance in the regional species pool and the rate of dispersal between the local and regional pools. Thus, to understand patterns of diversity and the effects of human impacts at local scales, they should be viewed

[1] www.marineregions.org

against the regional context. And, vice versa, understanding the regional effects of human impacts on biodiversity requires changing scale to look at the cumulative effects of local impacts that lead to regional changes, e.g. local extinctions leading to regional extinction.

The boundaries of regional species pools are often determined by the physical-chemical environment and these assemblages have characteristics which reflect their isolation during their evolution. For example, many extant teleost fishes evolved after the K-T mass extinction that marked the end of the Mesozoic Era (about 65.5 Myr), at the K-Pg (Cretaceous-Paleogene) boundary. At that time, the expanding North Atlantic—which had started to open approximately 200 Ma ago in the Early or Lower Jurassic—was surrounded by a continental shelf which ran continuously from the South of the North American continent to the South of Africa, via Greenland and the European seafront of the Eurasian continent. To the west, the Atlantic was largely connected with the East Pacific between the two Americas, linked by island arcs which would later become Central America and the West Indies. On the opposite shore, the Atlantic Ocean was linked to the Tethys Sea between Northern Africa and Europe as well as by the Indian Ocean—to the south of Africa (Scotese 1997). Polar climates had also cooled before the K-T mass extinction event (Spicer and Parrish 1986) and thus reinforcing global latitudinal temperature gradients (Frakes et al. 1994) and restricting the tropics to these low-medium latitudes. It is likely that biological communities were able to move into the belt of shallow seas surrounding what would later become North America. Ecosystems then changed substantially and some of them have only existed for the past 10 Ma. Today, therefore, even for taxa with relatively high ability to disperse, the expansion of regional species pools in the main ocean basins is often bounded by warm equatorial waters forming a natural barrier to the movement of temperate and cold water pelagic species. The land masses create an obstacle to zonal dispersal.

Causes of Pressures

Although fishing activity is quite variable over time and space, it is one of the strongest pressures brought to bear on marine biodiversity (see Chap. 3).

Other human activities also significantly affect marine and coastal ecosystems. Oil and gas extraction and their transport, with their attendant degassing and oil spills (Bordenave et al. 2004), chemical discharges and spills, river pollution (from industrial, urban and agricultural sources) ending up in estuarine and coastal zones (Kennish 2002) and aquaculture activities (Páez-Osuna et al. 1998) -including species introduction and invasions (Bax et al. 2003)—are also critical factors for the cumulative impacts on marine biodiversity. More globally, increasing urbanisation along coasts is a major driver (UNEP 2009).

The relative contribution of these different impacts on biodiversity will vary across space and time and depends on their spatial extent, duration and intensity and the environment where they occur, e.g. with respect to the type of habitat. For instance, a given level of fishing disturbance will have lower impacts on benthic species diversity in a highly mobile sand habitat than a complex biogenic habitat in an area subject to low natural disturbance (like the belt of cold-water corals on the continental slope in the North-East Atlantic). Pressures that are patchy (e.g. direct fishing impacts from dredging or bottom-trawling) rather than diffuse (as some forms of chemical pollution) will create a mosaic of local habitats that are impacted to different extents and may be in different stages, ranging from deeply degraded to quasi-recovery. There is an obvious relationship between the type of impact and response, but both ecological factors (populations' ability to migrate, life history traits, demographic strategies, and so on) and the oceanographic context (hydrodynamics and renewal times for water bodies, regional circulation features, etc.) must also be taken into account.

Despite the ecological processes that influence diversity acting across multiple scales, there are relatively few studies on the relationships between diversity at local and regional scales. Overall, regional gradients in species diversity tend to be influenced by environmental factors such as temperature and depth, rather than the human pressures that may be locally predominant (Callaway et al. 2002). However, cumulative local impacts can lead to significant regional change in areas subject to high pressures (Bianchi et al. 2000).

Importance of Disturbance: Biodiversity, Resilience and Robustness of Marine Ecosystems

The Scientific Challenge

Despite the growing implementation of ecosystem-based approaches in managing marine environment uses, the range and simultaneous nature of multiple stressors on natural ecosystems make it necessary to quantitatively assess the probability of ecosystem collapse or recovery, taking projected environmental change into account (Hendriks et al. 2010; Ling et al. 2009; Thrush et al. 2009). For sustainable ecosystem services, the various ecological components and the intricate interdependence across many levels must be maintained despite disruptions. Thus, marine ecosystems must be robust and resilient, i.e. retain the capacity to continue functioning even when exposed to stress or disturbances, whether gradual or rapid. The key challenge lies in the complexity of natural and socio-economic systems alike. Both are characterised by multiple possible outcomes and by a capacity for quick response or even major regime shifts.

These reactions can result from slower and smaller changes in exogenous and endogenous influences. The inherent nature of ecosystems means that the dynamics of small-scale interactions can spread to macroscopic levels as well as affecting smaller scales. Understanding the linkages between the various scales, and how to most effectively incorporate such knowledge into public awareness, management actions and policy decisions, are priority needs in biodiversity research.

Knowns

It is well established that the effects of such factors as climate change and other global changes like coastal zone development, overexploitation of marine resources, discharges of pollutants and nutrients, along with other anthropogenic influences, can result in major disturbances in marine ecosystems (Levin and Lubchenco 2008).

Such effects can alter the quality and quantity of ecosystem services, for instance through lower fisheries yields, poorer water quality, increased incidence of disease and arrival of alien species. While in many cases these pressures entail a sharp drop in the benefits derived from biodiversity, there are also cases in which ecosystem changes will lead to a different distribution of costs and benefits amongst users. An example of the management challenges this can produce is that of fisheries targeting multi-species, where increased landings of certain species will mean reduced landings of others, or the trade-offs involved in designating Marine Protected Areas (Boncoeur et al. 2002). Key to these trade-offs are the uncertainty attached to expected impacts of alternative exploitation patterns, time required to observe the actual impacts and reversibility of these impacts.

The capacity of a system to sustain changes and remain functional is known as resilience. The key components of resilience are: (1) the amount of change the system can undergo (and implicitly, therefore, the amount of extrinsic forcing the system can sustain) and still remain within the same domain of attraction (i.e. retain the same controls on structure and function); (2) the degree to which the system is capable of self-organisation (versus lack of organisation, or organisation forced by external factors); and (3) the degree to which the system can build the capacity to learn and adapt (Carpenter et al. 2001).

There is usually a positive relationship between biodiversity, resilience and capacity for recovery. This means that attempting to generate precise forecasts is in many cases inaccurate due to the inherent complexity of interactions across different ecosystem components. Some effects may be predicted with an acceptable degree of uncertainty when single environmental or human drivers are sufficiently strong to force an ecosystem into an alternative state. Increased temporal variability can provide a proxy for early-warning signals of an approaching regime shift or disruption to ecosystem services (Carpenter et al. 2006; Beaugrand et al. 2009). Nevertheless, there is increasing evidence that interactions among intrinsic ecological dynamics and numerous chronic, cumulative or multiple stressor effects can increase the uncertainty in any predictive framework, whatever the scenario, leading to a loss in resilience and greater risk of regime shift (e.g. far-reaching changes in species composition and function). These factors are summarised in Table 6.1.

Biodiversity is closely linked to biogeography, since its state is the consequence of physical and biological constraints operating over a range of spatial and temporal scales. Parts of the world whose climates are similar today may have very different biota as a result of contrasting evolutionary histories, severity of previous extreme conditions (e.g. geologically-recent glacial events—Graham et al. 2003),

Table 6.1 Community dynamics, feedbacks and thresholds in resilience of coastal marine ecosystems. (From Thrush et al. 2009)

Key processes	Mechanisms	How increased
Potentially containing thresholds	Maintaining resilience	Stress or disturbance can influence transitions
Functional loss of key species	Key species form habitats, and drive fluxes of energy and matter, patterns of species interaction	Density, size or spatial arrangement of key species drop below threshold for functional performance
Loss of diversity within functional groups	Diversity within functional groups maintains stable function in the face of change	Stress or disturbance affects all species within functional group; other aspects of the natural history of individual species limit the potential for replacement
Recovery to ambient conditions slow and variation in recovery of disturbed areas increases	Intrinsic interactions between species and local habitat during recovery processes facilitate recovery dynamics. Neighbouring habitats supply colonists with diverse functional traits	Variability in community structure increases moving away from a basin of attraction
Decrease in β-diversity and meta-community connectivity	Low β-diversity and high connectivity in a landscape ensure continuous supply of species to recover disturbed patches	Late successional-stage species are limited in distribution across the landscape

geographical barriers and species dispersal capabilities. An assessment of past dynamic processes and stochastic events determining present distributions of species is therefore critical to explain observed genetic structures and ecological interactions. Natural climate variability has recently—in geological terms- encompassed both rapid warming (at end of last ice age) and cooling (like the Younger Dryas or Big Freeze a little over 1,000 years after the Würm glacial stage ended, approximately 11,000 years ago).

Predicted climate changes over the next 50–100 years are of similar rates and magnitude, with ecological responses of species and communities depending on a complex mix of global and local forcing factors (Parmesan and Yohe 2003). Thus understanding of past events may contribute to the forecasting of future scenarios.

Analysis of DNA genealogies in a biogeographical context has greatly improved our understanding of climate effects on terrestrial biodiversity (e.g. Avise 2000; Hewitt 2000). A multi-species, comparative phylogeographic approach is particularly powerful, and can identify the boundaries of regional biota and elucidate the forces which structure genetic diversity and promote speciation. However, relatively few studies (except those by CORONA[2] partners—Coordinated Research on North Atlantic NSFDEB-0130275; Wares and Cunningham 2001) have applied this

[2] http://www.biology.duke.edu/corona/.

approach to marine biodiversity, despite the ecological importance and complexity of colonisation patterns in the Atlantic.

One major outcome of multi-species phylogeography is that intraspecific genetic breaks and areas of high (or low) genetic diversity are detected in the same geographic location for groups with diverse ecological requirements and taxonomic affinities (Avise 2000). These results suggest that similar factors (most likely linked to Pleistocene glacial and interglacial cycles) were involved, identifying the genetic background against which local adaptation and competition occur. The importance of specific morphological or life-history characteristics may also be revealed by identifying groups that have responded differently to the same environmental constraints.

Furthermore, climate change can be considered as a driver of evolutionary change. In future climate scenarios, both an increase in global mean temperatures and greater frequency of extreme climatic events are predicted. Many marine species will have to be able to adapt to such conditions. This will require individuals to possess near 'perfect' plasticity, enabling them to tolerate significant climate variability with no apparent fitness costs (DeWitt et al. 1998). Ongoing distributional changes and reports of climate-related species diebacks demonstrate that such plastic tolerance to changing climate is not widely shared (Chamaille-Jammes et al. 2006; Lagarde et al. 2008).

Unknowns

Ecosystem shifts are typically impossible to predict (de Young et al. 2008). Although these events are only identified after the fact, their consequences are known, i.e. general homogenisation of communities and ecosystems due to a reduction in food web complexity, lower diversity within functional groups and simplified habitat structure. Developing the capacity to anticipate regime shifts (via risk assessment) would provide a valuable resource for environmental managers. A major obstacle to forecasting is the disparity between theory and our ability to empirically investigate the effects of change of state under ecologically realistic conditions. There is an urgent need to improve methods and tools for the systematic assessment of marine ecosystem status in order to develop recovery scenarios (similarly to what is done for populations of marine species).

Natural disturbance tends to be relatively short in duration, while anthropogenic disturbances increasingly tend to be permanent. This creates a bias in the way we address these phenomena, by focusing on short-term disturbances, which may not necessarily help to understand more long-term effects (Fig. 6.1).

One priority in seeking to understand the scale and dynamics of marine biodiversity is to integrate population-level processes within ecologically predictive frameworks, for example, the prediction of extinction risks under climate change by coupling stochastic population models with dynamic bioclimatic habitat models and the integration of GIS-based environmental data in evolutionary biology

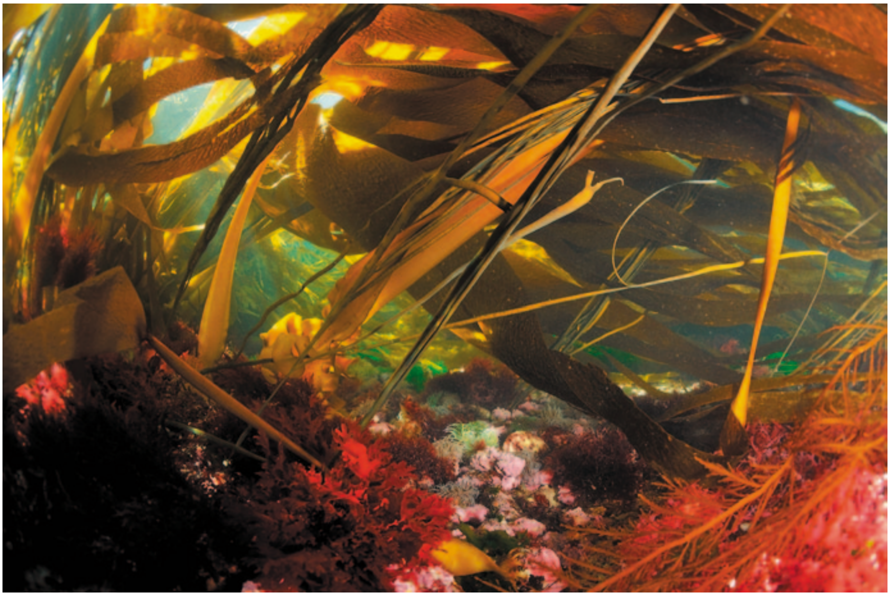

Fig. 6.1 Kelp forest, Molène island, Brittany. (© Ifremer, Olivier Dugornay)

(e.g. Keith et al. 2008). Many evolutionary processes are influenced by environmental variation over space and time, including genetic divergence among populations, speciation and evolutionary change in morphology, physiology and behaviour.

Yet, evolutionary biologists have generally not taken advantage of the extensive environmental data available from geographic information systems (GIS). For example, studies on phylogeography, speciation and character evolution often ignore or use only crude proxies for environmental variation (e.g. latitude and distance between populations). The integration of GIS-based environmental data, along with new spatial tools, can transform evolutionary studies and reveal new insights into the ecological causes of evolutionary patterns of biodiversity (Kozak et al. 2008).

Human Drivers

Knowns

Human societies are dependent upon a vast range of goods and services provided by coastal and marine ecosystems. Using these resources has altered ecosystems and directly or indirectly modified marine biodiversity, sometimes leading to unexpected outcomes. The drivers of these changes can be direct (like demographic growth, product value, pollution, for instance), but they may also combine diffuse, complex interactions between the numerous processes underpinning collective decision-making in a wide range of economic, cultural or institutional contexts.

Drivers are considered to be direct when they have a fundamental influence on how marine ecosystems evolve and indirect when their contribution is only secondary to direct drivers' effects. These forces driving change exist on various scales and maintain a range of interactions. This is truly a field of variability, uncertainty and complexity.

The Millennium Ecosystem Assessment analysed the key interactions between the driving forces of change in marine ecosystems and policy-makers (MA 2005). A difference is drawn between exogenous drivers which cannot be controlled by managers and endogenous ones which can. As in genetics or biology, the rapid growth of global databases of human uses of marine ecosystems cross-checked with ecological data has made new, ocean-scale assessments possible. Halpern et al. (2008, see also Selkoe et al. 2008) published a global map of human impacts to marine ecosystems. They developed a sophisticated multiscale spatial model to synthesize 17 global data sets of anthropogenic drivers of ecological change for 20 marine ecosystems with a vulnerability ranking for each stressor. This ranking was obtained through an e-mail survey of international experts (135 experts responded; some types of ecosystems were assessed by only one expert). Then the judgements of experts were mapped, attempting to ground-truth them by comparing the impact scores attributed to stressors combined with those from another assessment of the "ocean condition" based on a small sample of coral reef ecosystems. This made it possible to provide a geographical analysis of the distribution of pressures (except those for which insufficient data is available, including tourism, recreational fisheries, river inputs and aquaculture) and the degree of impact affecting the ecosystems. The objective was to integrate these results with flexible tools to identify the efforts to implement on regional or global scales in order to: (1) allocate the means needed to protect resources; (2) implement ecosystem-based management; and (3) produce data for spatial planning, education and research. The approach outlines a structured framework to quantify and compare the impacts and threats created by uses of ecosystem services and possibly define mitigation strategies with the goal of maintaining sustainable use. Additionally, this approach uses varied data (e.g. species distribution or diversity data) in order to identify the spatial dimension of hot spots with both high diversity and high cumulative human impacts and take account of them for monitoring purposes. This orientation will benefit from the deployment of observational data acquisition programmes.

In keeping with the rationale of the Millennium Ecosystem Assessment (MA 2005), we can take five main categories of anthropogenic drivers of global change. They are institutional, demographic, economic, social and cultural. The net outcomes of drivers, their interactions and the incentives they create can be positive or negative, or have ambiguous and complex effects on marine biodiversity.

Institutional Drivers Institutional drivers are generally described as a complex set of laws, customs, markets, standards and the related organisations which channel human activities towards social objectives. For individuals and societies alike, this is the general framework for uses and protection of marine ecosystems. These institutional arrangements interlink and overlap on international, national, regional and local levels. International agreements and conventions—the first and foremost being the United Nations Convention on the Law of the Sea and its legal instruments—are there to guide environmental protection towards responsible practices, and also concern other sectors of maritime activities, such as trade and transport. National policies and regulations are generally designed (and backed by incentives) to protect goods and services and control their use. In the United Kingdom, for example, the Marine and Coastal Access Act aims to protect and manage marine systems as well as extending the scope of protected areas called Marine Conservation Zones. Marine planning is expected to provide effective help for management, especially to identify zones of encroachment and competition between uses, e.g. between fisheries, offshore wind farms or marine aggregate extraction. The British government and its administrations, such as DEFRA[3] (Department for Environment, Food & Rural Affairs), work with different stakeholders to organise access to marine areas for a range of activities and for environmental protection. Modernised legislation on marine fisheries will clarify inshore fisheries management and conservation of habitats and biodiversity. Lastly, the Marine Management Organisation[4] regulates maritime activities and facilitates the implementation of environmental protection laws. This shows that national legislation can be strengthened (or weakened) by regional policies making use of positive (or negative) incentives for capital investments, developing technologies, population growth and guidelines for labour management work and trade. An example of this is the "basic" regulation of the European Union's Common Fisheries Policy (CFP, created in 1983 and revised in 2002, then in 2012), providing a framework for fisheries in the Exclusive Economic Zone of EU Member States. Its first instrument was the fixing of TACs and quotas for catches, progressively accompanied by fishing capacity of national fleets, then limitations set on fishing effort and regulated access for some so-called sensitive species (like deep-water species). The 2002 reforms "*ensure exploitation of living aquatic resources that provides sustainable economic, environmental and social conditions*" and for this purpose, the Community shall "*apply the precautionary approach*" *and* "*aim at a progressive implementation of an ecosystem-based approach to fisheries management*" underpinned by "*principles of good governance*". The 2002 reform of the CFP can be credited with several successes. For the first time, recourse to the precautionary principle and implementing an ecosystem approach became part of the basic regulation. It included increased stakeholder involvement and dialogue, consolidated and more transparent scientific foundations for the CFP, setting up multiannual management and means to efficiently combat Illegal, Unregulated and Unreported fisheries (entering into force in January 2010). However, chronic over-

[3] http://www.defra.gov.uk/.

[4] http://www.marinemanagement.org.uk/.

capacity of the fishing fleet was not curbed, the state of numerous fish stocks was degraded and the sector's economic resilience remained low (energy dependence, subsidies, short-term strategies). Taking account of these successes and failures, the Commission published a Green Paper (CEC 2009) in April 2009, as the basis of broad consultation culminating in the reformed CFP project in 2012 (CCE 2011).

It is well established that overfishing derives from the "common pool" nature of marine fish resources, leading to what economists have termed reciprocal negative externalities between fishers commonly—and incorrectly—called the 'tragedy of the commons' (Hardin 1968)—see e.g. Feeny et al. (1996), and to the development of the "race for fish" phenomenon with its potentially negative effect on fish stocks as well as on other species (Conservation international[5] 2010). Understanding such drivers of ecosystem uses thus provides a key in building future scenarios for sustainable fisheries.

This analysis can be generalised to multiple uses of marine biotic resources and more complex processes including the management of conflicts over common-pool resources and their impact on biodiversity (Adams et al. 2003). It is a way to highlight how important global environmental issues or the (international) public good concepts are in biodiversity conservation (Perrings et al. 2002), especially through mega-databases and information systems (Gaikwad and Chavan 2006). Many human uses of ecosystem services interact in ways that affect ecological processes. These interactions entail collective costs and benefits. But there is no incentive for individuals take these interactions into account in deciding how they use ecosystem services.

Demographic Drivers The human population, which exceeded 7 billion people in October 2011, is a major indirect driver of changes in biodiversity (Palumbi, 2001). Two billion human beings live less than 100kilometres from the coast, and the majority of the largest cities, with over 500,000 inhabitants[6] are located within 50 km from the coastline. Coastal population densities are nearly three times those of inland areas (Kay and Alder, 2005). The rising number of people moving into coastal zones exerts direct pressure on coastal marine resources. Population size, composition, distribution density, rate of growth, and level of development can also indirectly affect biodiversity through changes in demand for marine products, industrial, farm and urban waste, pollution and alteration of coastal habitats. Today, more than 50 % of the total U.S. population resides in coastal areas, and this is projected to increase to 60 % by 2020 (Crossert et al., 2004). In 2000, there were over 11,000 beach closings or advisories (freshwater and marine beaches) in the United States, a number that had almost doubled from the previous year, and a majority of these closings were due to wastewater pollution (NRCD 2001). On a global scale, coastal development is twice that of inland sites, with approximately 90 % of the generated wastewater being released untreated into marine waters (Henrickson et al., 2001). The growth of urban cen-

[5] http://www.conservation.org/Pages/default.aspx.

[6] http://www.citymayors.com/statistics/urban_2006_1.html.

tres and rise in incomes worldwide is also expected to increase global demand both for fish products and for recreational uses of marine biodiversity, whether extractive or non-extractive (Delgado et al., 2003).

Other demographic factors such as age, gender, and education can indirectly affect the patterns of resource consumption and use. While the individual trajectories of most of these demographic factors are relatively well known (UN, 2004), what is less well understood is how the effect of these different factors (size, age) combined with economic factors and in particular increased wealth (see below) will affect the ability of future societies to control their impacts on marine biodiversity.

Economic Drivers At international, regional, national, and local levels, economic activities are critical drivers of societal impact on marine ecosystems. Economic growth increases demand for ecosystem goods and services (Hoyer and Macinnis 2010). The structure and performance of international seafood commodity markets influences the type and level of ecosystem use by fisheries on both global and local scales. Price fluctuations of inputs (e.g. fuel) and outputs (e.g. food commodities) underpin changes in activities and in resource use (Abernethy et al. 2010). Technological innovations also influence demand for ecosystem goods and services by lowering costs and improving productivity, and may lead to changes in how ecosystems are used. Labour markets determine the attractiveness of employment in marine occupations. Change in wealth has also been documented as one of the main factors affecting change in consumption patterns and consumer choice, with an overall (although not systematic) positive correlation between societal wealth and willingness to pay for biodiversity conservation (Chukwuone and Okorji 2008). And finally, while globalisation of markets and trade leads to greater integration of producers and consumers over large geographic scales (Gereffi 1999; Mullon et al. 2009), the effects on biodiversity remain poorly understood and documented (Toly 2004; Zimmerer 2006).

Social Drivers

Governance Reforms Globalisation of markets and trade has been accompanied by a shift of governance away from national-level institutions towards the international arena, made official through international bodies, conventions and treaties. Simultaneously, reforms for devolution or decentralisation and the related subsidiarity principle, which places governance at the lowest (least centralised) effective level, have been adopted in developed and developing countries alike, especially with regard to the governance of natural resources (Ribot 2002). Market-based governance, including Fair Trade, tradable quotas, corporate responsibilities and public-private partnerships, is also increasingly presented as having the potential to address issues related to the conservation of natural resources and biodiversity (Raynolds 2004; Wilenius 2005).

The devolution of responsibilities towards the users, as well as towards international bodies, has often been presented as a positive shift to create a more supportive policy environment for resource management and biodiversity conservation (Pinkerton 1989; OECD 2003). However, current decentralisation experiences show rather mixed results and ambiguous outcomes (Dupar and Badenoch 2002; Béné et al. 2009). As for international treaties, Kyoto and Copenhagen provide striking evidence that global governance does not ensure effective interventions. Market-based management approaches, while offering the promise of positive incentives for biodiversity conservation, are still too new to provide definitive evidence of performance.

Poverty Levels and Food Security The poverty-environment nexus is one of the most heated debates in current scientific literature, with proponents and opponents of the equation "poverty = environmental threat" still unable to find consensus (Duraiappah 1998; Adams et al. 2004). While poverty certainly should not have a positive effect on resource/biodiversity conservation, the negative relationship between poverty and the environment seems to be context-specific and not a universal rule (Cavendish 2000). It should also be remembered that, in the fisheries context, high levels of capital investments and wealth of large-scale fleets in developed countries do not necessarily create a more effective context to sustain marine biodiversity, as borne out by the status of ecosystems in the European Exclusive Economic Zone. The link between food security and biodiversity is even more complex. When a food-insecure population is exploiting aquatic resources at levels which are unsustainable in the short-term, it is important to not lose sight of the fact that the root causes generally relate to structural issues beyond the local management scale, highlighting the complex and multi-dimensional nature of the problem (Béné and Friend 2010).

Cultural Drivers Culture is the expression of shared knowledge, values, beliefs, and norms. It may be shared within communities of various types, including national, regional, ethnic, occupational or organisational. Scientific knowledge, through its various dimensions, is also an expression of culture. Because it shapes world views, influences social priorities and determines the bounds of what is acceptable in terms of values, culture is a primary driver of biodiversity conservation (Yamin 1995)—but also conversely of biodiversity decline.

More subtly, culture and norms can also influence the way biodiversity is perceived, and therefore the tools, conventions and laws used for dealing with conservation issues. To take the example of Threatened and Endangered Species, the tendency has been to spotlight the most remarkable or iconic of them (large marine mammals, turtles or seabirds), to the detriment of other less remarkable species or other "ordinary" components of biodiversity. Moreover, the fact that some ecosystem components are considered as resources or goods in certain cultural contexts, but not in others, will influence the way societies or communities deal with biodiversity.

Unknowns

There is much we do not know about the net outcomes of human drivers, their complex interactions and the incentives they create.

Institutional Although the need for consistency in institutional layering is well-established, not enough is known about how to design and secure that consistency. Additional research is needed (1) to document the decision-making processes which determine the day-to-day governance of marine biodiversity uses, (2) to examine the incentives created by alternative institutional tools, and (3) to design control policies at different levels that have the appropriate incentives for natural resources and biodiversity protection (Yamin 1995; Ruddle 1998; Baland and Platteau 1999), as well as the related key performance indicators. That said, the example of community-based co-management in forestry and fisheries shows that the design and application of such indicators is not necessarily straightforward (Thompson 1999; Borrini-Feyerabend et al. 2000).

Demographic Aggregated estimations of population size and their rates of increase in coastal zones are available worldwide. What is less understood are the specific pathways of influence between human demographics, demand for marine resources and impacts on marine biodiversity Further research is required on how age, gender, education and other demographic factors influence patterns of marine resource use, understanding of risks to marine biodiversity and responses to conservation regulations. Better understanding of the interactions of demographic factors with economic factors and their resulting impact on marine biodiversity is a true priority.

Economic The dynamics underlying economic drivers on different scales of time, space and economic organisation are poorly understood. The effect that global integration of producers and consumers has on biodiversity on local scales needs further documentation and analysis (O'Hara and Stagl 2001; Zimmerer 2006). The potential for international markets and trade agreements to either promote or erode biodiversity is not well known, and the same holds true for the economic policies of governments and inter-governmental organisations (Lawn 2008; Czech 2008). The different values society places on changes in the availability and quality of ecosystem goods and services, many of them not subject to market exchanges, remain unquantified in many cases. Although growing emphasis is placed on the need to understand changes in ecosystems, models for marine resource use still often offer an aggregated and static representation of human interactions with nature. The need is there to develop more dynamic representations that take full account of the diversity of agents and of their interactions with marine ecosystems on different scales. These include short-term and local-scale decisions such as allocation of fishing effort, choice of shipping routes or compliance with environmental regulations by individual ship's masters, as well as longer-term and wider-ranging actions, such as choices by firms to invest or disinvest, taking the influence of economic, social, institutional and ecological contexts on such decisions into account. How economic incentives of fishery regulations and other extractive activities can be designed to promote biodiversity conservation should also be given more study in specific contexts. The impact of policies to control the negative external effects of economic activities on biodiversity is another area where further investigation is needed. The link between levels of wealth and preference for a type or level of marine resource

Fig. 6.2 Taken from Duhamel du Monceau and La Mare 1769, *Traité général des pêches*, Sect. 1, plate XVIII

use is poorly understood in specific contexts. And finally, more research is needed on labour force mobility and labour markets in marine sectors.

Social

The social and political elements of governance reform need to be better understood, including the distributional impacts of new governance arrangements (Johnson 2001). Distribution of costs and benefits of decentralisation is often not fully understood, nor is the link between decentralisation and incentives to conserve or destroy biodiversity (Ribot 2002). The role of normative and social influences in compliance with regulations aimed at protecting biodiversity should also be better studied (Hatcher et al. 2000). What are the impacts of subsidiarity in developing and developed countries? Many aspects of the social impacts of market-based governance need to be better understood. What factors contribute to the success or failure of decentralised approaches to resource management (Campbell et al. 2001; Béné et al. 2009) and what outcomes will they have for biodiversity? The policy context of food security and its link to marine biodiversity also needs further understanding. The relationships between poverty and environmental outcomes are directly related to biodiversity, and there is still much room for research in this field.

Cultural

Ultimately, the cultural dimension shaping the values given to uses of marine ecosystem services should be better analysed within specific local and national contexts. The influence of culture on behaviour is of direct relevance in designing effective biodiversity protection measures (Fig. 6.2).

Chapter 7
Integrated Scenarios and Policies

Policies and Decision Support

A wide range of options is available to society, especially stakeholders and managers, to move towards the goals of sustainable use, conservation, and restoration of marine biodiversity. Management approaches include protected areas to preserve ecosystem state and habitat restoration to maintain ecosystem services. A series of additional tools is being developed to enable decision-makers to (1) take ecosystem value into account in their decisions (e.g. in managing recreational uses and tourism), (2) recognise diffuse ecosystem benefits on a local scale, (3) establish property or access rights for living resources (e.g. ITQs or managed quotas) and (4) disseminate knowledge and raise public awareness about ecological footprints, life-cycle analyses, green- or eco-labelling, etc. Choosing from alternative policy options will lead to specific consequences, depending on the indirect and direct drivers of social demand for ecosystem services, and outcomes on marine ecosystems and human well-being. Achieving these policies will rely on standard management instruments (quotas, restricted access, etc.), economic incentives (taxes, subsidies), enforcement schemes, partnerships and cooperation, sharing of information and knowledge and public and private action.

As emphasised in Chap. 3, one of the main obstacles to sustainable exploitation of marine living resources is due to two attributes of these resources. First, since their renewal is closely connected to ecosystem state and dynamics, it involves risk and uncertainty. Second, they are what are called common-pool resources. Based on this duality, policies for restoration and sustainable resource use can be classified into two broad categories, i.e. conservation measures and access regulations (Troadec et al. 2003; Thébaud et al. 2007). The purpose of the former is to preserve the attributes of marine biodiversity which give it its value, for instance, the growth and renewal potential of fisheries stocks. These conservation measures have been widely adopted internationally, on various scales and under many different forms in practice. The latter—i.e. measures to regulate access- aim at explicitly resolving the problems generated by this "common-pool" nature of marine living resources, by designing mechanisms which limit the negative aspects of competitive use. In both categories of policy measures, approaches can be based on norms and administration, on the use of economic mechanisms (incentives, or taxes) or on combinations of these options.

P. Goulletquer et al., *Biodiversity in the Marine Environment*,
DOI 10.1007/978-94-017-8566-2_7, © Éditions Quæ, 2014

Given the multiplicity of uses of marine living resources, the complexity of their impacts on ecosystem processes, and the ensuing uncertainty regarding interactions between human activities via the ecosystem, this is a particularly challenging task.

What is more, using World Bank data (Kaufman et al. 2009), as their basis, Smith et al. (2010) stressed that since most of the world's seafood comes from regions with weak governance, this must be strengthened and improved to ensure food security. The choice of policies and tools will be greatly influenced by both the temporal and the physical scale of the systems as well as the political and legal context. It will also be affected by the uncertainty of outcomes, climate changes, cultural contexts and the desired goals. Management bodies at different levels have different response options available to them, so special care will be required to ensure coherent policies and coherent governance across scales.

Developing Scenarios

Scenarios can help summarise what is known about various options and policies, while highlighting and communicating the possible trajectories of marine ecosystems and biodiversity in coming decades. Scenarios take the uncertainty inherent to the simplification of complex system dynamics into account. They do not provide predictions, but rather describe likely alternative future states, each one depicting an outcome corresponding to a particular set of assumptions. "Status Quo" and "Business As Usual" scenarios constitute relevant baselines in this respect. Generally speaking, scenarios can be used systematically for thinking creatively about complex, uncertain futures. When defining and implementing present and future remediation and adaptation policies, they are invaluable in assessing the consequences of choices and trade-offs adopted (Fig. 7.1).

Five main types of complementary approaches are used to explore the variety of possible outcomes indicated by different scenarios.

Qualitative Learning from Past Experience

Developing regulation and control policies to guide the use of marine living resources towards sustainability objectives is not new, and past experience can inform the analysis. It is useful when designing alternative approaches and exploring them through scenarios. Availability of adequate data may be an issue, but we known that significant qualitative information can be obtained from hindcasts of empirical data, with a multidisciplinary perspective—as exemplified in the famous fisheries science lectures given in 1994 by R.J.H. Beverton (Anderson 2002). Typically, retrospective analyses or hindcasting make it possible to estimate the lead time needed to implement effective management measures. This can be illustrated by the examples of economic, social and ecological costs incurred by the chronic overcapacity of fishing fleets. For instance, in the 1990s a macroeconomic study by FAO (carried

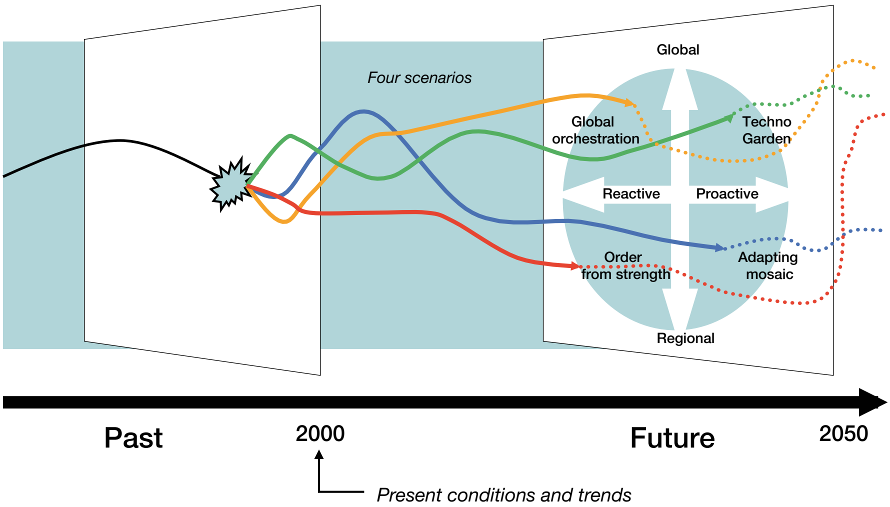

Fig. 7.1 An example of global scenarios for biodiversity: (MA 2005). *Four scenarios* are considered for the period from 2000 to 2050: *Global orchestration*, *Order from strength*, *Adaptive mosaic* and "*TechnoGarden*" or green-tech management

out by F.T. Christy, C.H.B. Newton and S.M. Garcia, who took account of costs not borne by professional fishers and thus not apparent on their financial statements) concluded that the cost of world fleet overcapacity was US$ 50 billion per year. When calculated using (unsubsidised) real costs, the approximate estimate obtained highlighted the extent of compensation by state aid. Taking this analysis further, in 1998 the World Bank estimated that subsidies to the fishing sector ranged from US$ 14 to 21 billion per year (Milazzo 1998). These pioneering studies ushered in the still-ongoing process of reducing subsidies. For instance, in the European Union, subsidies to replace fishing vessels were stopped on the 1st January 2005. More recently, the World Bank and FAO estimated that the confidence interval for the difference between the actual and potential net economic benefits of marine fisheries would be between $ 26 and 72 billion per year if they were sustainably managed. The aggregate deficit would be in the vicinity of US$ 2,200 billion for the period from 1974–2007. These "sunken billions" exclude consideration of losses to recreational fisheries and to marine tourism and losses attributable to illegal fishing are not included. Echoing the conservation and regulated access measures mentioned above, the World Bank and FAO made several recommendations to see the necessary reform of the sector through successfully, without social hardship, particularly indicating that "*the most critical reform is the effective removal of the open access condition from marine capture fisheries and the institution of secure marine tenure and property rights systems. Reforms in many instances would also involve the reduction or removal of subsidies that create excess fishing effort and fishing capacity*" (World Bank and FAO 2008).

As for the European Union, Cappell et al. (2010) assessed the environmental and social impacts of the Financial Instrument for Fisheries Guidance (FIFG, with total allocations of 4.9 billion € in funding between 2000 and 2006). The FIFG, created in 1994 to support the fisheries sector in Europe, was replaced in 2007 by the European Fisheries Fund (EFF). Under the FIFG, the objective was to bring the fishing capacity of the European fleet into line with available biological resources. Cappell et al. (2010) concluded that the FIFG had not achieved the intended net fishing capacity reduction. In fact, the contrary is true, where some fleet segments increased their capacity thanks to this funding and thus contributed to making the status of some stocks worse and hindered the recovery of other stocks. Indeed, apart from the measures considered as having a neutral effect on fishing capacity (the case for 54 % of subsidies), 29 % of FIFG funding fostered overcapacity (vessel construction and modernisation), whereas only 17 % went towards measures promoting sustainable fisheries (scrapping and temporary cessation of fishing activities).

Other examples of the inertia in fisheries sector governance can be cited (*inter alia*, reduction of discards), where the common point is that more than a decade has gone by between the moment that a management measure has been recognised as being appropriate and when it is effectively applied.

Quantitative Learning from Past Experience

As indicated in Chap. 3, statistical analysis of past ecological or socio-economic data and time series can help quantify the role of environmental or human stressors on biodiversity Possible approaches include fitting dynamic model results to historical data, the most rudimentary of them being virtual population analysis (VPA) or simple ecosystem models like EwE (Ecopath with Ecosim). Calibrating such models provides a quantitative basis—often orders of magnitude—as well as margins of uncertainty to explore the potential impacts of different management policy scenarios.

Learning from Analytical and Mathematical Reasoning

Theoretical analysis of simplified, aggregated or stylised models can help in understanding the role played by certain processes and how they generate counter-intuitive outcomes. It can help in conceptualising strategies, identifying tipping points and even in assessing and ranking management policies. For instance, for fisheries, widespread use has been made of dynamic equilibrium or optimality approaches to gauge relevant strategies and determine where stocks stand with respect to management targets. The slow adoption and implementation of Maximum Sustainable Yield, or MSY, provides a good example of this. According to the FAO definition, MSY is the highest theoretical equilibrium yield that can be continuously taken (on average) from a stock under existing (average) environmental conditions without significantly affecting the reproduction process[1] (Fig. 7.2).

The MSY concept was developed in the 1930s by several authors, but its use as one of the possible management reference points—and then as an international

[1] http://www.fao.org/fi/glossary/.

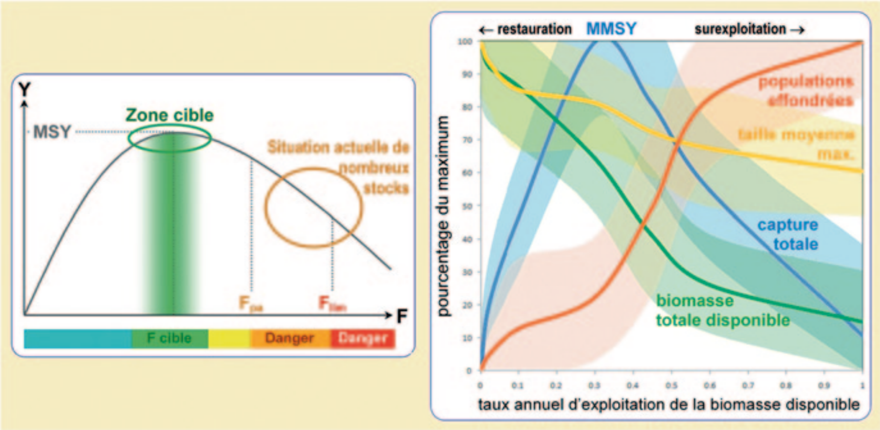

Fig. 7.2 Trends in fisheries resource-catch management targets: from a "stock by stock" approach to take account of impacted species communities. *Left*, single-species case: equilibrium or long-term average total yield Y (landing + discards) *vs.* fishing mortality rate. The maximum catch value at equilibrium is called *MSY* (Maximum Sustainable Yield). *Right*, multispecies case: equilibrium *MMSY* (Multispecies Maximum Sustainable Yield) and covariates (maxima scaled at 100 %) *vs.* the yearly fished proposition of the exploited community biomass. (From Worm et al. 2009)

standard for stock rebuilding strategies—became established two decades later with the advent of Surplus Production models.

As Mace (2001) indicated, much has been written and published about how MSY is used—and misused—all the more so seeing its use in the important United Nations Convention on the Law of the Sea (UNCLOS)[2] in 1982, thus paving the way for its inclusion in national fisheries acts and laws. Several conventions, policy instruments and other international laws related to fisheries management refer to MSY, *inter alia*: Chap. 17 of Agenda 21 (1992), the 1995 agreement for the conservation and management of straddling fish stocks and highly migratory fish stocks, the FAO Code of Conduct for Responsible Fisheries (1995), the World Summit on Sustainable Development plan of implementation (2002) and the Green paper for the reform of the European Common Fisheries Policy (European Commission 2009).

MSY is currently regarded as a robust indicator of the required direction of change in fishing mortality rates in order to achieve biologically optimal exploitation. In spite of its drawbacks (particularly the fact that it is a single-species indicator), MSY is a readily understood and operational concept (Gros et al. 2008). Furthermore, while not omitting counterexamples, there have been several 'success stories' of substantial increase in biomass of fish, shellfish and crustacean stocks following significant reduction in fishing pressure, as in cases reported by Mace (2004). More recently, Beddington et al. (2007) published a review of fisheries management systems where economic rent is taken into account in the MSY, an approach which leads to the concept of Maximum Economic Yield (MEY). In order to include fisheries' impacts on the ecosystem, Worm et al. (2009) expanded this

[2] http://www.un.org/depts/los/convention_agreements/texts/unclos/UNCLOS-TOC.htm

comprehensive overview by using the Ecosim model to simulate assessments of some thirty communities exploited, according to various scenarios, with reference to the Multispecies Maximum Sustainable Yield (MMSY; Fig. 7.2). Contrary to MSY and MEY, the MMSY indicator is not yet fully operational.

Learning from Virtual Experiments (in silico)

The exponential growth of computing power has greatly broadened our ability to explore the dynamics of complex systems which do not lend themselves to controlled experiments. Over the past two decades, research has rapidly developed in formal modelling applied to the analysis of marine social-ecological systems. We can now test our understanding of these systems and predict possible outcomes of alternative management options (see ANR Chaloupe project[3]).

Learning by Doing

In some cases, particularly where levels of uncertainty are such that qualitative assumptions cannot be made about key processes, knowledge can be progressively acquired through monitoring and research programmes closely related to the definition of policy scenarios. In this case too, modelling can be used to formulate hypotheses about social-ecological systems and how knowledge about them can be brought up to date with new information.

Quantitative Methods, Models and Integrated Assessment

There is a recognised need to develop, integrate, and promote the development of scenarios for marine ecosystems under anthropogenic and natural forcing. It proves the interest of an integrated strategy providing guidance in the "post-Aberdeen" context (see 'EurOCEANS 2007' Conference[4] to organise research in coordination with key groups and programmes (e.g. ICES, IMBER, MarBEF, Eur-Ocean). In 2011, the European Commission proposed a joint programming initiative (2011/C-276/01[5]) called *Healthy and Productive Seas and Oceans.*

Likewise, an integrative research study focusing on changes in marine ecosystems could be launched under the auspices of the IPBES (Intergovernmental Platform on Biodiversity and Ecosystem Services).

[3] http://wwz.ifremer.fr/guyane/Nos-activites/Viabilite-des-systemes-halieutiques/CHALOUPE.

[4] http://ec.europa.eu/maritimeaffairs/declaration_en.html. A New Deal for Marine and Maritime Science, June 2007. Aberdeen Declaration.

[5] http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=OJ:C:2011:276:0001:0003:FR:PDF.

Coupling Ecological, Environmental and Socioeconomic Models

Ecological models can be used to measure the effects of various management policies (quotas, marine reserves, consequences of climate change on species distribution, etc.). There is no overarching global model of marine systems, and for the moment such a model remains out of reach. However, numerous models based on an ecosystem approach throughout the world do exist and are applied to different regions. While dealing with only relatively simple models focused to a great extent on trophic relations, Plaganyi (2007; see also Hoggarth et al. 2006) analysed the advantages and limits—from the ecosystem approach viewpoint—of different modelling using Ecopath and Ecosim software or the statistical method called MSVPA (Multi-Species Virtual Population Analysis, actually a VPA which is generalised by taking predator-prey relationships into account). These modelling approaches which can be useful in planning and assessing fisheries management were also discussed by Marasco et al. (2007).

Garcia and Charles (2007, 2008) provided some perspective in reviewing the growing complexity of scientific representation of the "fishery system" during the twentieth century. Dynamic models were developed concomitantly as fisheries management and fisheries research evolved together. They were built on the basis of inputs from various disciplines involved in natural resource management and became increasingly complex in response to changing societal demands (Fig. 7.3). Reinforcing the streams of relationships between science, policy-making, and society within complex fishery systems, and between those systems and their environment, has led to increasing scope, detail, realism, and interdisciplinarity.

By using scenarios, the future of marine biodiversity can be explored against the backdrop of climate change. On the world ocean scale, Cheung et al. (2008, 2009) used a bioclimate envelope model to project changes in species distribution ranges for 1,066 marine fish and invertebrate species[6]. The authors identified the ocean regions which could become the most highly impacted by 2050 by calculating the average frequency of immigration and local extinction from 2040–2060, relative to the mean species richness for the period from 2001–2005 (changes are expressed as percentages). The intrinsic population growth rate of these 1,066 species was determined by the species' environmental preferences, varying as a function of the environmental conditions calculated for each cell of a 30' latitude × 30' longitude model grid of the world ocean. Thus the projections take account of population dynamics in bioclimate envelope models for various greenhouse gas emission scenarios (in this case, the IPCC's A1B scenario). Cheung et al. (2010) also tried to assess the potential impact on fisheries yields by 2055. However, gaps in knowledge make it necessary to apply a number of simplifications. In their present form, bioclimate envelope models applied to marine populations do not take account of the following: (1) biological interactions (e.g. predator-prey); (2) the factors shaping the spatial dynamics (*inter alia*, connectivity, which is not well known); (3) intra- and inter-community interactions and community response to synergistic effects of

[6] Data from *FishBase* (http://www.fishbase.org) and *SealifeBase* (http://www.sealifebase.org).

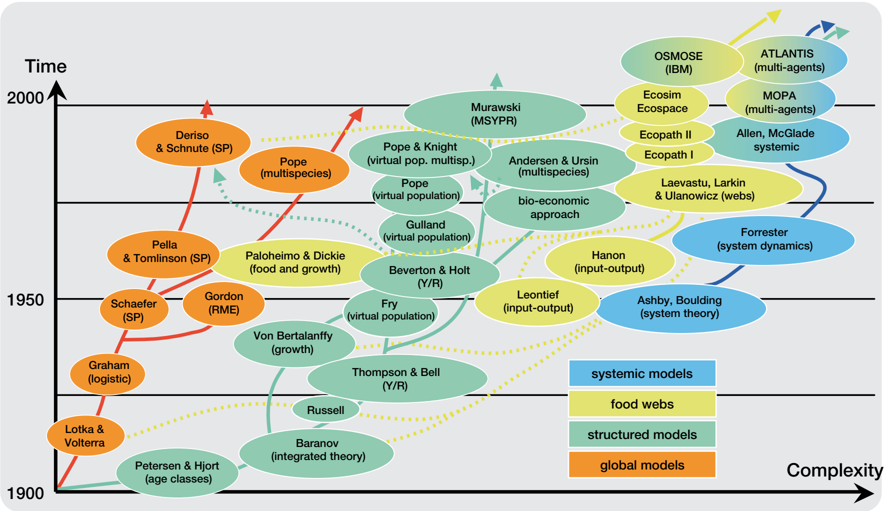

Fig. 7.3 Evolution (1900–2005) of fishery population dynamics models towards models of ecosystem dynamics and resources uses. After the pioneering research work in the early twentieth century (Lotka, Volterra, Hjort, Petersen) came the deterministic "stock assessment" models in the 1940–1950s used to estimate various indicators (maximum sustainable yield, yield-per-recruit). Global models (in *orange*: a single state variable, stock biomass, then organised by age and/or size classes in *green*) led to the development of virtual population analyses (VPA). In the 1980s broader concepts were taken (system approach, in *blue*) encompassing climate variability, trophic interactions between exploited and non-exploited populations (in *yellow*) and bio-economic coupling. Current developments are incorporating the dynamics of food webs and their exploitation in a changing environment (*OSMOSE*: Object-oriented Simulator of Marine ecOSystem Exploitation platform) or addressing the complex dynamics of marine resources and the drivers of their uses (ATLANTIS). (From Garcia and Charles 2007, 2008)

global change, climate change being just one of the components; and (4) phenotype plasticity and species' potential to adapt. Even though Cheung et al. (2011) included some additional elements of complexity (abundance of two plankton size categories, oxygen content, pH, fisheries yields) in their focus on the North-East Atlantic, the limitations indicated in points (1) to (4) still hold true.

The previously mentioned simplifications highlight the need for greater knowledge in several fields, both in terms of monitoring (to identify current distribution and species' preferred habitats at each stage of their ontogenic development) and research, e.g. to understand the linkages between demographic and environmental stochasticity in marine populations, to cite just two examples (Gros 2011). In other words, the projections presented can be taken as a "null hypothesis" which can be used to identify both monitoring system developments and avenues of research.

In human and social sciences, for example, modelling can be used to study the impacts of global change on yields, consumption and investment decisions in the fisheries sector. Market dynamics models which cover the exploitation of marine resources, i.e. both fished stocks and aquaculture production as well as individual transferable quotas, can provide diagnostic tools. For instance, the global fishmeal and fish oil market can be modelled based on small pelagic fish yields in a changing context (including looking for substitute food products, etc.) of global constraints, ranging from world market prices for fishmeal to local productions (Mullon et al. 2009). A key challenge lies in taking account of social, cultural and political drivers in bio-economic models.

Integration and scale-shifting are also being extended to integrated coastal zone management models, which combine the processes affecting both catchments and marine areas. More generally speaking, more systematic applications of integrated models which combine the interactions between ecological, human and climate systems will be seen on global and local scales.

Diversity vs. Homogeneity of Models

We argue that there should be a variety of modelling approaches to choose from, depending on the questions asked and the scales considered. Whatever the approach, modelling provides a powerful way to formalise ideas, concepts and assumptions that different people have about the systems under study and their key drivers. It is also a way of pooling knowledge from different disciplines in natural, social and economic sciences.

But above and beyond the expected benefits of being able to choose amongst models, comparative analyses should be encouraged. Just as there are a limited number of platforms used internationally, a small number of generic ecosystem models should be used to compare evolutionary scenarios in Large Marine Ecosystems. This should be a globally consolidated approach to be developed on regional scales, similar to that used by the IPCC.

Along the same lines, special attention should be devoted to calibration methods and to reconstructing past trajectory dynamics, (such as the states and trends described in Chap. 3). This will require developing permanent, integrated databases that can show the interlinkages of marine biodiversity and its uses, as is done in fisheries information systems (like SIH at Ifremer).

Modelling: Scenarios and Assessment Challenges

How useful scenarios are depends on the ability of the underlying models to cope with the complexity of drivers and mechanisms of integrated systems. The recognition, and even the confidence, shown to scenarios is closely linked to their own effectiveness in making assessments. Indeed, a range of tools is now available to help decision-makers choose among strategies and interventions, particularly cost-benefit analysis, cost-effectiveness, impact assessment, risk assessment, multi-criteria approaches or game theory. The choice of analytical tool should be determined by the context of the decision, key characteristics of the issue raised and the criteria considered to be important by the decision-makers. Integrated assessments should also reinforce the credibility of models and scenarios and make their use relevant, seeing their ability to incorporate ecological and socio-economic aspects (and related performance criteria). In order to obtain clear and useful assessments, it is essential to deal consistently and rigorously with uncertainty, in spite of the inherent difficulties in doing so (e.g. see the IPCC experience). This means that modelling of scenarios and assessing marine biodiversity raise a number of cross-cutting questions and topics in the following fields:

- complex dynamic systems;
- multi-criteria issues;
- sustainability issues;
- risk analysis and management;
- governance and coordination issues.

Complex Dynamic Systems

Single-species models (e.g. MSY, or the precautionary approach) for the management and regulation of fisheries and marine biodiversity remain insufficient to answer the questions raised by the ecosystem-based fisheries management (EBFM) approach Today the need to use complex models, typically covering multi-species and multi-fleet fisheries for their management, is widely recognised. One example, i.e. enlarging the MSY concept to MMSY, is mentioned above. More generally, modelling to support the management of marine biodiversity must take account of the complexity of ecological mechanisms, including community dynamics, food webs, meta-populations, geographical processes, and environmental (habitat, climate) features. It is also important to focus on the complexity of socio-economic drivers. This is particularly true in fisheries, as exemplified by efforts to adjust catch capacity in a multi-fleet context cross-checked with market dynamics for ITQ systems (spatial scales being yet another source of complexity here). All this advocates the use of tools which can be applied to non-linear dynamic systems and complex systems, such as network theory or graph theory (Fig. 7.4).

Multi-Criteria Issues

The need to adopt a multi-criteria perspective in defining sustainability has been stressed in many contexts, including marine fisheries (Charles 1994). The implicit scope of the definition covers ecological, economic and social objectives, to which a number of authors have added the political objective of reducing conflicts (Hilborn 2007). In this context, use of a cost-benefit approach that would apply the same metrics (i.e. monetary value) to all variables runs up against the major difficulty of trying to model biodiversity in financial terms.

Thus the challenge in modelling social-ecological systems is to develop approaches that explicitly comprise the diversity of sustainability objectives and provide a formal assessment of trade-offs. This may involve the use of multi-criteria decision-making methods in response to the difficulty of identifying, quantifying,

Fig. 7.4 Shoal of yellowfin tuna. (Seychelles islands; © Ifremer, Marc Taquet)

ranking and optimising objective indicators which are considered relevant across a range of stakeholders at different scales. In other cases, it is advisable to identify thresholds for certain key variables beyond which the system is no longer considered viable, and maintain long-term sustainability by setting specified constraints to stay below threshold values. Exchanges between model developers and users are central to the effectiveness of such decision support tool applications, as are the tools and approaches to design these interactions, from the phases of model framing and development to calibration and presentation of model results.

Sustainability and Intergenerational Equity

It is vital to adopt the perspective of sustainability in assessments and modelling of marine biodiversity so that both short- and long-term objectives can be reconciled. This means designing scenarios that are relevant in setting mid- and long-term targets for management policies. In this respect, approaches focusing on inter-generational equity should be given particular attention. Economic projections use discounting rates, which usually leads to depreciating the future and preferring the present—even, in some cases, as emphasised by Clark (1976), to the point of justifying a rationale that can lead to extinctions. Equilibria, steady states and classic sustainable yield indicators like MSY or MEY are more natural choices of criteria for intergenerational equity. However, the way to apply these reference points in more complex and uncertain contexts, as in the ecosystem approach, remains unclear. The maximin or Rawlsian approach (Solow 1974) or the viability framework focusing on constraints to be satisfied simultaneously along time (Béné et al. 2001; Delara and Doyen 2008; Baumgartner and Quaas 2009) could also be appropriate in addressing sustainability.

Precaution, Risk Analysis and Management

Establishing realistic estimates of the magnitude and direction of change in biodiversity under a range of scenarios is constrained by the interplay between the ongoing unpredictable and continuous alterations to marine ecosystems driven by natural and man-made change. For example, some species become locally extinct, others are introduced, the abiotic environment is modified through inputs of chemicals or acidification, habitat alteration affects distribution and abundance, and major extractive pressure is often imposed on most trophic levels. Moreover, the speed of such drivers is many orders of magnitude higher than the evolutionary time that it has taken to derive communities and ecosystem function. Amongst the numerous uncertainties related to understanding marine biodiversity dynamics are environmental stochasticity (habitat, climate), socio-economic uncertainties like market fluctuations and measurement errors.

This makes analysing vulnerability and bio-economic risks of prime importance. For mitigation purposes, irreversible "disaster scenarios" must be avoided. These issues are directly linked to aspects of resilience and the ability of systems to recover from disturbances and stress. In this field, one key approach which has proved to be especially insightful is the merging of food web research and biodiversity research to enhance our understanding of resilience (Worm and Duffy 2003).

More generally, risk assessment and management can play a major role in the decision process. Risk assessment is a well-established discipline, which identifies thresholds and evaluates possible damage. Similarly, environmental impact assessments designed to evaluate the impact of specific projects and strategic environmental assessments designed to evaluate the impact of policies are two mechanisms for making findings of ecosystem assessments part of decision-making processes. Population viability analysis (PVA), which focuses on extinction risk, is a major quantitative method for conservation biology (Fig. 7.5).

Several difficulties in operational risk assessment should be highlighted. The first is that in many cases the probability of achieving the different conditions of the alternatives is unknown, which is an impediment to the use of stochastic models. Typically IPCC climate scenarios describe different possibilities for the future, and an approach which jointly examines the most probable and the worst-case scenarios can be highly productive. Another difficulty is due to very high degrees of uncertainty which can make the effects of management scenarios impossible to detect in *ex ante* assessments. The Precautionary Principle emerged in this context, holding that the lack of adequate scientific information should not be used as a reason for failing to take conservation or management measures (although how to put this into operation is often the subject of debate).

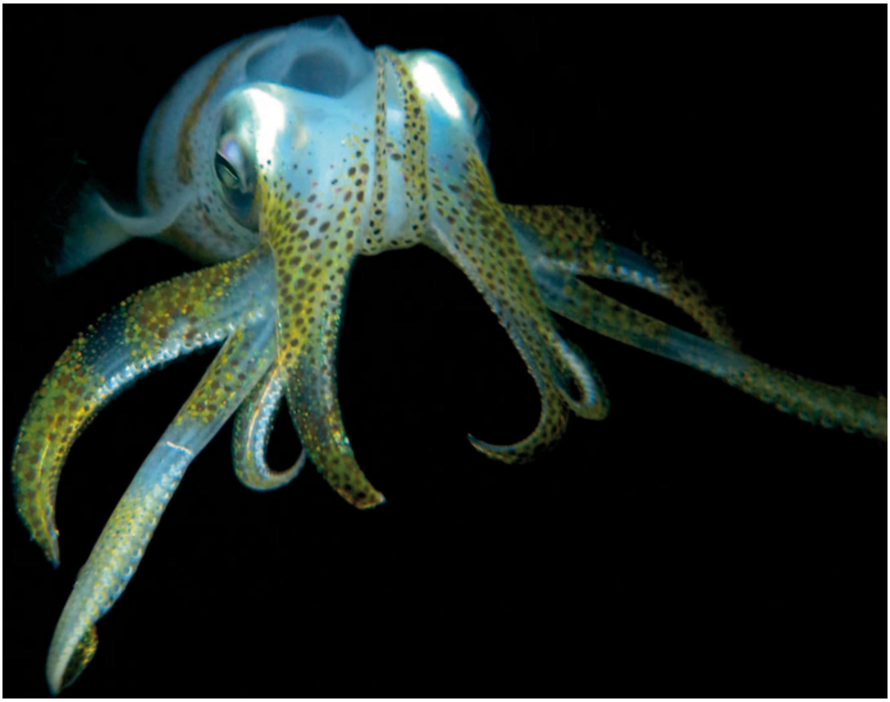

Fig. 7.5 New Caledonian cephalopod. (© Ifremer, Hugues Lemonnier)

Adaptive Management

In terms of ecosystem services and human well-being, adaptive management (AM) is expected to improve the outcomes of policy decisions. This approach draws on past experience to hedge against risk and uncertainty (i.e. our understanding of marine ecosystem dynamics will always be limited, socio-economic systems will continue to evolve and outside drivers can never be fully anticipated).

Decision-makers must therefore consider whether a course of action is reversible and whether or not it should incorporate, when possible, procedures to evaluate the outcomes and future benefits. Adaptive management is a structured, iterative process of decision making in the face of uncertainty, with a goal of reducing uncertainty over time via system monitoring (Holling 1978; Walters 1986).

In this way, decision-making simultaneously aims at reaching objectives or limiting risks, as well as accruing the information needed to improve future management. AM is often characterised as "learning by doing". Key methodological features of adaptive management are feedback decisions and Bayesian inference. Adaptive management is particularly applicable to systems in which learning via experimentation is impractical.

Fig. 7.6 Alcyon (*Alcyonium digitatum*): seafloor landscape in Brittany. (© Ifremer, Olivier Dugornay)

Governance, Coordination and Compliance

The shift of governance toward more participation and the inclusion of the civil society and other 'non-expert' stakeholders in the planning and management of natural resource uses are based on two main assumptions. One is that including end-users in the decision-making process will improve the relevance and quality of the decisions made, and the second is that compliance with and acceptability of these decisions by these end-users will be greater. This principle is increasingly promoted in scenario and modelling exercises where end-users and stakeholders other than the traditional resource managers are now often invited to the "experts' table" to test, use, and discuss decision-support tools. When so employed, modelling exercises can help these non-expert stakeholders to engage more actively in the decision-making process and thereby contribute to reinforcing the acceptability and accountability of these scenarios and models as tools in discussing critical trade-offs (especially those affecting inter-generational equity). Finally, using these participatory modelling exercises can also be viewed as the first step towards establishing adaptive management processes and to strengthening stakeholders' capacities. Governance issues related to the heterogeneity of end-users and stakeholders should also be highlighted using modelling, like multi-agent or game theory models, to focus on strategic (coordination, cooperative or non-cooperative) interactions in decision-making processes (Figs. 7.6, 7.7).

Fig. 7.7 Traité général des pêches, section 1, plate IX. (Taken from Duhamel du Monceau and La Mare 1769)

Chapter 8
Research Needs

The requirements for research listed here are overarching and should be organised in future along five key structuring priorities.

- *Defining and cataloguing the characteristics of marine biodiversity and its geographical range of distribution* studying and inventorying variety in genes, species and ecosystems (habitats and landscapes) and developing tools to describe this diversity. Measuring and mapping human uses and their impact on ecosystems, goods and services.
- *Understanding the ecological and evolutionary processes* that account for the variety, quantity and quality of genes, populations, communities and ecosystems over space and time, as well as the economic and political consequences of interactions between these processes and uses. How was Nature able to engender over 1.5 billion species in less than 4 billion years? And how can Nature's response to human and environment pressures be assessed on the basis of past and present analyses and future scenarios?
- *Elucidating the mechanisms* whereby components of biodiversity influence the functioning of populations, communities and ecosystems and ensuring the sustainable production of ecosystem services, including biogeochemical processes, major cycles and relations with the non-living or mineral world, as well as the related socio-economic benefits for people.
- *Understanding and predicting* how biodiversity and ecosystem functions and services respond to human and environmental stressors, and how human uses respond and adapt to changes in biodiversity. Relying jointly on retrospective analyses, comparative approaches and developing scenarios for the future.
- *Proposing incentives, management tools, processes and policies* to effectively and efficiently protect, conserve or restore biodiversity. Determining the status and trends of services rendered by biodiversity and the effective management of their utilisation, using ecological, economic, social and organisational indicators over various timescales.

P. Goulletquer et al., *Biodiversity in the Marine Environment,*
DOI 10.1007/978-94-017-8566-2_8,

The Framework: Environmental Research

Oceans are subjected to numerous forms of stress. The world population has grown six-fold since 1800 and nearly doubled since 1970. It will increase again by at least 50 % before its growth finally experiences a downswing. We now have proof of ocean acidification which can affect the processes of biomineralisation, particularly for microscopic marine organisms like zooplankton for example. Everywhere in the world, corals are threatened or affected. Nearly half of fisheries stocks worldwide are fully exploited and 30 % of these stocks are overexploited. Aquaculture production is growing by over 5 % each year, so will double in less than 12 years, and is contributing to a notable increase in pressures on coastal ecosystems. Shipping has increased more than three-fold since 1970, disseminating invasive species. Continents of trash, made up of plastic litter, have formed in the Pacific and Atlantic Oceans and probably in other oceans as well. The increasing exploitation of offshore oil has led to significant amounts of seafloor being fitted with equipment and to risks, like the very bad spill in the Gulf of Mexico; unfortunately, oil being spread by the Gulf Stream across the Atlantic cannot be ruled out. The mining of seafloors has begun and deep-sea ecosystems are threatened by serious disruptions. Uses of marine areas are becoming more numerous and, to date, all of them have had unexpected or even unknown impacts on the ecosystems there. For instance, assessments of environmental impacts of marine tidal and offshore wind turbines are not yet fully available.

Confronted with these changes and developments, the responses from political and managerial systems still seem quite limited in scope. Decision-making processes are fragmented and decisions concerning fisheries, aquaculture or pollution still fall under specific, sector-based approaches. Despite the consensus on developing ecosystem-based approaches, existing decision-making structures cannot deal with the complexity of interactions between the trends and developments mentioned above.

Research is trying to evolve within structures which have a very hard time adapting to new issues, consequences for methodologies and material and logistic requirements. They are not just administrative and budgetary structures, but also disciplinary and academic organisations. Current scientific challenges require new methods and new tools, as well as new multidisciplinary approaches. And yet, the interdisciplinary factor which is so essential, from the very outset of research projects, remains the exception to the rule. Beyond scientific challenges, administrative and financial innovations are needed to support the development of partnerships and scientific project formats.

The current research needs have arisen from the recent, far-reaching change in innovation-related design, which lies in designing new systems rather than developing specific actions. In industrial systems, upcoming products come from particularly innovative approaches. For example, one car maker who is worried about the carbon market has decided to build a plant using zero fossil fuel and 70 % less water. The constraints the firm has defined have led them to work with other firms and to a new production system using olive pits and a eucalyptus plantation, newly-invented paint booths and new paints. The new system meets their requirements and has led to innovations in both processes and products. In an industrial context, the con-

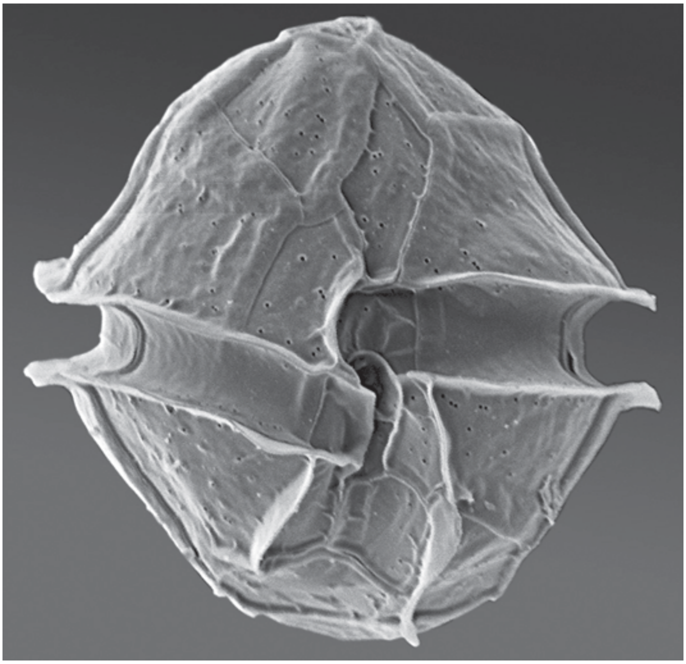

Fig. 8.1 Toxic microalgue (*Alexandrium minutum*). (© Ifremer, Nicolas Chomérat)

straints of economic and environmental viability have brought numerous companies to work together on new production systems, whose design requirements have led to innovative concepts or products. Ideally, we could imagine that the same thing could be done for the uses humans make of marine areas.

Throughout this book, we have emphasised the complexity of marine ecosystems, like that of systems of exploitation, ecological dynamics and economic and social dynamics alike (Fig. 8.1).

The weight given to research requirements must therefore reconcile both their intrinsic scientific importance and their importance with respect to the "research system" needed to understand the dynamics of biodiversity and its interactions with climate changes and human societies. Therefore, research needs must be resituated in the context of major political and scientific stakes related to the fate of biodiversity, ecosystems and ecosystem services.

Research Systems

In the face of the growing complexity of scientific, political, economic and social stakes and challenges interlinked on a multiplicity of spatial and social scales, cross-disciplinary approaches are being developed, and with them more or less well-adapted mediation measures. Scenarios, in particular, are increasingly used to enable multiple points of view to be compared and possible future outcomes to be inferred. The Millennium Ecosystem Assessment (MA 2005) has especially highlighted the interest of this approach, even though—as Hannah Arendt (1972) stressed—the global scenarios coming to the fore for the pathways assumed to be the most sustainable are based on representations corresponding to North American culture, thus reflecting the sociological make-up of collectives.

Modelling is increasingly perceived as a rigorous approach for integrating the perspectives of various disciplines which can be used to mediate different points of view. Recent developments combining role-playing and spatialised modelling also illustrate the need for mediating systems in order to move from an discipline-based approach to a research-system organisation (Janssen et al. 2010; Bousquet and Lepage 2004).

Mediation can lead to positive incentives, as illustrated by calls for projects by the European Commission and the French ANR national research agency which require cooperation between teams and across disciplines.

The large number of mediating arrangements shows that system-based research management is still in its infancy in terms of methodology. For the moment, it is materialised either through collective responses of the Millennium Ecosystem Assessment type and its extension to agriculture or through discipline-based responses: for instance, ecology has become a science, sometimes perceived as being situated at the crossroads of social sciences, earth sciences and life sciences (Couvet and Teyssèdre-Couvet 2010). In the social science realm, geography has long claimed a role equivalent to that proposed today for ecology. The "ecological economy" development also emphasises the growing efforts made to incorporate our understanding of economic and ecological systems. While these efforts should indeed be noted, they highlight the fact that research systems, as currently organised, provide only a partial response to cope with the challenges.

It is becoming more and more important to develop research on how to organise, conduct and manage "system-based research". This need for research is overarching and appears to be especially significant for research bodies with a role to play in the fate of marine biodiversity over the entire French EEZ and in Europe.

Sustaining Ecosystem Services

The overarching objective for marine biodiversity research is to conserve and, when possible, restore the availability of ecosystem services. This means shifting from the management of one service at a time to that of several services through protection of natural biodiversity. Maintaining biodiversity by conserving species richness, genetics and diverse habitats, is a prerequisite for ecosystem integrity and stability. Management practices and decision-making processes should be focused on these major stakes.

Meeting this challenge requires developing specific tools and methods, particularly to assess the costs of conserving and restoring ecosystem services, so that they can be compared to the benefits generated by degrading these services (CAS 2009). Until now, most of the monetary valuation methods for natural assets were based on stated preferences or revealed preferences specific to certain components of biodiversity. However, since services are the result of interactions between biodiversity components—of ecosystem function—they cannot be calculated as a sum of the values of these elements (TEEB 2009; Kinzig et al. 2007; MA 2005).

Conserving ecosystem services requires thorough analysis of the causes for their degradation and the "causes behind the causes" (Fig. 8.2). In the marine realm, this includes pressures such as overfishing or pollution which may themselves ultimately result from human, economic, political, social, cultural, short-term profit (preference for the present), lack of regulation or weak governance, globalisation and poverty. The environmental and ecological phenomena which can be observed must be analysed jointly with the economic, social and political contexts on different scales, from the most local to the most global. In particular, resource appropriation regimes, which go from *res nullius* or open access to private ownership, with a range of other intermediate forms of common property or common-pool resources in between, are a significant institutional dimension. Thus research is needed to understand the modes of regulation, access to and management of ecosystem services (Poteete et al. 2010).

Naturalistic Dimensions

Linking Ecological Functions and Ecosystem Services

The challenge is to understand the link between increased biological diversity and stable ecosystem function (Dulvy et al. 2003; Worm et al. 2006), meaning that both new approaches and appropriate experimental studies must be developed. Use of validated, standardised models to measure biodiversity will also be essential to secure a baseline to be used for comparisons.

Biodiversity should be measured in relation to its key components. First of all, the basic measurement can be made at the seascape level, by mapping habitats across regions. Next, an estimation of the species in these habitats will provide a baseline for estimates of richness and biological diversity. Finally, conspecific diversity estimators, including genetic variability, life history diversity or phenotype disparity, supply a range of options for adaptation to environmental change and stress. These levels, and above all their interactions, are not sufficiently taken into account, in spite of the fact that they drive the higher level response to environmental variation. A special challenge lies in linking the type and scope of marine ecosystem services to the biodiversity of habitats and communities, along with the levels, speed and nature of disturbance that they can sustain. Research efforts should be significantly increased to study the interactions and feedbacks amongst these factors.

In economic terms, the challenge is to assess the economic and social consequences of one or several ecosystem services being degraded. It also lies in assess-

Fig. 8.2 Benthic fauna of New Caledonia. (© Ifremer, Hugues Lemonnier)

ing the feasibility, when plausible, of replacing an ecosystem service by an "artificial" solution. Should this not be possible, the only solution will be to maintain the availability of ecosystem services, and therefore of ecosystem function, at a cost to be assessed. For instance, aquaculture is often presented as being a measure to replace fisheries, but it also relies on the availability of ecosystem services like water quality, absorption of waste and even production of fish feed.

In terms of improving our ability to place our understanding of ecosystem function into a predictive framework for forecasting resilience and recovery from environmental change, there is a need to identify thresholds and design studies that allow theoretical assumptions to be tested by using empirical data on key processes. The rates of speed which shape the way communities and ecosystems respond to various sources of stress should also be evaluated. The focus should be on interactions between species and the processes which determine dynamics with regard to environmental change. In this respect, Marine Protected Areas can provide relevant choices for monitoring and experimentation.

Experimentalists and theoreticians from numerous fields need to work together to develop the ability to predict when cumulative effects will exceed ecological thresholds, beyond which recovery is limited and ecosystem services compromised. Identifying these thresholds is also a high priority in economic and social terms, in order to limit situations of economic and ecological irreversibility like those seen in numerous cases where exploited marine resources have been depleted.

Measuring the Genetic Basis of Biodiversity

New tools make it possible to monitor species richness and dynamics, like the Barcode of life which also provides the opportunity to get involved in a global project using standardised methods. Seeing the range and variety of French biodiversity over the entire planet, these tools are essential for the building of global databases with open access. This is of great importance in view of the French EEZ's scope and the biological diversity it holds.

In particular, these tools can be used to describe and monitor the diversity of plankton, which are the basis of the entire food web. They can be quite useful for communities which are critically important for the structure and functioning of benthic ecosystems. They also make it possible to recover data from a wide range of sources, including gut content, otoliths, eggs, and even processed seafood to provide traceability of specific products from their source. This means that DNA barcoding offers the opportunity to draw links between species, their location and human activities all the way to the consumer (Costa and Carvalho 2007; Machida et al. 2009; Creer et al. 2010; Carvalho et al. 2010).

Many new analytical approaches have been developed that significantly improve geographical resolution (Ruzzante et al. 2000) and statistical power (Kalinowski 2005) thus making detection faster while reducing costs. Otolith chemistry can be used to link an otolith's composition to that of the seawater in the fish's place of origin (Thorrold et al. 2001). Likewise, recent advances in molecular biology have led to powerful tools for the genetic analysis of marine fish population structures (Kochzius et al. 2010). Genomic tools can now be used to directly explore the extent and dynamics of adaptive variations and to link "genetic variation" and "phenotype response".

Differentiating Evolutionary and Ecological Time Scales

A key issue underpinning much recent activity in marine biodiversity, and a necessary corollary for assessing the impact of environmental change, is to examine whether the current trends in marine biodiversity differ from historical trends. Historical perspective is needed to compare rates of change across evolutionary and ecological timescales in the absence of human disturbance.

The evolutionary timescale provides a baseline for estimating the extremes of changes in global marine diversity against which anthropogenic effects can be scaled: ecological timescales are relevant for examining the role that human-induced drivers have had on recent biodiversity change. Human activities have driven many extinctions of species and it is important to know how threshold effects could result in rapid collapses with apparently little warning. Indeed, the study of past global warming events and their impact on biodiversity can be highly informative in generating predictions based on current trends (Kenneth and Stott 1991; Crouch et al. 2001).

The relationship between biodiversity and climate is another important question, particularly for France, whose territories cover multiple climate zones. Hypotheses that species richness is greater in the tropics remain controversial, particularly for some compartments of marine biodiversity. Over a broad spatial and temporal scale, various factors interact to modulate the way biodiversity is structured. Thus, a sharp rise in temperature could lead in the long run (multi-secular time steps of thermohaline circulation) to a drop in climate gradients, and hence, water bodies with relatively homogeneous heat content. Similar phenomena occurred in geological periods long ago, and went hand in hand with a homogenisation of fauna.

Statistical phylogeographic studies provide a robust framework for testing the drivers which influenced population divergence and speciation (Richards et al. 2007). Coupling of geo-referenced occurrence localities with genetic approaches is opening new horizons for phylogeography.

Putting Fish Stocks Back in Their Ecosystems

In Chap. 2 it was shown that human activities have deeply modified genetic and species diversity, that direct impacts come mostly from overexploitation and degradation of habitats and that indirect impacts are the result of trophic cascades. In terms of biodiversity, research needs focus on the relationships between fisheries and ecological dynamics on one hand, and ecosystem-based management of fisheries activities on the other. These relationships must be explored and modelled across all scales of time and space, depending on the issues being examined (Levin and Lubchenko 2008). Once again, conceptual and methodological innovations will depend on our ability to design and implement suitable research systems (Fig. 8.3).

Fig. 8.3 Snow crabs (*Chionoecetes opilio*), Saint-Pierre-et-Miquelon, 2012. (© Ifremer, Stéphane Robert)

Impacts of Physical Amenities and Pollution on Biodiversity

As in the case of fisheries, pollution acts differently depending on trophic levels and the connectivity between communities. Previous oil spills have demonstrated the remarkable capacity of local ecosystems to recover and rebuild, although the differences between the baseline and later states have not been clearly established. Connectivity between communities and between ecosystems is a determining factor in the impact that pollution will have on biodiversity. Ifremer showed that fish in marine coastal habitats in the eastern English Channel had levels of PCB contamination which were as high as those in fish in the Seine River, leading to a ban on sardine fisheries over the entire Seine Bay (from Barfleur to Dieppe) in February 2010. This shows the importance of connectivity between ecosystems and communities in understanding biodiversity's dynamics, its sensitivity to disturbances and its resilience.

In the case of physical amenities and developments, whether located in coastal areas like wind turbines and plants using the power of currents or tides or on deep seafloors, identifying the possible impacts on communities and understanding the processes at work require both basic and applied research studies. One of the challenges is to determine and calculate the cost of maintaining the availability of ecosystem services in cases where the developments could disrupt them (*Centre d'analyse stratégique* 2009). Little is known about the impacts that physical developments in the benthic domain have on the structuring of communities and habitats. It is all the more important to develop research on the subject, since mining activities at sea are slated to begin soon. In the case of polymetallic sulphides and other strategic mineral resources, the first to be affected by the impacts will be deep-sea fauna for which we know very little about how they function and the timescales for their turnover and even less about their resilience capacities.

Human Dimensions of Research

Data Issues

Contrary to what is widely believed, there is a great amount of data related to economic, social and institutional structures and cultures. However, these data are quite scattered and depend on stakes and challenges where the sea is not significantly taken into account (*International Encyclopedia of the Social and Behavioral Sciences*; Descola and Palsson 1996). Social science publications related to coastal communities and activities are not found, for the most part, in journals specifically devoted to them, but rather in discipline-based journals (*Marine Policy*, *Marine Resource Economics*, as well as *Land Economics*, *Ecological Economics,* or *Ecology and Society*, *Applied Ecology*, *Human Ecology*, etc.).

The problem with social science data stems more from their quality and their accessibility than from their existence. Since they are often gathered for administrative purposes or by professional sectors, it is rarely possible to certify their quality and they are not collected for scientific purposes. Moreover, there is such diversity in these data that the idea of pooling them on a *Barcode of Life* model is sheer utopianism.

And yet, existing studies in various social science fields have shown that they can be used profitably, once access to them is made possible: indeed the stakes of confidentiality for economic data are quite different from those for biological data.

Seeing the usual confusion between the terms "institution" and "organisation" (Ostrom 1989; Weber and Bailly 1993) a reminder that here by institution is meant any arrangement between at least two individuals or groups that is binding on more than themselves. It can also be considered that an institution is a set of rules in use.

Although administrative data are likely to be available via databases, for institutional data, it is another story. Many of these institutional data, i.e. from multiple arrangements and sets of rules in use, are not even explicit. For instance, this is the case for consumer tastes or collective disgust for a given seafood product (e.g. crustaceans), that refer to more or less conscious norms, as well as for the impact of advertising and media, prompting at least in part a mimetic standardisation of consumption patterns.

Rules for access to and use of resources often fall under more or less formal social norms, affecting the roles assigned to age groups, genders and many other norms produced by value systems.

Cultures, Institutions, Appropriation

The representational systems of nature, and particularly marine life, reflect relationships between people with respect to this marine life. From this are derived the popular taxonomies that more or less overlap with scientific taxonomy, but which reflect the representation that local populations have forged for themselves (Johannes 1992; Helmreich 2005). Johannes showed that local knowledge could, provided that distances with academic knowledge were calibrated, be used to monitor lagoon ecosystems. Helmreich demonstrated that academic taxonomy can also result from scientists' biased representations of local cultures. Relations between people and nature are in fact the relations between humans with respect to nonhumans, so taxonomy may also reveal the role that scientific representation can play within social and political relationships.

Research has put much emphasis on cultures in the relationship between biodiversity and society. Social anthropology has shown the specificity of the social representation of Nature in the Judeo-Christian-Muslim world, that of the so-called "religions of the Book", which tends to consider nonhumans as "things to" (Descola 2005; Descola and Palsson 1993). This cultural family casts nonhumans as things or objects, at the disposal of whims or desires of humans, who alone are subjects. All other cultures consider nonhumans as sentient, thinking entities with strategic abilities and with which relationships are most often modelled on types of relations between humans, i.e. cooperation, antagonism, hostility, etc.

Thus research would benefit from treating the representations of biodiversity built by administrations as cultural facts. Indeed, they are value systems where the economy takes precedence over living organisms and where ecological and economic interests are in opposition, where norms and established procedures take precedence over organisational innovations and where there is a preference for applying command and control type regulations over market-based regulation. Administrative cultures form coherent sets of representations, values and behaviours which are far from having been sufficiently studied as such. And yet this sort of research is essential to understanding decision-making processes.

Another research need which is important for the fate of marine biodiversity is related to how resources and spaces are appropriated. In 1989, it was noted at the Montpellier symposium on "Research and small-scale fisheries" (Durand et al. 1990) that free access was the exception generally created by States, concluding that hereafter proof of free access must demonstrated if it is to be used as an acceptable working hypothesis. The situation of fisheries and coastal zones in Europe and in French overseas counties and regions illustrates this point, where open access and regulations preferred to market-driven approaches engender overexploitation of resources and environments. Appropriation regimes are a key field of research, where appropriation is not limited to property alone, whether private, public or collective/cooperative (Ostrom 1989; Weber and Reveret 1993; Weber 1996; Poteete et al. 2010). An appropriation regime refers to a series of levels of analysis:

- perceptions of the appropriated resources and areas;
- alternative uses of spaces and resources;

- methods of allocation and controlling access (Boncoeur and Troadec 2003);
- means of transferring rights to resources and areas, and outputs (physical, monetary and symbolic) across a group or across generations. Transferability can exist through the market (transferable or tradeable rights), administratively, or through donations, inheritance or alliances by marriage, depending on societies;
- sanctions or punishments in use and how they are implemented (Boyd et al. 2010; Janssen et al. 2010).

Modes of appropriation are good ways of summing up cultural differences in exploiting resources and environments. In particular, the reactions of coastal and island communities with respect to marine protected areas can be interpreted through the appropriation methods which are usual in these cultures. The further the protected area plan is from local modes of appropriation, the stronger tensions will be. When this is understood, insofar as conflicting oppositions can block the creation of a marine protected area or threaten its ability to function, the NGOs or the administration in charge of management are tending to move towards forms of co-management or decentralised, bottom-up management. Mutual presuppositions about representations of society-biodiversity relationships often lead to misunderstandings in MPA management. That is why role-playing, along with object-oriented modelling, approaches have been developed to provide mediation systems. These systems allow each stakeholder to understand the way the others perceive things, thus fulfilling one of the conditions for the viability of the protected area (Lawas et al. 2009).

Demographics and Economics

Globalisation and Biodiversity

As this study has highlighted, the effects of global integration of those who exploit and consume biodiversity on local situations are creating new needs for research and data collection to complement ecological analyses on various scales (Chap. 6). International trade's potential for erosion or conservation is not well known, nor is the local impact of national policies and international agreements on local ecosystems. The distinction between "local" and "global" is no longer sufficient in the face of phenomena which interact across all scales, giving rise to the coining of "glocal". From the economic viewpoint, dynamics which interact over various scales of time, space and economic organisation are poorly understood. Methodologies like modelling must be developed for them.

Economic Assessment and Preserving Biodiversity

A significant research effort is underway worldwide for the monetary valuation of costs and benefits associated with the conservation of biodiversity (Chap. 5) by using price signals (TEEB 2009). Very little has been done yet to assess the costs of conserving the availability of ecosystem services or restoring them if they have been degraded. The commission of the Strategic analysis centre (CAS 2009) recommended that methods be developed to calculate these conservation or restoration costs and that the costs be included when analysing projects. This will be an increasingly important research need as more and more projects to exploit ocean energy and mineral resources are launched.

Poverty and Biodiversity

It is often suggested that poverty has an impact on degrading biodiversity, but demonstrations are not always convincing. In fact, poor people exploit the environment within their reach, whereas rich populations exert their predation on a global scale as can be seen by fishmonger's stalls in rich countries or the catalogues of travel agencies. Also, the availability of living resources accessible to the poorest communities represents food and monetary security which can reduce poverty. Seeing the scope of the French Exclusive Economic Zone, this is an important research theme for the regions and territorial entities of overseas France.

Decision-Making Processes

Decision-making processes are complex objects within interacting multiple interests will carry different weight in the final decision. This means that it is no longer possible to consider a decision as the result of a choice made by the "decision-makers". Because these processes comprise multiple interactions and feedbacks, having recourse to modelling tools (see below) can shed light on the full range of possibilities. These modelling studies can be used in conjunction with experimental schemes in the fields of game theory or role-playing approaches.

The stakes for understanding and simulating decision-making processes are very important for marine biodiversity conservation, not just in applied science, but also as a major challenge in research on methodology (Sumaila and Armstrong 2006).

Developing Modelling: A Summarising Approach

Modelling is a core component in designing research systems which enhance interdisciplinarity. By enabling conceptual and methodological frameworks that are common to various disciplines to come to the fore, modelling leads to improved productivity for research as a whole. At the outcome of this collective expert review, we can clearly say that developing analytical and predictive models is the most important of research needs.

Building models which combine the oceans' physical-chemical functioning, biotic community dynamics and the impacts of human activities is of strategic

Fig. 8.4 Fishery at sunset in Guiana. (© Ifremer Guyane)

importance, due to the rapid growth of human activities at sea and in coastal zones. This strategic requirement is sadly illustrated by the Gulf of Mexico oil spill and the ongoing oil pollution of the Niger delta (*The Guardian*, 30 May 2010).

Developing models is decisive for the understanding of dynamic interactions within biological diversity, in contexts where ecosystem resilience is at stake (de Lara and Doyen 2008). Understanding, measuring and predicting the impacts of human activities on biodiversity also requires that models be developed in order to test the potential effects of administrative or market systems based on constraints or incentives. From genetic diversity to that of ecosystems, from the most local to global levels, highlighting the drivers of biodiversity dynamics and resilience will only be possible through improving existing models and designing new types. Increased model capacity has been especially useful for exploring complex, dynamic systems where experimentation is not possible, as is notably the case for economic, social and institutional systems. Analysing decision-making processes, assessing the feasibility, scope and limitations of sets of economic incentives are greatly facilitated and streamlined by using mathematical or rule-based simulations: in an uncertain world, simulation models are essential to validate whether scenarios to explore future possibilities are consistent.

A recent avenue of research in modelling is using role-playing games to involve stakeholders like users, administrators or conservation activists in the decision-mak-

Fig. 8.5 Hard and soft corals and a crinoid, on a reef drop-off between îlot Mato and îlot Kuaré, Great South Lagoon in New Caledonia. (© Lionel Loubersac)

ing process (Becu et al. 2008). Processes emerging from the role-playing approach make it possible to develop IT simulation models (multiagent systems) which can be discussed by the stakeholders. This pathway for the "co-construction" of models is particularly interesting for analysing decision-making processes and the feasibility of management tools, in metropolitan and overseas France alike.

If we consider that the main challenge of marine biodiversity research is to understand the dynamics of impacts from human activities, direct impacts from various uses of the ocean and indirect impacts such as the effect of climate change, then research on modelling clearly stands out as the main need (Figs. 8.4 and 8.5).

Sources

Databases

WoRMS(marine species, global footprint) (*www.marinespecies.org***)** The World Register of Marine Species aims to provide an authoritative and comprehensive list of names of marine organisms, including information on synonymy. While highest priority goes to valid names, other names in use are included so that this register can serve as a guide to interpret taxonomic literature. The content of WoRMS is controlled by taxonomic experts, not by database managers. WoRMS has an editorial management system where each taxonomic group is represented by an expert who has the authority over the content, and is responsible for controlling the quality of the information. Each of these main taxonomic editors can invite several specialists of smaller groups within their area of responsibility to join them.

This register of marine species grew out of the European Register of Marine Species (ERMS) (*http://www.marbef.org/data/erms.php*) and its combination with several other species registers coordinated together by the Flanders Marine Institute (VLIZ). Rather than building separate registers for all projects, and to make sure taxonomy used in these different projects is consistent, VLIZ developed a consolidated database called Aphia. The WoRMS objective, currently being developed, is to combine information from Aphia with other more specialised databases (e.g. AlgaeBase, FishBase—*www.fishbase.org*, Hexacorallia, NeMys).

GBIF (marine and terrestrial species, global footprint) (*http://lis-upmc.snv.jussieu.fr/lis/?q=reseau/gbif***)** The Global Biodiversity Information Facility is a decentralised network of biodiversity information facilities (BIF), developed in the framework of a multi-partner agreement (*http://www. gbif.org/*). It provides free and open access via Internet to all known global biodiversity data, with the aim of promoting sustainable development. In France, GBIF is coordinated by the National museum of natural history (MNHN). The priorities set (mobilising numerous partners) are to make accessible and disseminate biodiversity data, develop protocols and quality assurance procedures, interoperability and develop information architecture to connect data of various types and origins. GBIF also provides assistance

P. Goulletquer et al., *Biodiversity in the Marine Environment,*
DOI 10.1007/978-94-017-8566-2, © Éditions Quæ, 2014

for local development of expertise and decision-making capabilities (one of the Content Needs Assessment Task Group's fields of action).

GEO BON (Group on Earth Observations—Biodiversity Observation Network) Its scope covers biodiversity in all continental and oceanic environments (*http://www.earthobservations.org/geobon.shtml*). Along with maintaining time series (for presence, abundance) and observing the state of biodiversity, GEO-BON also collects information about interactions between species, on human use of biodiversity as well as metadata (abiotic environment, taxonomic status, drivers of change in biodiversity). GEO BON conducts preliminary analyses (detecting changes, identifying trends, interpolations and projections and modelling of ecosystem service production). It contributes to studies led by organisations in charge of assessing biodiversity and ecosystems.

OBIS (Ocean Biogeographic Information System, *http://www.iobis.org/***)** This is an international information system focused on marine biodiversity worldwide. It provides spatially referenced marine life data and, as of 15 June 2011, more than 31 million of these georeferenced data related to 116,603 accurately identified marine species, had been accessed via the OBIS portal. Search tools provide a visual display of how species occupy their environment together. OBIS is progressively incorporating oceanographic (biological, physical and chemical) data from numerous sources, supplying the tools to test various hypotheses and assist ecosystem research. Users, who include researchers, students and environmental managers, can gain a dynamic view of the distribution of marine species over space and time. OBIS was set up by the international Census of Marine Life (see below) programme and hosts data produced by CoML. The OBIS portal opens access to a growing federation of sites, many of them more specialised geographical portals for the regions, taxa or tools they cover.

EoL (global, all species) The Encyclopedia of Life is the unprecedented outcome of a global partnership between the scientific community and the general public. It is a freely-accessible collaborative encyclopaedia at *http:// www.eol.org/*whose ambition is to document the 1.8 million known species. EoL compiles information from existing databases and contributions from experts and non-experts from all over the world, in order to create a page for each species which can be expanded indefinitely and including video and sound recordings, illustrations and text. Furthermore, the Biodiversity Heritage Library containing digitized copies of the main printed collections of natural history libraries can also be found in EoL. The project is supported by funding of US $50 million supplemented by the MacArthur Foundation (*http://www.macfound.org/*) and Sloan Foundation (*http://www.sloan.org/*).

Mar-BOL (*http://www.marinebarcoding.org/***)** Marine Barcode of Life is an international initiative to enhance our capacity to identify marine life by utilizing DNA Barcoding. MarBOL is a joint effort of the consortium Consortium for the Barcode of Life (CBOL) (*http://www.barcoding.si.edu/*) and the international Census of Marine Life (CoML) programme, the latter federating a network of research scientists from over 80 countries.

FISH-BOL (*http://www.fishbol.org/index.php***)** The Fish Barcode of Life initiative is operated by the CBOL consortium. It is a global effort to coordinate an assembly of a standardised reference sequence library for all fish species, especially those from voucher specimens with authoritative taxonomic identifications. As of 23 August 2011, a total of 8,293 species of fish had been barcoded. FISH-BOL benefits include facilitating species identification for all potential users, including taxonomists; highlighting specimens that represent a range expansion of known species; flagging previously unrecognised species; and enabling identifications where traditional methods are not applicable. Barcode sequences (68,061 of them on 23 August 2011), images, and geospatial coordinates of examined specimens and a large amount of other information are made available to the public. FISH-BOL complements and enhances information from existing sources like FishBase (*www.fishbase.org/*) and various genomics databases.

Census of Marine Life (CoML, 2000–2010, *http://www.coml.org/***)** This international programme has brought together over 2,700 scientists from over 80 countries. During the decade from 2000–2010, 538 expeditions were made (at a cost of US $650 million, and no less than 1,200 species new to science were discovered (Costello et al. 2010). Several theme-based databases have been produced (*e.g.*, Census of Marine Zooplankton www.cmarz.org). Apart from Field Projects, the CoML has been organised into project to study the History of Marine Animal Populations, or HMAP, and the Future of Marine Animal Populations, or FMAP. The data are managed by the OBIS biogeographical information system already mentioned above. A special project dealt with outreach, conveying knowledge to the public via numerous media. In 2010, the valuable scientific output was made tangible in more than 2,600 publications, amongst which should be mentioned: 1) regional comparison overviews (O'Dor et al. 2010), which can be accessed free of charge at *http://dx.doi.org/10.1371/issue.pcol.v02. i09*, 2) the results of theme-based projects (including HMAP and FMAP), at *http://www.ploscollections.org/static/comlCollections.action*, and 3) special issues of scientific journals: *http://www. coml.org/scientific-papers*.

FishBase (*http://www.fishbase.org/***)** This relational database was developed at the WorldFish Center, in collaboration with the Food and Agriculture Organization of the United Nations (FAO) and many other partners, and with support from the European Commission. Today it is supported by a consortium of nine organisations (including the national museum of natural history MNHN).It is consulted by numerous professional users, like research scientists, fisheries managers and others, providing access to information (backed up by 45,800 bibliographical references) on almost all known fish species. As of August 2011 it held descriptions of 32,000 species, the list of their 291,200 common names and 50,400 pictures.

SeaLifeBase (*http://www.sealifebase.org/***)** The long-term goal of SeaLifeBase is to create and maintain a FishBase-like information system for all aquatic living organisms, both marine and freshwater, but beginning with just marine species in the first phase. The project aims to make biological information necessary to conduct biodiversity studies available for each species. This means: (1) current scientific accepted names, and synonyms in the sources used, (2) distribution by EEZ, country and FAO area, and (3) published references used (both hard copy and on-line).Additional information concerns the common names (in several languages, distribution by region, province or State (when possible), by ecosystem, by depth, maximum length and weight, as well as their IUCN and CITES status. As in FishBase, several other categories are documented, i.e. habitats; diet (food items, trophic level); growth parameters (length-weight ratios); reproduction (age at first maturity, fecundity) and supplemented by drawings and/or photos.

Biocéan (geographically global in scope, but limited to sites explored by French research cruises, deep-water species) (*http://www.ifremer.fr/biocean/indexgb.html***)** Biocean is designed to gather the extremely large volume of data collected from different deep-sea ecosystem studies jointly conducted by Ifremer's deep environment department (*Fig. 9*).The database has 6 specific applications: two of them used aboard research vessels to collect operational data (Alamer) and the others for linkage with a core database back on land. The latter are used to: (1) manage the taxonomic nomenclature (Bioclass); (2) monitor the identification of faunal collections (Gescol); (3) complete the results of chemical analyses or measurement data files (Donenv); and (4) add or extract data from the database (EchangeTM).The goals of the Biocean database are: (1) collect and maintain operational data from research cruises; (2) organise faunal and environmental data in a standardised form; and (3) conserve data for studies of long-term temporal changes. BIOCEAN is also used by several European and international bodies.

ERMS (European waters, marine species) The European Register of Marine Species (ERMS) is an authoritative taxonomic list of species occurring in the European marine environment (up to the strandline or splash zone above the high tide mark and down to 0.5 psu, ppt) salinity in estuaries. The project has compiled a list of marine species in Europe and a bibliography for them, including marine species identification guides. ERMS has also contributed to European taxonomic expertise, species identification and the current status of marine species collections in Europe. A total of 29,713 species-level taxa were catalogued. Ninety percent of taxa were satisfactorily inventoried, but non-halacarid Acarina, diatoms, lichens and cyanobacteria were not included, and geographical coverage was incomplete for Rotifera and Brachiopoda. Lists that would benefit from further input include (1) those that have not yet been checked by an expert on European fauna, (e.g. lists of non-epicarid Isopoda, Cephalochordata, Appendicularia, Hemichordata, Hirudinea, Gnathostomulida, Ctenophora and Placozoa; (2) preliminary lists, including some of the above and lists of protists; and (3) lists with many species which need to be reviewed (the obstacle beingthe small number of experts).Improvements are currently being made via the on-line version of ERMS (*www.marbef.org/data/erms.php*).

SCAR MarBIN (SCAR Marine Biodiversity Information Network) SCAR MarBIN establishes and supports a distributed system of interoperable databases, forming the Antarctic Regional OBIS Node, under the aegis of the Scientific

Committee on Antarctic Research (SCAR). SCAR-MarBIN compiles and manages existing and new information on Antarctic marine biodiversity by coordinating, supporting, completing and optimizing database networking. The data is transmitted to global scale systems like OBIS and GBIF. They included georeferenced sampling sites, taxonomic inventories, interactive keywords and bibliographic sources. SCAR-MarBIN is interconnected to Google Earth.

ITIS (North America, all terrestrial and marine species; *http://www.itis.gov/***)** The Integrated Taxonomic Information System (ITIS) is a partnership designed to provide consistent and reliable information on the taxonomy of biological species. ITIS was originally formed in 1996 as an inter-agency group within the U.S. Federal government, involving agencies ranging from the Department of Commerce to the Smithsonian Institution. It has now become an international body, with Canadian and Mexican government agencies participating. The primary focus of ITIS is North American species, whether or not they have a global range of distribution. ITIS continues to collaborate with other international agencies to increase its global coverage. It provides an automated reference database of scientific and common names for animals, plants, fungi, and microbes. In 2009, ITIS contained 592,000 scientific names, synonyms and common names.

Data presented in ITIS are considered public information, and may be freely distributed and copied, though appropriate citation is requested. In fact, ITIS is frequently used as the de facto source of taxonomic data in biodiversity informatics projects. ITIS couples each scientific name with a stable and unique taxonomic serial number, or TSN, as the "common denominator" for accessing information on issues like invasive species, declining amphibians, migratory birds, fishery stocks, pollinators, agricultural pests, and emerging diseases. It presents the names in a standard classification that contains author, date, distributional, and bibliographic information. Common names are available in the main official languages of the Americas (English, French, Spanish, and Portuguese).

GenBank The GenBank sequence database (Global, all species, sequences) is an open access, annotated collection of all publicly available nucleotide sequences and their protein translations. This database is managed by the National Center for Biotechnology Information (NCBI) as part of the International Nucleotide Sequence Database Collaboration, or INSDC. GenBank receives sequences produced in laboratories throughout the world and is growing at an exponential rate, doubling every 18 months. Release 155, produced in August 2006, contained over 65 billion nucleotide bases in more than 61 million sequences. GenBank is built by direct submissions from individual laboratories, as well as from bulk submissions from large-scale sequencing centres. The taxonomy database contains the names and phylogenetic lineages of more than 160,000 organisms whose molecular data is stored in NCBI databases. New taxa are added to the taxonomy database as data are deposited for them.

ICES There are also specific databases for fisheries. For instance, at ICES, a network of more than 1,600 scientists from 200 institutes linked by an intergovernmental agreement, with expertise in marine ecology and environment as well as in fisheries science and aquaculture (*http://www.ices.dk/ indexfla.asp*). ICES maintains some large databases in these fields and particularly on marine fisheries. The ICES Secretariat holds the data and software needed for use by ICES Working Groups for fisheries and environmental management, e.g.:

- STATLANT 27A, containing official statistics on nominal catches of fish and shellfish;
- ICES Fisheries Assessment Package, which is used by some 20 Working Groups for ICES stock assessments. It includes catches in tonnes, fishing effort, catch in number at age and relevant biological data;
- International Bottom Trawl Survey (IBTS), results from an international survey conducted each year in the North Sea since the 1970s by scientific fleets from neighbouring countries. It produces, amongst others, an annual index of abundance for the main demersal fish stocks.
- North Sea Databank (originally set up by the EU) contains information about catches and fishing effort;
- North Sea Multispecies Databank holds descriptions of stomach contents for the main predatory species.

SAUP (*http://www.seaaroundus.org/***)** The Sea Around Us Project is also worth noting here. It proposes integrated data and expertise on topics dealing with the interactions between fisheries resources and biodiversity in large marine ecosystems.

DAISIE (Delivering Alien Invasive Species Inventories for Europe; *http://www.europe-aliens.org/***)** This EU FP6 project is one of the pillars in developing a European strategy to deal with biological invasions, aiming to ensure the consistency of studies on taxa in marine, freshwater and terrestrial environments. The DAISIE database gives access to European data on invasive (marine and inland) species in habitats of over 93 countries/ecoregions (including overseas regions). The general objectives of DAISIE are to:

- create an inventory of invasive species in Europe;
- structure the inventory to provide the basis for prevention and control of biological invasions through the understanding of the environmental, social, economic and other factors involved;
- assess and summarise the ecological, economic and health risks and impacts of the most widespread and/or noxious invasive species; and
- use distribution data and the experiences of the individual Member States as a framework for considering indicators for early warning. There is a comparable database on a global scale, i.e. the Global Invasive Species Database (**GISD**) (*http://www.issg.org/database/welcome/*). It was developed from the Global Invasive Species Program and is managed by the Invasive Species Specialist group of the IUCN. The **NOBANIS** (North European and Baltic Networks on Invasive Alien Species) database deals with identification of invasive species in the Northern European and Baltic Sea region (*http://www.nobanis.org/*).

QUADRIGE QUADRIGE is a component of the Water information system (SIE) and thus contributes to work carried out by the national water data administration (SANDRE) (*http://wwz.ifremer.fr/envlit/resultats/quadrige*, *http://sandre.eaufrance.fr/*). This information system is a national reference for monitoring networks dedicated to the coastal environment like RNO, REMI, REPHY and IGO), (e.g. *http://www.ifremer.fr/delst/ surveillance/rephy.htm*). It contributes to fulfilling national commitments under EU Directives, especially the Water Framework Directive (WFD), and those dealing with public health issue, such as regulations governing shellfish production areas and recreational waters. The QUADRIGE system is now used for a series of new applications, such as SINP and for the implementation of the MSFD.

The stakes and possibilities are numerous and varied. The most important of them is the need to store data collected from coastal environment monitoring networks in compliance with international standards and the Inspire (Infrastructure for Spatial Information in Europe) Directive. The storage models used ensure optimised data storage. Other challenges then arise, particularly making a range of data available that enables significant cross-checking of information.

System functions include:

- launching and integrating new monitoring networks, e.g. REBENT, *http://www.rebent.org/*; REMORA,*http://www.ifremer.fr/remora/* ;the shellfish farming observatory *http://wwz.ifremer.fr/observatoire_ conchylicole;* REPER, RSL, and so on;
- taking spatial data and mapping functions into account;
- dissemination of data and communication to the general public;
- exchange of qualified data between national (Sandre format) and international partners.

Group of experts

Chairman Gilles Boeuf is a biologist, physiologist, professor at Pierre & Marie Curie University (UPMC) and the president of the French national museum of natural history (MNHN). *He was Chairman of the collective expertise group for this publication.* After working as a specialist in the physiology of organisms at Ifremer for 24 years, he was the director of the Arago Oceanology Observatory Laboratory in Banuyls from 1999 to 2005. Dr. Boeuf currently chairs several scientific boards, including that of Agropolis international in Montpelier, and is a member of Ifremer's scientific board and the Scientific Council on Natural Heritage and Biodiversity at the French Ministry of Ecology, Sustainable Development, Transport and Housing (MEDDTL). He is also a member of the Biodiversity Task Force of the Secretariat of Ecology, the Precaution and Ethics committee (Inra-Cirad) and the Scientific Committee of the Marine Science Centre in Monaco. He is president of the Massane Nature Reserve in the Eastern Pyrenees region, and has authored 330 national and international publications and papers and spoken at symposia in France and worldwide.

Experts Christophe Béné, Senior Advisor, WorldFish Center (Penang, Malaisie) on small-scale fisheries and development, external expert for FAO, the World Bank, UK-DFID and the international Challenge Programme on Food and Water. The WorldFish Center is one of 15 members of the Consultative Group on International Agricultural Research, or CGIAR. Dr. Béné's work focuses on socio-economic issues and developing policies for resource management, particularly in small-scale activities (fisheries and aquaculture) regarding quality of life for rural populations and specialises on the topics of reducing poverty, governance methods and rural development. He holds a PhD in Fisheries Sciences from the University of Paris VI and a Post-Graduate Diploma in Development Economics from the School of Development Studies, University of East Anglia (UK). As a scientific advisor at the WorldFish Center, his duties include coordinating a research initiative panel on development and management at national and international levels. He has written over 40 articles published in international journals and several chapters of book, and has worked in more than 20 countries in Latin America, the Caribbean, South and South-East Asia, sub-Saharan Africa and the Pacific.

Gary Carvalho, biologist and systematist, a professor at the School of Biological Sciences, University of Wales, Bangor, United Kingdom. He is a professor of molecular ecology and directs the Molecular Ecology and Fisheries Genetics Laboratory (MEFGL), one of Europe's largest centres focusing on population and species diversity of aquatic animals. Research interests include the ecology and evolution of population differentiation, fisheries genetics and the evolution of adaptive variation in the wild.

He has led numerous international projects in the capacity of coordinator or project leader (e.g. NERC, Leverhulme Trust, FishPopTrace). His studies include one of the first scientific demonstrations of fine-scale structuring, decadal analyses of cod and the analysis of biocomplexity in marine pelagic fish. He is Editor of *Fish and Fisheries*, and serves on the editorial boards of scientific journals (*Molecular Ecology*, *Conservation Genetics and Proceedings of the Royal Society, London*). He is currently member of the ICES WG AGFM Working Group on the Application of Genetics in Fisheries and Mariculture, marine expert for the Pew Environment Group and Chair of the European Regional Working Group of *Fish-Bol* (developing DNA barcoding of fishes). Dr. Carvalho has served on several international panels, including thematic programmes of the NERC, the Ecology panel of the Norwegian Research Council, the Biosciences and Environment Peer Review Panel, the Academy of Finland, and member of the ERA-Net BiodivERsA Evaluation Committee.

Philippe Cury is a biologist and ecologist, with a PhD in biomathematics, Director of research at the IRD (institute of research for development) and Director of the CRH (Mediterranean and tropical fisheries research centre) based in Sète, France (*www.CRH-sete.org*). He directs one of the largest research units working on marine ecosystems and the ecosystem-based approach applied to fisheries in (UMR-EME 212, employing 110 people).He is also the scientific coordinator of the Eur-Oceans Consortium which provides guidance for research on marine ecosystems in Europe. Since 1980, he has worked in Senegal, Côte d'Ivoire, California and South Africa to analyse the climate's effect on fisheries and how to implement the ecosystem approach to fisheries. During his career, Dr. Cury has received several distinctions, including the National Scientific Philip Morris Prize obtained in 1991 (Life Science Prize), the French oceanography medal awarded in 1995 by the scientific committee of the Prince Albert Monaco Museum of Oceanography and the Gilchrist Medal in 2002 from Sancor in South Africa. He participates in numerous national and international scientific committees and has organised several symposia, the most recent of which was the Globec-Euroceans symposium on Coping with Global Change in Marine Social-Ecological Systems held in 2008 at FAO headquarters in Rome. He has published more than 110 peer-reviewed articles in major international journals (*The Tree*, *Canadian Journal of Fisheries and Aquatic Sciences, Ecology Letters*, *Fish and Fisheries, Marine Ecology Progress Series*, *Fisheries Oceanography,* etc.) and is the author of 8 books or chapters of books.

Bruno David, an evolutionary (paleo)-biologist, obtained his PhD in Paleontology at the university of Franche-Comté (Besançon) and his D.Sc. in paleontology and evolutionary biology at the university of Burgundy (1985). He is CNRS director of research at the Biogeosciences laboratory, University of Burgundy, Dijon, and president of the scientific council of the National museum of natural history (MNHN). Originally trained as a geologist and a paleontologist, his work focuses on the evolution of life forms, both as patterns (phylogeny, large-scale structuring of biodiversity) and processes (symbiosis, evolution-development relationships and emergence of phenotypes).

Echinoderms, and especially echinoids, are the biological models studied. Dr. David has written a book and edited a database on Antarctic echinoids, specifically focusing on the origin of radial symmetry in echinoderms and how the fivefold pattern is intimately related to seriality (repetition of elements along arms) and anteroposterior polarization (A/P axis).More recently, he has become interested in the functioning and evolution of symbiosis between tropical echinoid hosts and parasitic crabs. In this field of research, works principally with scientists in Brussels (ULB), San Francisco (California Academy of Sciences), and more recently Santiago (University of Chile). Dr. David has published about 100 peer-reviewed papers and 5 chapters in books, edited 3 books, authored 2, and been a contributor to 6 scientific films and many public conferences or popular science papers. He has organised several international conferences and has been involved in many scientific bodies or committees throughout his career. Currently, he is president of the scientific council of the MNHN national museum of natural history, leader of a cross-cutting programme devoted to databases and collections (Trans'Tyfipal) and member of several scientific councils (CNRS, INEE, FRB foundation for research on biodiversity and Natural heritage & Biodiversity). He is also in charge of a trans-regional and trans-university project devoted to ecology and environment in Burgundy—Franche Comté. He is the founder and former director of the Biogeosciences lab.

Daniel Desbruyères, abyssal environmental biologist, senior research scientist at Ifremer in Brest. For 22 years, he directed the department of deep-sea ecosystem studies. His career began with research on the benthos of the French sub-Antarctic shelf. After an initial study on the deep benthos in the Bay of Biscay, he carried out experiments to study the colonisation dynamics of the abyssal plains. He participated in discovering and studying hydrothermal vent communities in the Pacific and Atlantic Oceans (East Pacific Rise, Back Arc Basins, Mid-Atlantic Ridge) and took part in more than 30 deep-sea dives in French and American manned submersibles (*Cyana, Alvin* and *Nautile*). He has published over 80 articles in international scientific journals on the zoology and ecology of deep-dwelling species. He described (co-described) 35 new taxa of annelid worms (species, genera, sub-families and families), the best known of which are the Pompeii worms (*Alvinella pompejana*) which live on the walls of active hydrothermal vent systems and the "ice worms" (*Hesiocaeca methanicola*) which live in methane hydrates.

Luc Doyen, mathematician and research director at CNRS-MNHN, Paris. Luc Doyen received HDR accreditation to supervise research in applied mathematics. He studied control theory, optimization and mathematical economics. He currently holds a permanent position at the CNRS national centre for scientific research, and is particularly involved in bio-economic modelling, viable management of biodiversity and the mathematics of sustainability His book, *Sustainable Management of Natural Resources. Mathematical Models and Methods* was published by Springer and he has written more than 30 publications in international peer-reviewed journals; highlighting the mix of applied and theoretical dimensions of his research activity and its interdisciplinary nature balancing ecology, environmental economics, modelling and mathematics. The applied component of his research has involved research contracts (ANR Chaloupe, ANR Systerra, ACI MEDD, IFB-GICC, STICAmsud, and FEAST programme), highlighting the transfer of results from theoretical studies to national or international institutions such as INRA, IFREMER, MNHN, WorldFish Center and CSIRO which work in fisheries, agriculture and biodiversity management. He is the coordinator of the ANR research project called Adhoc, focusing on modelling the co-viability of fisheries and marine biodiversity, as well as the leader of the RTP INEE-CNRS interdisciplinary network called MOBIS on modelling of biodiversity scenarios. To facilitate outreach for research, he was coordinator of a seminar on "Viable development" for 4 years at the ENS in Paris. He is reviewer for various international journals and has sat on research assessment committees (University of Paris VI, IRD, ACI and CEMAGREF). The interdisciplinary nature of his activities, at the boundaries of mathematics and numerical modelling, economics and ecology, is further illustrated by courses taught (Master's degree curricula and engineering schools) and supervision (PhD, Master's, etc.).

PhilippeGoulletquer, *Scientific Secretary of this Expert panel review*, holds a PhD in Oceanography (1989) and HDR accreditation to supervise research (2000) from the University of western Brittany (UBO) and the University of Caen. He is currently in charge of biodiversity issues as a senior scientist at Ifremer's Prospective & Scientific Strategy Division (Nantes). He headed the national research programme on aquaculture sustainability until 2007, and previously in charge of the genetics &

pathology research laboratory in La Tremblade, France. He was an associate professor at the University of Maryland (CBL, MD, USA) from 1989–1992. He was member of the ICES working group on Introduction and Transfer of Marine Organisms' (ITMO) and the *Mariculture Committee*. Dr. Goulletquer is a member of the CSPNB national scientific council on natural heritage and biodiversity. He has been involved in several international expert reviews on risk assessment in introducing a new oyster species in Chesapeake Bay (USA NAS), the status of the Eastern oyster (NOAA-CIE), the Convention on Biological Diversity (chairing the WG on mariculture's impact on biodiversity and member of the WG defining 2010 Targets, as well as the European Environment Agency (SEBI meeting—CoML) on invasive species. He contributed to developing EU regulations regarding the use of exotic species in aquaculture (EU 2007) and drawing up Marine Strategy Framework Directive (MSFD) indicators for the "Invasive species" descriptor. He is currently co-editor of the on-line scientific journal Aquatic Invasions (www.aquaticinvasions.net). Rapporteur of WG2 on sustainable development of marine activities for the French Ministry of the Environment during the '*Grenelle de la Mer*' fora and summit meetings in 2009, he is the author & co-author of more than 55 publications in refereed journals.

Philippe Gros, biomathematician, scientific division at Ifremer. After qualifying as Agrégé in biological sciences at the prestigious École normale supérieure teaching college in the 1970s, he became an associate professor at the University of western Brittany (UBO, France) and in 1980 joined Cnexo, which became Ifremer in 1984. His research work has focused on mathematical modelling of harvested marine fish population dynamics and marine and coastal ecosystems. He took part in coordinating the French National Programme on Coastal Oceanography in the mid-90s. From 2001 to 2004, he was Head of the Living resources (fisheries & aquaculture) division at Ifremer. In the field of European research cooperation, he promoted interdisciplinary programmes whose outcomes have provided the scientific basis for the Ecosystem Approach to Fisheries (EAF).After Ifremer was restructured in 2005, Dr. Gros was the scientific manager of fisheries research and is currently a senior scientist in Ifremer's scientific division.

Susan Hanna is professor of marine economics at Oregon State University (USA), affiliated with the Coastal Oregon Marine Experiment Station and Oregon Sea Grant. Her research and publications are in the areas of marine policy and management, with an emphasis on assessing fishery management performance, particularly through the ecosystem-approach, and incentive-based management instruments, property rights and institutional design. Dr. Hanna worked with the Pacific Regional Fishery Management Council as member of the Scientific and Statistical Committee as well as on various ad-hoc committees. She is also a Senior Scientific Advisor to working groups of NOAA (National Oceanic and Atmospheric Administration): the US Commission on Ocean Policy, National Marine Fisheries Service, Minerals Management Service, the boards of Northwest Power and Conservation Council, and the Oregon Ocean Policy Advisory Council. She has been a member of the National Research Council's Ocean Studies Board and several NRC Committees, including that to review Individual Quotas in Fisheries and the Committee on Protection and Management of Pacific Northwest Anadromous Salmonids.

Simon Jennings is Lead Scientist in charge of the Environment & Ecosystems department of Cefas (Centre for Environment, Fisheries and Aquaculture Science, GB). Research conducted by Simon and his colleagues focuses on (1) describing and predicting the structure and function of marine populations, communities and ecosystems, (2) measuring and predicting human and environmental impacts on structures and functions to assess the sustainability of impacts and (3) developing and applying tools to support environmental management. The research spans the continuum from fundamental to applied science, to ensure that ideas tested are operational and to improve them for environmental management. Dr. Jennings is an advisor to the committee on marine protection, biodiversity, environmental management and fisheries and is former Chair of the ICES Advisory Committee on ecosystems. As a research scientist, he has worked in the Indian, Pacific and Atlantic Oceans. He is an Honorary Professor and teaches environmental and marine science at the University of East Anglia and the University of Newcastle.

Harold Levrel is an economist and ecologist in the maritime economics department of Ifremer and belongs to the Amure joint research unit (UMR Ifremer/UBO). His research focuses on indicators of interactions between biodiversity and society, assessment of ecological services and monitoring of coastal recreational activities. After defending his thesis on indicators for interactions between conservation and development issues at EHESS graduate school in social sciences and at the National museum of natural history (host laboratory: Species conservation, restoring and monitoring of populations), he spent a post-doc year at CNRS, working to set up an atlas of biodiversity and ecological services. While working on his PhD and post-doc, he was a consultant for the UNESCO MAB programme, effecting numerous assignments in Western Africa to implement the indicators of biodiversity-societal interactions in the Biosphere reserves; working for the Ecofor Public Interest Group to coordinate a national network of biodiversity indicators and carrying out the French Millenium Ecosystem Assessment feasibility study on behalf of the MEEDDM. Today he is in charge of several actions within Ifremer, mainly involving the coordination of a national and international working network (ICES working group) on recreational fisheries, setting up marine biodiversity indicators in the framework of the SINP nature and landscapes information system, developing a research programme based on assessing the costs of eroding biodiversity and marine and coastal ecological services, and analysing compensatory measures for aquatic environments. He has published some fifteen scientific articles, mostly in interdisciplinary journals dealing with conservation issues (*Ecological Economics*, *Ecology and Society*, *Society and Natural Resources*, *Biodiversity and Conservation*, *Responsabilité et Environnement*, *Environmental Modelling & Software*, *Interdisciplinary Science Reviews* and *Ambio*), as well as a dozen articles and chapters of books for non-specialists. He is also the author of a book on biodiversity management indicators.

Olivier Thébaud holds a PhD from EHESS (graduate school of social sciences), Paris, and HDR accreditation to supervise research from the University of western Brittany (UBO, Brest, France). Prior to joining CSIRO in November 2009 as a senior economist for the Marine and Atmospheric Research Division, Dr. Thébaud was Head of Ifremer’s maritime economics department, Director of the AMURE research group (one of the largest research groupings in Europe for maritime economics and law associating UBO and Ifremer), and associate professor at the University of Western Brittany (UBO). His research focuses on the development and empirical assessment of models of fishermen's behaviour in response to economic, ecological and institutional change. He also worked extensively in the area of bioeconomic modelling and the economics of ecosystem-based approaches to marine resources. Key applications from this research include the regulation of commercial and recreational fisheries, aquaculture, multiple ecosystem uses, including accidental pollution, as well as biodiversity conservation policies such as Marine Protected Areas (MPA).Olivier Thébaud has coordinated several multidisciplinary research projects, and supervised a many staff and students in this field. He has served as an expert for regulatory bodies on fisheries and MPAs in France, and contributed to several expert groups on research needs to support the Ecosystem Approach to the management of marine resources. He is author or co-author of 33 peer-reviewed publications.

Jacques Weber, an economist and anthropologist, Director of research at Cirad (centre for international cooperation on agricultural and development issues), a lecturer at the EHESS (public research institute graduate school in the social sciences) and UPMC (University Pierre & Marie Curie). He has supervised 12 PhD theses and more than 20 DEA and Master's theses. He was a research scientist at Orstom (now IRD) from 1971 and 1983, the founder and director of Ifremer’s maritime economics department (1984–1992), and researcher since 1993 at Cirad, where he created a research unit called Green, then became scientific director of social sciences. From 1998 to 2001, he created and managed the Expertise and valorisation department at IRD, and organised collective appraisals called “expert panel reviews”. He then became director of the IFB French institute of biodiversity from 2002 to 2008. There are two red threads running throughout his career. The first is management of the renewable resources (forests, wildlife, oceans and atmosphere) which all fall under the concept of common property. The second involves the various ways that societies determine goods as signs of “wealth” and the ways that “wealth” circulates between societies and between cultures. The inherent challenge common to these two concerns is to understand the drivers of poverty and sustainability in numerous societies and cultures. Member of several national and international scientific committees, corresponding member of the French academy of agriculture, member of the council for sustainable development (CEDD), the scientific council on natural heritage and biodiversity (CSPNB) and vice president of the French MAB board. He is also a member of several editorial committees of scientific journals and several learned societies including IIFET, IISEE, and IASC. In 1988, he founded the European Association of Fisheries Economists (EAFE). He has published a hundred articles and chapters of publications, as well as two books. He is the author of dozen forewords and has given numerous lectures to audiences from both academia and the general public.

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Printing
Beta, Barcelona

Registration of copyright: January 2013

References

Abernethy, K. E., Trebilcock, P., Kebede, B., Allison, E. H., & Dulvy, N. K. (2010). Fuelling the decline of UK fishing communities? *ICES Journal of Marine Science, 67*, doi:10.1093/icesjms/fsp289.

Adams, W., Aveling, R., Brockington, D., Dickson, B., Elliott, J., Hutton, J., Roe, D., Vira, B., & Wolmer, W. (2004). Biodiversity conservation and the eradication of poverty. *Science, 306*(5699), 1146–1149.

Adams, W., Brockington, D., Dyson, J., & Vira B. (2003). Managing tragedies: Understanding conflict over common pool resources. *Science, 302,* 1915–1916.

Adger, W. N., Benjaminsen, T. A., Brown, K., & Svarstad, H. (2001). Advancing a political ecology of global environmental discourses. *Development and Change, 32,* 681–715.

Agnew D. J., Pearce, J., Pramod, G., Peatman, T., Watson, R., Beddington, J. R., & Pitcher, T. J. (2009). Estimating the worldwide extent of illegal fishing. *PLoS ONE, 4*(2), e4570.

Alder J., & Wood, L. (2004). Managing and protecting seamounts. In T. Morato & D. Pauly (Eds.), *Seamounts: biodiversity and fisheries*. *Fisheries Center Research Report* (Vol. 12(5), pp. 67–73). Canada: University of British Columbia.

Alheit, J., & Bakun, A. (2010). Population synchronies within and between ocean basins: Apparent teleconnections and implications as to physical-biological linkage mechanisms. *Journal of Marine Systems, 79*(3–4), 267–285.

Alverson, D., Freeberg, M. H., Pope, J. G., & Murawski, S. A. (1994). A global assessment of fisheries by-catch and discards. *FAO Fisheries Technical Papers, 339,* 235.

Andersen, T., Carstensen, J., Hernández-García, E., Duarte, & C. M. (2008). Ecological thresholds and regime shifts: Approaches to identification. *Trends in Ecology and Evolution, 24*(1), 49–57.

Anderson, E. D. (ed.). (2002). The Raymond J. H. Beverton lectures at Woods Hole, Massachusetts. Three lectures on fisheries science given May 2–3, 1994. U.S. Dep. Commer., *NOAA Technical Memorandum* NMFS-F/SPO-54, p. 161.

Anderson, L. (2002). A bioeconomic analysis of marine reserves. *Natural Resource Modelling, 15*(3), 311–334.

Angel, M. V. (1992). Managing biodiversity in the oceans. In M. N. A. Peterson (Ed.), *Diversity of ocean life* (pp. 23–62). Washington, D.C.: Center for Strategic and International Studies.

Angly, F. E., Felts, B., Breitbart, M., Salamon, P., Edwards, R. A., Carlson, C., Chan, A. M., Haynes, M., Kelley, S., Liu, H., Mahaffy, J. M., Mueller, J. E., Nulton, J., Olson, R., Parsons, R., Rayhawk, S., Suttle, C. A., & Rohwer, F. (2006). The marine viromes of four oceanic regions. *PLoS Biology, 4*(11), e368.

Arendt, H. (1972). *Du mensonge à la violence. Essais de politique contemporaine* (French trans: Durand G.). Paris: Calmann-Lévy, 1972, Pocket, 1989.

Attrill, M. J., & Edwards, M. (2008). Reply to Haddock, S.H.D. Reconsidering evidence for potential climate-related increases in jellyfish. *Limnology and Oceanography, 53*(6), 2763–2766.

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DOI 10.1007/978-94-017-8566-2, © Éditions Quæ, 2014

Austen, M., Billett, D. S. M., Bishop, J. D. D., Brownlee, C., Burrows, M. T., Day, J. G., Edwards, M., Fox, C. J., Gachon, C. M. M., Gilbert, F. J., Gooday, A. J., Hall, A. J., Hill, A. E., Lampitt, R. S., Leakey, R., Mieszkowska, N., Schroeder, D. C., Sharpies, J., Sims, D. W., Somerfield, P. J., Tyler, P. A., & Williamson, P. (2010). Major issues in marine biodiversity-related research topics that address NERC theme challenges and current theme actions. NERC-OCEANS2025 report, NOC- Southampton, 1 October 2009.

Avise, J. C. (2000). *Phylogeography*. Cambridge: Harvard University Press.

Bajjouk, T. (coord.). (2010). *Réseau de surveillance des biocénoses benthiques côtières (Rébent). Bilan des actions sur la région Bretagne pour l'année 2009*. RST/Ifremer/Dyneco/AG/10-12/TB, p 69.

Baland, J. M., & Platteau, J. P. (1999). The ambiguous impact of inequality on local resources management. *World Development, 27*(5), 773–788.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K., & Turner, R. K. (2002). Economic reasons for conserving wild nature. *Science, 297*, 950–953.

Barbault, R. (2006). Un éléphant dans un jeu de quilles. *L'homme dans la biodiversité* (p. 266). Seuil, Science ouverte.

Barbier, E. B. (2009). Ecosystem service trade-offs. In K. McLeod & H. Leslie (Eds.), *Ecosystem-based management for the oceans* (pp. 129–144). Washington, D.C.: Island Press.

Barcelos e Ramos, J., Müller, M. N., & Riebesell, U. (2010). Short-term response of the coccolithophore *Emiliania huxleyi* to an abrupt change in seawater carbon dioxide concentrations. *Biogeosciences, 7,* 177–186.

Bariche, M., Letourneur, Y., Harmelin-Vivien, M. (2004). Temporal fluctuations and settlement patterns of native and Lessepsian herbivorous fishes on the Lebanese coast (eastern Mediterranean). *Environmental Biology of Fishes, 70*(1), 81–90.

Barrientos, S. (2000). Globalization and ethical trade: Assessing the implications for development. *Journal of International Development, 12*(4), 559–570.

Baumgärtner, S., & Quaas, M. F. (2009). Ecological-economic viability as a criterion of strong sustainability under uncertainty. *Ecological Economics, 68,* 2008–2020.

Bax, N., Williamson, A., Aguero, M., Gonzalez, E., & Geeves, W. (2003). Marine invasive alien species: A threat to global biodiversity. *Marine Policy, 27,* 313–323.

Beaufort, L., Probert, I., Garidel-Thoron, T. de, Bendif, E. M., Ruiz-Pino, D., Metzl, N., Goyet, C., Buchet, N., Coupel, P., Grelaud, M., Rost, B., Rickaby, R. E.M., & Vargas, C. de. (2011). Sensitivity of coccolithophores to carbonate chemistry and ocean acidification. *Nature, 476,* 80–83.

Beaugrand G., & Kirby R. R. (2010). Spatial changes in the sensitivity of Atlantic cod to climate-driven effects in the plankton. *Climate Research*, *41*, 15–19.

Beaugrand, G., Reid, P. C., Ibañez, F., Lindley, J. A., Edwards, M. (2002) Reorganization of North Atlantic marine Copepod biodiversity and climate. *Science, 296,* 1692–1694.

Beaugrand, G., Brander, K. M., Lindley, J. A., Souissi, S., Reid, P. C. (2003). Plankton effect on cod recruitment in the North Sea. *Nature, 426,* 661–664.

Beaugrand, G., Luczak, C., Edwards, M. (2009). Rapid biogeographical plankton shifts in the North Atlantic Ocean. *Global Change Biology, 15*(7),1790–1803.

Becu, N., Neef, A., Schreinemachers, P., & Sangkapitux, C. (2008). Participatory computer simulation to support collective decision-making: Potential and limits of stakeholder involvement. *Land Use Policy, 25*(4), 498–509.

Beddington, J. R., Agnew, D. J., & Clark, C. W. (2007). Current problems in the management of marine fisheries. *Science, 316,* 14713–1716.

Beebee, T., & Rowe, G. (2008). *An introduction to molecular ecology* (2nd ed.). Oxford: Oxford Univ. Press.

Behrenfeld, M. J., O'Malley, R. T., Siegel, D. A., McClain, C. R., Sarmiento, J. L., Feldman, G. C., Milligan, A. J., Falkowski, P. G., Letelier, R. M., & Boss, E. S. (2006). Climate-driven trends in contemporary ocean productivity. *Nature, 444,* 752–755.

Bekkevold, D., André, C., Dahlgre, T. G., Clausen, L. A., Torstensen, E., Mosegaard, H., Carvalho, G. R., Christensen, T. B., Norlinder, E., & Ruzzante, D. E. (2005). Environmental correlates of population differentiation in Atlantic herring. *Evolution, 59*(12), 2656–2668.

Belmonte, G., & Potenza, D. (2001). Biogeography of the family Acartiidae (Calanoida) in the Ponto-Mediterranean Province. *Hydrobiologia, 453/454,* 171–176.

Ben Rais Lasram, F., & Mouillot, D. (2009). Increasing southern invasion enhances congruence between endemic and exotic Mediterranean fish fauna. *Biological Invasions, 11*(3), 697–711.

Ben Rais Lasram, F., Guilhaumon, F., Albouy, C., Somot, S., Thuiller, W., & Mouillot, D. (2010). The mediterranean sea as a 'cul-de-sac' for endemic fishes facing climate change. *Global Change Biology, 16*(12), 3233–3245.

Béné, C., & Doyen, L. (2008). Contribution values of biodiversity to ecosystem performances: A viability approach. *Ecological Economics, 68*(1–2), 14–23.

Béné, C., & Friend, R. (2010). Poverty in small-scale inland fisheries: Old issues, new analysis. *Progress in Development Studies, 11*(2), 119–144.

Béné, C., Doyen, L., & Gabay, D. (2001). A viability analysis for a bio-economic model. *Ecological Economics, 36,* 385–396.

Béné, C., Belal, E., Baba, M. O., Ovie, S., Raji, A., Malasha, I., Njaya, F., Russell, A., & Neiland, A. (2009). Power struggle, dispute and alliance over local resources: Analysing "Democratic" decentralization of natural resources through the lenses of Africa Inland Fisheries. *World Development, 37*(12), 1935–1950.

Benn, A. R., Weaver, P. P., Billet, D. S. M., van den Hove, S., Murdock, A. P., Doneghan, G. B., & Le Bas, T. (2010). Human activities on the deep seafloor in the North East Atlantic: An assessment of spatial extent. *PLoS ONE, 5*(9), e12730.

Bianchi, G., Gislason, H., Graham, K., Hill, L., Jin, X., Korateng, K., Manickchand-Heileman, S., Paya, I., Sainsbury, K., Sanchez, F., & Zwanenburg, K. (2000). Impact of fishing on size composition and diversity of demersal fish communities. *ICES Journal of Marine Science,* 57, 558–571.

Bindoff, N. L., Willebrand, J., Artale, V., Cazenave, A., Gregory, J., Gulev, S., Hanawa, K., Le Quéré, C., Levitus, S., Nojiri, Y., Shum, C. K., Talley, L. D., & Unnikrishnan, A. (2007). Observations: Oceanic climate change and sea level. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, & H. L. Miller (Eds.), *Climate change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge: Cambridge University Press.

Blaikie, P. (2006). Is small really beautiful? Community-based natural resource management in Malawi and Botswana. *World Development, 34*(11), 1942–1957.

Block, B. A., Teo, S. L. H., Walli, A., Boustany, A., Stokesbury, M. J. W., Farwell, C. J., Weng, K. C., Dewar, H., Williams, T. D. (2005). Electronic tagging and population structure of Atlantic Bluefin Tuna. *Nature, 434,* 1121–1127.

Blondel, J., Aronson, J., Bodiou, J. Y., & Boeuf, G. (2010). *The Mediterranean region: biological diversity in space and time* (p. 376). New York: Oxford University Press Inc.

Boeuf, G. (2007). Océan et recherche biomédicale. *Journal de la Société de Biologie, 201*(1), 5–12.

Boeuf, G. (2008). Quel avenir pour la biodiversité? In J. P. Changeux & J. Reisse (Eds.), *Un monde meilleur pour tous, projet réaliste ou rêve insensé*? (pp. 47–98). Collège de France: Odile Jacob.

Boeuf, G. (2010a). *The gifts of marine biodiversity*. Unesco IYB Biodiversity Science-Policy Conference, 25–29 January 2010, Paris–Abstract. http://www.unesco.org/mab/doc/iyb/scConf_abstracts.pdf.

Boeuf, G. (2010b). The Biodiversity knowledge base: Taxonomy today and tomorrow for environmental sustainability and human well-being. Unesco IYB Biodiversity Science-Policy Conference, 25–29 January 2010.

Bœuf, G. (2011). *Marine biodiversity characteristics*. doi:10.1016/j.crvi.2011.02.009.

Boeuf, G., & Kornprobst, J. M. (2009). Bio et chimio-diversités marines. *Biofutur, 301,* 28–32.

Boncoeur, J., & Troadec, J. P. (2003). La régulation de l'accès. In L. Laubier (Ed.), *Exploitation et surexploitation des ressources marines vivantes* (pp. 355–394). Paris: Académie des Sciences, RST n°17, éditions Lavoisier.

Boncoeur, J., Alban, F., Guyader, O., Thébaud, O. (2002). Fish, fishers, seals and tourists, economic consequences of creating a marine reserve in a multi- species, multi-activity context. *Nature Resource Modelling, 15*(4), 387–411.

Bordenave, S., Jézéquel, R., Fourçans, A., Budzinski, H., Merlin, F.-X., Fourel, T., Goñi-Urriza, M., Guyoneaud, R., Grimaud, R., Caumette, P., Duran, R. (2004). Degradation of the Erika oil. *Aquatic Living Resources, 17*(3), 261–267.

Borrini-Feyerabend, G., Farvar, M. T., Nguinguiri, J. C., Ndangang, V. A. (2000). *Co-management of Natural Resource: Organising, negotiating and learning-by- doing* (p. 95). GTZ and IUCN, Kasparek Verlag, Heidelberg (Germany).

Bouchet, P. (2006). The magnitude of marine biodiversity. In C. M. Duarte (Ed.), *The exploration of marine biodiversity scientific and technological défis* (pp. 33–64). Paris: Fundación BBVA.

Bouchet, P., & Cayré, P. (2005). Richesse et fragilité de la biodiversité marine: monts sous-marins et récifs coralliens. In R. Barbault, B. Chevassus-au-Louis, & A. Teyssèdre (coord.) (Ed.), *Biodiversité et changements globaux: enjeux de société et défis pour la recherche* (pp. 73–74). Paris: ADPF, 2005. Biodiversité, Science et Gouvernance, Conférence mondiale, Paris (France).

Bousquet, F., & Le Page, C. (2004). Multi-agent simulations and ecosystem management: A review. *Ecological Modelling*, *176*, 313–332.

Boyd, R., Gintis, H., & Bowles, S. (2010). Coordinated punishment of defectors sustains cooperation and can proliferate when rare. *Science, 328,* 617.

Boyer, D. E., Kertesz, J. S., & Bruno, J. F. (2009). Biodiversity effects on productivity and stability of marine macroalgal communities: The role of environmental context. *Oikos, 118*(7), 1062–1072.

Branch, T. A., Watson, R., Fulton, E. A., Jennings, S., McGilliard, C. R., Pablico, G. T., Ricard, D., & Tracey, S. R. (2010). The trophic fingerprint of marine fisheries. *Nature, 468,* 431–435.

Brander, K. (1981). Disappearance of common skate, *Raia batis*, from Irish Sea. *Nature, 290,* 48–49.

Brown, J. H., & Maurer, B. A. (1989). Macroecology: The division of food and space among species on continents. *Science, 243,* 1145–1150.

Bruno, J. F., Stachowicz, J. J., & Berteness, M. D. (2003). Inclusion of facilitation into ecological theory. *Trends in Ecology & Evolution, 18*(3), 119–125.

Brusca, R. C., & Brusca, G. J. (2003). *Invertebrates* (2nd ed., p. 936). Sunderland: Sinauer Associates.

Bucklin, A., Steinke, D., & Blanco-Bercial, L. (2011). DNA barcoding of marine Metazoa. *Annual Review of Marine Sciences, 3,* 471–508.

Bulling, M. T., White, P. C., Raffaelli, D., & Pierce, G. J., (2006). Using model systems to address the biodiversity–ecosystem functioningprocess. *Marine Ecology Progress Series, 311,* 295–309.

Bundy, A., & Fanning, P. L. (2005). Can Atlantic cod (*Gadus morhua*) recover? Exploring trophic explanations for the non-recovery of the cod stock on the eastern Scotian Shelf. Canada, *Canadian Journal of Fisheries Aquatic Sciences, 62,* 1474–1489.

Burke, L., Reytar, K., Spalding, M., Perry, A., (2011). *Reefs at risk revisited* (p. i-x, 1–114). World Resources Institute. http://www.wri.org/publication/reefs- at-risk-revisited.

Byrne, M. (1996). Starfish wanted, dead or alive. *New Scientist*, *2052*, 53 p

Byrne, M., Morrice, M., & Wolf, B. (1997). Introduction of the northern Pacific seastar, Asterias amurensis, to Tasmania: Reproduction and current distribution. *Marine Biology, 127,* 673–685.

Byrnes, J., Stachowicz, J. J., Hultgren, K. M., Hughes, A. R., Olyanik, S. V., & Thornber, C. S. (2006). Predator diversity strengthens trophic cascades in kelp forests by modifying herbivore behaviour. *Ecology Letters, 9,* 61–71.

Callaway, R., Alsvag, J., De Boois, I., Cotter, J., Ford, A., Hinz, H., Jennings, S., Kröncke, I., Lancaster, J., Piet, G., & Prince, P., (2002). Diversity and community structure of epibenthic invertebrates and fish in the North Sea. *ICES Journal of Marine Science, 59,* 1199–1214.

Campbell, B., Mandondo, A., Nenarundwe, N., Sithole, B., De Jong, W., Luckert, & M., & Matose, F. (2001). Challenges to proponents of common property resource systems: despairing voices from the social forests of Zimbabwe. *World Development*, *29*(4), 589–600.

Campbell, B., Sayer, J. A., Frost, P., Vermeulen, S., Perez, M. R., Cunningham, A., & Prabhu, R. (2001). Assessing the performance of natural resource systems. *Conservation Ecology, 5*(2), 22.

Cappell, R., Huntington, T., & Macfadyen, G. (2010). FIFG2000-2006. Shadow evaluation. Report to the Pew Environment group.

Carlton, J., & Geller, J. (1993). Ecological roulette: The global transport of non-indigenous marine organisms. *Science, 262,* 78–82.

Carlton, J. T., Geller, J. B., Reaka-Kudla, M. L., & Norse, E. A. (1999). Historical extinctions in the sea. *Annual Review of Ecology & Systematics, 30,* 515–538.

Carpenter, S. R., Brock, S. R., & Brock, W. A. (2006). Rising variance: A leading indicator of ecological transition. *Ecology Letters, 9*(3), 311–318.

Carpenter, S., Walker, B., Anderies, J. M., & Abel, N. (2001). From metaphor to measurement: Resilience of what to what? *Ecosystems, 4*(8), 765–781.

Carvalho, G. R. (1998). Molecular ecology: Origins and approaches. In Carvalho, G. R. (Ed.), *Advances in molecular ecology* (pp. 1–23). Amsterdam: NATO Science Series, IOS Press.

Carvalho, G. R., Creer, S., Allen, M., Costa, F. O., Tsigenopoulos, C. S., Le Goff-Vitry, M., Magoulas, A. M., Medlin, L., & Metfies, K. (2010). Genomics in the discovery and monitoring of marine biodiversity. Advances in Marine Genomics, Vol. 1. In J. M. Cock, K. Tessmar-Raible, C. Boyen, & F. Viard (Eds.), *An introduction to marine genomics: Advances in Marine Genomics, Vol. 1.* (p. 370). New York: Springer-Verlag.

Casini, M., Hjelm, J., Molinero, J.-C., Lövgren, J., Cardinale, M., Bartolino, V., Belgrano, A., & Kornilovs, G. (2009). Trophic cascades promote threshold-like shifts in pelagic marine ecosystems. *Proceedings of the National Academy of Sciences, 106*(1), 197–202.

CAS. (2011). *Les aides publiques dommageables à la biodiversité* (p. 336). Rapport & Documents—Mission présidée par G. Sainteny, Paris, La Documentation française.

CAS. (2009). *Approche économique de la biodiversité et des services liés aux écosystèmes* (p. 376). CAS Report & Documents—Mission presided by B. Chevassus au Louis, Paris, La Documentation française.

Cavendish, W. (2000). Empirical regularities in the poverty-environment relationship of rural households: Evidence from Zimbabwe. *World Development, 28*(11), 1979–2003.

CBD. (2009). Scientific synthesis of the impacts of ocean acidification on marine biodiversity. *Montreal*, *Technical Series, 46,* 61.

CBD. (2010). Global biodiversity outlook 3. Montréal, p. 94.

Chamaille-Jammes, S., Massot, M., Aragon, P., & Clobert, J. (2006). Global warming and positive fitness response in mountain populations of common lizards Lacerta vivipara. *Global Change Biology, 12,* 392–402.

Chao, A., & Shen, T. J. (2005). SPADE (Species Prediction and Diversity Estimation) programme. Program and user's guide at http://chao.stat.nthu.edu. tw.

Chao, A., Chazdon, R. L., Colwell, R. K., & Shen, T. J. (2005). A new statistical approach for assessing similarity of species composition with incidence and abundance data. *Ecology Letters, 8,* 148–159.

Charles, A. T. (1994). Towards sustainability: The fishery experience. *Ecological Economics, 11,* 201–211.

Chassot, E., Bonhommeau, S., Dulvy, N. K., Mélin, F., Watson, R., Gascuel, D., & Le Pape, O. (2010). Global marine primary production constrains fisheries catches. *Ecology Letters, 13,* 495–505.

Cheung, W. W. L., Close, C., Lam, V., Watson, R., & Pauly, D. (2008). Application of macroecological theory to predict effects on climate change on global fisheries potential. *Marine Ecology Progress Series, 365,* 187–197.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., & Pauly, D. (2009). Projecting global marine biodiversity impacts under climate change scenarios. *Fish and Fisheries, 10*(3), 235–251.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., Zeller, D., & Pauly, D. (2010). Large-scaleredistribution of maximum fisheries catch potential in the global ocean under climate change. *Global Change Biology, 16*(1), 24–35.

Cheung, W. W. L., Dunne, J., Sarmiento, J. L., & Pauly, D. (2011). Integrating ecophysiology and plankton dynamics into projected maximum fisheries catch potential under climate change in the Northeast Atlantic. *ICES Journal of Marine Science, 68*(6), 1008–1018.

Christophersen, Ø., Mathisen, R., Niesing, H., & Weiss, A. (2009). *Assessment of climate change mitigation and adaptation* (p. 40). OSPAR Commission Reports, *Monitoring and assessment Series*.

Chukwuone, N. A., & Okorji, C. E. (2008). Willingness to pay for systematic management of community forests for conservation of non-timber forest products in Nigeria's rainforest region: Implications for poverty alleviation. In R. B. Dellink & A. Ruijs (Eds.), *Economics of poverty, environment and natural resource use: Wageningen UR Frontis Series Vol. 25* (pp. 117–137). Springer.

Cimmaruta, R., Bondanelli, P., Ruggi, A., & Nascetti, G. (2008). Genetic structure and temporal stability in the horse mackerel (Trachurus trachurus). *Fisheries Research, 89,* 114–121.

Cimmatura, R., Bondanelli, P., & Nascetti, G. (2005). Genetic structure and environmental heterogeneity in the European hake (*Merluccius merluccius*). *Molecular Ecology, 14,* 2577–2591.

Clark, C. W. (1976). *Mathematical bio-economics: The optimal management of renewable resources*. New York: Wiley (1990).

Clark, M. R. (2010). Effects of trawling on seamounts.*Oceanography*, *23*(1), 132–133.

Clarke, K. R., & Warwick, R. M. (1994). *Change in marine communities: An approach to statistical analysis and interpretation*. Natural Environment Research Council, Plymouth, UK.

Clarke, K. R., & Warwick, R. M. (1998). A taxonomic distinctness index and its statistical properties. *Journal of Applied Ecology, 35,* 523–531.

Clarke, K. R., & Warwick, R. M. (2001). *Change in marine communities: An approach to statistical analysis and interpretation* (2nd ed., p. 190). Plymouth: Primer-E Ltd.

Cochrane, K., De Young, C., Soto, D., & Bahri T. (Eds.). (2009). Climate change implications for fisheries and aquaculture: Overview of current scientific knowledge. *FAO Fisheries and Aquaculture Technical Paper*, 530 (p. 212). Rome, FAO.

Coleman, M. L., Sullivan, M. B., Martiny, A. C., Steglich, C., Barry, K., DeLong, E. F., & Chisholm, S. W. (2006). Genomic islands and the ecology and evolution of Prochlorococcus. *Science, 311*(5768), 1768–1770.

Coll, M., Libralato, S., Tudela, S., Palomera, I., & Pranovi, F. (2008). Ecosystem overfishing in the ocean. *PLoS ONE, 3*(12), e3881.

Coll, M., Piroddi, C., Steenbeek, J., Kaschner, K., Ben Rais Lasram, F., Aguzzi, J., Ballesteros, E., Bianchi, C. N., Corbera, J., Dailianis, T., Danovaro, R., Estrada, M., Froglia, C., Galil, B. S., Gasol, J. M., Gertwagen, R., Gil, J., Guilhaumon, F., Kesner-Reyes, K., Kitsos, M.-S., Koukouras, A., Lampadariou, N., Laxamana, E., López-Fé de la Cuadra, C. M., Lotze, H. K., Martin, D., Mouillot, D., Oro, D., Raicevich, S., Rius-Barile, J., Saiz-Salinas, J. I., San Vicente, C., Somot, S., Templado, J., Turon, X., Vafidis, D., Villanueva, R., & Voultsiadou, E. (2010). The biodiversity of the Mediterranean Sea: Estimates, patterns, and threats. *PLoS ONE, 5*(8), e11842.

Collins, S. L., & Benning, T. L. (1996). Spatial and temporal patterns in functional diversity. In K. J. Gaston (Ed.), *Biodiversity: A biology of numbers and difference* (pp. 253–280). Oxford: Blackwell Science.

Comeau, S., Gorsky, G., Jeffree, R., Teyssié J.-L., & Gattuso, J.-P. (2009). Impact of ocean acidification on a key Arctic pelagic mollusk (*Limacina helicina*). *Biogeosciences, 6,* 1877–1882.

Commission des communautés européennes. (2009). *Livre vert. Réforme de la politique commune de la pêche* (p. 30). COM (2009) 163 final.

Commission des communautés européennes. (2011). *Proposition de règlement du Parlement européen et du Conseil relatif à la politique commune de la pêche* (p. 93). COM (2011) 425 final.

Commission européenne. (2011). Recommandations de la Commission du 16 septembre 2011 concernant l'initiative de programmation conjointe en matière de recherche "Healthy and Productive Seas and Oceans" (Des mers et des océans sains et productifs), C.276/1–3.

Condon, R. H., Steinberg, D. K., del Giorgio, P. A., BouvierT. C., Bronk, D. A., Graham, W. M., & Ducklow, H. W. (2011). Jellyfish blooms result in a major microbial respiratory sink of carbon in marine systems. *Proceedings of the National Academy of Sciences, 108*(25), 10225–10230.

Condon, R. H., Graham, W. M., Duarte, C. M., Pitt, K. A., Lucas, C. H., Haddock, S. H. D., Sutherland, K. R., Robinson, K. L., Dawson, M. N., Decker, M. B., Mills, C. E., Purcell, J. E., Malej, A., Mianzan, H., Uye, S. I., Gelcich, S., & Madin, L. P. (2012). Questioning the rise of gelatinous zooplankton in the world's oceans, *BioScience, 62*(2), 160–169.

Conservation International. (2010). Millions of sea turtles are collateral damage in the race for fish. www.conservation.org/newsroom/pressreleases/Pages/SEATURTLE_BYCATCH_REPORT.aspx.

Costa, F. O., & Carvalho, G. R. (2007). The barcode of life initiative: Synopsis and prospective societal impacts of DNA barcoding of fish. *Genomics, Society and Policy, 3,* 29–40.

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

Costello, M., Bouchet, P., Emblow, C., & Legakis, A. (2006). European marine biodiversity inventory and taxonomic resources: State of art and gaps in the knowledge. *Marine Ecology Progress Series, 316,* 257–268.

Costello, M. J., Coll, M., Danovaro, R., Halpin, P., Ojaveer, H., & Miloslavich, P. (2010). A census of marine biodiversity knowledge, resources, and future challenges. *PLoS ONE, 5*(8), e12110.

Couvet, D., & Teyssèdre-Couvet, A. (2010). *Écologie et biodiversité* (p. 336). Paris: Belin Editions.

Cowen, R. K., Paris, C. B. & Srinivasan, A. (2006). Scaling of connectivity in marine populations. *Science, 311*(5760):522–527.

Crain, C. M., Kroeker, K., & Halpern, B. S. (2009). Interactive and cumulative effects of multiple human stressors in marine systems. *Ecology Letters, 11,* 1304–1315.

Creer, S., Fonseca, V. G., Porazinska, D. L., Giblin-Davis, R. M., Sung, W., Power, D. M., Packer, M., Carvalho, G. R., Blaxter, M. L., Lambshead, P. J., Tomas, W. K. (2010). Ultrasequencing of the meiofaunal biosphere: Practice, pitfalls and promises. *Molecular Ecology, 19*(1), 4–20.

Crossett, K. M., Culliton, T. J., Wiley, P. C., & Goodspeed, T. R. (2004). Population Trends Along the Coastal United States: 1980–2008 National Oceanic and Atmospheric Administration, National Ocean Service, Management and Budget Office Special Projects. http://oceanservice.noaa.gov/programs/mb/pdfs/coastal_pop_trends_complete.pdf.

Crouch, E. M., Heilmann-Clausen, C., Brinkhuis, H., Morgans, H. E.G., Rogers, K. M., Egger, H., Schmitz, B. (2001). Global dinoflagellate event associated with the late Paleocene thermal maximum. *Geology, 29,* 315–318.

CSD (Commission on Sustainable Development). (2001). Indicators of sustainable development: Guidelines and methodologies. Division for Sustainable Development, United Nations, New York.

CSPNB. (2007). *La biodiversité à travers des exemples* (p. 104). MEDD/D4E.

CSPNB. (2008). *La biodiversité à travers des exemples, les réseaux de la vie* (196 pages). MEEDAD.

Cury, P., & Morand, S. (2005). Biodiversité marine et changements globaux: une dynamique d'interactions où l'humain est partie prenante. In R. Barbault, B. Chevassus-au-Louis, & A. Teyssèdre (coord.) *Biodiversité et Changements globaux: enjeux de société et défis pour la recherche* (pp. 50–79).

Cury, P., Shannon, L. J., & Shin, Y. J. (2003). The functioning of marine ecosystems: A fisheries perspective. In M. Sinclair & G. Valdimarsson. *Responsible fisheries in the marine ecosystem* (pp. 103–123). CAB International Wallingford.

Cury, P. M., Jai Shin, Y., Planque, B., Durant, J. M., Fromentin, J. M., Kramer Schadt, S., Stenseth, N. C., Travers, M., & Grimm, V. (2008). Ecosystem oceanography for global change in fisheries. *Trends in Ecology and Evolution, 23*(6), 338–346.

Cushing, D. H. (1982). *Climate and fisheries* (p. 373). London: AcademicPress.

Czech, B. (2008). Prospects for reconciling the conflict between economic growth and biodiversity conservation with technological progress. *Conservation Biology, 22*(6), 1389–1398.

D'Amato, M. E., Harkins, G. W., de Oliveira, T., Teske, P. R., & Gibbons, M. J. (2008). Molecular dating and biogeography of the neritic krill Nyctiphanes. *Marine Biology, 155,* 243–247.

Danovaro, R., Gambi, M. C., Dell'Anno, A., Corinaldesi, C., Fraschetti, S., Vanreusel, A., Vincx, M., & Gooday, A. J. (2008). Exponential decline of deep-sea ecosystem functioning linked to benthic biodiversity loss. *Current Biology*, *18*(1), 1–8 + supplemental data: S1–S6; 1–16.

Darwin, C. (1896). *The variation of animals and plants under domestication* (Vol. II). New York: D Appleton & Co.

Daskalov, G. M., Grishin, A. N., Rodionov, S., & Mihneva, V. (2007). Trophic cascades triggered by overfishing reveal possible mechanisms of ecosystem regime shifts. *Proceedings of the National Academy of Sciences, 104*(25), 10518–10523.

De Lara, M., & Doyen, L. (2008). *Sustainable management of natural resources. Mathematical models and methods*. Berlin: Springer-Verlag.

De Young, B., Barange, M., Beaugrand, G., Harris, R., Perry, R. I., Scheffer, M., & Werner, F. (2008). Regime shifts in marine ecosystems: Detection, prediction and management. *Trends in Ecology and Evolution*, *23*, 402–409.

De Young, C., Charles, A., & Hjort, A. (2008). Human dimensions of the ecosystem approach to fisheries: An overview of context, concepts, tools and methods. *FAO Fisheries Technical Papers*, 489. Food & Agriculture Organization of the United Nations, Rome, 2008.

De Young, B., Harris, R., Alheit, J., Beaugrand, G., Mantua, N., & Shannon, L. (2004). Detecting regime shifts in the ocean: Data considerations. *Progress in Oceanography, 60,* 143–164.

Deconinck, J. F. (2006). *Paléoclimats* (p. 198). Vuibert.

Del Monte-Luna, P., Lluch-Belda, D., Serviere-Zaragoza, E., Carmona, R., Reyes-Bonilla, H., Aurioles-Gamboa, D., Castro-Aguirre, J. L., Guzman del Proo, S. A., Trujillo-Millan, O. B., & Brook, B. W. (2007). Marine extinctions revisited. *Fish & Fisheries, 8*(2), 107–122.

Delgado, C. L., Wada, N., Rosegrant, M. W., Meijer, S., & Ahmed, M. (2003). *Fish to 2020. Supply and demand in changing global market* (p. 226). International Food Policy Research Institute. Washington, D. C.

Denman, K. L., Brasseur,G., Chidthaisong, A., Ciais, P., Cox, P. M., Dickinson, R. E., Hauglustaine, D., Heinze, C., Holland, E., Jacob, D., Lohmann, U., Ramachandran, S., da Silva Dias, P. L., Wofsy, S. C., & Zhang, X. (2007). Couplings Between Changes in the climate system and biogeochemistry. In S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, & H. L. Miller (Eds.), *Climate Change 2007: The Physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Chang*e. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Descola, P. (2005). *Par delà Nature et Culture* (p. 623). Paris: Galllimard.

Descola, P., & Palsson, G. (Eds.). (1996). *Nature and society: Anthropological perspectives* (p. 324). London: Routledge.

Deser, C., Alexander, M. A., Xie, S. P., & Phillips, A. S. (2010). Sea surface temperature variability: patterns and mechanisms. *Annual Review of Marine Sciences, 2,* 115–143.

DeWitt, T. J., Sih, A., & Wilson, D. S. (1998). Coasts and limits of phenotypic plasticity. *Trends in Ecology and Evolution, 13,* 78–81.

Diaz, R. J., & Rosenberg, R. (2008).Spreading dead zones and consequences for marine ecosystems, *Science, 321,* 926–929.

Doney, S. C. (2006). Plankton in a warmer world. *Nature, 444,* 695–696.

Doney, S. C. (2010). The growing human footprint on coastal and open-ocean biogeochemistry. *Science, 328,* 1512–1516.

Doney, S. C., Fabry, V. J., Feely, R. A., & Kleypas, J. A. (2009). Ocean acidification: The other CO_2 problem. *Annual Review of Marine Sciences, 1,* 69–92.

Dong, B., Sutton, R. T., & Woollings, T. (2010). Changes of interannual NAO variability in response to greenhouse gases forcing. *Climate Dynamics*, doi:10.1007/s00382-010-0936-6.

Drinkwater, K., Schrum, C., & Brander, K. (Eds.). (2010). Cod and future climate change. *ICES Cooperative Research Report, 305,* 88.

Duhamel du Monceau H. L., & Delamarre, R. P. (1769). Traité général des pêches et histoire des poissons qu'elles contiennent (en collaboration avec Delamarre), Paris, Saillant et Nyons, 1769-1777, 3 vol. in folio

Dulvy, N. K., Sadovy, Y., & Reynolds, J. D. (2003). Extinction vulnerability in marine populations. *Fish and Fisheries, 4,* 25–64.
Dulvy, N. K., Jennings, S., Rogers, S. I., & Maxwell, D. L. (2006). Threat and decline in fishes: An indicator of marine biodiversity. *Canadian Journal of Fisheries and Aquatic Sciences, 63*(6), 1267–1275.
Dupar, M., & Badenoch, N. (Eds.). (2002). *Environment, livelihoods, and local institutions: Decentralization in Mainland Southeast Asia* (p. 77). Washington: World Resources Institute.
Duraiappah, A. (1998). Poverty and environmental degradation: A review and analysis of the nexus. *World Development*, *26*(12), 2169–2217.
Durand, J. R., Lemoalle, J., & Weber, J. (Eds.). (1990). *La recherche face aux pêches artisanales*. Symposium international, OSTOM—IFREMER, Montpellier, 1–7 juillet 1989. Paris, Orstom, coll. et docs.
Dziegielewska, D., Tietenberg, T., & Seo, S. N. (2007). Total economic value, In C. J. Cleveland (Ed.), Encyclopedia of earth, Washington D. C., Environmental Information Coalition, National Council for Science and the Environment. http:// www.eoearth.org/article/Total_economic_value.
Edwards, M., & Richardson, A. (2004). Impact of climate change on marine pelagic phenology and trophic mismatch. *Nature, 430,* 881–884.
Edwards, M., Johns, D. G., Leterme, S. C., Svendsen, E., & Richardson, A. J. (2006). Regional climate change and harmful algal blooms in the Northeast Atlantic. *Limnology and Oceanography, 51*(2), 820–829.
Edwards, M., Beaugrand, G., Hays, G. C., Koslow, J., & Richardson, A. J. (2010). Multi-decadal oceanic ecological datasets and their application in marine policy and management. *Trends in Ecology and Evolution, 25*(10), 602–610.
Essington, T. E., Beaudreau, A. H., & Wiedenmann, J. (2006). Fishing through marine food webs. *Proceedings of the National Academy of Sciences, 103*(9), 3171–3175.
European Environment Agency (EEA). (2003). Europe's environment: The third assessment. EEA, Copenhagen.
European Environmental Agency (EEA). (2006). Progress towards halting the loss of biodiversity by 2010. EEA Report N°5.
European Environmental Agency (EEA). (2007). Europe’s environment—The fourth assessment (452 p.). Copenhagen.
European Environmental Agency (EEA). (2010). The European environment—State and outlook 2010 (SOER 2010). Synthesis, p. 222;Thematic assessments: Biodiversity, p. 45; Marine and coastal environment, p. 55; Understanding climate change, p. 34.
European Commission. (2010). Options for an EU vision and target for biodiversity beyond 2010. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions.COM (2010) 4 final.
European Commission. (2008). Communication from the commission to the council and the European parliament. The role of the CFP in implementing an ecosystem approach to marine management. COM (2008) final & SEC (2008) 449.
European Science Foundation. (2009). Impacts of ocean acidification. *Science Policy Briefing*, *37*, 12. http://www.esf.org/nc/publications/science-policy- briefings.html.
European Union (EU). (2008). Directive 2008/53/EC of the European Parliament and of the Council of 17June 2008 establishing a framework for community action in the field of marine environmental policy (Marine Strategy Framework Directive).
Fabri, M. C., Galéron, J., Larour, M., & Maudire, G. (2006). Combining the biocean database for deep-sea benthic data with the on-line ocean biogeographic information system. *Marine Ecology Progress Series, 316,* 215–224.
Fabry, V. J., McClintock, J. B., Mathis, J. T., & Grebmeier, J. M. (2009). Ocean acidification at high latitudes: the bellweather, *Oceanography*, *22*(4), 160171. htttp://www.tos.org/oceanography/issues/issue_ archive/22_4.html.
FAO. (2003). *L’approche écosystémique des pêches*. Technical guidelines for responsible fisheries—N° 4 Suppl. 2, Rome, p. 133.

FAO. (2009). Agreement on port state measures to prevent, deter and eliminate illegal, unreported and unregulated fishing. http://www.fao.org/Legal/ treaties/037s-e.html.
FAO. (2010). SOFIA. The state of world fisheries and aquaculture 2010, Rome, p 197.
FAO. (2011). *Review of the state of world marine fishery resources* (p. 334). FAO fisheries and aquaculture technical Paper No. 569. Rome: FAO. http://www.fao. org/docrep/015/i2389e/i2389e00.html.
Feeny, D., Hanna, S., & McEvoy, A. (1996). Questioning the assumptions of the 'Tragedy of the Commons' model of fisheries. *Land Economics, 72*(2), 187–205.
Fernand, L., & Brewer, P. (Eds.). (2008). Changes in surface CO_2 and ocean pH in ICES shelf sea ecosystems. *ICES Cooperative Research Report, 290,* 35.
Finney, B. P., Gregory-Eaves, I., Douglas, M. S.V., & Smol, J. P. (2002). Fisheries productivity in the Hortheastern pacific ocean over the past 2,200 years. *Nature, 416,* 729–733.
Fisher, B., Turner, K., Zylstra, M., Brouwer, R., de Groot, R., Farber, S., Ferraro, P., Green, R., Hadley, D., Harlow, J., Jefferiss, P., Kirkby, C., Morling, P., Mowatt, S., Naidoo, R., Paavola, J., Strassburg, B., Yu, D., & Balmford, A. (2008). Ecosystem services and economic theory: Integration for policy-relevant research. *Ecological Applications, 18*(8), 2050–2067.
Fisher, B., Turnier, R. K., & Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics, 68,* 643–653.
Fisher, J. A.D., Frank, K. T., Petrie, B., Leggett, W. C., & Shackell, N. L. (2008). Temporal dynamics within a contemporary latitudinal diversity gradient. *Ecology Letters, 11,* 883–897.
Frakes, L. A., Probst, J. L., & Ludwig, W. (1994). *Latitudinal distribution of paleotemperatures on land and sea from early Cretaceous to middle Miocene*. Comptes Rendus de l'Académie des Sciences, *Séries II, 318,* 1209–1218.
Frank, K. T., Petrie, B., Fisher, J. A.D., & Legget, W. C. (2011). Transient dynamics of an altered large marine ecosystem. *Nature,* online 2011 07 27, doi:10.1038/nature10285.
Frid, C. L., Garwood, P., & Robinson, L. A. (2009). The North sea benthic system: A 36 year time-series. *Journal of Marine Biological Association, 89,* 1–10.
Fromentin, J. M. (2009). Lessons from the past: investigating historical data from bluefin tuna fisheries. *Fish and Fisheries, 10,* 197–216.
Gaikwad, J., & Chavan, V. (2006). Open access and biodiversity conservation: Challenges and potentials for developing world. *Data Science Journal, 5,* 1–17.
Gailhard, I., Gros, Ph., Durbec, J. P., Beliaeff, B., Belin, C., Nézan, E., & Lassus, P. (2002). Variability patterns of microphytoplankton communities along the French coasts. *Marine Ecology Progress Series, 242,* 39–50.
Garcia, S. M., & Charles, A. T. (2007). Fishery systems and linkages: From clockworks to soft watches. *ICES Journal of Marine Science, 64,* 580–587.
Garcia, S. M., & Charles, A. T. (2008). Fishery systems and linkages: Implications for science and governance. *Ocean & Coastal Management, 51,* 505–527.
Garcia, S. M., & de Leiva Moreno, J. I. (2001). Global overview of marine fisheries. In *Reykjavik Conference on Responsible Fisheries in the Marine Ecosystem*, Reykjavik, Iceland, p. 23 (1–4 October 2001).
Garcia, S. M., de Leiva Moreno, J. I., & Grainger, R. J.R. (2005). Global trends in the state of marine fisheries resources 1974–2003. In Review of the state of world marine fishery resources. *FAO Fisheries Technical Papers*, *457*, 10–14. Rome: FAO.
Gaston, K. J., & Blackburn, T. M. (2000). *Pattern and process in macroecology*. Blackwell Science.
Gattuso, J.-P., & Hansson, L. (Eds.). (2011). *Ocean acidification* (p. 352). OUP Oxford.
Gattuso, J. P., Orr, J., Pantoja, S., Pörtner, H. O., Riebesell, U., & Trull, T. (Eds.). (2008). The ocean in the high- CO_2 world, II. http://www.biogeosciences.net/special_ issue44.html.
Gawarkiewicz, G. S., Monismith, S., & Largnier, J. (2008). Observing larval transport processes affecting population connectivity. Progress & Challenges.*Oceanography*, *20*, 40–53.
Gereffi, G. (1999). International trade and industrial upgrading in the apparel commodity chain. *Journal of International Economics, 48*(1), 37–70.
Glover, A. G., & Smith, C. R. (2003). The deep-sea floor ecosystem: Current status and prospects of anthropogenic change by the year 2025. *Environmental Conservation, 30,* 219–241.

Glover, A. G., Gooday, A. J., Bailey, D. M., Billett, D. S. M., Chevaldonné P., Colaço, A., Copley, J., Cuvelier, D., Desbruyères, D., Kalogeropoulou, V., Klages, M., Lampadariou, N., Lejeusne, C., Mestre, N. C., Paterson, G. L. J., Perez, T., Ruhl, H., Sarrazin, J., Soltwedel, T., Soto, E. H., Thatje, S., Tselepides, A., Van Gaever, S., & Vanreusel, A. (2010). Temporal change in deep-sea benthic ecosystems: A review of the evidence from recent time-series studies, *Advances in Marine Biology, 58,* 1–97.

Gomez, A., Wright, P. J., Lunt, D. H., Cancino, J. M., Carvalho, G. R., & Hughes, R. N. (2007). Mating trials validate the use of DNA barcoding to reveal cryptic speciation of a marine bryozoan taxon. *Proceedings of the Royal Society of London, B. 274*, 199–207.

Goulletquer, P., Bachelet, G., Sauriau, P. G., & Noël, P. (2002). Open Atlantic coast of Europe—a century of introduced species into French waters. In E. Leppäkoski et al. (Eds.), *Invasive aquatic species in Europe* (pp. 276–290). London: Kluwer Academic Publishers.

Graham, M. H., Dayton, P. K., & Erlandson, J. M. (2003). Ice ages and ecological transitions on temperate coasts. *Trends in Ecology and Evolution, 18,* 33–40.

Grassle, J. F., & Maciolek, N. J. (1992). Deep-sea species richness: Regional and local diversity estimates from quantitative bottom samples. *American Naturalist, 139,* 313–341.

Greene, C. H., & Pershing, A. J. (2007). Climate drives sea changes. *Science, 315,* 1084–1085.

Greenstreet, S. P., Spencer, F. E., & McMillan, J. A. (1998). Fishing effects in Northeast atlantic shelf seas: Patterns in fishing effort, diversity and community structure. V. Changes in the structure of the North sea groundfish species assemblage between 1925 and 1996. *Fisheries Research, 40,* 153–183.

Groombridge, B., & Jenkins, M. D. (Eds.). (2000). *Global biodiversity: Earth's living resources in the 21st century*. Cambridge: World Conservation Press.

Gros, Ph. (2010). Vers une pêche et une aquaculture viables dans un monde changeant. In Aliments de demain et santé: réalités et fantasmes, *Dossier scientifique* (p. 158, pp. 93–115). n° 16, Institut français pour la nutrition. http://archimer. ifremer.fr/doc/00027/13845/10991.pdf.

Gros, Ph. (2011). Écosystèmes marins. In N. Massu & G. Landmann (coord.). (Eds.), *CCBio, Connaissance des impacts du changement climatique sur la biodiversité en France métropolitaine—synthèse de la bibliographie* (Chap. 5, p. 181, pp. 127–165). Ecofor, MEDDTL. http://archimer.ifremer.fr/doc/00043/15394/12749.pdf.

Gros, Ph., Biseau, A., Fromentin, J. M., & Thébaud, O. (2008). *Strategic research priorities to the Common fishery policy (CFP) with regard to global commitments (MSY, EAF, MFSD)* (p. 34). European Parliament's Committee on Fisheries, PE 408.936. http://archimer.ifremer.fr/doc/00013/12408/9194.pdf.

Guarinello, M. L., Shumchenia, E. J., & King, J. W. (2010). *Marine habitat classification for ecosystem-based management: A proposed hierarchical framework*. Environmental Management (online).

Haddock, S. H. D. (2008).Reconsidering evidence for potential climate-related increases in jellyfish. *Limnology and Oceanography, 53*(6), 2759–2762.

Hall, S. J. (1999). *The effects of fishing on marine ecosystems and communities* (p. 274). Oxford: Blackwell Science ed.

Hall, S. J., Collie, J. S., Duplisea, D. E., Jennings, S., Bravington, M., & Link, J. (2006). A length-based multi-species model for evaluating community responses tofishing. *Canadian Journal of Fisheries and Aquatic Sciences, 63,* 1344–1359.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R., & Watson, R. (2008). A global map of human impact on marine ecosystems. *Science, 319*(5865), 948–952.

Hames, R. (2007). The ecologically noble savage debate. *Annual Review of Anthropology, 36,* 177–190.

Hanna, S. (2004). The economics of protected areas in marine fisheries management: an overview of issues. In J. B. Shipley (Ed.), *Aquatic protected areas as fisheries management tools* (pp. 259–265). American Fisheries Society Symposium 42, American Fisheries Society, Bethesda, MD.

Hanna, S., & Sampson, D. (2009). *Report of the ocean policy advisory council's scientific and technical advisory committee technical workshop on economic data and analysis of marine reserves*. Oct 21–22 2008, Hatfield Marine Science Center Newport, OR.

Hansen, J., Sato, M., Ruedy, R., Lo, K., Lea, D. W., & Medina-Elizade, M. (2006). Global temperature change. *Proceedings of the National Academy of Sciences*, *103*(39), 14288–14293.

Hardin, G. (1968). The tragedy of the commons. *Science, 162,* 1243–1248.

Hatcher, A., Jabbar, S., Thébaud, O., & Bennett, E. (2000). Normative and social influences affecting compliance with fishery regulations. *Land Economics, 76,* 448–462.

Hauser, L., & Carvalho, G. R. (2008). Paradigm shifts in marine fisheries genetics: Ugly hypotheses slain by beautiful facts. *Fish and Fisheries, 9,* 333–362.

Hebert, P. D., Cywinska, A., Ball, S. L., & DeWaard, J. R. (2003). Biological identifications through DNA barcodes. *Proceedings of the Royal Society of London, B. 270*, 313–321.

Heino, M., & Dieckmann, U. (2004). *Exploitation as a driving force of life history evolution: methods and empirical analyses*. International Council for Exploration of the Sea (ICES) CM 2004/K:35.

Helmreich, S. (2005). How Scientists think: About "Natives", for example. A Problem of Taxonomy among biologists of Alien Species in Hawaii. *Journal of the Royal Anthropological Institute* (*N.S.), 11,* 107–128.

Hemmer-Hansen, J., Nielsen, E. E., Grønkjær, P., & Loeschcke, V. (2007). Evolutionary mechanisms shaping the genetic population structure of marine fishes; lessons from the European flounder (*Platichthys flesus* L). *Molecular Ecology, 16,* 3104–3118.

Hendriks, I. E., Duarte, C. M., Àlvarez, M. (2010). Vulnerability of marine biodiversity to ocean acidification: A meta-analysis. *Estuarine, Coastal and Shelf Science, 86*(2), 157–164.

Henrickson, S. E., Wong, T., Allen, P., Ford, T., & Epstein, P. R. (2001). Marine swimming-related illness: implications for monitoring and environmental policy. *Environmental Health Perspectives, 109*(7), 645–650.

Hermant, M., Lobry, J., Bonhommeau, S., Poulard, J. C., & Le Pape, O. (2010). Impact of warming on abundance and occurrence of flatfish populations in the Bay of Biscay (France). *Journal of Sea Research, 64,* 45–53.

Hewitt, G. M. (2000). The genetic legacy of the quaternary ages. *Nature, 405,* 907–913.

High Seas Task Force. (2006). *Closing the net: Stopping illegal fishing on the high seas* (p. 112). Governments of Australia, Canada, Chile, Namibia, New Zealand, and the United Kingdom, WWF, IUCN and the Earth Institute at Columbia University.

Hilborn, R. (2007). Defining success in fisheries and conflicts in objectives. *Marine Policy, 31,* 153–158.

Hilborn, R., Quinn, T., Schindler, D., & Rogers, D. E. (2003). Biocomplexity and fisheries sustainability. *Proceedings of the National Academy of Sciences*, *100*(11), 6564–6568.

Hilborn, R., Branch, T. A., Ernst, B., Magnusson, A., Minte-Vera, C. V., Scheuerell, M. D., & Valero, J. L. (2003). State of the world's fisheries. *Annual Review of Environment and Resources, 28,* 15.1–15.40.

Hjort, J. (1914). Fluctuations in the great fisheries of northern Europe viewed in the light of biological research. *Rapports et procès-verbaux des réunions/Conseil permanent international pour l'exploration de la mer, 20,* 1–228.

Hoegh-Guldberg, O., & Bruno, J. F. (2010). The impact of climate change on the world's marine ecosystems. *Science, 328,* 1523–1528.

Hoegh-Guldberg, O., Mumby, P. J., Hooten, A. J., Steneck, R. S., Greenfield, P., Gomez, E., Harvell, C. D., Sale, P. F., Edwards, A. J., Caldeira, K., Knowlton, N., Eakin, C. M., Iglesias-Prieto, R., Muthiga, N., Bradbury, R. H., Dubi, A., & Hatziolos, M. E. (2007). Coral reefs under rapid climate change and ocean acidification.*Science, 318,* 1837–1742.

Hofmann, G. E., Barry, J. P., Edmunds, P. J., Gates, R. D., Hutchins, D. A., Klinger, T., & Sewell, M. A. (2010). The effect of ocean acidification on calcifying organisms in marine ecosystems: An organism-to-ecosystem perspective. *Annual Review of Ecology and Systematics, 41,* 127–147.

Hoggarth, D. D., Abeyasekera, S., Arthur, R. I., Beddington, J. R., Burn, R.W., Halls, A. S., Kirkwood, G. P., McAllister, M., Medley, P., Mees, C. C., Parkes, G. B., Pilling, G. M., Wakeford, R. C., & Welcomme, R. L. (2006). Stock assessment for fishery management. A framework guide to the stock assessment tools of the Fisheries Management Science Programme (FMSP). *FAO Fisheries Technical Papers*, *487*, *261*. Rome, FAO.

Holling, C. S. (1978). *Adaptive environmental assessment and management*. (Editor) London: Wiley.

House of Lords. (2008). Report into systematics & taxonomy, science & technology. 5th report July 2008. http://www.parliament.the-stationery-office. co.uk/pa/ld200708/ldselect/ldsctech/162/16202.htm.

Hoyer, W. D., & Macinnis, D. J. (2010). *Consumer behavior* (5th ed., p. 672). Cengage Learning.

Hughes, A. R., & Stachowicz, J. J. (2004). Genetic diversity enhances the resistance of a seagrass ecosystem to disturbance. *Proceedings of the National Academy of Sciences*, *101*, 8998–9002.

Hukkinen, J. (2003). From groundless universalism to grounded generalism: Improving ecological economic indicators of human-environmental interaction. *Ecological Economics, 44,* 11–27.

Humes, A. G. (1994). How many copepods? *Hydrobiologia, 292/293*, 1–7.

Hurrell, J. W., & Deser, C. (2010). North Atlantic climate variability: The role of the North Atlantic Oscillation. *Journal of Marine Systems*, *79*(3–4), 231–244.

Hutchings, J. A., & Fraser, D. J. (2007). The nature of fisheries- and farming- induced evolution. *Molecular Ecology, 17*(1), 294–313.

Ibañez, F., & Dauvin, J. C. (1988). Long-term changes (1977 to 1987) in a muddy fine sand *Abra alba*—Melinna palmata community from the Western English Channel: Multivariate time-series analysis. *Marine Ecology Progress Series, 49,* 65–81.

ICCAT. (2006). ICCAT report of the 2006 Atlantic Bluefin Tuna Stock Assessment Session.

ICES. (2005). Guidance on the application of the ecosystem approach to management of human activities in the european marine environment. *ICES Cooperative Research Report*, 273.

ICES. (2006). ICES report on Ocean Climate 2005.DKK 70.00. *ICES Cooperative Research Report, 280,* 53.

ICES. (2008). The effect of climate change on the distribution and abundance of marine species in the OSPAR maritime area.2008. DKK1700.00- *ICES Cooperative Research Report, 293,* 49.

ICES-JRC. (2010). *Marine strategy framework directive: Task group 1 report, biological diversity*. Luxembourg: Office for Official Publications of the European Communities.

Iglésias, S. P., Toulhoat, L., & Sellos, D. Y. (2010). Taxonomic confusion and market mislabelling of threatened skates: Important consequences for their conservation status. *Aquatic Conservation: Marine and Freshwater Ecosystems, 20*(3), 319–333.

International convention for the control and management of ships' ballast water and sedi ments. 13 February 2004. http://www.imo.org/Conventions/mainframe.asp?topic_id=867.

Jackson, J. B. (2008). Ecological extinction and evolution in the brave new ocean. *Proceedings of the National Academy of Sciences*, *105*(suppl. 1), 11458–11465.

Jackson, J. B., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. (2001). Historical overfishing and the recent collapse of coastal ecosystems.*Science*, *293*(5530), 629–638.

Janssen, M. A., Holahan, R., Lee, A., & Ostrom, E. (2010). Lab experiments for the study of social-ecological systems. *Science, 328*(30 April), 613–617.

Jaume, D., & Duarte, C. (2006).*The exploration of marine biodiversity* (pp. 17–30). Edit. CSIC Press.

Jennings, M., & Kaiser, M. J. (1998). The effects of fishing on marine ecosystems.*Advances in Marine Biology, 34,* 201–352.

Jennings, S., & Reynolds, J. D. (2000). Impacts of fishing on diversity: From pattern to process. In M. J. Kaiser & S. J. de Groot (Eds.), *Effects of fishing on non-target species and habitats: Biological conservation and socio-economic issues* (pp. 235–250). Oxford: Blackwell Science.

Jennings, S., Kaiser, M. J., & Reynolds, J. D. (2001). *Marine fisheries ecology* (p. 417). Blackwell Science ed.

Jennings, S., Mélin, F., Blanchard, J. L., Forster, R. M., Dulvy, N. K., & Wilson, R. W. (2008). Global-scale predictions of community and ecosystem properties from simple ecological theory.*Proceedings of the Royal Society of London, B275,* 1375–1383.

Jiang, S., Bralower, T. J., Patzkowsky, M. E., Kump, L. R., & Schueth, J. D. (2010). Geographic controls on nanoplankton extinction across the Cretaceous/Palaeogene boundary. *Nature Geoscience, 3,* 280–285.

Johannes, R. E. (1992). *Words of the Lagoon: Fishing and marine lore in the Palau district of Micronesia* (p. 320). Univ. of California Press.

Johnson, C. (2001). Local democracy, democratic decentralisation and rural development: Theories, challenges and options for policy. *Development Policy Review, 19*(4), 521–532.

Johnson, D. (1994). Seastar fight gains momentum. *Australian Fisheries*, *53*, 25–27.

Jones, G. P., Srinivasan, M., & Almany, G. R. (2008). Population connectivity and conservation of marine biodiversity.*Oceanography, 20*(3), 100–111.

Jones, J. C., & Reynolds, J. D. (1997). Effects of pollution on reproductive behaviour of fishes. *Reviews in Fish Biology and Fisheries, 7,* 463–491.

Kalinowski, S. T. (2005). Do polymorphic markers need large sample sizes to estimate genetic distances? *Heredity, 94,* 33–36.

Kappel, E. (Ed.). (2009). Special Issue "The future of ocean biogeochemistry in a high-CO_2 world". *Oceanography, 22*(4), 16–201.

Karlberg, L., Gustafsson, D., & Jansson, P. E. (2006). Modeling carbon turnover in five terrestrial ecosystems in the Boreal zone using multiple criteria of acceptance. *AMBIO, 35,* 448–458.

Kaufman, D., Kraay, A., & Mastruzzi, M. (2009). Governance matters VIII: Aggregate and individual governance indicators 1996–2008. *World Bank Policy Research Working Paper*, 4978, World Bank, Washington, D.C. (p. 103).

Kay, R., & Alder, J. (2005). *Coastal planning and management* (p. 375, 2nd ed.). London: Taylor & Francis.

Keeling, R. F., Körtzinger, A., & Gruber, N. (2010). Ocean deoxygenation in a warming world. *Annual Review of Marine Science, 2,* 199–229.

Keith, D. A., Akçakaya, H. R., Thuiller, W., Midgley, G. F., Pearson, R. G., Phillips, S. J., Regan, H. M., Araújo, M. B., & Rebelo, T. G. (2008). Predicting extinction risks under climate change: coupling stochastic population models with dynamic bioclimatic habitat models. *Biology Letters, 4,* 560–563.

Kelleher, K. (2005). Discards in the world's marine fisheries: An update. *FAO Fisheries Technical Papers*, *470*, 131. Rome, FAO ftp://ftp.fao.org/docrep/fao/008/y5936e/y5936e00.pdf.

Kemp, A. C., Horton, B. P., Donnelly, J. P., Mann, M. E., Vermeer, M., & Rahmstorf, S. (2011). Climate related sea-level variations over the past two millennia. *Proceedings of the National Academy of Sciences*, doi:10.1073/pnas.1015619108.

Kenneth, J. P., & Stott, L. D. (1991). Abrupt deep-sea warming, paleoceanographic changes and benthic extinctions at the end of the Paleocene. *Nature, 353,* 225–229.

Kennish, M. J. (2002). Environmental threats and environmental future of estuaries. *Environmental Conservation, 29*(1), 78–107.

Kier, P. M. (1974). Evolutionary trends and their functional significance in the post-Paleozoic echinoids. *Journal of Paleontology, 48*(Suppl.), 1–96.

Kiker, G. A., Bridges, T. S., Varghese, A., Seager, T. P., & Linkov, I. (2005). Application of multicriteria decision analysis in environmental decision making. *Integrated Environmental Assessment & Management, 1,* 95–108.

Kimmel, D. G., & Hameed, S. (2008). Update on the relationship between the North Atlantic Oscillation and Calanus finmarchicus. *Marine Ecology Progress Series, 366,* 111–117.

Kinzig, A., Perrings, C., & Scholes, B. (2007). *Ecosystem services and the economics of biodiversity conservation* (p. 21). Working paper, School of Life Sciences, Arizona State University.

Klok, C., Hille Ris Lambers, R., de Vries, P., Tarnis, J. E., Wisjman, J. W. M. (2009). Model instruments for marine biodiversity policy. A quick scan; Wagenigen, Statutory Research Task Unit for Nature & Environment, *WOT*-werkdocument (p. 148.73).

Klyashtorin, L. B. (2001). Climate change and long-term fluctuations of commercial catches: The possibility of forecasting. *FAO Fisheries Technical Papers*, p. 410,86, FAO Rome.
Knowlton, N. (1999). The evolution of mutualisms: Exploring the paths between conflict and cooperation. *Trends in Ecology & Evolution, 14,* 49–53.
Knutson, T. R., McBride, J. L., Chan, J., Emanuel, K., Holland, G., Landsea, C., Held, I., Kossin, J. P., Srivastava, A. K., & Sug, M. (2010). Tropical cyclones and climate change. *Nature Geoscience, 3,* 157–163.
Kochszius, M., Seidel, C., Antoniou, A., Kumar-Botla, S., Campo, D., Cariani, A., Garcia-Vazquez, E., Hauschild, J., Hervet, C., Hjörleifsdottir, S., Hreggvidsson, G., Kappel, K., Landi, M., Magoulas, A., Marteinsson, M., Nölte, M., Planes, S., Tinit, F., Turan, C., Venugopal, M. N., Weber, H., & Blohm, D. (2010). Identifying fishes through DNA barcodes and microarrays. *PLoS ONE, 5*(9), e12620, doi:10:1371/journal.pone.0012620.
Korobeinikov, A. (2009). Stability of ecosystem: Global properties of a general predatory-prey model. *Mathematical Medicine & Biology, 26*(4), 309–321.
Korobeinikov, A., & Petrovskii, S. V. (2008). Toward a general theory of ecosystem stability: Plankton-nutrient interaction as a paradigm. In R. J. Hosking & E. Venturino (Eds.), *Aspects of mathematical modelling* (pp. 27–40). Birkhäuser, Basel.
Kozak, K. H., Graham, C. H., & Wiens, J. J. (2008). Integrating GIS-based environmental data into evolutionary biology. *Trends in Ecology & Evolution*, *23*(3), 141–148.
Kunin, W. E., & Lawton, J. H. (1996). Does biodiversity matter? Evaluating the case for conserving species. In K. J. Gaston (Ed.), *Biodiversity. A biology of numbers and difference* (pp. 283–308). Oxford: Blackwell Science.
Lagarde, F., Guillon, M., Dubroca, L., Bonnet, X., Ben Kaddour, K., Slimani, T., & El Mouden, H. E. (2008). Slowness and acceleration: A new method to quantify the activity budget of chelonians. *Animal Behaviour, 75,* 319–329.
Lambdon, P. W., Lloret, F., & Hulme, P. E. (2008). Do non-native species invasions lead to biotic homogeneization at small scales? The similarity and functional diversity of habitats compared for alien and native components of Mediterranean floras. *Diversity and distribution, 14,* 774–785.
Lance, E. M., & Chuenpagdee, R. (2003). *Shifting gears. Addressing the collateral impacts of fishing methods in U.S. waters* (p. 42). Pew science series on conservation and the environment, Island Press Publication Services.
Lavelle, J. W., & Mohn, C. (2010). Motion, commotion and biophysical connections at deep ocean seamounts.*Oceanography, 23*(1), 90–103.
Law, R., & Rowell, C. A. (1993). Cohort structured populations, selection responses, and exploitation of the North Sea cod. In T. K. Stokes, J. M. McGlade, & R. Law (Eds.), *The exploitation of evolving resources* (pp. 155–173). Berlin: Springer.
Law, R. (2000). Fishing, selection, and phenotypic evolution. *ICES Journal of Marine Science*, *57*(3), 659–668.
Lawas, T. P., Tirol, M. S.C., Cardenas, V. R., & Jamias, S. B. (2009). Communication resource mapping for coastal resources management of Barangay Malabrigo, Lobo, Batangas, Philippines. *Journal of environmental science and managemen*t, *12*(2), 38–56.
Lawn, P. (2008). Macroeconomic policy, growth, and biodiversity conservation. *Conservation Biology, 22*(6), 1418–1423.
Lefkaditou, E., Corsini-Foka, M., Kondilatos, G. (2009). Description of the first Lessepsian squid migrant, Sepioteuthis lessoniana (CEPHALOPODA: Loliginidae), from the Aegean Sea (Eastern Mediterranean). *Mediterranean Marine Science*, *10*, 87–97.
Lehodey, P. (2004). Climate and fisheries: An insight from the central pacific ocean. In N. C. Stenseth, G. Ottersen, J. W. Hurrell, & A. Belgrano (Eds.), *Marine ecosystems and climate variation. The North atlantic. A comparative perspective* (p. 252, pp. 137–146). Oxford: Oxford University Press.
Lehtonen, M. (2002). *Les indicateurs d'environnement et de développement durable de l'OCDE: quel rôle dans la mondialisation?* presentation at seminar Axe Mondialisation, Institution et Développement Durable, C3ED, Université de Saint- Quentin-en-Yvelines, 29 March 2002.

Lejeusne, C., Chevaldonne, P., Pergent-Martini, C., Boudouresque, C. F., & Perez, T. (2009). Climate change effect on a miniature ocean: The highly diverse highly impacted Mediterranean Sea. *Trends in Ecology and Evolution, 25*(4), 250–260.

Leppäkoski, E., Gollasch, S., & Olenin, S. (Eds.). (2002). *Invasive aquatic species in Europe. Distribution, impacts and management* (p. 583). London: Kluwer Academic Publishers.

Lessios, H. A. (2008). The Great American Schism: Divergence of marine organisms after the rise of the central American Isthmus. *Annual Review of Ecology, Evolution, and Systematics, 39,* 63–91.

Leterme, S. C., Edwards, M., Seuront, L., Attrill, M. J., Reid, P. C., John, A. W. G. (2005). Decadal basin-scale changes in diatoms, dinoflagellates and phytoplankton colour across the North Atlantic. *Limnology and Oceanography, 50*(4), 1244–1253.

Leterme, S. C., Pingree, R. D., Skogen, M. D., Seuront, L., Reid, P. C., & Attrill, M. J. (2008). Decadal fluctuations in North Atlantic water inflow in the North Sea between 1958–2003: Impacts on temperature and phytoplankton populations. *Oceanologia, 50*(1), 59–72.

Levin, S. A., & Lubchenco, J. (2008). Resilience, robustness, andmarine ecosystem-based management.*Bioscience, 58*(1), 27–32.

Levitus, S., Antonov, J. I., Boyer, T. P., Locarnini, R. A., Garcia, H. E., & Mishonov, A. V. (2009). Global ocean heat content 1955–2008 in light of recently revealed instrumentation problems. *Geophysical Research Letters*, *36*, L07608, doi:10.1029/2008GL037155,2009. http://www.nodc.noaa.gov/OC5/3M_HEAT_CONTENT/.

Levrel, H., & Bouamrane, M. (2008). Instrumentallearning and sustainability indicators: Outputs from co-construction experiments in West African biosphere reserves. *Ecology and Society*, *13*(1), 28. http://www.ecologyandsociety.org/vol13/iss1/art28/.

Levrel, H., Fossat, J., & Pelletier, D. (2010). Les indicateurs de biodiversité marine et côtière: état des lieux institutionnels. *VertigO*, *10*(2), septembre 2010, 1–22.

Levrel, H., Kerbiriou, C., Couvet, D., & Weber, J. (2009). OECD pressure-state-response indicators for managing biodiversity: A realistic perspective for a French biosphere reserve. *Biodiversity and Conservation, 18*(7), 1719–1732.

Levrel, H., Thébaud, O., & Mongruel, R. (2008). *Le Millenium Ecosystem Assessment (MEA): Quelles perspectives pour le domaine marin et côtier?* (p. 10). Document de travail IFREMER, Prospective biodiversité marine.

Libralato, S., Coll, M., Tudela, S., Palomera, I., Pranovi, F. (2008). Novel index for quantification of ecosystem effects of fishing as removal of secondary production. *Marine Ecology Progress Series, 355*, 107–129. doi:10.3354/meps07224.

Licandro, P., Conway, D. V.P., Daly Yahia, M. N., Fernandez de Puelles, M. L., Gasparini, S., Hecq, J. H., Tranter, P., & Kirby, R. R. (2010). A blooming jellyfish in the Northeast Atlantic and Mediterranean. *Biology Letters*, doi:10.1098/rsbl.2010.0150.

Lindley, J. A., & Kirby, R. R. (2010). Climate-induced changes in the North Sea Decapoda over the last 60 years. *Climate Research, 42*(3), 257–264.

Ling, S. D., Johnson, C. R., Frusher, S. D., & Ridgway, K. R. (2009). Overfishing reduces resilience of kelp beds to climate-driven catastrophic phase shift. *Proceedings of the National Academy of Sciences*, *106*, 22341–22345.

Little, C. T.S., & Vrijenhoek, R. C. (2003). Are hydrothermal vent animals living fossils? *Trends in Ecology and Evolution, 18,* 582–588.

Llope, M., Chan, K. S., Ciannelli, L., Reid, P. C., Stige, L. C., & Stenseth, N. C. (2009). Effects of environmental conditions on the seasonal distribution of phyto-plankton biomass in the North Sea. *Limnology and Oceanography, 54*(2), 512–524.

Loh, J., Green, R. E., Ricketts, T., Lamoreux, J., Jenkins, M., Kapos, V., & Randers, J. (2005). The living planet index: Using species population time series to track trends in biodiversity. *Philosophical Transactions of the Royal Society, B. 360,* 289–295.

Mace, P. M. (2001). A new role for MSY in single-species and ecosystem approaches to fisheries stock assessment and management. *Fish & Fisheries, 2,* 2–32.

Mace, P. M. (2004). In defense of fisheries scientists, single-species models and others scapegoats: Confronting the real problems. *Marine Ecology Progress Series, 274,* 285–291.

Machida, R., Hashiguchi, Y., Nishida, M., & Nishida, S. (2009). Zooplankton diversity analysis through single-gene sequencing of a community sample. *BMC Genomics 2009, 10,* 438, doi:10.1186/1471-2164-10-438.

Maggs, C. A., Castilho, R., Foltz, D., Henzler, C., Jolly, T., Kelly, J., Olsen, J., Perez, K. E., Stam, W., Vainola, R., Viard, F., & Wares, J. (2008). Evaluating signatures of glacial refugia for North Atlantic marine organisms. *Ecology, 89,* S108–S122.

Magurran, A. E. (1988). *Ecological diversity and its measurement* (p. 179). London: Croom Helm.

Marasco, R. J., Goodman, D., Grimes, C. B., Lawson, P. W., Punt, A. E., & Quinn, T. J. (2007). Ecosystem-based fisheries management: Some practical suggestions. *Canadian Journal of Fisheries and Aquatic Sciences, 64,* 928–939.

Martel, C., Viard, F., Bourguet, D., & Garcia-Meunier, P. (2004). Invasion by the marine gastropod Ocinebrellus inornatus in France. I. Scenario for the source of introduction. *Journal of Experimental Marine Biology and Ecology, 305*(2), 155–170.

Mavruk, S., & Avsar, D. (2008). Non-native fishes in the Mediterranean from the Red Sea, by way of the Suez Canal. *Reviews in Fish Biology and Fisheries, 18,* 251–262.

May, R. M. (1994). Biological diversity: Differences between land and sea. *Philosophical Transactions of the Royal Society, B. 343,* 105–111.

McNeil, B.I, & Matear, R. J. (2008). Southern ocean acidification: A tipping point at 450-ppm atmospheric CO_2. *Proceedings of the National Academy of Sciences, 105*(48), 18860–18864.

Medlin, L. K. (2007). If everything is everywhere, do they share a common gene pool?*Gene, 405,* 189–183.

Medlin, L., & Zingone, A. (2007). A taxonomic review of the genus Phaeocystis.*Biogeochemistry, 83,* 3–18.

Melzner, F., Gutowska, M. A., Langenbuch, M., Dupont, S., Lucassen, M., Thorndyke, M. C., Bleich, M., & Pörtner, H. O. (2009). Physiological basis for high CO_2 tolerance in marine ectothermic animals: Pre-adaptation through lifestyle and ontogeny? *Biogeosciences, 6,* 2313–2331.

Mieszkowska, N., Genner, M. J., Hawkins, S. J., & Sims, D. W. (2009). Effects of climate change and commercial fishing on Atlantic cod *Gadus morhua*. *Advances in Marine Biology, 56,* 213–273.

Milazzo, M. (1998). Subsidies in world fisheries—a re-examination. *World Bank Technical Paper*, *406*, 86. Washington, D.C. International Bank forReconstruction & Development/World Bank.

Millar, A. J. (2002). The world's first recorded extinction of a seaweed. In *Proceedings of the 17th International Seaweed Symposium*, Cape Town, 28 January–2 February 2001.

Millennium Ecosystem Assessment. (2005). Millennium ecosystem assessment: Synthesis report. Washington, D.C.: Island Press. http://www.millenniumassessment.org/.

Mills, L. C., Soule, M. E., & Doak, D. F. (1993). The keystone-species concept in ecology and conservation. *Bioscience, 43*(4), 219–226.

Miossec, L., Le Deuff, R. M., & Goulletquer, P. (2009). Alien species alert: *Crassostrea gigas* (Pacific oyster). *ICES Cooperative Research Report, 299,* 42.

Miranda, L. S., Collins, A. G., & Marques, A. G. (2010). Molecules clarify a cnidarian life cycle—The "Hydrozoan" Microhydrula limopsicola is an early life stage of the staurozoan Haliclystus antarcticus. *PLoS ONE, 5*(4), e10182, doi:10.1371/journal. pone.0010182.

Moalic, Y., Arnaud-Haond, S., Perrin, C., Perrin, G. A., Pearson, G. A., & Serrao, E. A. (2011). Travelling in time with networks: Revealing present day hybridization versus ancestral polymorphism between two species of brown algae. *Fucus vesiculosus* and *F. spiralis*. *BMC Evolutionary Biology, 11,* 33.

Morato, T., & Pauly, D. (Eds.). (2004). Seamounts: Biodiversity and fisheries. *Fisheries Centre Research Reports, 12*(5), 78.

Morato, T., Hoyle, S. D., Allain, V., & Nicol, S. J. (2010). Seamounts are hotspots of pelagic biodiversity in the open ocean. *Proceedings of the National Academy of Sciences*, *107*(21), 9707–9711. www. pnas.org/cgi/doi/10.1073/pnas.0910290107.

Morgan, L. E., & Chuenpagdee, R. (2003). *Shifting gears: Addressing the collateral impacts of fishing methods in U.S. waters. Pew science series on conservation and the environment* (42 pp.). Washington, DC: Island Press.

Mullon, C., Freon, P., Cury, P., Shannon, L., & Roy, C. (2009). A minimal model of the variability of marine ecosystems. *Fish and Fisheries, 10*(2), 115–131.

Mullon, C., Mittaine, J. F., Thébaud, O., Peron, G., Merino, G., & Barange, M. (2009). Modeling the global fishmeal and fish oil markets. *Natural Resource Modeling, 22*(4), 564–609.

Munns, W. R. Jr., Gervais, J., Hoffman, A. A., HommenU., Nacci, D. E., Nakamaru, M., Sibly, R., Topping, C. (2007). Chapter 9: Modeling approaches to population-level ecological risk assessment. In W. Lawrence, W. Barnthouse, R. Munns, Jr., & M. T. Sorensen (Eds.), *Population-level ecological risk assessment* (pp. 179–210). USA: CRC/SETAC press, USA.

Myers, R. A., & Worm, B. (2003). Rapid worldwide depletion of predatory fish communities. *Nature, 423,* 280–283.

Myers, R. A., Baum, J. K., Shepherd, T. D., Powers, S. P., & Peterson, C. H. (2007). Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science, 315*(5820), 1846–1850.

Mylonas, C. C., de la Gándara, F., Corriero, A., & Belmonte Ríos, A. (2010). Atlantic Bluefin tuna (*Thunnus Thynnus*) farming and fattening in the Mediterranean Sea. *Reviews in Fisheries Science, 18*(3), 266–280.

Naeem, S. (2006). Expanding scales in biodiversity-based research: défis and solutions for marine systems. *Marine Ecology Progress Series, 311*, 273–283.

National Ocean Economics Program. (2008). Environmental & Recreational (Non-Market) values—Research Methodologies. http://noep.mbari.org/nonmarket/methodologies.asp.

National Research Council (NRC). (1993). *Managing wastewater in coastal urban Areas*. Committee on Wastewater Management for Coastal Urban Areas National Academy Press. National Research Council, Washington, D.C.102.

National Research Council (NRC). (2010). *Ocean acidification: A national strategy to meet the challenges of a changing ocean*. National Research Council, Washington, D.C., p. 175.

National Resources Defense Council. (2001). Clean water and oceans: Oceans: In brief: News: Beach closings and advisories soar in 2000, increased monitoring and reporting reveals widespread pollution at U.S. beaches. National Resources Defense Council, Washington, D.C.

Nelson, W. G. (1993). Beach restoration in the southeastern U.S.: environmental effects and biological monitoring. *Ocean Coastal Management*, *19*, 157–182.

NOAA Coastal Services Center and National Marine Protected Area Center. (2005). Social Science for Marine Protected Areas.

Nyström, M., & Folke, C. (2001). Spatial resilience of coral reefs. *Ecosystems*, *4*, 406–417.

O'Dor, R., Miloslavich, P., & Yarincik, K. (2010). Marine biodiversity and biogeography—regional comparisons of global issues, an introduction. *PLoS ONE*, *5*(8), e11871.

O'Hara, S., & Stagl, S. (2001). Global food markets and their local alternatives: A socio-ecological economic perspective. *Population & Environment*, *22*(6), 533–554.

Olenin, S., Alemany, F., Gollasch, S., Goulletquer, P., Lehtiniemi, M., McCollin, T., Minchin, D., Occhipinti Ambrogi, A., Ojaveer, H., Jensen, K. R., Wallentinus, I., Aleksandrov, B., Cardoso, A. C., Miossec, L., & Stankiewicz, M. (2009). *Good Environmental Status (GES) Descriptor: "Non-indigenous species introduced by human activities are at levels that do not adversely alter the ecosystems"* (p. 49). EU Commission—Joint Research Center.

Olsen, E. M., & Moland, E. (2011). Fitness landscape of Atlantic cod shaped by harvest selection and natural selection. *Evolutionary Ecology*, *25*(3), 695–710.

Organisation for Economic Co-operation and Development (OECD). (1979). *Inter futures: Facing the future* (p. 438).

Organisation for Economic Co-operation and Development (OECD). (1993). OECD Core set of indicators for environmental performance reviews: A synthesis report by the group on the state of the environment. *Environment Monographs*, *83*, Paris.

Organisation for Economic Co-operation and Development (OECD). (1994). *Environmental indicators: OECD Core Set*. Paris: OECD.

Organisation for Economic Co-operation and Development (OECD),). (1997). *OECD Environmental Performance Reviews —A practical introduction*. OCDE/GD(97)35, Paris

Organisation for Economic Co-operation and Development (OECD). (2003). *Harnessing markets for biodiversity: Towards conservation and sustainable use*.

OrganisationforEconomic Co-operation and Development (OECD)-DAC. (2003). *Synthesis study on supporting decentralisation and local governance: lessons learned, good practices and emerging issues*. DAC Network on Development Evaluation, DCD/DAC/EV (2003) 3/REV1, OECD, Paris, 20 October (p. 73).

OSPAR (2010). Quality status report 2010. OSPAR Commission (p. 176). London. http://qsr2010. ospar. org/fr/cgh03_01.html.

Ostrom, E. (1989). *Governing the commons*. Cambridge: Cambridge University Press.

Ottersen, G., Planque, B., Belgrano, A., Post, E., Reid, P. C., & Stenseth, N. C. (2001). Ecological effects of the North Atlantic Oscillation. *Oecologia*, *128*, 1–14.

Páez-Osuna, F., Guerrero-Galván, S. R., & Ruiz-Fernández, A. C. (1998). The environmental impact of shrimp aquaculture and the coastal pollution in Mexico. *Marine Pollution Bulletin*, *36*(1), 65–75.

Palumbi, S. R. (2001). Humans as the world's greatest evolutionary force. *Science, 293*, 1786–1790.

Palumbi, S. R., McLeod, K. L., & Grunbaum, D. (2008). Ecosystems in action: Lessons from marine ecology about recovery, resistance, and reversibility. *Bioscience*, *58*, 33–42.

Palumbi, S. R., Sandifer P, A., Allan, J. D., Beck, M. W., Fautin, D. G., Fogarty, M. J., Halpern, B. S., Incze, L. S., Leong, J., Norse, E., Stachowicz J, J., & Wall, D. H. (2009). Managing for ocean biodiversity to sustain marine ecosystem services. *Frontiers in Ecology and the Environment*, *7*(4), 204–211.

Parmesan, C., & Yohe, G. (2003). A globally coherent fingerprint of climate change impacts across natural systems. *Nature*, *421*, 37–42.

Parry, G., Cohen, B., McArthur, M., & Hickman, N. (2000). Victorian Incursions Report Number 2. Asterias amurensis incursions in Port Phillip Bay: Status at May 1999. Marine and Freshwater ResearchInstitute, Australia, internal report no 19, p. 21.

Pauly, D., Christensen, V., Dalsgaard, J., Froese, R., & Torres, F. (1998). Fishing down marine food webs. *Science*, *279*, 860–863.

Pauly, D., Graham, W., Libralato, S., Morissette, L., & Deng Palomares, M. L. (2009). Jellyfish in ecosystems, online databases and ecosystem models. *Hydrobiologia*, *616*(1), 67–85.

Pelejero, C., Calvo, E., & Hoegh-Guldberg, O. (2010). Paleo-perspectives on ocean acidification. *Trends in Ecology & Evolution*, *25*(6), 332–344.

Pereira, H. M., Leadley, P. W., Proença, V., Alkemade, R., Scharlemann, J. P.W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W.L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., & Walpole, M. (2010). Scenarios for global biodiversity in the 21st century. *Science*, *330*, 1496–1501.

Perrings, C., Williamson, M., Barbier, E. B., Delfino, D., Dalmazzone, S., Shogren, J., Simmons, P., & Watkinson, A. (2002). Biological invasion risks and the public good: An economic perspective. *Conservation Ecology*, *6*(1), 1. http://www.consecol.org/ vol6/iss1/art1/.

Petitgas, P., Alheit, J., Peck, M. A., Raab, K., Irigoien, X., Huret, M., van der Kooij, J., Pohlmann, T., Wagner, C., Zarraonaindia, I., & Dickey-Collas, M. (2012). Anchovy population expansion in the North Sea. *Marine Ecology Progress Series*, *444*, 1–13.

Pinkerton, E. (Ed.). (1989). *Co-operative management of local fisheries. New directions for improved management and community development* (p. 229). Vancouver: University of British Columbia Press.

Plagányi, E. E. (2007). Models for an ecosystem approach to fisheries. *FAO Fisheries Technical Papers*, *477*, 108. Rome, FAO.

Pogson, G. H., Taggart, C. T., Mesa, K. A., & Boutilier, R. G. (2001). Isolation by distance in the Atlantic cod, Gadus morhua, at large and small geographic scales. *Evolution*, *55*, 131–146.

Polovina, J. J., Howell, E. A., & Abecassis, M. (2008). Ocean's least productive waters are expanding. *Geophysical Research Letters*, *35*, L03618. http://www.melanie- abecassis.eu/recherche.html.

Poore, G., & Wilson, G. (1993). Marine species richness. *Nature*, *361*, 597–598.

Por, F. D. (1978). *The influx of Red Sea biota into the Mediterranean by way of the Suez Canal*, *23*, 228. Berlin: Springer.

Poteete, A. R., Janssen, M. A., & Ostrom, E. (2010). *Working together: Collective action, the commons and multiple Methods in Practice* (p. 370). Princeton: Princeton University Press.

Purvis, A., & Hector, A. (2000). Getting the measure of biodiversity. *Nature*, *405*, 212–219.

Queffelec, B., Cummins, V., & Bailly, D. (2009). Integrated management of marine biodiversity in Europe: Perspectives from ICZM and the evolving EU Maritime Policy framework. *Marine Policy*, *33*, 871–877.

Raven, P. H. (2002). Science, sustainability and the human prospect. *Science*, *297*, 954–958.

Ravier, C., & Fromentin, J. M. (2004). Are the long-term fluctuations in Atlantic bluefin tuna (Thunnus thynnus) population related to environmental changes? *Fisheries Oceanography*, *13*(3), 145–160.

Ray, G. C. (1991). Coastal-zone biodiversity patterns. *BioScience, 41*, 490–498.

Raynolds, L. T. (2004). The globalization of organic agro-food networks. *World Development*, *32*(5), 725–743.

Rees, S. E., Rodwell, L. D., Attrill, M. J., Austen, M. C., & Mangi, S. C. (2010). The value of marine biodiversity to the leisure and recreation industry and its application to marine spatial planning. *Marine Policy*, *34*(5), 868–875.

Reid, P. C., Edwards, M., Hunt, H. G., & Warner, A. J. (1998). Phytoplankton change in the North Atlantic. *Nature*, *391*, 546.

Reinfelder, J. R. (2011). Carbon concentrating mechanisms in eukaryotic marine phytoplankton. *Annual Review of Marine Science*, *3*, 291–315.

Reusch, T. B., Ehlers, A., Hammerli, A., & Worm, B. (2005). Ecosystem recovery after climatic extremes enhanced by genotypic diversity. *Proceedings of the National Academy of Sciences*, *102*(8), 2826–2831.

Reynolds, J. D., Dulvy, N. K., Goodwin, N. B., & Hutchings, J. A. (2005). Biology of extinction risk in marine fishes. *Proceedings of the Royal Society of London, B.272*, 2337–2344.

Ribot, J. C. (2002). *Democratic decentralization of natural resources: Institutionalizing popular participation*. Washington: World Resources Institute.

Richards, C. L., Carstens, B. C., & Knowles, L. L. (2007). Distribution modeling and statistical phylogeography: An integrative framework for generating and testing alternative biogeographical hypotheses. *Journal of Biogeography*, *34*(11), 1833–1845.

Richardson, A. J., Bakun, A., Hays, G. C., & Gibbons, M. J. (2009). The jellyfish joyride: Causes, consequences and management responses to a more gelatinous future. *Trends in Ecology & Evolution*, *24*, 312–322.

Ricklefs, R. E. (2004). A comprehensive framework for global patterns in biodiversity. *Ecology Letters*, *7*, 1–15.

Riebesell, U., Körtzinger, A., & Oschlies, A. (2009). Sensitivities of marine carbon fluxes to ocean changes. *Proceedings of the National Academy of Sciences*, *106*(49), 20602–20609.

Rijnsdorp, A. D., Peck, M. A., Engelhard, G. H., Möllmann, C., & Pinnegar, J. K. (Eds). (2010). Resolving climate impacts on fish stocks. *ICES Cooperative Research Report*, *301*, 371.

Rolshausen, G., Segelbacher, G., Hobson, K. A., & Schaefer, H. M. (2009). Contemporary evolution of reproductive isolation and phenotypic divergence in sympatry along a migratory divide. *Current Biology*, *19*(24), 2097–2101.

Rooker, J. R., Secor, D. H., DeMetrio, G., Schloesser, R., Block, B. A., & Neilson, J. D. (2008). Natal homing and connectivity in Atlantic Bluefin tuna populations. *Science*, 322, 742–744.

Rosa, R., & Seibel, B. A. (2008). Synergistic effects of climate-related variables suggest future physiological impairment in a top oceanic predator. *Proceedings of the National Academy of Sciences*, *105*(52), 20776–20780.

Rosenzweig, C., Karoly, D., Vicarelli, M., Neofotis, P., Wu, Q., Casassa, G., Menzel, A., Root, T. L., Estrella, N., Seguin, B., Tryjanowski, P., Liu, C., Rawlins, S., & Imeson, A. (2008). Attributing physical and biological impacts to anthropogenic climate change. *Nature*, *453*, 353–357.

Rost, B., Zondervan, I., & Wolf-Gladrow, D. (2008). Sensitivity of phytoplankton to future changes in ocean carbonate chemistry: Current knowledge, contradictions and research directions. *Marine Ecology Progress Series*, *373*, 227–237.

Rozenfeld, A. F., Arnaud-Haond, S., Hernandez-Garcia, E., Eguiluz, V. M., Serrao, E. A., & Duarte, C. M. (2008). Network analysis identifies weak and strong links in a metapopulation system. *Proceedings of the National Academy of Sciences*, *105*, 18824–18829.

Ruddle, K. (1998). The context of policy design for existing community-based fisheries management in the Pacific Islands.*Ocean and Coastal Management*, *40*, 105–126.

Ruzzante, D. E., Wroblewski, J. S., Taggart, C. T., Smedbol, S. K., Cook, D., & Goddarde, V. S. (2000). Bay-scale population structure in coastal Atlantic cod in Labrador and Newfoundland Canada. *Journal of Fish Biology*, *56*, 408–431.

Sabine, C. L., Feely, R. A., Gruber, N., Key, R. M., Lee, K., Bullister, J. L., Wanninkhof, R., Wong, C. S., Wallace, D. W.R., Tilbrook, B., Millero, F. J., Peng, T. H., Kozyr, A., Ono, T., & Rios, A. F. (2004). The oceanic sink for anthropogenic CO_2. *Science*, *305*, 367–371.

Sabine, C. L., & Tanhua, T. (2010). Estimation of anthropogenic CO_2 inventories in the Ocean. *Annual Review of Marine Science*, *2*, 175–198.

Sala, E., & Knowlton, N. (2006). Global marine biodiversity trends.*Annual Review of Environment and Resources*, *31*, 93–122.

Sanchirico, J. N., & Wilen, J. E. (2002). The impacts of marine reserves on limited-entry Fisheries.*Natural Resource Modeling*, *15*(3), 291–310.

Santelli, C. M., Orcutt, B. N., Banning, E., Bach, W., Moyer, C. L., Sogin, M. L., Staudigel, H., & Edwards, K. J. (2008). Abundance and diversity of microbial life in ocean crust. *Nature, 453*, 653–657.

Sarrazin, J., Robigou, V., Juniper, S. K., Delaney, J. R. (1997). Biological and geological dynamics over four years on a high-temperature sulfide structure at the Juan De Fuca Ridge hydrothermal observatory. *Marine Ecology Progress Series*, *153*, 5–24.

Savolainen, V., Cowan, R. S., Vogler, A. P., Roderick, G. K., & Lane, R. (2005). Towards writing the encyclopaedia of life: An introduction to DNA barcoding. *Philosophical Transactions of the Royal Society, B.360*, 1805–1811.

Schlüter, M. H., Merico, A., Reginatto, M., Boersma, M., Wiltshire, K. H., & Greve, W. (2010). Phenological shifts of three interacting zooplankton groups in relation to climate change. *Global Change Biology*, *16*(11), 3144–3153.

Schrope, M. (2006). The real sea change. *Nature, 443*(12), 622–624.

Scotese, C. R. (1997). *Continental drift* (7th edn, p. 79). PALEOMPA project, Arlington.

Screen, J. A., & Simmonds, I. (2010). The central role of diminishing sea ice in recent Arctic temperature amplification. *Nature*, *464*, 1334–1337.

Selkoe, K. A., Henzler, C., & Gaines, S. D. (2008). Seascape genetics and the spatial ecology of marine populations. *Fish and Fisheries*, *9*, 363–377.

Shank, T. M. (2010). Seamounts— deep-ocean laboratories of faunal connectivity, evolution and endemism. *Oceanography*, *23*(1), 108–122.

Shank, T. M., Fornari, D. J., Von Damm, K. L., Lilley, M. D., Haymon, R. M., & Lutz, R. A. (1998). Temporal and spatial patterns of biological community development at nascent deep-sea hydrothermal vents (9°50'N, East Pacific Rise). *Deep Sea research II*, *45*, 465–515.

Smith, K. L., Baldwin, R. J., Ruhl, H. A., Kahru, M., Mitchell, B. G., & Kaufmann, R. S. (2006). Climate effect on food supply to depths greater than 4,000 meters in the northeast Pacific. *Limnology and Oceanography, 51*, 166–176.

Smith, C. R., De Leo, F. C., Bernardino, A. F., Sweetman, A. K., Martinez Arbizu, P. (2008). Abyssal food limitation, ecosystem structure and climate change. *Trends in Ecology & Evolution*, *23*, 518–528.

Smith, M. D., Roheim, C. A., Crowder, L. B., Halpern, B. S., Turnipseed, M., Anderson, J. L., Asche, F., Bourillón, L., Guttormsen, A. G., Khan, A., Liguori, L. A., McNevin, A., O'Connor, M. I., Squires, D., Tyedmers, P., Brownstein, C., Carden, K., Klinger, D. H., Sagarin, Selkoe, K. A. (2010). Sustainability and Global Seafood.*Science*, *327*, 784–786.

Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K. B., Tignor, M., & Miller, H. L. (eds.). (2007). *Climate Change 2007: The physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change* (Cambridge University Press), Cambridge, United Kingdom and New York, NY, USA.

Solow, R. (1974). Intergenerational equity and exhaustible resources. *Review of Economic Studies*, *41*, 29–45.

Spicer, R. A., & Parrish, J. T. (1986). Paleobotanical evidence for cool North Polar climates in the mid-Cretaceous (Albian-Cenomanian). *Geology*, *14*, 703–706.

Stachowicz, J. J., & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, *314*(5800), 787–790.

Stachowicz, J. J., Fried, H., Osman, R. W., & Whitlatch, R. B. (2002). Biodiversity, invasion resistance, and marine ecosystem function: Reconciling pattern and process. *Ecology, 83*, 2575–2590

Steinacher, M., Joos, F., Frolicher, T. L., Plattner, G. K., & Doney, S. C. (2009). Imminent ocean acidification in the Arctic projected with the NCAR global coupled carbon cycle-climate model. *Biogeosciences, 6*, 515–533.

Steinacher, M., Joos, F., Frölicher, T. L., Bopp, L., Cadule, P., Cocco, V., Doney, S. C., Gehlen, M., Lindsay, K., Moore, J. K., Schneider, B., & Segschneider, J. (2010). Projected 21st century decrease in marine productivity: A multi-model analysis. *Biogeosciences*, *7*(3), 979–1005.

Steinmetz, F., Thébaud, O., Blanchard, F., Le Floch, P., & Bihel, J. (2008). A bio-economic analysis of long term changes in the production of French fishing fleets operating in the Bay of Biscay. *Aquatic Living Resources*, *21*(3), 317–327.

Stige, L. C., Ottersen, G., Brander, K., Chan, K. S., & Stenseth, N. C. (2006). Cod and climate: Effect of the North Atlantic Oscillation on recruitment in the North Atlantic. *Marine Ecology Progress Series*, *325*, 227–241.

Streftaris, N., Zenetos, A., & Papathanassiou, E. (2005). Globalisation in marine ecosystems: the story of non-indigenous marine species across European seas. *Oceanography and marine biology: An annual review*, *43*, 419–453.

Strong, D. R., & Frank, K. T. (2010). Human involvement in food webs. *Annual Review of Environment and Resources*, *35*, 1–23.

Sumaila, U. R., & Charles, A. (2002). Economic models ofmarine protected areas: An introduction. *Natural Resource Modeling*, *15*(3), 261–272.

Sumaila, U. R., & Armstrong, C. W. (2006). Distributional and efficiency effects of marine protected areas: a study of the Northeast Atlantic cod fishery. *Land Economics*, *82*(3), 321–332.

Tallis, H., Levin, P. S., Ruckelshaus, M., Lester, S. E., McLeod, K. L., Fluharty, D. L., & Halpern, B. S. (2010). The many faces of ecosystem-based management: making the process work today in real places. *Marine Policy*, *34*, 340–348.

Talmage, S. C., & Gobler, C. J. (2010). Effects of past, present, andfuture ocean carbon dioxide concentrations on the growth and survival of larval shellfish. *Proceedings of the National Academy of Sciences*, *107*(40), 17246–17251.

Tasker, M. L. (Ed.). (2008). The effect of climate change on the distribution and abundance of marine species in the OSPAR Maritime Area.*ICES Cooperative Research Report*, *293*, 45. http://www.ices.dk/ products/cooperative.asp.

TEEB. (2009). The economics of ecosystems and biodiversity for national and international policy makers—summary: Responding to the value of nature. http://www.teebweb.org.

Teyssèdre, A. (Coord.). (2005a). *Biodiversité et changements globaux: enjeux de société et défis pour la recherche*. ADPF, 2005, (pp. 50–68). Biodiversité, Science et Gouvernance: Conférence Mondiale, Paris (FRA), 2005/01/24.

Teyssèdre, A. (2005b). Vers une sixième grande crise d'extinctions? In R. Barbault B. & Chevassus-au- Louis (Eds.), *Biodiversité et changements globaux. Enjeux de société et défis pour la recherche*, édition Adpf, (pp. 24–36).

Thackeray, S. J., Sparks, T. H., Frederiksen, M., Burthe, S., Bacon, P. J., Bell, J. R., Botham, M. S., Brereton, T. M., Bright, P. W., Carvalho, L., Clutton-Brock, T., Dawson, A., Edwards, M., Elliott, J. M., Harrington, R., Johns, D., Jones, I. D., Jones, J. T., Leech, D. I., Roy, D. B., Scott, W. A., Smith, M., Smithers, R. J., Winfield, I. J., & Wanless, S. (2010). Trophic level asynchrony in rates of phenological change for marine, freshwater and terrestrial environments. *Global Change Biology*, *16*(12), 3304–3313.

The Royal Society. (2005).Ocean acidification due to increasing atmospheric carbon dioxyde. Policy document 12/05, p. 58. http://royalsociety.org/document.asp?tip=0 & id=3249.

Thébaud, O., & Soulié J. C. (2008). *Fishing through fish communities: A simple bio-economic model*. In Proceedings of the International Congress on Modelling and Simulation, Christchurch, New Zealand, December 2007. International Society for Computer Simulation (CD).

Thébaud, O., Boncoeur, J., & Berthou, P. (2007). *Regulating access to marine fisheries in the coastal zone: key issues*. AMURE working Pap. Ser.D22-2007. http://www.univ-brest.fr/gdr-amure/docu- ments/gdr-amure-D-22-2007.pdf.

Thompson, P. M., Sultana, P., Islam, M. N., Kabir, M. M., Hossain, M. M., & Kabir, M. S. (1999). An assessment of co-management arrangements developed by the Community Based Fisheries Management Project in Bangladesh. Paper presented at the international workshop on fisheries co-management, 23–28 August 1999, Penang, Malaysia.

Thorrold, S. R., Latkoczy, C., Swart, P. K., Jones, C. M. (2001). Natal homing in a marine fish metapopulation.*Science*, doi:10.1126/science.291.5502.297.

Thrush, S. F., Hewitt, J. E., Dayton, P. K., Coco, G., Lohrer, A. M., Norkko, A., Norkko, J., & Chiantore, M. (2009). Forecasting the limits of resilience: Integrating empirical research with theory. *Proceedings of the Royal Society of London, B.276*, 3209–3217.

Ting, M., Kushnir, Y., Seager, R., & Li, C. (2009). Forced and internal twentieth-century SST trends in the North Atlantic, *Journal of Climate*, *22*(6), 1469–1481.

Toly, N. J. (2004). Globalization and the capitalization of nature: A political ecology of biodiversity in Mesoamerica. *Bulletin of science Technology and Society*, *24*(1), 47–54.

Troadec, J. P., Boncoeur, J., & Boucher, J. (2003). Le constat. In L. Laubier (Ed.), *Exploitation et surexploitation des ressources marines vivantes* (pp. 15–56). Académie des Sciences, RST n°17, Editions Lavoisier, Paris.

Turley, C., Eby, M., Ridgwell, A. J., Schmidt, D. N., Findlay, H. S., Brownlee, C., Riebesell, U., Fabry, V. J., Feely, R. A., & Gattuso, J. P. (2010). The societal challenge of ocean acidification.*Marine Pollution Bulletin*, *60*(6), 787–792.

Tyrrell, T., & Merico, A. (2004). Emiliania huxleyi: bloom observations and the conditions that induce them. In H. R. Thierstein & J. R. Young *Coccolithophores: From molecular processes to global impact* (pp. 75–97). Berlin: Springer.

UNEP. (2009). Coastal areas pollution: the role of cities, United Nation Environment Programme. www.unep. org/urban_environment/PDFs/Coastal_Pollution.

UNEP—CDB. (2003). *Report of expert meeting on indicators of biological diversity including indicators for rapid assessment of inland water ecosystems* (p. 82). Montreal, 10–14 November 2003.

UNPD—United Nations Population Division. (2011). World Population Prospects: The 2010 Revision, United Nations, New York; http://esa.un.org/unpd/wpp/ index.htm.

Venter, J. C., Remington, K., Heidelberg, J. F., Halpern, A.L, Rusch, D., Eisen, J.A, Wu, D., Paulsen, I., Nelson, K. E., Nelson, W., Fouts, D. E., Levy, S., Knap, A.H., Lomas, M. W., Nealson, K., White, O., Peterson, J., Hoffman, J., Parsons, R., Baden-Tillson, H., Pfannkoch, C., Rogers, Y. H., & Smith, H. O. (2004). Environmental genome shotgun sequencing of the Sargasso Sea. *Science*, *304*, 66–74.

Vézina, A. F., & Hoegh-Guldberg, O. (coord.). (2008). Theme section "Effects of ocean acidification on marine ecosystems". *Marine Ecology Progress Series*, *373*, 199–309.

Walli, A., Teo, S. L.H., Boustany, A., Farwell, C. J., Williams, T., Dewar, H., Prince, E., & Block, B. A. (2009). Seasonal movements, aggregations and diving behavior of Atlantic bluefin tuna (*Thunnus thynnus*) revealed with archival tags. *PLoS ONE*, *4*, e6151.

Walters, C. J. (1986). *Adaptive management of renewable resources* (p. 374). New York: MacMillan Pub. Co.

Walther, G. R., Roques, A., Hulme, P., Sykes, M. T., Pysek, P., Kühn, I., Zobel, M., Bacher, S., Botta-Dukat, Z., Bugmann, H., Czucz, B., Dauber, J., Hickler, T., Jarosik, V., Kenis, M., Klotz, S., Minchin, D., Moora, M., Nentwig, W., Ott, J., Panov, V. E., Reineking, B., Robinet, C., Semenchenko, V., Solarz, W., Thuiller, W., Vila, M., & Settele, J. (2009). Alien species in a warmer world: risks and opportunities. *Trends in Ecology and Evolution*, *24*(12), 686–693.

Wang, M., & Overland, J. E. (2009). A sea ice free summer Arctic within 30 years? *Geophysical Research Letters*, *36*, L07502. http://www.agu.org/pubs/crossref/2009/2009GL037820.shtml.
Waples, R. S. (2002). Evaluating the effect of stage- specific survivorship on the N-e/N ratio.*Molecular Ecology*, *11*, 1029–1037.
Ward, R. D., Woodwark, M., & Skibinski, O. F. (1994). A comparison of genetic diversity levels in marine, freshwater and anadromous fishes. *Journal of Fish Biology*, *44*, 213–232.
Wares, J. P., & Cunningham, C. W. (2001). Phylogeography and historical ecology of the North Atlantic intertidal. *Evolution*, *55*, 2455–2469.
Warwick, R. M., & Clarke, K. R. (1998).Taxonomic distinctness and environmental assessment.*Journal of Applied Ecology*, *35*, 532–544.
Warwick, R. M., & Clarke, K. R. (2001). Practical measures of marine biodiversity based on relatedness of species. *Oceanography and Marine Biology: An Annual Review*, *39*, 207–231.
Weber, J. (1996). Conservation, développement et coordination: peut-on gérer biologiquement le social? Colloque panafricain "Gestion communautaire des ressources naturelles renouvelables et développement durable". Harare, 24–27 juin.
Weber, J., & Bailly, D. (1993). Prévoir c'est gouverner. *Nature, Sciences, Sociétés*, n°1, janvier.
Weber, J., & Reveret, J. P. (1993). la gestion des relations sociétés-natures: modes d'appropriation et processus de décision. *Le Monde Diplomatique*, coll. Savoirs, 2, Environnement & développe ment, octobre.
Weidema, I. (2000). *Introduced species to the Nordic countries* (p. 244). Nord 2000:13. Nordic Council of Ministers. Copenhagen.
Wessel, P., Sandwell, D. T., & Kim, S. S. (2010). The global seamount census.*Oceanography*, *23*(1), 24–33.
Whithnam, T. G., DiFazio, S., Schweitzer, J. A., Shuster, S. M., Allan, G. J., Bailey, J. K., & Woolbright, S. A. (2008). Extending genomics to natural communities and ecosystems. *Science*, *25*, 492–495.
Whitmarsh, D., & Palmieri, M. G. (2011). Consumer behaviour and environmental preferences: A case study of Scottish salmon aquaculture. *Aquaculture Research*, *42*, 142–147.
Wilenius, M. (2005). Towards the age of corporate responsibility? Emerging challenges for the business world. *Futures*, *37*, 133–150.
Williamson, P. (2010). OCEANS 2025 NERC. www.oceans2025.org.
Wolfslehner, B., & Vacik, H. (2008). Evaluating sustainable forest management strategies with the Analytic Network Process in a Pressure-State-Response Framework. *Journal of Environmental Management*, *88*(1), 1–10.
World Bank. (2002a). World development indicators 2002. World Bank, Washington, D.C.
World Bank. (2002b). *World development report 2003: Sustainable development in a dynamic world: Transforming institutions, growth, and quality of life*. New York: Oxford University Press.
World Bank & FAO. (2008). The Sunken Billions. The economic justification for fisheries reform, Agriculture & Rural Dept., The World Bank, Washington D.C., Advance web edition, xvii + p. 67.
World Resources Institute, UNEP, United Nations Development Programme & the World Bank. (1996). World Resources 1996–97: The Urban Environment.
Worm, B., & Duffy, J. E. (2003). Biodiversity, productivity, and stability in real food webs. *Trends in Ecology and Evolution*, *18*, 628–632.
Worm, B., Barbier, E. B., Beaumont, N., Duffy, J., Folke, C., Halpern, B. S., Jackson, J. B., Lotze, H. K., Micheli, F., Palumbi, S. R., Sala, E., Selkoe, K. A., Stachowicz, J. J., & Watson, R. (2006). Impacts of biodiversity loss on ocean ecosystem services. *Science*, *314*(5800), 787–790.
Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., Fogarty, M. J., Fulton, E. A., Hutchings, J. A., Jennings, S., Jensen, O. P., Lotze, H. K., Mace, P. M., McClanahan, T. R., Minto, C., Palumbi, S. R., Parma, A. M., Ricard, D., Rosenberg, A. A., Watson, R., & Zeller, D. (2009). Rebuilding global fisheries. *Science*, *325*, 578–585.
Yamin, F. (1995). Biodiversity, ethics, and international law. *International Affairs*, *71*(3), 529–546.
Zimmerer, K. (2006). Conclusion: Rethinking the compatibility, consequences, and geographic strategies of conservation and development. In K. Zimmerer (Ed.), *Globalization and new geographies of conservation* (pp. 315–346). Chicago: University of Chicago Press.