

Marine Historical Ecology

Informing the Future by Learning from the Past

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We all learn about history in school. Knowing about our past broadens our view, helps us understand past actions and their consequences, and can prevent us from repeating past mistakes. As George Santayana (1905) said, “Those who cannot remember the past are condemned to repeat it.” Usually we learn about the history of people on this planet: how we evolved, how we spread around the world, and how we fought wars and conquered one another. History is generally about our culture and societal changes, but rarely about our effects on the environment. And if it is about human effects on the environment, then it is commonly focused on the terrestrial environment. For example, J. Donald Hughes’s excellent book *An Environmental History of the World* (2001) deals almost exclusively with human effects on the terrestrial world, such as deforestation, soil salinization, and the loss of large game. What about the history of human interactions with and influences on the ocean?

Humans are exerting many pressures on marine ecosystems today that can be readily observed. Just think about the overfishing, pollution, nutrient loading, habitat alterations, invasive species, and climate change that are occurring in most parts of the global ocean to varying degrees (see Chapters 19 and 20). Yet the ocean, people’s interactions with it, and its living creatures have undergone a long history of change that is often much less visible. Over the past hundreds, thousands, and millions of years, the ocean and life within it have naturally evolved and changed. People throughout history have harvested marine resources for food, fuel, clothing, medicine, and ornaments, transformed coastal habitats, influenced water quality, and transplanted marine species. Over time, such changes have altered the abundance of many marine populations and the structure and functioning of marine ecosystems (**Figure 8.1**). Therefore, to fully understand the current state of marine ecosystems, we need to know their long-term history. Moreover, to properly manage and protect today’s oceans and to make informed decisions about their future, we need to understand the causes and consequences of past changes. In this way, marine historical ecology is similar to climate change research, for which we need long-term records of past temperatures or CO₂ concentrations to make inferences about current states and potential future trends.

There are three major reasons why we need to know about the ocean’s past:

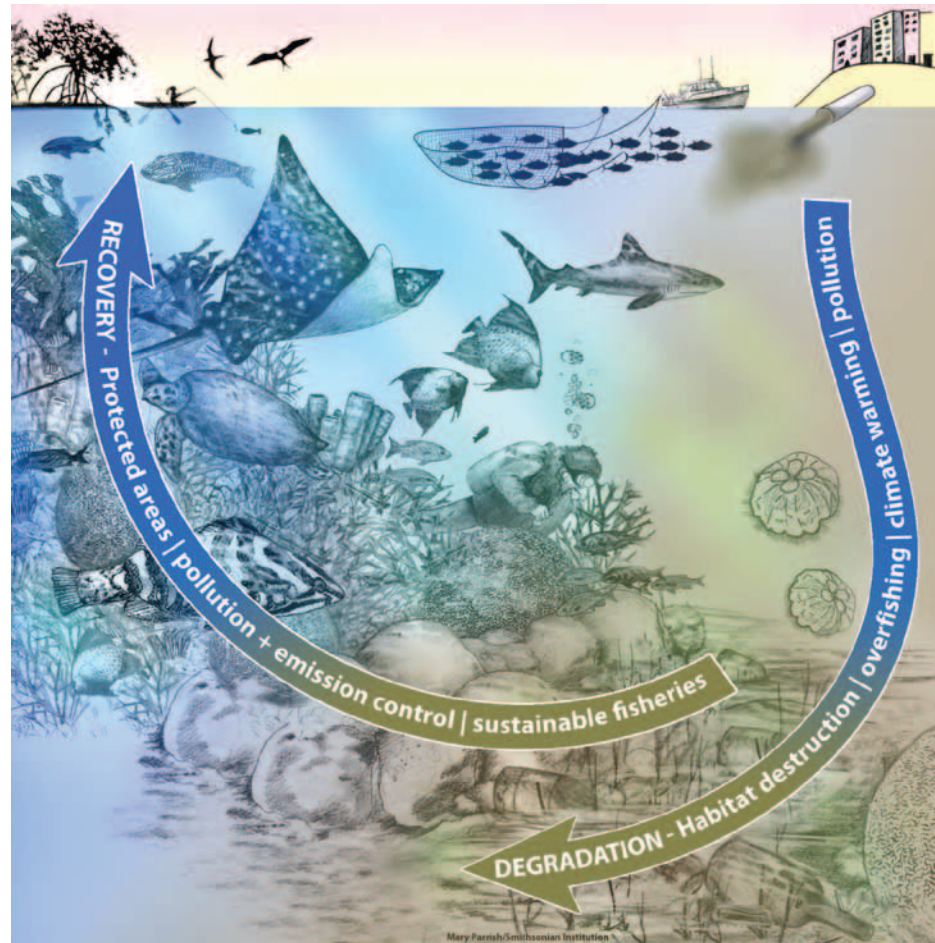
1. To determine historical reference points and long-term trajectories of change, we need to understand the past abundance and distribution of marine species and the structure of ocean ecosystems.
2. To judge the current state of marine ecosystems, we need to understand the magnitude and range of changes that have already occurred.
3. To better inform the future, we need to understand the drivers and consequences of past changes.

Figure 8.1 What was natural? Illustration of the degradation of coral reef ecosystems from past (left) to modern (right) conditions due to human impacts including the effects of overexploitation, habitat destruction both on land and in the sea, nutrient loading, and water pollution. (Based on Pandolfi et al. 2003, 2005. Image courtesy of the Smithsonian Institution.)

If we don't understand the history of human effects on the ocean, we are condemned to repeat it, but how can we know the ocean's past? Fortunately, many past human activities have left traces in paleontological, archaeological, and historical records, whereas others can be reconstructed using modern scientific methods such as genetic analyses, hindcasting, or modeling. This chapter introduces you to the new multidisciplinary research field of marine historical ecology, which aims to reconstruct past ecosystems and the history of change in the ocean. First, we describe the origin and development of marine historical ecology. Next we outline the different disciplines, data records, and analytic methods used to reconstruct the past. We then present a range of major results and insights gained from historical case studies and syntheses. Finally, we discuss the application of this historical knowledge in science, management, and conservation as well as in teaching and communication. Overall, it is our hope that you will learn about the important and fascinating insights one can gain from studying the ocean's past, appreciate the long-term history of human-induced changes in marine ecosystems, and use that knowledge to contribute to a better future for the blue planet.

Origin and Development of Marine Historical Ecology

The new research field of marine historical ecology has its roots in both the natural sciences and the humanities. Researchers from such different disciplines as ecology, fisheries science, history, archaeology, and paleontology now work together and inspire one another to unravel the ocean's past. In the late twentieth century, several researchers began to think about the longer-term history of the sea (Pauly 1995; Jackson 1997; Carlton 1998; Dayton et al. 1998). With the increasing signs of human impacts, it



became clear that the ecosystems we are studying today are not at all “natural” anymore, but heavily altered by human activities. For example, paleoecologist Jeremy Jackson (1997) demonstrated in his paper “Reefs since Columbus” that Caribbean coastal ecosystems were severely degraded by about 1800, long before ecologists began to study them, and had suffered severe losses in large vertebrates such as green turtles, hawksbill turtles, manatees, and the now-extinct Caribbean monk seal. He remarked that studying grazing and predation on corals reefs today is like trying to understand the ecology of the Serengeti by studying the termites and the locusts while ignoring the elephants and the wildebeests. Two years earlier, fishery scientist Daniel Pauly (1995) hypothesized in his paper “Anecdotes and the Shifting Baseline Syndrome of Fisheries” that historical amnesia has contributed to a shift in our perception of what is natural toward more and more degraded ecosystems. Thus things that we perceive as natural today, such as the abundance or size of fish, may be drastically different from what our parents and grandparents thought of as natural (Figure 8.2).

In 1999 Jeremy Jackson brought together ecologists, historians, archaeologists, and paleontologists in a multi-year working group called “Long-Term Ecological Records

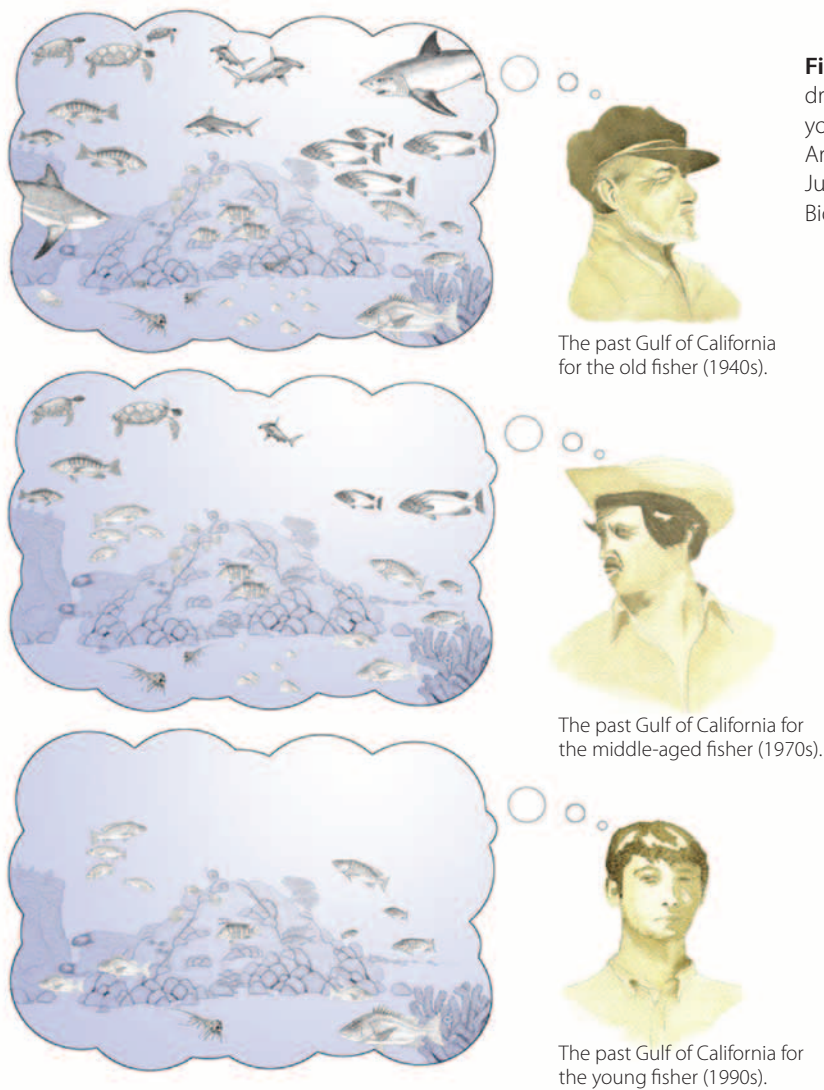


Figure 8.2 Illustration of the shifting baseline syndrome and the change in living memory from old to young fishermen in the Gulf of California. (Based on Sáenz-Arroyo et al. 2005. Courtesy of Anne Randall, Pier Thiret and Juan Jesus Lucero, 2005, with permission of Comunidad y Biodiversidad, A.C.)

Currently, the field of marine historical ecology continues to grow as more and more scientists incorporate a historical perspective into their research projects to place current results in a larger temporal context. Parallel developments have occurred in the field of marine environmental history as more historians have brought in scientific perspectives and included the changing oceans in their analysis of social changes in coastal communities. Many conferences host special symposia on historical ecology or environmental history. Several scientific journals publish special issues highlighting historical perspectives, and an increasing number of scholarly and popular books concerning historical ecology (e.g., Roberts 2007; Rick and Erlandson 2008; Starkey et al. 2008; Jackson et al. 2011; Bolster 2012) are reaching a growing audience in academia and the general public. And slowly but surely, marine historical ecology is making its way into textbooks, such as this one.

of Marine Environments, Populations and Communities” at the National Center for Ecological Analysis and Synthesis (NCEAS) in California. In 2001 their first seminal paper, “Historical Overfishing and the Recent Collapse of Coastal Ecosystems,” was published in *Science* and made news around the world. Because of the wealth of interest and research possibilities, the group expanded and split into three subgroups publishing on the long-term history of coral reefs (Pandolfi et al. 2003), kelp forests (Steneck et al. 2002), and estuaries and coastal seas (Lotze et al. 2006).

At about the same time, historian Poul Holm, together with several other historians, ecologists, and archaeologists, founded the History of Marine Animal Populations (HMAP) program under the Census of Marine Life (Holm et al. 2010). Since 2000, HMAP has grown into a network of more than 100 scientists around the world, working on research projects on specific regions or taxa and building a new discipline by teaching summer schools, training graduate students, publishing scientific articles and books, organizing the biannual Ocean’s Past conference, and engaging in outreach.

How We Can Learn about the Past

There are many different ways to gain insight into the ocean’s past. This section outlines the different research disciplines involved in studying the past and highlights the kinds of information that can be extracted, the methods that can be used, and how the available data can be analyzed. Paleontologists, for example, analyze fossils and sediment cores to derive tracers of environmental change; archaeologists excavate animal remains and human artifacts at former settlement sites; and historians investigate early illustrations or narrative descriptions of what people saw, caught, and sold. Each discipline can gain insight into a particular time period (**Figure 8.3**). For example, historical data usually span the past few centuries, since they consist of written sources, whereas archaeological data can reach back several thousand years, depending on the length and density of former human occupation. Each discipline has its strengths and weaknesses (**Table 8.1**; Lotze et al. 2011c), as there are different biases in the availability, preservation, collection, and analysis of data. However, they all

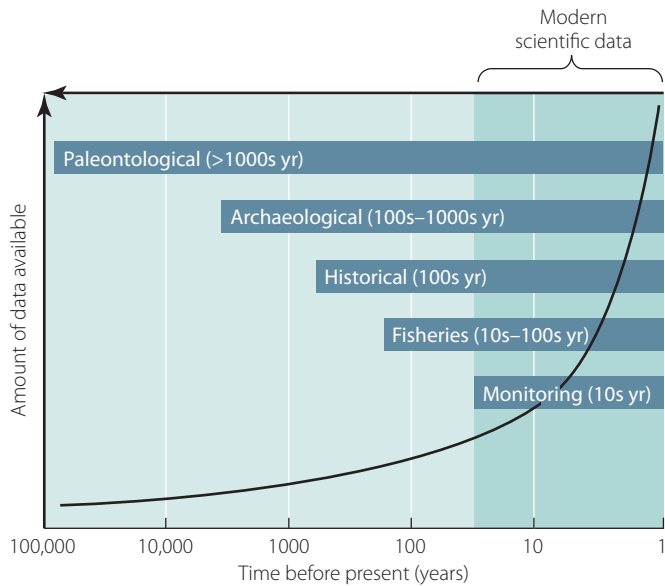


Figure 8.3 The availability of data has exponentially increased over time (solid line); however, modern scientific data usually cover only the last 20–50 years (dark green bar), yet data from different disciplines can be used to reconstruct the past and expand the timeline of information. (After Lotze and Worm 2009.)

provide valuable data about the past, which complement one another and can be pieced together to reconstruct a long-term history of ecological changes. We highlight here how results from different disciplines can be analyzed and integrated to develop a comprehensive understanding of the past that stretches from deep time to the present (**Table 8.2**). As we will see, results from different disciplines or analytic approaches can corroborate one another, thereby making the conclusions drawn more reliable, but they can

also contradict one another and thus open up more research questions. We start with disciplines analyzing the deep past and then move toward modern times.

Paleontological data

Paleontologists generally analyze data that span the deep or geologic past over thousands and millions of years. Their primary data source is the sedimentary record, from which they extract fossils, fish scales, shells, plant seeds and pollen in distinct layers across time, as well as biochemical tracers of organisms that are not fossilized, and in some cases even ancient DNA. Moreover, minerals, trace elements, and isotopes can be used as proxies to reconstruct past changes in climate and productivity. Such sedimentary records can be gathered either from sediment cores drilled into the seafloor or on land where former seafloor sediments have been exposed at some point in the past due to geologic uplift. Because layers of sediment are deposited over distinct periods, the age of each layer can be determined by various methods such as radiometric dating, the first and last occurrences of fossils, and climate records based on stable isotopes or pollen. Most sediments are mixed to some extent by burrowing organisms, but some annually laminated deep-sea sediments in anoxic basins have been shown to preserve bimonthly environmental records that span many thousands of years (Haug et al. 2003). Another source of data for paleontologists is the skeletal growth bands of long-lived organisms such as corals and molluscs. Because they grow incrementally every year, they show growth layers just like tree rings. In addition, the isotopic or trace elements in the layers of a coral core can be analyzed to determine the environmental conditions when each layer was formed (Swart and Grottoli 2003).

The relative abundance of isotopes and minerals in different sedimentary or coral layers can be used to re-

TABLE 8.1 Comparisons of data sources used by the different disciplines involved in marine historical ecology

DATA SOURCE	DATA TYPE ^a	BIAS ^b	TIME SCALE ^c	RESOLUTION ^d	SPATIAL SCALE ^e	EXAMPLE IN THIS CHAPTER
Paleontological data	O, R	B, E, M	Short–long	High–low	Small	Figure 8.4
Archaeological data	O, R	B, E, H, M	Short–long	Medium–high	Medium	Figure 8.5
Historical data	O, R, A	H, M	Short–long	High	Medium–large	Figures 8.6, 8.7
Fisheries/hunting records	O, R, A	H, M	Short–medium	High	Medium	Figures 8.8, 8.9
Scientific surveys	O, R, A	H, M	Short	High	Medium	Figure 8.10
Living memory	O, R	H, M	Short	Medium–high	Medium	Figure 8.11
Genetic data	O, R, A	B, M	Long	Low	Large	Figure 8.12

Source: Lotze et al. 2011c.

^aData type: O = occurrence (presence, absence), R = relative abundance, A = absolute abundance

^bBiases: B = biological (depending on species, e.g., selective preservation), E = environmental (depending on environment, e.g., selective preservation), H = human (depending on activities, e.g., selective use / interest), M = methodology (depending on analytic method)

^cTime scale: long (>1000 years), medium (100–1000 years), short (<100 years)

^dResolution: high (yearly or less), medium (decadal to century), low (millennial)

^eSpatial scale: large (>1000 km), medium (1–1000 km), small (<1 km)

TABLE 8.2 Different approaches to combining or comparing data to reconstruct the past

APPROACH	DESCRIPTION	EXAMPLES
Temporal comparison	Point estimates of past and present species abundance, distribution, or size can be compared as “Then” versus “Now.” Such comparisons provide valuable insight into past changes, yet cannot decipher temporal variation or whether the past estimate represents a true baseline without considering the historical context.	Figures 8.6, 8.7B, 8.10
Time-series analysis	Time series of absolute or relative abundance, distribution, or size can indicate trends and fluctuations over time, which can be analyzed statistically, along with records of putative drivers such as fishing or climate. Time series can be combined into longer or more robust series or compared meta-analytically in search of general patterns. The length (tens, hundreds, or thousands of years) and historical context of the series need to be considered when making inferences about baselines.	Figures 8.4, 8.5, 8.7A, 8.8, 8.11, 8.14, 8.15
Hindcasting	If estimates of present species abundance, historical catch data, and information on life history, such as recruitment, growth rate, or natural mortality, are available, we can back-calculate former abundance using simple population models. Related approaches include the calculation of virgin biomass or carrying capacity. Past abundance estimates can also be calculated based on historical habitat availability or ecosystem configuration. Abundance–body mass relationships (size spectra) have also been used to estimate the potential abundance of marine animal populations under unexploited conditions.	Figures 8.9, 8.12, 8.17B
Space-for-time comparisons	Unexploited regions in the ocean should reflect former abundance, size, and species composition in exploited regions, assuming that other conditions are similar. Surveys across spatial gradients of exploitation can therefore provide insight into how exploitation changes population abundance and ecosystem structure through time.	Friedlander and DeMartini 2002; Sandin et al. 2008

Source: Lotze and Worm 2009.

construct past climatic and environmental variation. For example, the oxygen isotope ^{18}O is a common proxy for determining past temperatures and reconstructing paleoclimates (Figure 8.4A; Haug and Tiedemann 1998). Likewise, strontium and titanium can be used as proxies for paleosalinity and the amount of precipitation and river runoff over thousands of years (Haug et al. 2003). The concentration of ^{15}N isotopes in lake sediment cores from Kodiak Island in Alaska has been used to reconstruct the abundance of sockeye salmon (*Oncorhynchus nerka*) over the past 2200 years (Finney et al. 2002). Because sockeye salmon feed in the marine environment and die after spawning in lakes, they transport marine-derived ^{15}N isotopes upstream, where they are deposited in lake sediments. The results showed a multi-century decline in sockeye salmon from 100 BC to AD 800 and consistently higher abundances from AD 1200 to AD 1900. These shifts were linked to changes in climate and productivity in the Pacific Ocean and provide a baseline for natural long-term variation against which recent human-induced changes can be compared.

When fossils or other remains of species are found in the sedimentary record, they provide important information about when and how rapidly species first appeared or went extinct and how relative abundances of species and higher taxa changed over time. The sedimentary record thus represents a rich archive that can be used to analyze past population and community changes. Some fossil records show large fluctuations in population abundance, such as that of the Caribbean planktonic foraminiferan

Globorotalia miocenica. Between 2.8 and 2.3 million years ago (see Figure 8.4A), it was more abundant during glacial periods, repeatedly dipped to low population abundances during interglacial periods, and ultimately became extinct globally during one of these episodes (Norris 1999). Using such data from the fossil record for a wide range of species, background rates of extinction have been estimated at 0.1–1.0 marine species per thousand species per millennium over evolutionary time scales, whereas the current rate of species extinctions is thought to be about a thousand times higher (Millennium Ecosystem Assessment 2005). Past mass extinctions were most likely linked to sudden changes in climatic or environmental conditions, whereas most marine extirpations and extinctions in the nineteenth and twentieth centuries were caused by exploitation and habitat loss, with lesser effects of pollution, species invasions, disease, or climate change (Harnik et al. 2012). Thus the fossil record can provide an important baseline for placing recent changes in perspective.

In addition to natural changes, the sedimentary record can be used to analyze human-induced changes in marine environments over past centuries and millennia. For example, the concentrations of terrestrial pollen in each sediment layer can be used to determine changes in sediment loading due to land clearing (Cooper and Brush 1993). Eutrophication can be measured by analyzing the ratio of centric to pennate diatoms in sediment cores. Centric diatoms are generally planktonic and more prevalent in nutrient-rich, eutrophic waters, whereas pennate diatoms inhabit the seafloor and prefer more oligotrophic

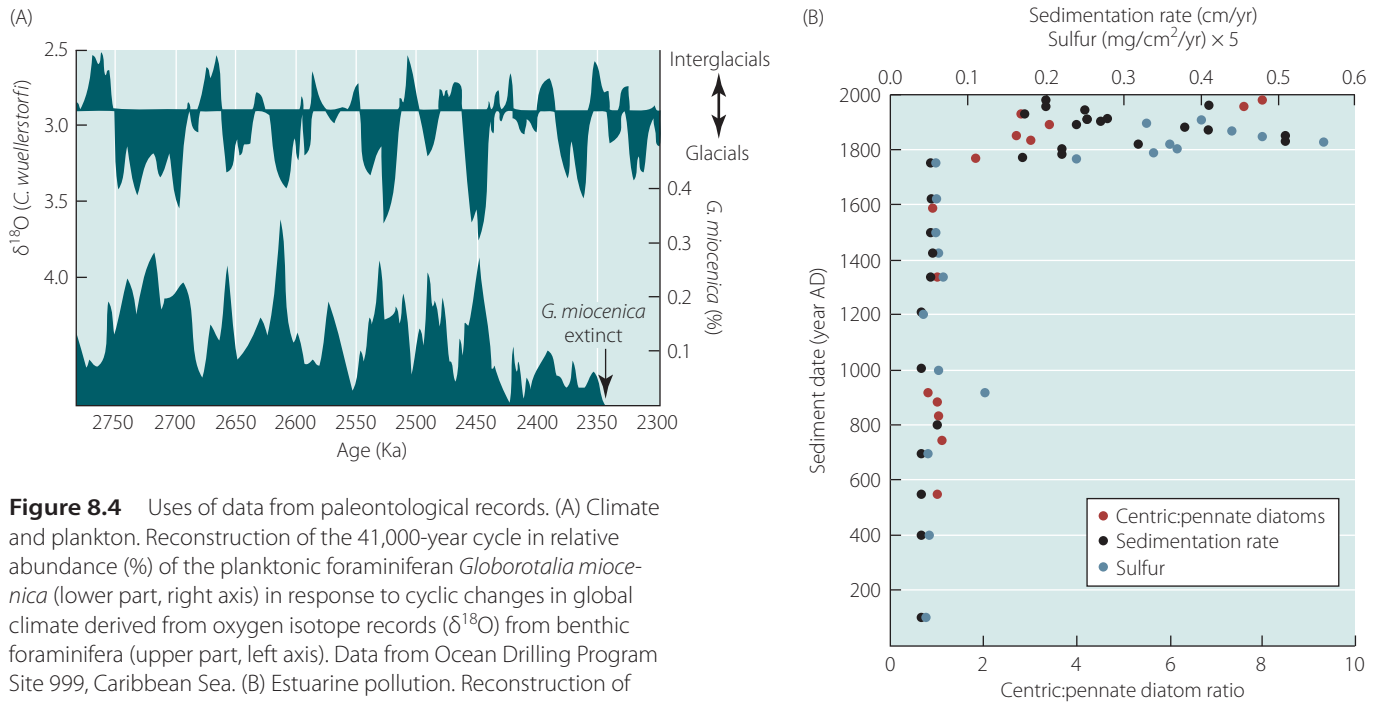


Figure 8.4 Uses of data from paleontological records. (A) Climate and plankton. Reconstruction of the 41,000-year cycle in relative abundance (%) of the planktonic foraminifer *Globorotalia miocenica* (lower part, right axis) in response to cyclic changes in global climate derived from oxygen isotope records ($\delta^{18}\text{O}$) from benthic foraminifera (upper part, left axis). Data from Ocean Drilling Program Site 999, Caribbean Sea. (B) Estuarine pollution. Reconstruction of anthropogenic effects on water quality in Chesapeake Bay over the past 2000 years. Data from sediment cores indicate strong increases in the sedimentation rate (black), the ratio of centric to pennate diatoms as a proxy for eutrophication (red), and the content of sulfur as a proxy for anoxia (blue) after European settlement in the seventeenth and eighteenth centuries. (A after Norris 1999 and Haug and Tiedemann 1998; B, data from Cooper and Brush 1993 for Core R4-50; figure after Lotze et al. 2011c).

conditions (Cooper and Brush 1993). The total amount of organic carbon and nutrients in sediment cores provides a further indication of eutrophication, and the amounts of sulfur and pyrite serve as proxies for hypoxia and anoxia. Using all these indicators, Cooper and Brush (1993) showed that water quality in the Chesapeake Bay deteriorated quickly in the seventeenth and eighteenth centuries after permanent European settlement (**Figure 8.4B**). Similarly, McCulloch and colleagues (2003) used the ratio of barium to calcium in coral cores from the inner Great Barrier Reef as a proxy for sediment fluxes from river runoff to show that sedimentation strongly increased after permanent European settlement and land clearing.

All these paleontological data can provide quantifiable evidence of environmental and ecological change over very long time scales, which can be invaluable for placing current changes in context so that we may better distinguish the natural, long-term variability of ecosystems from more recent anthropogenic effects. There are some limitations, however (see Table 8.1). First, the spatial scale of drilled cores can be very small, and the temporal resolution of paleontological data varies among regions and environments. While deep-water microfossil deposits can provide high-resolution records of past environmental change on a global scale, they tend to include only a small

number of species. In comparison, shallower-water macrofossil assemblages are much coarser in temporal resolution, but can be more diverse and include many commercially important groups. Second, the fossil and subfossil record generally consists of species that have hard parts, such as shells and bones that are preserved over time; as a result, some groups have higher preservation probabilities than others. Furthermore, the quality of preservation depends on the environment of deposition and the extent of subsequent alteration (Valentine et al. 2006). Thus there is a clear bias in what kinds of species can be analyzed. Third, although the fossil record can provide essential data on the presence or absence of species, information about relative, and particularly absolute, abundances is more limited (Kidwell and Flessa 1995). Only where variations in sedimentation rates can be precisely determined and fossil preservation is relatively unchanging, as in many deep-sea sediments and lake environments, is it possible to determine the flux of organisms to the bottom and create highly accurate determinations of microfossils' relative and absolute abundances (Sexton and Norris 2008).

Archaeological data

Archaeologists excavate and analyze plant, animal, and human remains found in caves, graves, or buried in the sediment of former occupation sites. Plant remains, including pollen and seeds, provide information on wild or cultivated plant use as well as the natural vegetation in the area. Animal remains, including shells, bones, scales, teeth, and hairs, reflect the species that were used for food, medicine, ornaments, or other purposes. They can also reveal information on the size, age, relative abundance,

and spatial and seasonal distribution of the animals used. Cultural artifacts such as fishhooks or spears indicate the kinds of tools and technology that were used for hunting and fishing. In many so-called coastal shell middens, such remains were deposited year after year over several hundreds or thousands of years, sometimes by more than one culture. Therefore, archaeologists can uncover different layers of past occupation, which can be dated by various methods such as radiocarbon dating. Such stratified shell middens are great archives of past information about human technology, cultural practices, resource use, and environmental conditions.

One such archaeological study comes from the Channel Islands of southern California. In Daisy Cave, Erlandson and colleagues (2004) found the remains of more than 150 species of marine and terrestrial animals from over the past 11,000 years. Some of these species, which were used by former inhabitants as resources, are extinct today, and the relative abundances of others have changed over time. As human populations grew and technology improved, fishing increased in importance, and about 1500 years ago, fisheries expanded into deeper waters, targeting larger offshore species (Rick et al. 2008). Similar trends toward an increase

in marine fisheries and a spatial expansion farther offshore as well as in deeper waters have been documented in archaeological records from other parts of the world, such as southern Taiwan (Kuang-Ti 2001), Great Britain (Barrett et al. 2004), the Wadden Sea (Lotze 2007), and around the North Atlantic (Perdikaris and McGovern 2008).

Over the past decade, more and more archaeological studies have revealed telltale signs of local resource depletion that coincided with increasing human populations (Rick and Erlandson 2008). For example, in the Emeryville shell mound in San Francisco Bay, the highly valued white sturgeon (*Acipenser transmontanus*) and several goose species declined in relative abundance between 2600 and 700 years ago (Figure 8.5A) and were replaced by smaller, less valued species (Broughton 1997, 2002). Moreover, the average size and age of sturgeon, as indicated by their dentary width, also declined over time. Similarly, shell middens in Puerto Marqués, Mexico revealed the decline and subsequent disappearance of sea turtles (Cheloniidae) between 5500 and 2300 years ago (Figure 8.5B; Kennett et al. 2008). And in New Zealand, bone remains of fur seals (*Arctocephalus forsteri*) from 800 to 200 years ago suggest that Maori hunters extirpated fur seals from the North Is-

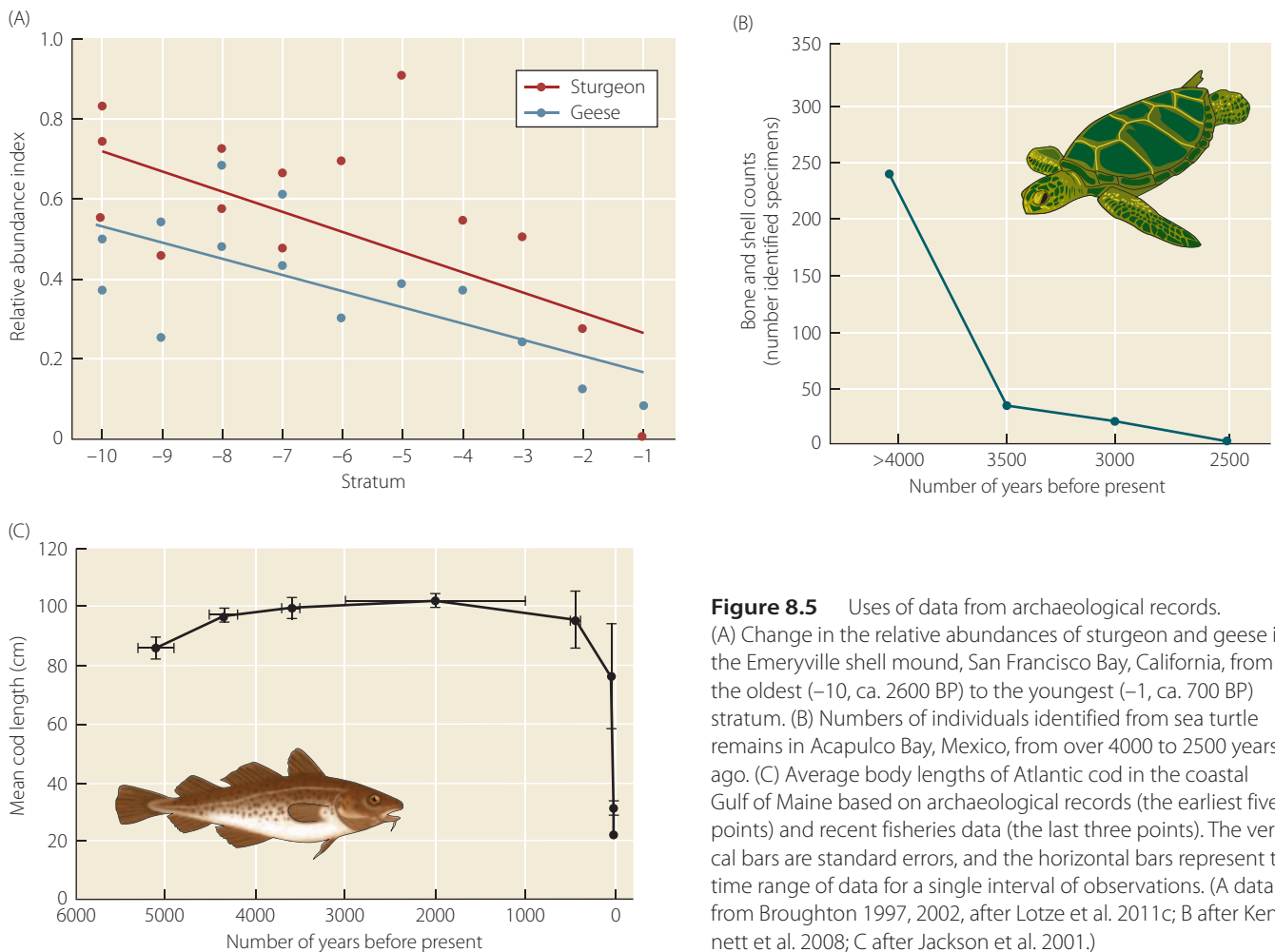


Figure 8.5 Uses of data from archaeological records. (A) Change in the relative abundances of sturgeon and geese in the Emeryville shell mound, San Francisco Bay, California, from the oldest (-10, ca. 2600 BP) to the youngest (-1, ca. 700 BP) stratum. (B) Numbers of individuals identified from sea turtle remains in Acapulco Bay, Mexico, from over 4000 to 2500 years ago. (C) Average body lengths of Atlantic cod in the coastal Gulf of Maine based on archaeological records (the earliest five points) and recent fisheries data (the last three points). The vertical bars are standard errors, and the horizontal bars represent the time range of data for a single interval of observations. (A data from Broughton 1997, 2002, after Lotze et al. 2011c; B after Kennett et al. 2008; C after Jackson et al. 2001.)

land by AD 1500, and then also from large parts of the South Island, resulting in a contraction of their range to less than 10% of its former extent by 1790 (Smith 2005). Historical exploitation by the European sealing industry further reduced the range and numbers of fur seals until protection was implemented in 1873. In contrast, in the Gulf of Maine, the average length of Atlantic cod (*Gadus morhua*) used by indigenous people remained at about 1 meter in length for 4000 years until intensive fishing by Europeans (Figure 8.5C; Jackson et al. 2001). Yet a detailed study in Penobscot Bay revealed a decrease of apex predators (mainly cod and swordfish) and an increase of mesopredators (flounder, sculpin) over the past 4350–400 years (Bourque et al. 2008).

From these and other records it has become clear that modern humans have heavily utilized and influenced coastal environments for much longer than previously thought (Rick and Erlandson 2008). Archaeological evidence of marine fishing, harvesting, and hunting has been dated back more than 100,000 years to the last interglacial period. Although prehistoric hunters and fishermen had simple tools and relatively small populations, an increasing number of studies show that their exploitation had substantial effects on local marine animal abundance, distribution, and size long before the onset of commercial and industrial exploitation. On the other hand, not all archaeological records indicate resource depletion, and not all fluctuations are related to humans. On the Aleutian Islands, for example, remains of seabirds showed long-term climate-driven population fluctuations that were superimposed on local overhunting of accessible breeding colonies (Causey et al. 2005). Similarly, in the North Sea, several fish species showed climate-related fluctuations over the past 9000 years, which mirror recently observed shifts related to warming (Enghoff et al. 2007).

Like paleontological records, archaeological data have advantages and disadvantages. These data provide some of the longest consistent time series describing human exploitation of coastal ecosystems, extending over hundreds to thousands of years, with a resolution of decades to centuries (see Table 8.1). The available records generally represent the local environment at small to medium spatial scales. Shell middens may represent one of the best data sources about coastal ecosystems over the last 15,000 years, since rising postglacial sea levels have drowned, damaged, or destroyed much of the paleontological record for such ecosystems except in areas of active tectonic uplift (Pandolfi 1999). The faunal record they contain, however, is biased toward species that have had some resource value to humans and toward those species whose remains have been well preserved, which depends on soil and other geomorphic characteristics. Variation in the excavation and analytic techniques applied can also influence the quality and quantity of data. Moreover, burrowing animals, plowing by farmers, and other physical disturbances can cause a mixing of the stratigraphic layers, thereby lim-

iting the temporal resolution of the record. These factors limit our ability to accurately estimate a species' former abundance, yet excavated animal and plant remains still provide highly valuable records of their occurrence, size, age, and relative importance (see Table 8.1).

Historical data

Historians use a wide range of surviving written accounts, such as historical reports by early naturalists, travelers, or government officials, logbooks from fishing and whaling vessels, trade or customs accounts, and even newspapers, cookbooks, and restaurant menus, which can be found in libraries, archives, and museum collections. Other important sources of information are illustrated materials, including maps, drawings, paintings, photographs, and older artwork such as cave paintings, wall carvings, and monuments made of whale bones and other animal remains. As well, more organized government records can provide important information on catches and trade, on taxes paid or the amount of salt used to cure the caught fish, on the gear and technology used, and on regulations that were implemented. Such records became more detailed and sophisticated over time, eventually leading to modern fisheries and hunting statistics (Smith 1994; Lotze 2007). Historians have typically used these sorts of records to reconstruct and trace the lives and activities of people in historical times, but they also can be used to infer changes in marine species and ecosystems (Al-Abdulrazzak et al. 2012).

Historical records can provide insight into the former occurrence, distribution, and abundance of a species as well as on the influences human activities had on species used for food, fuel, fashion, or other purposes. For example, McClenachan and colleagues (2006) used narratives and descriptions of former seafarers, pirates, and naturalists together with archaeological data to reconstruct the former distribution of major and minor nesting beaches of sea turtles across the Caribbean (Figure 8.6A,B). Comparing these historical data with modern records revealed that today's populations of about 300,000 green turtles (*Chelonia mydas*) and about 30,000 hawksbill turtles (*Eretmochelys imbricata*) represent only about 0.3% of their historical abundance. Similar data were used to reconstruct the former distribution of the Caribbean monk seal (*Monachus tropicalis*) as well as its gradual range contraction and final extinction (Figure 8.6C; McClenachan and Cooper 2008).

Other studies use old logbooks and maps from whaling vessels to reconstruct the former distribution and abundance of whales in the ocean. One such study suggests that the historical feeding range of humpback whales (*Megaptera novaeangliae*) in the North Atlantic may have extended to the Mid-Atlantic Ridge, which is very different from today's coastal migration routes (Reeves et al. 2004). Northern right whales (*Eubalaena glacialis*) used to occur on both sides of the North Atlantic, but a long history of intense whaling from AD 1000 to the early twentieth century extirpated the population on the eastern side, leaving

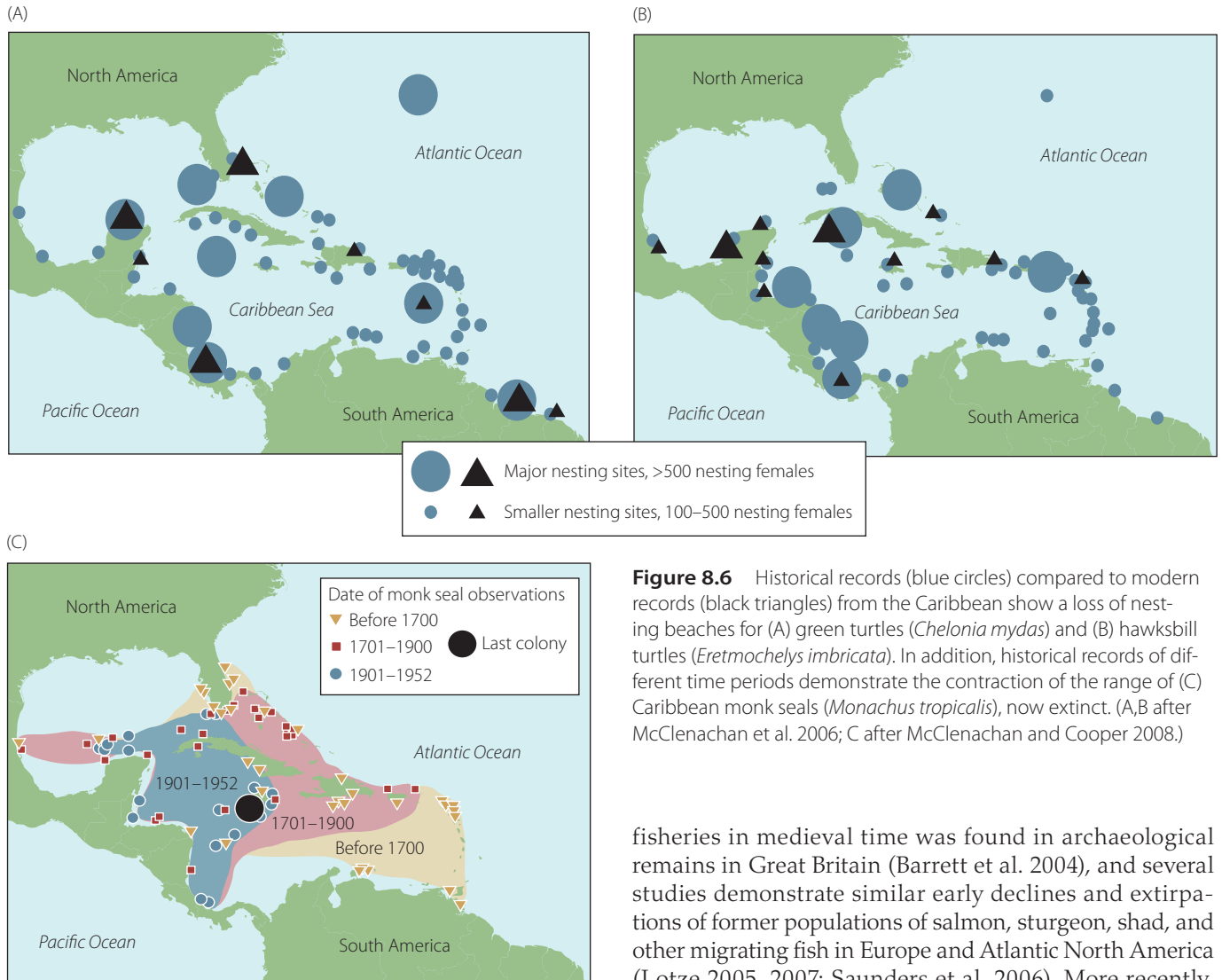


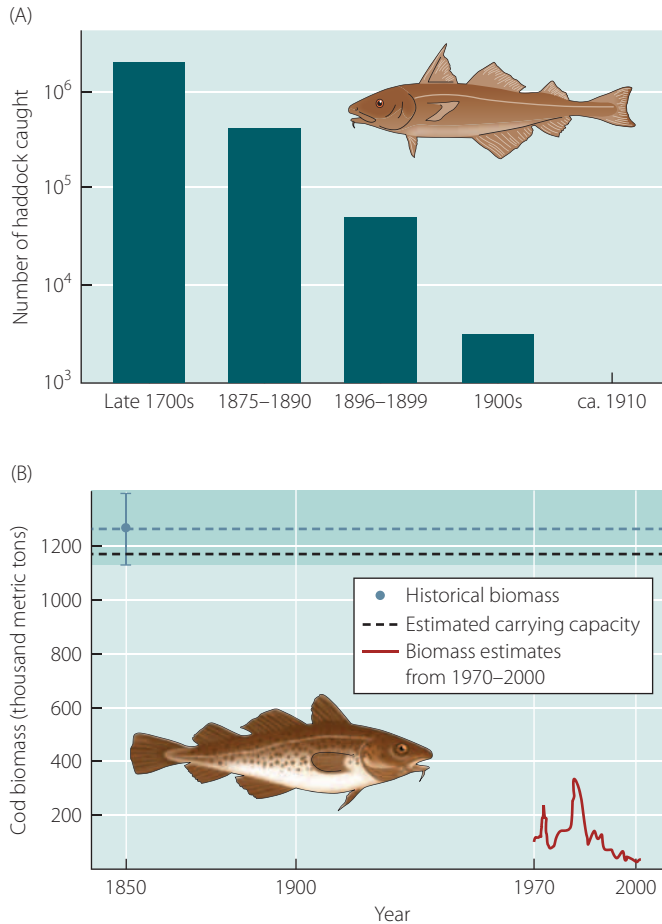
Figure 8.6 Historical records (blue circles) compared to modern records (black triangles) from the Caribbean show a loss of nesting beaches for (A) green turtles (*Chelonia mydas*) and (B) hawksbill turtles (*Eretmochelys imbricata*). In addition, historical records of different time periods demonstrate the contraction of the range of (C) Caribbean monk seals (*Monachus tropicalis*), now extinct. (A,B after McClenachan et al. 2006; C after McClenachan and Cooper 2008.)

only a small population of 300–400 individuals remaining on the western side today, about 3% of the estimated pre-whaling abundance (Reeves 2001). In a similar manner, whaling for southern (*Eubalaena australis*) and North Pacific right whales (*Eubalaena japonica*) rapidly expanded in the nineteenth century and quickly depleted their populations to very low numbers (Josephson et al. 2008).

Historian Richard Hoffmann (1996, 2001) used a range of illustrated and written sources to reconstruct changes in the marine and freshwater fisheries of medieval Europe. He was able to show that the formerly preferred freshwater and migrating fishes, such as salmon and sturgeon, became first depleted and then scarce in the Middle Ages due to overexploitation, habitat destruction, and water pollution. As a consequence, people implemented fisheries regulations, but also introduced carp aquaculture from Asia and expanded their fisheries for marine species, such as herring, cod, and tuna, around Europe (Hoffmann 1996, 2001). A similar shift from freshwater to marine

fisheries in medieval time was found in archaeological remains in Great Britain (Barrett et al. 2004), and several studies demonstrate similar early declines and extirpations of former populations of salmon, sturgeon, shad, and other migrating fish in Europe and Atlantic North America (Lotze 2005, 2007; Saunders et al. 2006). More recently, Bolster (2012) described the purposefully destructive nature of North Atlantic fisheries and how marine fisheries in the age of sail expanded from Europe throughout the North Atlantic.

Historical records of bankers, financiers, and tax collectors enabled Ravier and Fromentin (2004) to reconstruct a 300-year record of bluefin tuna (*Thunnus thynnus*) catches in the Mediterranean. While long-term fluctuations between 1650 and 1950 were probably driven by changes in temperature, strong recent declines were due to overexploitation (MacKenzie et al. 2009). Holm (2005) used fish inspectors' reports, tax records, and archival information to reconstruct changes in inshore haddock (*Melanogrammus aeglefinus*) stocks in the Danish Wadden Sea. In 1562, catches were 1200 metric ton and remained stable for a century before declining to 500 metric tons by the eighteenth century and zero in the twentieth century. Today the fishery operates far offshore because inshore stocks are nonexistent. Likewise, the handline haddock fishery around Helgoland in the southern North Sea caught more than a million haddock in the late eighteenth century,



while landings dropped to less than a hundred thousand in the late nineteenth century, to a few thousand around 1900, and ceased to exist by 1910 (**Figure 8.7A**; Lotze 2005). On the other side of the Atlantic, Hutchings and Myers (1995) reconstructed historical catches and harvest rates of Atlantic cod (*Gadus morhua*) in Newfoundland from 1500 to 1991 and showed that the first signs of overfishing appeared in the mid-1800s. Using detailed logbook information from fishing schooners, Rosenberg and colleagues (2005) estimated the biomass of cod on the Scotian Shelf in the 1850s at around 1.2 million metric tons, which is three to four times greater than their maximum abundance in the 30 years before the collapse of the fishery in 1992 (**Figure 8.7B**). This historical biomass estimate was corroborated by an analysis of cod carrying capacity on the Scotian Shelf (Myers et al. 2001). Finally, McClenachan (2009) used historical photographs to document the decline of large trophy fish in the Florida Keys from 1956 to 2007. The mean size of fish caught declined from 19.9 to 2.3 kg, and the species composition of catches changed from large groupers, sharks, and other large predatory fish in the 1950s to small snappers in the 2000s.

All of these examples show that historical data are invaluable for understanding the former distribution and abundance of species and the rise and fall of past fisher-

Figure 8.7 Historical records show declines in fish catches. (A) Decline of haddock landings in the handline fishery near Helgoland, in the southern North Sea, from the late eighteenth to early twentieth century. (B) Estimate of historical biomass of Atlantic cod on the Scotian Shelf based on logbook data from 1852 (with confidence interval), estimated carrying capacity (based on Myers et al. 2001), and recent biomass estimates from fisheries-independent research surveys from 1970 to 2000. (A after Lotze 2005; B after Rosenberg et al. 2005.)

ies. In some cases, if the records are detailed and precise enough, they can be used to estimate the relative or even absolute abundance of commercially important species. Historical data often have a high temporal resolution, yet consistent records usually cover only short periods (see Table 8.1). Moreover, historical records are generally biased by people's selective interest in species that could be utilized in one way or another. The spatial scale can range from very local to global depending on the record and species considered (see Table 8.1).

Fisheries and hunting records

In the nineteenth and twentieth centuries, people began to compile more detailed and systematic fisheries and hunting statistics, including records on the species and numbers caught, their size and weight, and in some cases the gear and effort used (Goode 1884–1887; Smith 1994). These data often provide the best and most consistent long-term quantitative data available for the recent and historical past (see Table 8.1), yet they are biased toward species of greatest commercial and recreational interest and depend on the accuracy of reporting and other factors. Unstandardized catches, for example, can be influenced by the gear used, the fishing effort, and the interests involved, as well as by market forces and management restrictions. Therefore, catch data alone do not directly reveal trends in abundance, but can still provide valuable insight into the beginning, peak, and end of a fishery.

Over time, fisheries scientists have developed a variety of analytic methods to more accurately infer past trends in abundance from catch data (Smith 1994). For example, records of fishing effort can be used in combination with catch size to calculate standardized rates of catch per unit effort (CPUE) as a measure of relative abundance over time. This approach was used in a study on historical changes in the groundfish and lobster fisheries in the outer Bay of Fundy (Lotze and Milewski 2004). In both fisheries, total catches increased in the late nineteenth century and declined in the first half of the twentieth century, while the number of fishing boats and handlines in the groundfishery and the number of traps in the lobster fishery continuously increased. Therefore, calculated CPUEs showed even stronger declines, indicating that both groundfish and lobster stocks were already severely depleted by the early twentieth century (**Figure 8.8**). In-

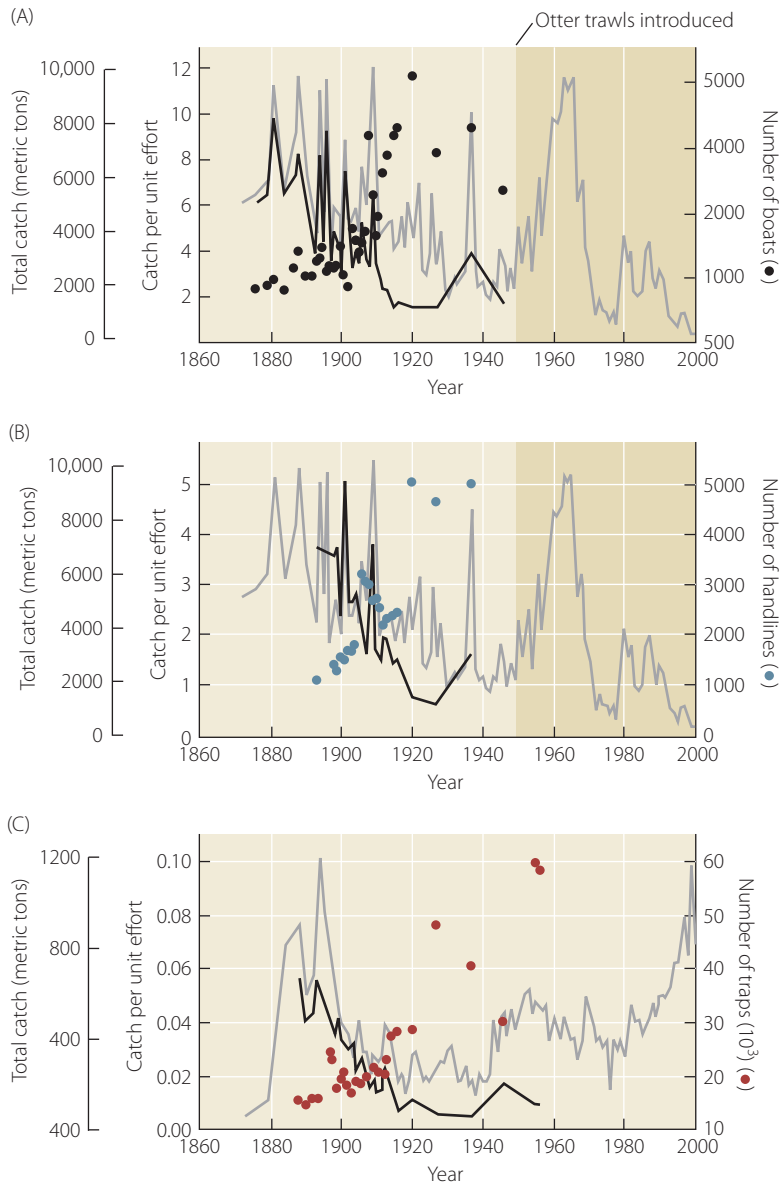


Figure 8.8 Historical fisheries records from the outer Bay of Fundy. Decline in the traditional handline fishery for groundfish (cod, pollock, haddock) measured as total catch from 1870 to 2000 (light gray lines) and catch per unit effort (CPUE; solid black lines) with respect to (A) the number of fishing boats (black dots) and (B) number of handlines (blue dots) from 1870 to 1950, after which otter trawls were introduced, resulting in an offshore expansion of the fishery. (C) Total catch (light gray line), number of traps (red dots), and catch per trap (CPUE; solid black line) for the lobster fishery. (After Lotze and Milewski 2004.)

creasing catches in the second half of the twentieth century were achieved only by moving farther offshore in both fisheries and by using more efficient otter trawls in the groundfishery, whereas for lobsters, release from predation due to declining groundfish stocks also contributed to the recent increase in landings (Lotze and Milewski 2004). In a study from the Mediterranean, Ferretti and colleagues (2008) combined several commercial and recreational fishing and sighting records to analyze changes in CPUE of large sharks over the past century. Their results suggest long-term declines of 96–99%. Similarly large declines in large sharks in the twentieth century in the northwestern Atlantic (Myers et al. 2007), the Gulf of Mexico (Baum and Myers 2004), and the tropical Pacific (Ward and Myers 2005) have been documented using different data sources.

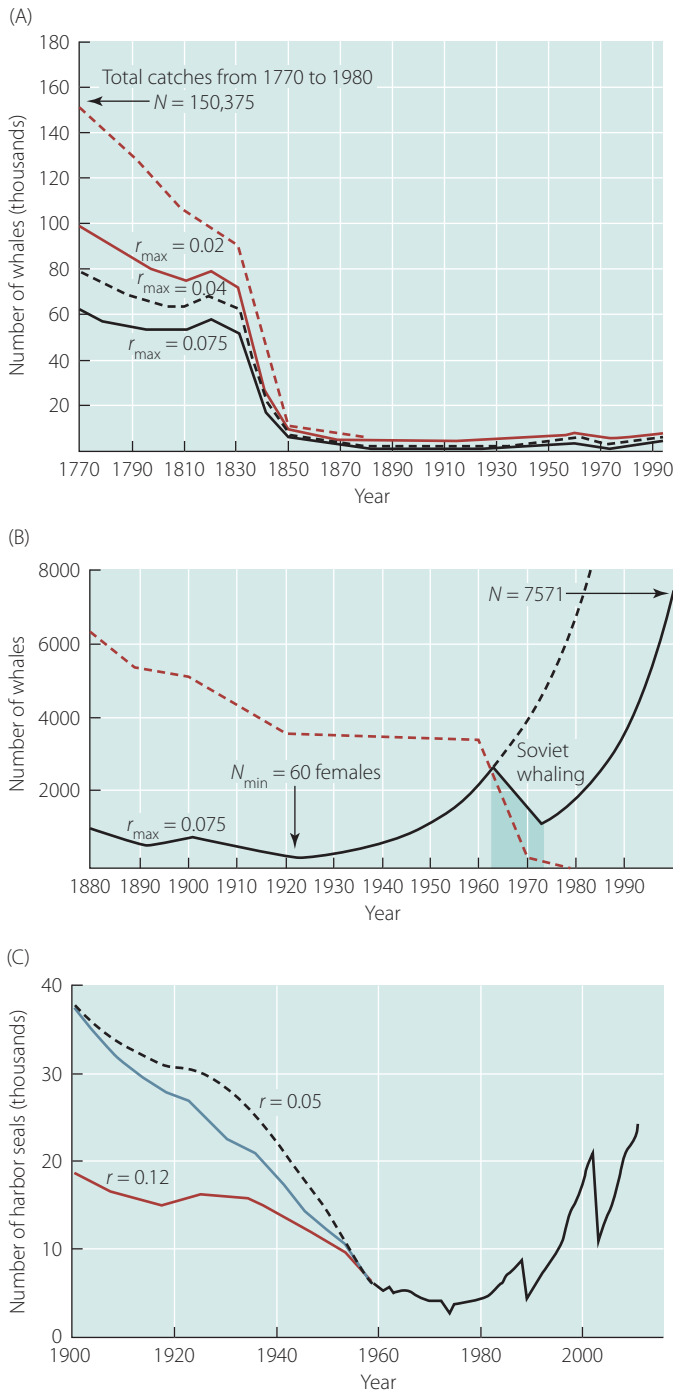
In the case of marine mammals, hunting records from the whaling and sealing industries have been used to re-

construct past trends in population abundance. This approach requires records on the number of individuals caught in the past in combination with current abundance estimates and information on population growth rates, which can be incorporated into population growth models. One such study of southern right whales (**Figure 8.9A**) estimated that their pre-exploitation abundance was about 80,000 individuals, that the strongest population decline occurred between 1830 and 1850, that the population was depleted to 0.1% of its former abundance by the early twentieth century, and that it has since recovered to 7600 individuals, or 9.5% of its former abundance following protection (**Figure 8.9B**; Baker and Clapham 2004). Another example using a similar approach comes from a study of harbor seals (*Phoca vitulina*) in the Wadden Sea (**Figure 8.9C**), which were hunted to very low levels in the 1960s and 1970s and have since recovered to more than 24,000 individuals in 2011 following hunting bans and habitat protection (Lotze 2005; Trilateral Seal Expert Group 2011). Two recent disease outbreaks raised the question of whether harbor seals had reached carrying capacity; yet a reconstruction of their former abundance based on hunting records suggested that there were probably around 37,000 individuals in 1900 (Reijnders 1992).

All these examples show that fisheries and hunting statistics, if used and analyzed in a meaningful way, can provide great insight into past trends in fisheries and changes in population abundance.

Scientific surveys

Beginning in the mid-nineteenth and early twentieth centuries, scientists began to study marine animals and plants more systematically. The resulting biological records include species lists and data on the abundance, distribution, and life history characteristics of certain species as well as on environmental parameters, such as water temperature. Such investigations are often episodic and short-term in scope (see Table 8.1), but useful when compared with more recent ones if similar sampling methods were applied, if one



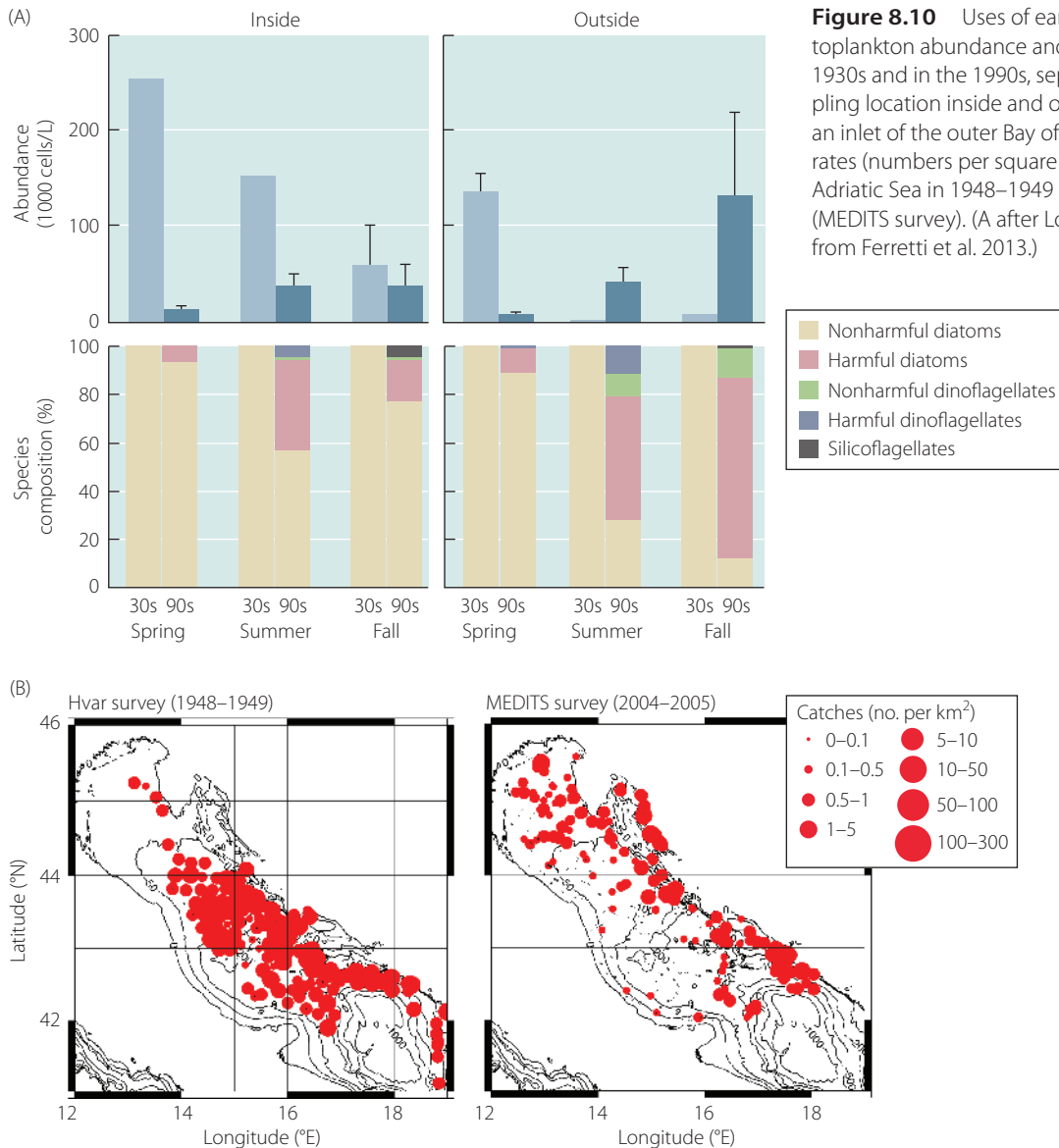
can correct for the differences in methods, or if modern methods can be adapted to simulate those in the past. For example, comparing historical and recent species lists can provide insight into the number of species extirpations or introductions (Lotze et al. 2005, 2006). A comparison of two plankton sampling episodes from the 1930s and 1990s in the outer Bay of Fundy (**Figure 8.10A**) revealed that patterns of phytoplankton abundance shifted from formerly dominant spring blooms to more prevalent fall blooms, and that species composition shifted from the past dominance

Figure 8.9 Uses of whaling and hunting records. (A) Reconstruction of the historical abundance of southern right whales from the start of whaling in 1770 to 1997, using estimates of current abundance and historical catch records along with a basic population growth model with different values of maximum net reproduction (r_{max}). Back-calculated total cumulative catches of 150,375 whales from 1790 to 1980 are plotted in red (dashed line). (B) An expanded scale of abundance for 1880 to 1997. The downward arrow points to the year of minimum population size, and the dotted line shows the predicted recovery if there had been no illegal hunting by the Soviet Union. (C) Back-calculation of the number of harbor seals in the Wadden Sea from 1900–1960 based on historical hunting records and a population model. The model assumes different rates of exponential population increase: $r = 0.05$ (black dashed line), $r = 0.12$ (red line), and a sliding r from 0.05 in 1900 to 0.09 in 1960 (blue line; data from Reijnders 1992). For comparison, monitoring data for the population (black solid line) are available since the 1960s (data from Lotze 2005, Trilateral Seal Expert Group 2011). (A,B after Baker and Clapham 2004; C after Reijnders 1992 and Lotze 2005.)

of nonharmful diatoms to a greater proportion of dinoflagellates and harmful species, which are common signs of eutrophication (Lotze and Milewski 2004). In another approach, Reise and colleagues (1989) rebuilt former sampling gear to repeat a historical oyster bank survey, thereby demonstrating that with the loss of former oyster banks, the complexity and diversity of the benthic community decreased.

Because such research survey or monitoring data from before the 1960s are rarely available, they typically reflect conditions long after exploitation or other human impacts began and therefore rarely provide insight into historical baselines. In some cases, however, research surveys were performed earlier and cover an extended period of time. For example, Ferretti and colleagues (2013) combined and standardized catch data from five scientific research surveys in the Adriatic Sea from 1948 to 2005 and used these data to estimate abundance trends of sharks and rays over time and space. They found that since 1948, catch rates have declined by 94%, and 11 species have ceased to be detected. Moreover, both catch rates and species richness were lower in the western, Italian part of the Adriatic, where industrial fishing developed earlier and fishing pressure is higher (**Figure 8.10B**). Even if survey data don't reach back that far, they are important reference points for the present-day abundances, distributions, and sizes of marine species (see, for example, Figures 8.7B and 8.9B), and they are essential for analyzing current population trends (Lotze et al. 2011b).

Present-day monitoring data can also be used to hindcast former conditions. For example, Jennings and Blanchard (2004) used biomass and size data from recent trawl surveys in the North Sea to estimate fish biomass under no fishing and under current fishing pressure. Their results suggest that medium-sized fish (4–16 kg) have declined by 97% and large fish (16–66 kg) by more than 99%



due to fishing. In a related study, Christensen and colleagues (2003) used ecosystem models to predict past and present biomasses of high-trophic level fish for the North Atlantic as a whole and found a 90% decline since 1900.

In the absence of any historical data, some studies have designed field surveys across a spatial gradient of fishing pressure to evaluate how much fish biomass has changed due to fishing (see Table 8.2). For example, Friedlander and DeMartini (2002) performed dive surveys in the remote and largely unpopulated Northwestern Hawaiian Islands and the highly populated main Hawaiian Islands to compare shallow-water reef fish assemblages. Their results indicate that overall fish biomass was 72% lower in the main Hawaiian Islands and that the biomass of large predators, such as sharks and jacks, was 94% lower. Similar results were reported in dive surveys across the Line Islands, where the biomass of large predators decreased by more than 80%

from protected and unpopulated Kingman Reef to highly populated and fished Kiritimati Island (Sandin et al. 2008).

Overall, standardized research surveys and monitoring programs provide detailed trends of biological or environmental changes over the past decades (see Table 8.1). Their temporal and spatial resolution can be very high, and their level of precision is typically greater than that of other types of historical data, which allows for estimation of relative or even absolute abundance and can help researchers to tease apart the causes and consequences of observed changes. However, the data can be biased by the purpose and design of the survey as well as by the methodology and effort used. Moreover, apparent trends may reflect phases of longer cycles, or even random fluctuations in ecosystems that were greatly altered by human disturbance before monitoring began. It is therefore important to place them in an adequate historical context.

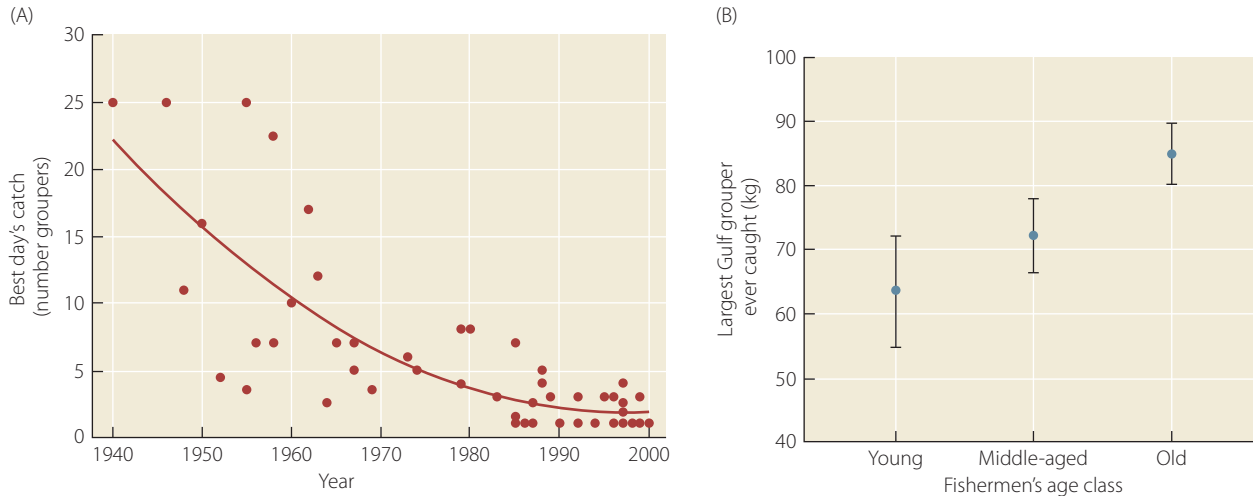


Figure 8.11 Living memory data based on interviews with fishermen of different ages in the Gulf of California. (A) Number of Gulf groupers in the best day's catch plotted against the year in which the fishermen remembered landing it. (B) Mean weight with 95% confidence interval of the largest Gulf grouper ever landed by fishermen from three generations (After Saenz-Arroyo et al. 2005.)

Living memory

Not all possible information about the past is written down, but many people have a memory of what things looked like in the past, and that information can be accessed through interviews. For example, Sáenz-Arroyo and colleagues (2005) interviewed fishermen of different ages in the Gulf of California to ask them when they had caught the largest number of groupers in one day (their best day's catch) in the past and the size of the largest grouper they had ever caught, information they usually remembered well. The results indicate that older fishermen remembered catching up to 25 groupers a day in the 1940s and 1950s, while younger fishermen's best day's catch was fewer than 5 groupers a day in the 1990s (**Figure 8.11A**). In addition, the largest groupers caught by older fishermen averaged between 80 and 90 kg, whereas those of younger fishermen averaged 60 to 70 kg (**Figure 8.11B**). This study provides a clear example of the “shifting baseline syndrome” described on p. 166 (see **Figure 8.2**; Pauly 1995). In a similar study of a traditional fishery on coral reefs on Rodrigues Island in the Indian Ocean, older fishermen remembered catching about three times more and five times larger groupers than younger fishermen (Bunce et al. 2008). Older fishermen had observed a decline in grouper catches over the past 25 years, which prompted them to expand their fishery out of the lagoon to areas farther offshore. They also reported more fish species in decline (18 vs. 8.5) over a longer period (15 vs. 5 years) than younger fishermen did. These studies are great examples of how valuable information about changes in a fishery can be gained from interviewing knowledgeable resource users, and this approach may be particularly important in

situations where other quantitative data are lacking, as is the case for many artisanal fisheries.

Molecular data

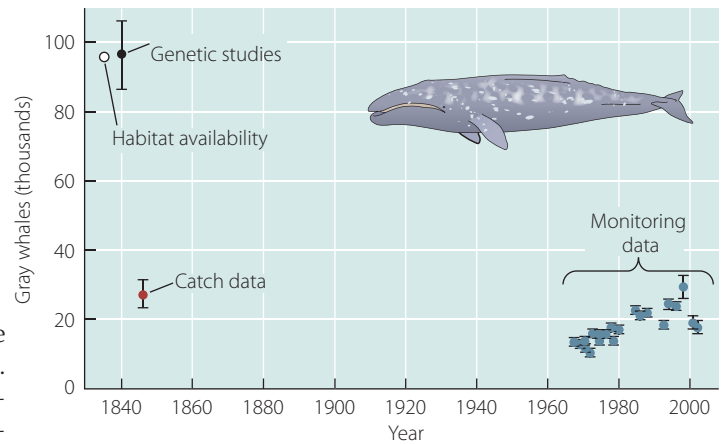
Another approach to reconstructing the past comes from the use of molecular data. Both the genes in organisms living today and ancient DNA from extinct species represent an archive of information over evolutionary time. Because levels of neutral genetic variation increase with population size, the genetic diversity found in a population today reflects its average population size throughout the past (Palumbi 2011). Using this approach, Roman and Palumbi (2003) estimated the average long-term population sizes of humpback, fin, and minke whales in the North Atlantic. Their results suggest that past populations might have been on the order of 240,000 humpback, 360,000 fin, and 265,000 minke whales before intensive whaling. This leaves today's population of humpbacks at 4%, fins at 15%, and minkes at 56% of their original size, far less than previous calculations based on current abundance and historical catch records. The difference may be due to uncertainties in catch records, such as the often high numbers of whales that were struck and killed but not landed, as well as uncertainties in the molecular approach, such as estimates of mutation rates (Baker and Clapham 2004; Palumbi 2011). A similar approach was used to estimate the pre-whaling abundance of Pacific gray whales (**Figure 8.12**). Monitoring records since the 1970s showed an increase in the population, and reconstructions of past abundance based on catch records suggested that the population had about fully recovered (Rugh et al. 2005). This observed population trend could have led to reduced management concern for the Pacific gray whale, but genetic analyses revealed a pre-whaling abundance that was three to five times higher than current population levels (Alter et al. 2007). Interestingly, a separate study that aimed to estimate pre-whaling abundance of Pacific gray whales based on a reconstruction of habitat availability reached a similar conclusion as the genetic approach (see **Figure 8.12**; Pyenson and Lindberg

Figure 8.12 Recent increase of gray whales in the northeastern Pacific based on monitoring data (means and SE, data from Rugh et al. 2005) compared with different historical population estimates based on catch data (data from Rugh et al. 2005), genetic studies (data from Alter et al. 2007), and habitat availability (data from Pyenson and Lindberg 2011). Commercial whaling for this species was banned by the League of Nations in 1936, and then in 1949 by the International Whaling Commission.

2011). Overall, molecular data can provide very valuable estimates of long-term population abundances in the past. However, they integrate population trends over long temporal and large spatial scales and cannot be used to identify trends at finer resolution (see Table 8.1).

Integrating different data sources and disciplines

Although each discipline and data source provides valuable information on its own, many of the examples we have mentioned in this section show that more insight can be gained when data from different sources are combined or compared to reconstruct the past (see Table 8.2). For example, McClenachan and colleagues (2006) used a variety of historical and archaeological data to reconstruct the past distribution and abundance of Caribbean sea turtles, then compared past with present-day estimates to determine the magnitude of change (see Figure 8.6A,B). Similarly, time-series analysis of modern monitoring data in combination with historical hunting records allowed the back-calculation of past harbor seal abundance in the Wadden Sea (see Figure 8.9B; Reijnders 1992; Lotze 2005). Ecosystems are dynamic, however, and one snapshot of the past is unlikely to represent an absolute baseline (Figure 8.13A). A historical estimate of abundance, distribution, or body size based on records from 50 years ago may be lower than one based on records from 100 or 500 years ago. It is therefore important to place the results of historical analyses in the larger historical context of what happened earlier and later, such as when exploitation started or environmental conditions changed. Another problem arises when different data sources or reconstruction methods result in different estimates of past abundance (Figure 8.13B). For example, estimates of historical abundances for



the Pacific gray whale based on genetic analyses and former habitat availability are similar and corroborate each other, yet both are very different from estimates based on the analysis of historical catch records (see Figure 8.12). These discrepancies can be resolved only with further research into the uncertainties in the data or methods (e.g., Palumbi 2011). If many different estimates exist, an average across all estimates, such as a numerical (e.g., Lotze and Worm 2009) or meta-analytic average (e.g., Myers and Worm 2005), together with a variance estimate or confidence interval, can derive a more synthetic result.

Many of the examples in this section show that, depending on the available data, it is sometimes possible to reconstruct the past abundance or distribution of an individual species for a certain period of time. But how do we extend timelines from archaeological records to the present? Or time series from the twentieth century to earlier centuries? And how do we compare data series across different species? Here, different data, methods, and disciplines can be seen as pieces of a puzzle. The individual pieces and the overall puzzle can be qualitative, conceptual images of the past or more quantitative reconstructions. In an effort to combine such different pieces, Jackson and colleagues (2001) used quantitative case studies from different ecosystems, including kelp forests, coral reefs, seagrass beds, and estuaries, and pieced them together into an overall narrative of historical changes in coastal ecosystems. Likewise,

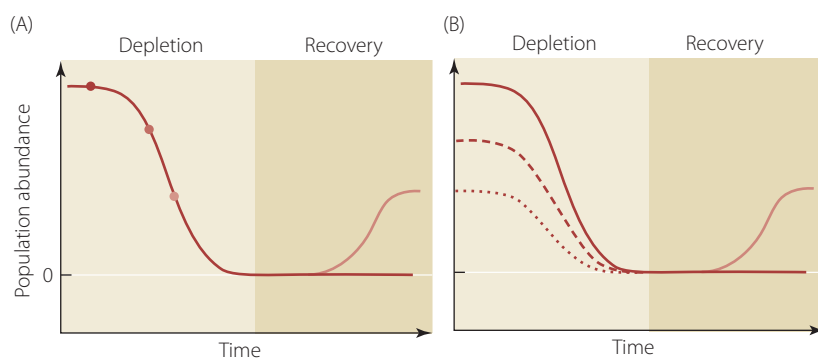


Figure 8.13 Conceptual diagram of historical depletion and recovery trends. The estimated historical baseline (left side of each diagram) can vary with (A) differences in the timing of a historical estimate (represented by solid dots of different colors), or (B) differences in reconstruction methods (represented by the different solid, dashed, and dotted lines). These different historical estimates will influence the judgment of the magnitude of recovery (right side of each diagram) as full recovery or partial recovery depending on which historical estimate is used.

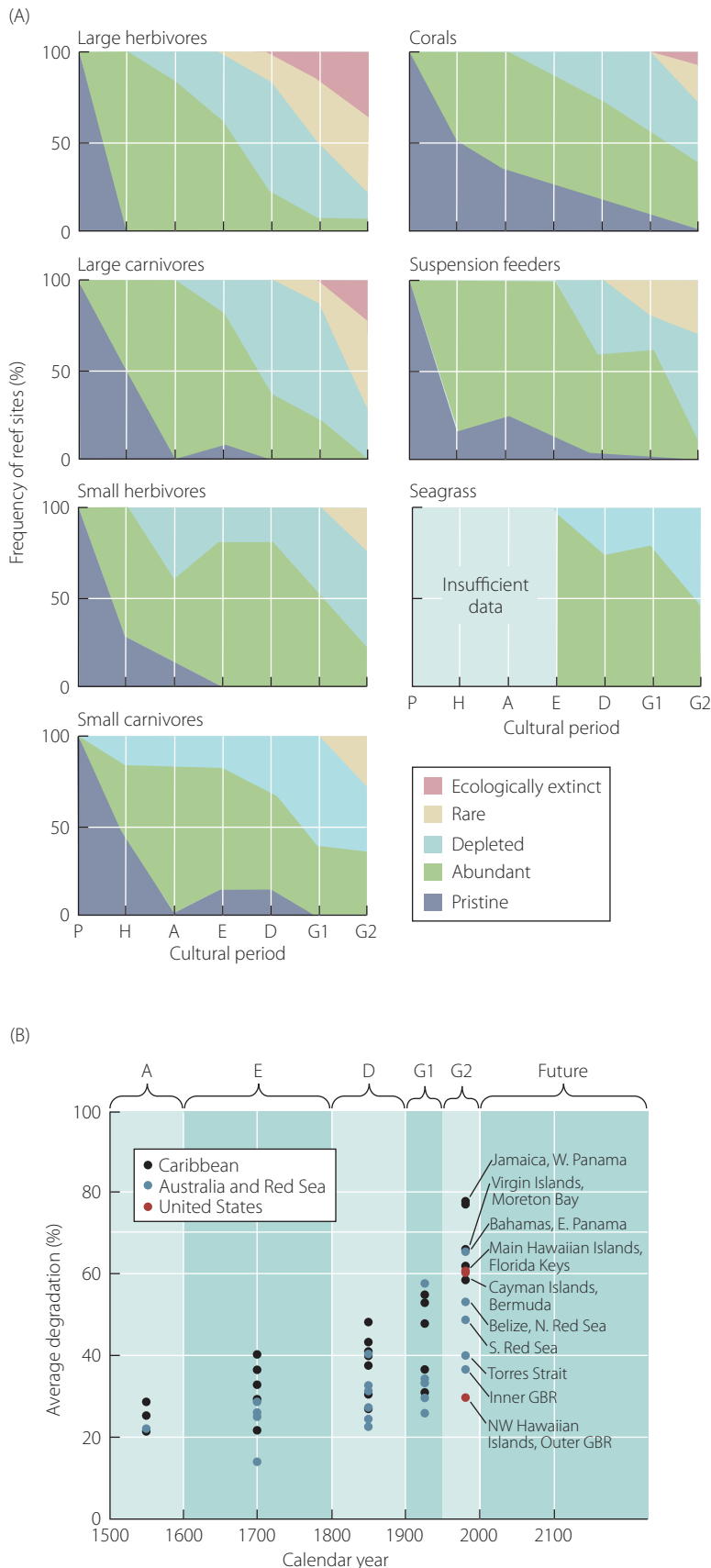


Figure 8.14 Historical changes in coral reef ecosystems. (A) Trajectories of change for seven functional groups, expressed as the percentage of reef sites in each ecological state of abundance from 14 regions in the Caribbean, Australia and Red Sea. (B) Average degradation over time of each of the 14 reef sites above plus an additional 3 in the United States. See Table 8.3 for cultural period abbreviations and definitions. (A after Pandolfi et al. 2003; B after Pandolfi et al. 2005.)

qualitative and quantitative pieces of information on different species groups were used to piece together the long-term history of changes in the Benguela upwelling system (Griffiths et al. 2004), the outer Bay of Fundy (Lotze and Milewski 2004), the Wadden Sea (Lotze 2005), and the Gulf of California (Sáenz-Arroyo et al. 2006).

Taking it a step further, Pandolfi and colleagues (2003) compiled paleontological, archaeological, historical, and recent scientific data for a range of ecologically important species from 14 coral reef ecosystems around the world. Individual species were pooled into functional groups that were comparable across reefs. Similarly, the dated histories of human settlement and activities at each site were grouped into cultural periods that could be compared across reefs (Table 8.3). The authors then defined categorical abundance estimates, such as pristine, abundant, depleted, rare, and ecologically extinct, which were assigned to each species or functional group for each cultural period based on quantitative or qualitative evidence. In this way, they could piece together trajectories of change from prehuman to recent time, showing strong degradation of many functional groups following European colonization (Figure 8.14A). Using principal component analysis, they could also rank the overall degradation of each ecosystem across time and across different regions, highlighting which reefs are most degraded compared with pristine conditions (Figure 8.14B; Pandolfi et al. 2005). A similar study aimed to reconstruct the history of ecological changes in 12 estuaries and coastal seas around the world (Figure 8.15; Lotze et al. 2006). The authors of this study incorporated more quantitative results, using both numerical and categorical estimates of abundance across species and cultural periods (see Table 8.3), and also combined estimates of changes in water quality and species invasions. Such interdisciplinary studies provide synthetic, highly aggregated timelines that are based on hundreds of individual estimates and allow broad comparisons across different regions and species groups. The main results of these studies are discussed later in this chapter.

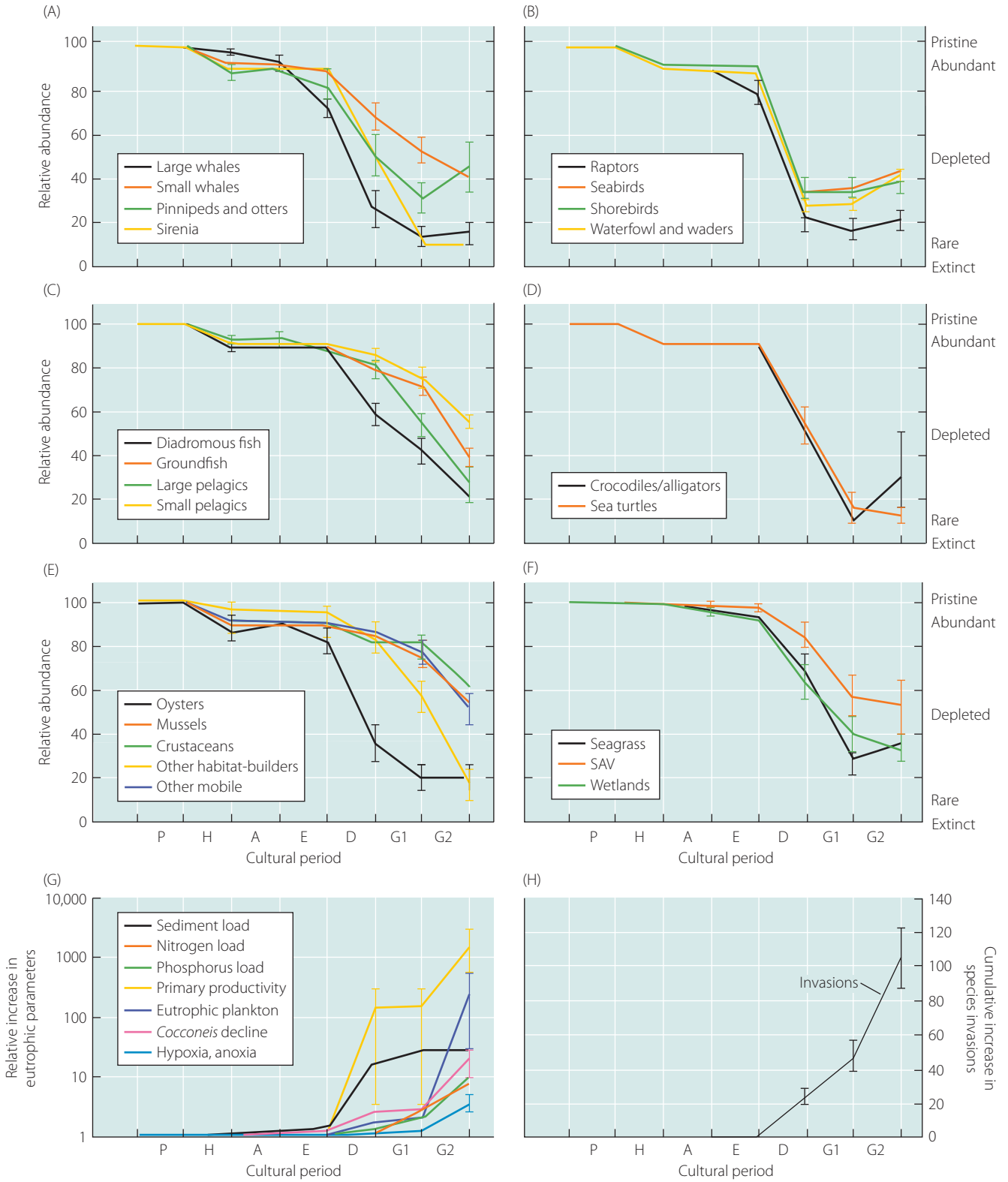


Figure 8.15 Historical changes in estuaries and coastal seas. (A–F) Trajectories of change in relative abundance for 22 species groups averaged over twelve study systems in Europe, North America, and Australia for (A) marine mammals, (B) coastal birds, (C) fish, (D) reptiles, (E) invertebrates, and (F) vegetation. (G) Degradation of

water quality as indicated by the relative increase in eutrophication parameters across eight study systems. (H) Cumulative increase in recorded species invasions across five study systems. Data are means and SE. See Table 8.3 for cultural period abbreviations and definitions. (After Lotze et al. 2006.)

TABLE 8.3 Definitions of cultural periods based on human presence, technology, and market conditions

CULTURAL PERIOD	HUMAN PRESENCE, TECHNOLOGY, MARKET CONDITIONS
Prehuman (P)	No evidence of or insignificant human presence; pristine ecosystems with only natural disturbance
Hunter-gatherer (H)	No permanent settlements; small populations; subsistence exploitation for individual resource use; pre-market with no major surplus for trade and no major system for distribution and exchange over large areas
Agricultural (A)	Agriculture enables permanent settlements; small populations; subsistence and artisan exploitation for individual or village-based resource use; pre-market with no major surplus for trade and no major system for distribution and exchange over large areas
Establishment of colony and market (E)	Establishment of local economy and market; European settlement in the New World and spread of Western values; still relatively low population numbers; trade between colonies and European empires; catch more than needed for own consumption; development of techniques for storage and transport; ship surplus to neighbors (up country, next village, etc.), exchange with barter
Development of colony and market (D)	Strong growth and expansion of economy, market, and trade; rapid population rise; people become centralized into larger cities; commercialization of resource use; development of luxury and fashion markets especially affecting birds (feather trade), mammals (whale baleen, seal and otter furs, walrus ivory), and reptiles (turtle shells, crocodile skins); onset of industrialization and technological progress; mass killing of mammals and birds with guns; fishing mostly inshore and seasonal with selective and nondestructive gear (e.g., hook and line, light trawls towed by sailboats over soft bottoms)
Early global (G1: 1900–1950)	Global economy and market develop; strong population increase; industrialization and technological progress toward more efficient, less selective, and more destructive gear (e.g., motorboats, steam trawlers), accelerating exploitation, bycatch, and habitat destruction; increasing fishing effort; fishing possible in any season but still mostly inshore and coastal
Late global (G2: 1950–2000)	Global economy and market; establishment of free-trade zones; increased industrial fishing (especially after World War II) extending offshore and toward the deep sea; multiple unselective and destructive gears (otter trawls, purse seines, long-lining, rockhopper gear, etc.); all habitats are being fished, all fish are detectable; mass fishing; other human impacts increase (eutrophication, pollution, fish farming); conservation efforts increase

Source: Pandolfi et al. 2003; Lotze et al. 2006.

Another approach to combining pieces of the puzzle to reconstruct the past comes from food web analyses. Food webs generally present networks of species that are linked through predator-prey relationships. A food web can be a conceptual drawing, describing which species have been present in the past compared with today, how they have interacted with other species, and how those interactions have changed over time (e.g., Jackson et al. 2001). But one can also create more quantitative food webs based on the simple presence of species and their diets, which link them into a “who-eats-whom” network. Such networks can then be analyzed for their emergent food web properties, such as the number of feeding links or average food chain length. This approach has been used to reconstruct Cambrian compared with modern food webs (Dunne et al. 2008), as well as food webs of different historical periods in the Adriatic Sea (Lotze et al. 2011a), to analyze changes in food web structure and robustness over time. If enough information on the abundance of individual species or functional groups is available or can be estimated, one can also build fully quantitative food webs based on biomass and energy flow, which has been done for past time periods in Newfoundland (Pitcher et al. 2002) and British Columbia (Ainsworth et al. 2002).

What We Can Learn from the Past

All the case studies presented in the previous section provide insight into aspects of historical changes in various species and ecosystems. With each individual study, and each new researcher interested in the past, the field of marine historical ecology has grown. Although there are still many gaps in our knowledge, there are an increasing number of results that allow us to look at the bigger picture and ask what we can learn from the ocean’s past. In this section we present an overview of major results and insights gained into different components of ecosystems, from environmental and species changes to changes in habitats, water quality, and overall ecosystem structure. We then summarize overarching patterns in the magnitude, direction, and rate of change.

Environmental changes

Change is a major part of life and of the history of this planet. Over geologic time scales, dramatic changes in environmental conditions have resulted in mass extinctions, followed by speciation and radiation of new life forms. Even less dramatic changes, such as temperature fluctuations during glacial and interglacial periods, have

caused extinctions at “background rates” (Harnik et al. 2012). Marine historical ecology can help us understand the natural variation of environmental parameters and how it has affected the abundance and extinction of species over geologic and historical time scales. Understanding the underlying influence of environmental parameters is critical if we are to place recent changes in context and distinguish between natural and anthropogenic changes. For example, changes in both climate and exploitation pressure have strongly influenced the abundance of seabirds on the Aleutian Islands over the past few millennia (Causey et al. 2005) and of cod in Newfoundland over the past few centuries (Rose 2004). Yet while overhunting occurred at small scales with few long-term effects in the Aleutian Islands, overfishing together with climate change resulted in the collapse of Newfoundland cod stocks in the 1990s, with no recovery to date. Generally, the Holocene (the 12,000 years since the last Ice Age) has been a period characterized by stable climatic conditions, which has allowed the growth, spread, and development of the human population and its impacts around the globe. Thus, in many cases, human pressures have been more significant drivers of historical change than environmental fluctuations. With increasing temperatures and CO₂ concentrations, the ocean is currently facing a combination of drivers that is unprecedented in the history of this planet, putting an increased number of species at risk of extinction (Figure 8.16).

Changes in species occurrence and abundance

Many case studies document the history of depletion of a marine population or species, including changes in abundance, distribution, and size or age structure over time. In some cases, the depletion has led to local or regional extirpation or even global extinction, as for the Caribbean monk seal (see Figure 8.6C). Most historical changes have been documented for species that have been of value to humans as a resource for food, fuel, fashion, or other products and could be easily found, caught, and processed. In several cases, the depletion happened during the time of hunters and gatherers, as for New Zealand fur seals, which were severely reduced by Maori hunters (Smith 2005). Overall, however, human pressures increased over time with growing human populations and demands as well as the commercialization and industrialization of exploitation (see Table 8.3). As a result, more and more species were affected, and the degree of depletion intensified. In order to summarize the overall historical changes in species occurrence and abundance, we discuss a few synthetic studies in this section.

Comparing trends for several functional groups across 14 coral reef ecosystems in the Caribbean, the Red Sea, and Australia revealed very similar historical trajectories of change (see Figure 8.14A; Pandolfi et al. 2003). Indigenous people during the hunter-gatherer and agricultural peri-

ods had limited effects on valued resources. Yet after European colonization, large carnivores, such as sharks and monk seals, and large herbivores, such as manatees and green turtles, were rapidly depleted, followed by smaller animals and architectural species, such as corals and seagrasses. Over the last 50–100 years, some groups became so depleted that they were deemed ecologically extinct. By the end of the twentieth century, Jamaica and Western Panama were the most degraded ecosystems, and the outer and inner Great Barrier Reef were the least degraded ecosystems (see Figure 8.14B; Pandolfi et al. 2005). Interestingly, a recent study reconstructing similar historical trajectories for Hawaiian coral reefs revealed that several groups showed some degree of recovery in recent decades, especially in the more remote and protected Northwestern Hawaiian Islands (Kittinger et al. 2011).

Similarly, comparing trends for many valued species across 12 estuaries and coastal seas also showed minor changes in resource abundance during the hunter-gatherer and agricultural periods (see Figure 8.15). However, depletions of marine mammals, birds, reptiles, fish, and invertebrates increased during the period of market or colonial development—namely, the Roman Empire in the Mediterranean Sea, the Middle Ages in northern Europe, and European colonization in North America and Australia (see Table 8.3; Lotze et al. 2006). By the late twentieth century, 91% of the recorded species were depleted, 31% were rare, and 7% were locally extirpated or globally extinct. On average, large whales were depleted earlier and more severely than small whales or pinnipeds, and raptors were the most heavily depleted species among birds (see Figure 8.15). Among reptiles, both crocodiles and sea turtles were heavily reduced. Diadromous fishes, such as salmon and sturgeon, were the most affected species among fish, and oysters were the most affected among the invertebrates. These patterns are in concordance with the history of whaling for oil and baleen, the strong persecution of raptors as competitors and for their feathers, the exploitation of crocodiles for their leather and sea turtles for their meat, eggs, and shells, the effects of fishing and river alterations on diadromous fishes, and the high value of oysters as a food, delicacy, and aphrodisiac. Interestingly, while abundances of marine mammals, birds, and reptiles began to stabilize or even increase in the twentieth century due to stricter conservation and management efforts, abundances of fish and invertebrates continued to decline (see Figure 8.15). Detailed descriptions of historical changes in individual species and areas can be found for the Wadden Sea (Lotze 2005), the Adriatic Sea (Lotze et al. 2011a), and across six U.S. estuaries (Lotze 2010).

To quantify the overall magnitude of historical depletions across a wide range of large marine animals, Lotze and Worm (2009) compiled 256 case studies that reported absolute abundance estimates of historical and recent population sizes. On average, populations were depleted by

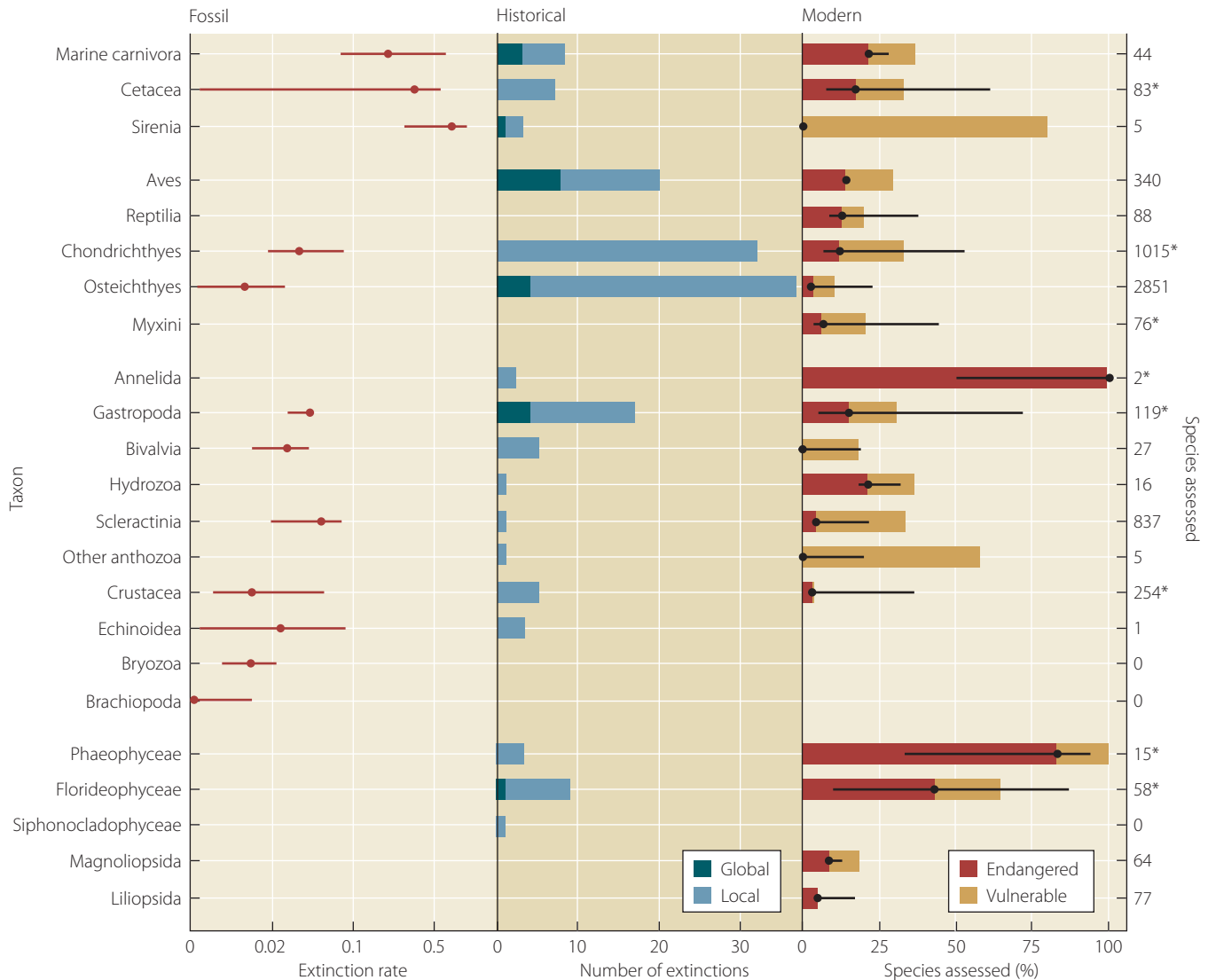


Figure 8.16 Comparison of extinction risks for different taxonomic groups through time. The first column shows extinction rates in the fossil record over the Cenozoic era (past 65 million years) expressed as medians (circles) and 1st and 3rd quartiles (lines). The second column shows the number of recorded historical global extinctions and local extirpations. The third column shows the percentages of modern species assessed in the International Union for the Conservation of Nature (IUCN) Red List as endangered or critically endangered and vulner-

able, excluding data-deficient species. Numbers on the right indicate the number of modern species assessed by the IUCN for each taxonomic group. Asterisks indicate taxonomic groups with more than 50% of assessed species considered data deficient. Line segments indicate upper and lower estimates of the fraction of endangered species if all data-deficient species were classified as endangered or not endangered, respectively. (Data from Dulvy et al. 2003, 2009; Carlton et al. 1999 and IUCN 2011. After Harnik et al. 2012.)

89% from their historical baseline when they had reached their low point in abundance (see Figure 20.5), usually sometime in the early to mid-twentieth century. Only 40 case studies reported some population recovery, especially among marine mammals and birds, but those recoveries only reduced the overall depletion to 84% of historical abundance levels. Finally, since 1500, at least 20 marine species became globally extinct (see Figure 8.16), including the Caribbean monk seal, Steller’s sea cow, and the great auk (Dulvy et al. 2009), and Dulvy and colleagues (2003)

reported 133 marine extirpations and extinctions on local or regional scales during the nineteenth and twentieth centuries. Some of these extirpations have been disputed (del Monte-Luna et al. 2007), but many have been confirmed, and the list continues to grow (Dulvy et al. 2009; Ferretti et al. 2010).

These syntheses show that many valuable species were severely depleted, locally extirpated, or globally extinct long before the mid-twentieth century, when modern management, monitoring, and ecological studies emerged. For

most of the recorded depletions and extinctions, overexploitation was the primary driver, and habitat loss the second most important factor, followed by pollution (Dulvy et al. 2003; Kappel 2005; Lotze et al. 2006; Harnik et al. 2012).

Habitat alterations

With increasing human settlements along coasts worldwide, human activities started to affect coastal and marine habitats. These habitats included wetlands, such as mangrove forests and salt marshes; submerged vegetation, such as seagrass beds and kelp forests; and reef systems built by corals and oysters. Some of these habitats were affected by direct exploitation. Oysters, for example, were harvested for food, but their reefs were also mined for shells, and mangroves were used for firewood and building supplies. Other habitats were affected by increasing sedimentation, as were many coral reefs and seagrass beds in the course of land clearing after European settlement, as well as by nutrient loading from agriculture, sewage, and industrial activities. Still others were directly transformed into farm or settlement land, as were many salt marshes, or destroyed by dredging and trawling. Like changes in species abundances, many of these habitat transformations and losses occurred much earlier than previously thought: many of them can be traced back to the beginning of colonial times in the New World and the beginning of a market economy in Europe (see Figures 8.14A and 8.15).

In estuaries and coastal seas, oyster reefs were already heavily affected during Roman times in the Adriatic Sea and in the course of commercial and industrial exploitation in the eighteenth and nineteenth centuries in Europe, North America, and Australia (see Figure 8.15; Kirby 2004; Lotze et al. 2006). For example, about 111,600 ha of natural oyster bar habitat originally existed on the Maryland side of Chesapeake Bay, but it declined by more than 50% from 1907 to 1982, with localized losses of up to 95% (Rothschild et al. 1994). Seagrass beds and wetlands, including salt marshes and mangroves, had already severely declined before 1900, and they continued to decline during the first half of the twentieth century, but trends have stabilized since 1950 with increasing habitat protection. In Europe, for example, many regions lost more than 50% of their original wetland and seagrass area, with highs of more than 80% in several regions (Airoldi and Beck 2007). Across six U.S. estuaries, average wetland loss amounted to 55% of the historical area, with a peak of 94% loss in San Francisco Bay (Lotze 2010). Other submerged vegetation, such as rockweeds and kelps, were less affected, although strong declines have occurred in some regions, such as the Adriatic Sea (Airoldi and Beck 2007; Lotze et al. 2011a). In coral reef ecosystems, some effects on habitat-building corals occurred before European colonization, but the strongest changes in corals, suspension feeders, and seagrasses were seen during the second half of the twentieth century (see Figure 8.14A; Pandolfi et al. 2003).

Many coastal habitats provide important nursery, breeding, staging, and foraging grounds as well as shelter from predation for a wide range of dependent animals (Beck et al. 2001). Moreover, coastal habitats are important natural filters and buffer zones between land and sea because they retain and cycle nutrients, sediments, and organic matter (Costanza et al. 1997). Their loss has therefore also contributed to the degradation of water quality in many coastal ecosystems.

Changes in water quality

Around the world, large-scale deforestation and land clearing for agriculture and human settlement has increased erosion and sediment loading of rivers, estuaries, and coastal waters (see Figure 8.15), affecting coastal environments such as Chesapeake Bay (see Figure 8.4B; Cooper and Brush 1993) and the inner Great Barrier Reef (McCulloch et al. 2003). Since many nutrients, including nitrogen and phosphorus, are bound to sediments, past sediment runoff has also increased nutrient loading of coastal waters, which in turn has enhanced primary productivity (see Figure 8.15; Lotze et al. 2006). Over time, other human activities have further enhanced the nutrient loading and chemical pollution of coastal waters, including sewage disposal, municipal and industrial wastewater discharges, and runoff from agriculture and aquaculture operations. Nutrient loading can cause eutrophication, which has several negative consequences (Cloern 2001), including increased phytoplankton productivity and shifts in species composition that favor pelagic over benthic species (e.g., centric over pennate diatoms; see Figure 8.4B) as well as fast-growing species that can form nuisance or harmful algal blooms (see Figure 8.10A). Increased pelagic productivity also enhances water turbidity and shading, commonly leading to a decline in seagrasses and other benthic vegetation. Such changes in the abundance of benthic vegetation over time can be measured in sediment cores by the presence of epiphytic diatoms, such as *Cocconeis* species, which decline with their hosts (see Figure 8.15). Among benthic species, there is often a shift to fast-growing ephemeral species, such as sea lettuce or filamentous algae, which can form benthic algal blooms that overgrow and smother seagrasses and other long-lived, slow-growing perennial benthic vegetation (Cloern 2001). Finally, the decomposition of this enhanced annual algal biomass by bacteria uses up large amounts of oxygen, causing hypoxia or anoxia (see Figure 8.4B). As Figure 8.15 indicates, all these measures of eutrophication increased by one to three orders of magnitude in estuaries and coastal seas in the twentieth century, especially since 1950 (Lotze et al. 2006, 2011a; Lotze 2010). Moreover, around the world, the number and extent of harmful algal blooms and oxygen-depleted dead zones has been increasing over the past decades (Gilbert et al. 2005; Diaz and Rosenberg 2008).

In addition to nutrient loading, many organic and inorganic chemicals have been added to the environment by a range of human activities, especially since the Industrial Revolution in the eighteenth and nineteenth centuries and increasingly in the twentieth century (Lotze and Milewski 2002; Lotze 2010). A range of organic substances, such as wood wastes and sawdust from lumber, pulp, and paper mills, offal from fish processing plants and the canning industry, and uneaten food and feces from aquaculture operations, have enhanced decomposition and oxygen depletion. In addition, a wide range of chemicals have been released from various industries (e.g., textiles, pulp and paper, oil, coal, and gas, mining, and ship building), including heavy metals, polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and other persistent pollutants that accumulate in the sediment and in animal tissues, causing long-term effects on health and survival. Furthermore, agriculture, forestry, and aquaculture operations have increasingly used chemicals such as pesticides (e.g., DDT) and antibiotics to enhance production and have inadvertently released them into the environment, where they cause harm to other species, from acute toxicity to chronic illnesses. For example, a wide range of coastal birds, including bald eagles and ospreys, were heavily affected by DDT in the 1960s and 1970s (Lotze 2010).

Species invasions

Throughout history, people have moved around the world and settled new areas, often bringing seeds, plants, and animals from their former homes. They have also traded goods, thereby exchanging and intentionally introducing new species. Most of the introduced species were terrestrial plants and animals; however, people also transported marine species, either intentionally, for food, aquaculture, aquarium display, or angling, or unintentionally, when species became attached to their boats or to aquaculture organisms or survived in a ship's ballast water. The first identified marine invasion occurred before AD 1245, when Norse voyagers brought the soft-shelled clam (*Mya arenaria*) from North America into the Baltic and North Seas (Petersen et al. 1992). In the following centuries, marine species invasions gradually increased with the expansion of global navigation, trade, and commerce (see Figure 8.15; Ruiz et al. 1997; Lotze et al. 2006). For example, the number of recorded marine and brackish-water invasions in Chesapeake Bay increased from zero in 1609 to 4 in 1739 and to 150 in 1999 (Fofonoff et al. 2003), and invasions in San Francisco Bay increased from the first, recorded in 1853, to 164 by 1995 (Cohen and Carlton 1998). Most of these invaders have been invertebrates, plants, and algae, but some have been viruses, fish, or mammals. Such exotic species change local and regional biodiversity both directly, by enhancing species richness, and indirectly, by affecting native species through biological interactions and thereby altering ecosystem structure and functioning (Steneck and Carlton 2001; Byrnes et al. 2007).

Changes in ecosystem structure, functioning, and services

Together, the changes in species occurrences and abundances, habitats, water quality, and invasions described in this section have changed the overall structure of marine ecosystems and with it, their functioning and services. Generally, species extinctions have reduced and species invasions have increased local species richness and diversity. In many coastal ecosystems, invaders actually outnumber extirpated species (Lotze et al. 2005, 2006; Byrnes et al. 2007). However, since many native species have been depleted by more than 90% of their historical abundance, they may be considered ecologically extinct, meaning that they no longer fulfill their ecological role, which would strongly reduce the functional species diversity in these ecosystems. Moreover, depletions and extinctions have occurred predominantly among large marine mammals, birds, reptiles, and fishes, whereas invaders mostly consist of smaller invertebrates, plants and microscopic algae, protozoans, viruses, and bacteria (Lotze et al. 2006). This taxonomic mismatch between species losses and gains has shifted species compositions away from larger, long-lived, slow-growing, and late-maturing species toward smaller, fast-growing, and high-turnover species (Byrnes et al. 2007).

These changes in the occurrence and abundance of species have not only altered the species composition of food webs and ecosystems, but also the abundances of functional groups. For example, entire guilds of large and small carnivores, herbivores, habitat builders, and species with filtration and buffering capacity have been depleted (see Figure 8.14A; Pandolfi et al. 2003; Lotze et al. 2006). This observation suggests that coastal ecosystems have lost a large amount of top-down control, while increasing nutrient loads have enhanced bottom-up control. Changing the strength of these basic ecological drivers affects species abundances and interactions as well as ecosystem productivity and functioning (Worm et al. 2002; Worm and Duffy 2003). For example, the loss of top predators has released several prey species and competitors from predation pressure, sometimes causing trophic cascades, whereas in other cases recovery of top predators may be limited by the depletion of their prey (Baum and Worm 2009; Ferretti et al. 2010; Altieri et al. 2012). Moreover, the loss of essential habitat and the degradation of water quality may have altered the carrying capacity for many species. Thus the composition and dynamics of coastal ecosystems have profoundly changed, with potentially severe consequences for the maintenance of their diversity, productivity, and stability.

One example of how historical species depletions and losses have affected food web structure and stability comes from a study of the Adriatic Sea. Lotze and colleagues (2011a) found two major changes over time: first, as species and groups at high trophic levels were overexploited, the food web became "shorter" and "fatter," and second, the food web structure became more simplified as the spe-

cies and groups became less connected and the network less complex. Moreover, the high number of species losses and ecological extinctions made the food web more vulnerable to further, secondary extinctions, suggesting that the Adriatic ecosystem has become less resilient. Similar overexploitation of high trophic levels and simplification of food web structure over time have also been found by comparing network models (based on presence/absence of species) with mass balance models (based on biomass and energy flow) for the Adriatic and Catalan Seas in the 1970s and 1990s, respectively (Coll et al. 2008). For comparison, the basic structure and underlying principles of food web assembly do not appear to have changed between the Cambrian and modern times (Dunne et al. 2008).

These documented changes in the structure and functioning of ecosystems further translate into changes in the ecosystem services that benefit people. Over historical time scales, the depletion and loss of species in coastal ecosystems has affected their provision of viable fisheries and seafood, spawning and nursery habitats to sustain marine resources, and filtration capacity to maintain water quality. The losses of these functions and services have brought rising health risks and costs to society, such as increasing beach closures, harmful algal blooms, fish kills, shellfish closures, oxygen depletion, coastal flooding, and species invasions (Worm et al. 2006; Lotze 2010).

Overarching patterns

The case studies and syntheses presented in this section illustrate the substantial progress made in marine historical ecology over the past two decades. Historical changes in marine ecosystems have now been traced back hundreds to thousands of years for a variety of species and regions. Although it is not always easy to establish historical baselines, which are continually shifting over time and with environmental changes, they can nevertheless be informative in placing more recent changes in perspective. In the rest of this section we summarize overarching patterns in the magnitude, direction, and rate of historical changes as well as their underlying drivers.

- *Magnitude of historical changes* Across 256 case studies that quantify population changes for different species and regions using different approaches, the average magnitude of historical depletion was 89% (± 1 SE), with individual estimates ranging between 11% and 100% (Lotze and Worm 2009). This magnitude is comparable to that of estimates of more recent depletions across 232 fish stocks made using modern stock assessment methods; those stocks declined on average by 83% from their maximum breeding population size over 25 (range 10–73) years (Hutchings and Reynolds 2004). Similarly, an analysis of aggregated catch, abundance, and size data for exploited marine mammal populations suggested an average decline of 81% for the great whales and 76% across all species since the beginning of exploitation (Christensen 2006). These figures suggest that
- an order-of-magnitude decline may be a general rule for exploited marine species. The late Ransom Myers termed this the “factor of ten hypothesis” (Myers and Worm 2005).
- *Spatial expansion* Around the world, most people first exploited local, nearshore, easily accessible species. After those species were depleted, a common response was to move farther offshore toward previously unexploited waters where the same or similar species were still in high abundance. Such spatial expansions from the coasts to increasingly offshore resources can be seen in archaeological records from California (Rick et al. 2008), Mexico (Kennett et al. 2008), southern Taiwan (Kuang-Ti 2001), the Wadden Sea (Lotze 2007), and the North Atlantic (Perdikaris and McGovern 2008). In historical times, similar spatial expansions occurred in the history of whaling (Christensen 2006; Josephson et al. 2008) and the North Atlantic groundfishery (Lear 1998; Lotze and Milewski 2004; Rose 2004; Bolster 2012). In some cases, spatial expansions occurred along shorelines, as in the expansion of oyster harvesting along the coasts of North America and Australia in the nineteenth century (Kirby 2004) or the more recent expansion of sea urchin and sea cucumber exploitation from Asia into a global industry (Berkes et al. 2006; Anderson et al. 2011). On a global scale, marine exploitation began in estuaries and coastal waters several hundred to thousands of years ago before spreading across the continental shelves in the nineteenth and twentieth centuries, moving into the open ocean in the mid-twentieth century, and recently expanding into the deep sea (Myers and Worm 2003; Pauly et al. 2003; Devine et al. 2006; Lotze and Worm 2009). Therefore, the historical changes that have occurred in estuaries and coastal seas might provide insight into some of the present and potential future changes offshore.
- *Temporal acceleration* Over the course of history, one factor driving accelerating marine exploitation was the growing human population and its demands for food, fuel, fashion, and other products. When local resources became depleted, another common response was to advance technology to catch more (Bolster 2012). These advances included not only shifts in gear (e.g., from a single to multiple to many thousands of hooks) and fishing vessels (e.g., from sail to steam power to diesel engines and increasingly larger and faster ships), but also the introduction of storage and freezing capacity on board and, more recently, fish-finding gear. Thus historical changes in coastal ecosystems caused by relatively small human populations with simple gear and small boats unfolded over hundreds to thousands of years (Pandolfi et al. 2003; Lotze et al. 2006), while modern changes have unfolded increasingly faster on continental shelves over the last 100 to 200 years (Christensen et al. 2003; Rosenberg et al. 2005), in the open ocean over the last 50 years (Myers and Worm 2003), and in

the deep sea over the last 20 years (Devine et al. 2006). Unfortunately, global fisheries catches have stagnated and decreased recently, indicating a limit to marine exploitation on a global scale (Pauly et al. 2003).

- *Serial exploitation* Human hunters and fishermen have usually preferred resources that are relatively easy to find and catch and that are large, nutritious, tasty, or otherwise highly valued. Thus, in the history of whaling, humans first hunted slow-swimming coastal species that would float when dead, which included the gray, right, and bowhead whales. When these species were depleted and technology advanced, whaling expanded to increasingly faster and more offshore species and, over time, from larger to smaller whale species (Christensen 2006). Such serial exploitation patterns have also been observed for other species. In the Florida Keys, for example, exploitation of highly valued turtles and sponges quickly moved to other species as depletions occurred (McClenachan and Kittinger 2012). The sequential shift from large, high-trophic level species to increasingly smaller and lower-trophic level species has been termed “fishing down the food web” (Pauly et al. 1998), although the serial replacement of one species by another does not necessarily represent a decline in average trophic level and can also reflect “fishing through the food web” (Essington et al. 2006). Shifts from species of high to low monetary value or, within a species, from larger to smaller individuals have also been observed (Anderson et al. 2011).
- *Cumulative drivers of change* Although long-term fluctuations in population abundance or ecosystem components can be related to climatic variation (Ravier and Fromentin 2004), rapid historical population declines have often been attributed to human impacts (Rose 2004; Causey et al. 2005). Among the range of human pressures, overexploitation generally ranked first in causing population depletions or extinctions, followed by habitat loss and pollution (Dulvy et al. 2003; Kappel 2005; Lotze et al. 2006; Harnik et al. 2012). All of these factors expanded spatially and accelerated temporally over time and, importantly, they do not act in isolation. In estuaries and coastal seas, for example, 45% of species depletions and 42% of extinctions were caused by cumulative human impacts, in most cases a combination of exploitation and habitat loss (Lotze et al. 2006). For population recoveries, cumulative impacts were even more important, with only 22% of recoveries resulting from the reduction of a single human impact, mostly exploitation, and 78% of recoveries resulting from the mitigation of at least two impacts, mostly habitat protection and restricted exploitation but also pollution mitigation (Lotze et al. 2006).
- *Recovery* Although most studies of historical changes document the depletion and degradation of marine ecosystems, several indicate some recent recovery due to increasing management and conservation efforts in

the twentieth century (Lotze et al. 2006; Kittinger et al. 2011). A recent review suggests that 10–50% of depleted populations and ecosystems show some recovery, but so far rarely to former levels of abundance (Lotze et al. 2011b). For example, among 256 populations for which historical depletion was estimated, only 40 populations experienced some recovery in the twentieth century, mostly marine mammals and some birds, and on average only to 13–39% of their former abundance (Lotze and Worm 2009). The protection of mammals and birds has been increasingly supported by the public and society at large, whereas the conservation of exploited fish and invertebrates is still at the beginning (Worm et al. 2009).

How the Past Can Inform the Future

Marine historical ecology is invaluable for understanding the past, but how can the past inform the future? If we want to predict future trends in marine populations and ocean ecosystems, we need to understand the long-term trajectories of past and current changes and their underlying drivers. If we want to develop better management plans and conservation targets, we need to incorporate historical baselines and learn from past management successes and failures. And if we want to change the way people view and societies interact with the ocean in the future, we can use the past to show how human activities, choices, and cultural values have influenced the ocean and how this could be changed in the future. In this section we outline examples of how the past can inform the future in science and ecology, management and conservation, and teaching and communication.

Science and ecology

One important task for scientists is to provide a range of scenarios for what is likely to happen in the future. This task is critical not only for science itself, but also for informing politics, management, and the public. In marine ecology, forecasting of future trends includes trends in species abundance and ecosystem structure, the likelihood of recovery or further decline, and how different environmental changes or management scenarios will resolve. In order to do this, we need to know long-term trends in the past and understand what drivers were responsible for the observed changes. Importantly, modern observations and recent ecological data rarely represent the full range of variation in ocean ecosystems. For this reason, one of the most fundamental roles of marine historical ecology is to provide a more accurate, long-term view of natural and human-induced variation in marine species and ecosystems.

PROJECTING FUTURE STATES OF MARINE POPULATIONS AND ECOSYSTEMS Understanding the magnitude and direction of past changes can provide a bounds for what we can reasonably expect in the future. Projections based on the past can be done conceptually or in a

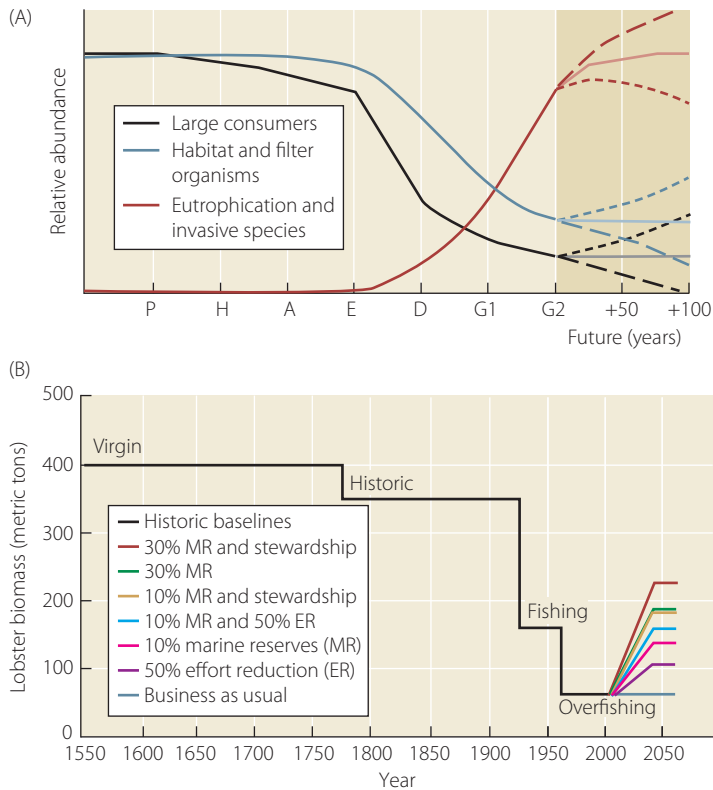


Figure 8.17 Projecting future states of marine populations and ecosystems. (A) Historical trajectories of change for large consumers, habitat-building and filter-feeding organisms, and eutrophication and invasive species in estuaries and coastal seas can be used to project potential trends for these structural and functional components of ecosystems into the future. Future scenarios depict stabilizing (solid lines), improving (short dashed), or worsening (long dashed lines) trends. See Table 8.3 for cultural period abbreviations and definitions. (B) Reconstruction of historical changes in the abundance of lobsters in the Juan Fernández Archipelago, Chile, and predictions of future abundance based on different management and conservation scenarios (MR = marine reserves, ER = fishing effort reduction). (A after Lotze et al. 2006; B after Eddy et al. 2010.)

quantitative modeling framework. For example, estuarine ecosystems have experienced strong declines in large consumers, habitat-building species, and filter feeders since prehistoric times, concurrently with increases in invasive species and levels of eutrophication (Figure 8.17A). Lotze and colleagues (2006) used the magnitude and timing of these past trajectories to project possible future scenarios in which conditions may stabilize, improve, or worsen. Using a quantitative modeling approach, Eddy and colleagues (2010) reconstructed historical changes in the abundance of lobsters (*Jasus frontalis*) in Chile over the past 400 years based on archival documents, fishermen's ecological knowledge, and underwater observations (Figure 8.17B). Modern lobster biomass was 85% less than the "pristine" abundances present from 1550 to 1750. The authors then modeled a range of future scenarios that explored the effects of different types and intensities of management efforts, including marine reserves, reduction of fishing effort, and stewardship. The results suggested that increased stewardship of catch coupled with 30% area closures would allow lobster biomass to recover to about 50% of historical levels within 40 years (Eddy et al. 2010). Such uses of historical data can help managers understand what goals different management efforts can achieve and how future expectations compare to historical baselines.

DISTINGUISHING NATURAL CHANGES FROM ANTHROPOGENIC CHANGES Historical studies describe a range of past population sizes and ecological conditions that can

help us to distinguish natural variation from anthropogenic change, interpret whether modern and expected future changes are within previously experienced bounds, and predict future responses to different stressors. For example, Rose (2004) reconstructed 500 years of cod biomass in Newfoundland using a surplus production model and analyzed the drivers behind the observed changes. The results suggest that the decline of cod biomass during the Little Ice Age (1800–1880) was caused by a climate-driven drop in productivity, that overfishing caused the decline in the 1960s, and that the collapse of biomass in the late 1980s was caused by both climate and fishing. This study suggests that future changes in fishing pressure and climate will interact in altering cod abundance. In the Caribbean, the fossil record revealed that coral reefs were dominated by two species of acroporids over the last 200,000 years (Pandolfi and Jackson 2001). In the early 1900s, however, the relative abundance of these corals declined concurrently with land clearing for agriculture, and a further sharp decline in the 1970s was related to increases in disease events and bleaching (Figure 8.18; Greenstein et al. 1998; Cramer et al. 2012). These recent declines are outside the range of natural variation observed in the fossil record and suggest that acroporid corals will continue to decline if these impacts are not mitigated. Finally, an analysis of extinction rates and drivers in the ocean over the last 500 million years revealed that naturally caused acidification, anoxia, warming, cooling, and habitat loss have driven marine extinctions

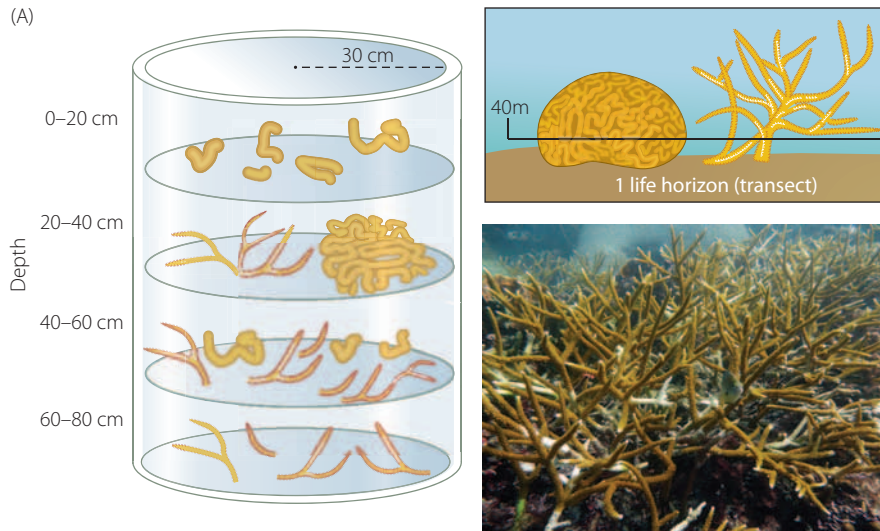
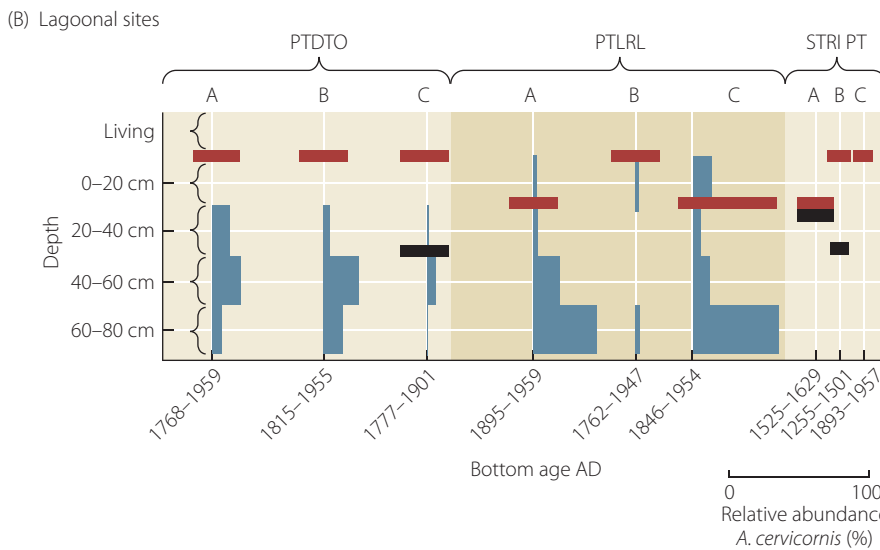


Figure 8.18 Distinguishing natural from anthropogenic changes. (A) Long-term data taken from “death horizons” in coral reef rubble (left) and compared with the modern “life horizon” (right top) from the Bocas del Toro region of western Caribbean Panama suggests that *Acropora cervicornis* was a common reef-building coral prior to the second half of the twentieth century. (B) The relative abundance of *A. cervicornis* declined from an average of 28% to 6% of the total community prior to 1960, after which it continued to decline to 0% of the coral community living today. A, B, and C depict three sampling replicates at each site. The black horizontal bars indicate boundaries between pre- and post-1900; the red horizontal bars indicate boundaries between pre- and post-1960. Blue bars represent the percentage of *A. cervicornis* found in the death horizons. (PTDTO = Punta Donato; PTLRL = Punta Laurel; STRI PT = Smithsonian Tropical Research Institute Point; After Cramer et al. 2012. Photo courtesy of Katie Cramer.)



in the fossil record, whereas the effects of human-caused overexploitation, habitat loss, and pollution are exclusive to the last 10,000 years (Table 8.4; Harnik et al. 2012). Today many of these drivers are present simultaneously, and this situation will probably intensify in the future with the effects of a growing human population and climate change. This will place an increasing number of marine species at risk of extinction, including those taxonomic groups strongly affected by human pressures in historical times (e.g., marine mammals, birds, and fish) as well as many invertebrates and plants (see Figure 8.16; Harnik et al. 2012). Such analyses provide a window into how specific organisms will respond to projected future environmental conditions and help to refine modern analyses of extinction risk in the world’s oceans.

REVEALING THE IMPORTANCE OF DIFFERENT HUMAN DRIVERS Defining which human activities have been most important in causing past changes in marine species

and ecosystems is crucial to predicting future changes in response to different perturbations. Historical studies indicate that overexploitation has frequently been the first and most severe human impact on marine populations, followed by habitat loss, pollution, and more recently, invasive species and climate change (Jackson et al. 2001; Lotze et al. 2006; Rick and Erlandson 2008). Over time, the range of human activities has multiplied, increasingly causing cumulative effects. For example, in the Bay of Fundy, the combination of fishing and river damming reduced wild salmon populations in the nineteenth century, while pollution and interactions with escaped farmed salmon caused further declines in the twentieth century (Lotze and Milewski 2004). In modern coral reefs, climate change and nutrient pollution often have the most visible impacts, as they result in a loss of live coral due to bleaching and overgrowth by macroalgae. Yet historical analyses have revealed that overfishing has been a significant long-term driver of change, as losses of large fish and other

TABLE 8.4 Past drivers of extinction in the ocean and current threats

TIME PERIOD ^a	DRIVERS AND THREATS ^b						
	ACIDIFICATION ^c	ANOXIA ^d	WARMING	COOLING	HABITAT LOSS ^e	OVER-EXPLOITATION	POLLUTION
Ordovician–Silurian (~444 Ma)		○	○	●	●		
Late Devonian* (Frasnian–Famennian; ~374 Ma)		●	○	●	●		
End Permian* (~251 Ma)	●	●	●	○	●		
Early Triassic (~245 Ma)	○	●	○	○	○		
Triassic–Jurassic* (~202 Ma)	●		●				
Early Jurassic* (Pliensbachian–Toarcian; ~183 Ma)	●	●	●	●			
Aptian–Albian (~112 Ma)	●	●		○	○		
Cenomanian–Turonian (~93.5 Ma)	●	●	○				
Cretaceous–Paleogene (~65.5 Ma)	●		○	●	○		
Paleocene–Eocene Thermal Maximum* (~56 Ma)	●	○	●				
Eocene–Oligocene (~34 Ma)				●	○		
Mid-Miocene Climatic Optimum (~14.7 Ma)			●		○		
Historical (~10 Ka)			○		●	●	○
Modern	●	●	●		●	●	●

Source: Hallam 2004; Harnik et al. 2012.

^aTime periods in boldface indicate mass extinctions; asterisks indicate global reef crises.

^bSolid circles highlight confident and open circles less confident drivers.

^cCauses of acidification include volcanism, bolide impacts, and release of methane clathrates in the past and burning of fossil fuels in the modern time period.

^dCauses of anoxia include warming, eutrophication, and ocean stratification in the past and eutrophication in the modern time period.

^eCauses of habitat loss include sea-level fall in the past and habitat degradation and coastal development in the modern time period.

herbivores over past centuries have reduced grazing pressure, establishing conditions in which macroalgae thrive (Pandolfi et al. 2005). Therefore, reefs that survive into the future will probably be characterized by robust populations of fish and other megafauna that can mediate the effects of eutrophication and climate change (Newman et al. 2006). Reconstructing the sequence and effects of multiple human impacts helps us to understand the timing of change and provides insight into how ecosystems may respond to different drivers in the future.

PREDICTING RECOVERY POTENTIAL Understanding and predicting the potential of species and ecosystems to recover from human or natural disturbance is one of the

most important tasks facing marine conservation biology today. Marine historical ecology provides critical information for assessing the success of recovery, particularly the degree, magnitude, and timeline of recovery (Lotze et al. 2011b). Several historical studies demonstrate that species recovery is possible when certain drivers of change are mitigated. For example, in the main Hawaiian Islands, an expansion of agriculture between AD 400 and AD 1820 reduced fishing pressure on reefs surrounding the islands, and archaeological evidence suggests that the average sizes of parrotfish (*Scarus* spp.) and intertidal limpets (*Cellana* spp.) increased during this time (Kittinger et al. 2011). More recently, some level of recovery has occurred in many marine vertebrate populations in response to the

reduction or cessation of exploitation, habitat protection, or pollution controls over the last century (see Figures 8.9, 8.12, and 8.15). Available summaries suggest that 10–50% of formerly depleted populations among marine mammals, birds, reptiles, and fish have recovered to some degree, albeit rarely to historical abundance levels (Lotze et al. 2011b). Here, historical baselines are essential to quantify the magnitude of recovery and judge recovery success (see Figures 8.13 and 8.17). Moreover, trajectories of historical depletion are needed to assess when the low point in abundance occurred and how long the recovery process has taken. The same is true for assessing and predicting the recovery of ecosystems (Lotze et al. 2011b).

Despite these instances of recovery, history also warns us that barriers to recovery exist. In spite of efforts to increase their abundances, many populations have remained depressed or continue to decline (Lotze et al. 2011b). Only 40 of 256 populations that showed historical population declines also showed some degree of recovery (Lotze and Worm 2009), and across 232 depleted fish stocks, 40% showed no signs of recovery 15 years after their collapse (Hutchings and Reynolds 2004). Lack of recovery can be due to the magnitude of depletion, with larger depletions reducing recovery potential; the species' life history, with fast-growing species recovering more rapidly than slow-growing ones; or the time available for recovery, with long-lived species needing decades or centuries for recovery (Hutchings and Reynolds 2004; Lotze et al. 2011b). Such information on historical recovery trends can be extremely helpful in predicting the future recovery potential of different species. In some cases, shifts in the abundances of other species or alteration of ecological conditions may prevent recovery. For example, in the Northwestern Hawaiian Islands, populations of the Hawaiian monk seal (*Monachus schauinslandi*) continue to decline despite protection under the U.S. Marine Mammal Protection Act, under the U.S. Endangered Species Act, and in the Papahānaumokuākea National Marine Monument. In this case, increases in the abundances of predatory fish and sharks may have resulted in competitive exclusion and higher mortality rates for juvenile seals, inhibiting recovery (Gerber et al. 2011). On the Scotian shelf, populations of Atlantic cod failed to recover after the Canadian government closed the fishery in 1992. Frank and colleagues (2005, 2011) found that planktivorous fish that prey on cod larvae increased and may have stalled the recovery of cod even after fishing was halted. Recent signs suggest that planktivorous fish may have stabilized or declined, possibly enhancing the recovery prospects of cod (Frank et al. 2011). These studies provide warnings that the trajectory of recovery may differ from that of decline.

Management and conservation

Historical baselines and trajectories of change are necessary to assess extinction risk for long-lived marine

species, to determine when endangered species can be considered recovered, to improve fisheries management and assessment, to set targets for ecosystem restoration, and to evaluate past management successes and failures. Thus historical information is increasingly used by conservation organizations and management bodies to set meaningful management targets and conservation goals (McClenachan et al. 2012).

EXTINCTION RISK ASSESSMENT Rapid population decreases are often a warning of extinction risk. The International Union for Conservation of Nature's IUCN Red List of Threatened Species considers species that have declined by more than 50% over three generations to be vulnerable to extinction. For long-lived species, that three-generation period can exceed 100 years, so the use of historical data is important, particularly for species that have a long history of exploitation. For example, the hawksbill turtle (*Eretmochelys imbricata*) has a generation time of over 30 years and has been intensively exploited for centuries. Modern data on its population abundance derived from beach counts of nesting females have been collected over the last few decades, but that is a much shorter time than necessary to evaluate trends over three generations. To address this problem, the 2008 IUCN Red List assessment sought customs data from the shell trade since the mid-nineteenth century and other historical information over the last few centuries (Mortimer and Donnelly 2008). Integrating this information with recent monitoring data allowed for a historically robust estimation of long-term population change and extinction risk, which suggested that hawksbill turtles have declined by more than 80% over three generations, and they are therefore listed as "critically endangered" (**Figure 8.19A**). Strikingly, data over the past 30 years tell a very different story. They suggest population increases, with similar patterns for green turtles (**Figure 8.19B**), indicating the need for long-term data to accurately describe long-term trends. Similar evaluations comparing historical with recent abundance trends have helped researchers to assess the conservation or management status of other large marine species (McClenachan et al. 2012). Large sharks, for example, have a history of exploitation spanning several centuries, but a chronic lack of data has hindered effective population assessment and management. In the Mediterranean, 22 nations comanage fisheries, but statistics are temporally and spatially restricted. In 2007 an IUCN regional assessment synthesized many of these sources, but few time series of longer than 50 years were available, and quantitative data existed for only a few species (Cavanagh and Gibson 2007). Thus Ferretti and colleagues (2008) compiled historical data on shark abundance, extending the temporal window of observation to 180 years and providing relevant information on trends in abundance for four large shark species previously classified as "data deficient" (**Table 8.5**). These examples

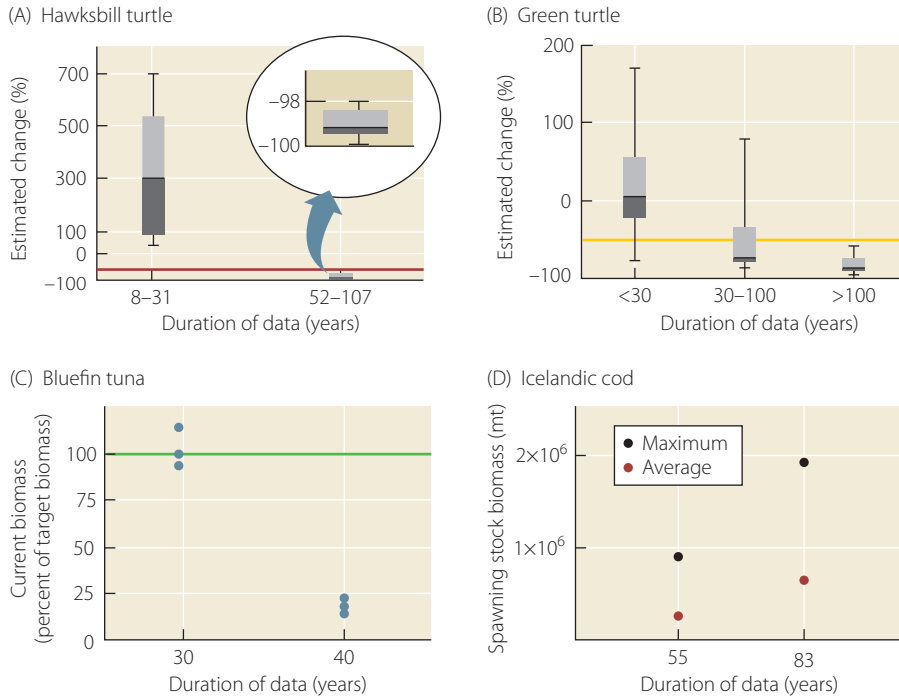


Figure 8.19 The use of historical data affects extinction risk assessments and fisheries reference points. (A) Population trends for hawksbill turtles from nesting beaches for which available data span 8–31 years and 52–107 years suggest median population changes of +308% (64 to 705%) and -96% (-80 to -99%), respectively. Populations from beaches with 52–107 years of data would be assessed as critically endangered by the IUCN (red line), whereas those from beaches with 8–31 years of data would not. (B) Population trends for green turtles from nesting beaches for which available data span less than 30, 30–100, and over 100 years suggest median population changes of +7% (range -77 to +170%), -73% (-88 to +75%), and -83% (-65 to -93%), respectively. Populations from beaches with data for more than 30 years would be assessed as endangered by the IUCN (yellow line), whereas those from beaches with data for less than 30 years would not. (C) The current stock size for western bluefin tuna exceeds the target biomass (green line) when shorter time series of 30 years are used, but constitutes only 15% of target biomass when 40 years of data are used. (D) Icelandic cod time series over the last 55 years suggested that average and maximum spawning stock biomasses (SSB) were less than half than those estimated when 83 years of data, which includes peaks in the 1920s and 1930s, were considered. (After McClenachan et al. 2012).

illustrate how historical information allows more accurate assessments of extinction risk in marine species.

ENDANGERED SPECIES RECOVERY Historical data are also necessary to assess when a formerly depleted species whose abundance is increasing can be considered recovered and how that recovery compares with its past population sizes. Such assessments are necessary to determine whether a species needs further protection or could potentially be harvested again. For example, following the end of whaling, several whale populations have shown signs of recovery. The Antarctic blue whale (*Balaenoptera musculus intermedia*), whose populations reached a low of 360 individuals, has increased by about 7% per year since 1974. A variety of data over the last century,

including logbook data from the early 1900s and sighting surveys since the late 1960s, suggested that despite this rapid increase, the population is still below 1% of its estimated historical abundance of over 200,000 whales and thus requires further protection (Branch et al. 2004). Similar reconstructions of historical population abundances were used to place the recent recoveries of the southern right whale (see Figure 8.9A) and Pacific gray whale (see Figure 8.12) in a long-term context. Likewise, green turtle (*Chelonia mydas*) populations in several locations have responded positively to protection efforts and have increased over the last three decades (McClenachan et al. 2006; Chaloupka et al. 2008). In the Northwestern Hawaiian Islands, one population on the French Frigate Shoals has increased by over 400% since 1973. As a

TABLE 8.5 Estimates of long-term change in shark populations in the Mediterranean Sea, with and without historical data

SPECIES		WITHOUT HISTORICAL DATA		WITH HISTORICAL DATA	
SCIENTIFIC NAME	COMMON NAME	BASELINE DATE	TREND	BASELINE DATE	TREND
<i>Prionace glauca</i>	Blue shark	1979	-38.50%	1950	-96.53%
<i>Alopias vulpinus</i>	Common thresher	Absent	Absent	1898	-99.99%
<i>Lamna nasus</i>	Porbeagle	Absent	Absent	1827	-99.99%
<i>Isurus oxyrinchus</i>	Shortfin mako	Absent	Absent	1827	-99.99%
<i>Sphyrna zygaena</i>	Smooth hammerhead	Absent	Absent	1827	-99.99%

Source: Cavanagh and Gibson 2007; Ferretti et al. 2008; McClenachan et al. 2012.

result, the Hawaiian subpopulation has been downlisted to “Least Concern” by the IUCN and is being considered for delisting under the U.S. Endangered Species Act. However, archaeological and historical records suggest that past populations were characterized by more spatial heterogeneity, with nesting sites distributed throughout the main and Northwestern Hawaiian Islands (Kittinger et al. 2013). Eighty percent of former nesting sites present between AD 1250 and 1950 have been eliminated or severely reduced, and recovery on the French Frigate Shoals has not resulted in their recolonization. This example shows that historical biogeographic patterns can provide another important variable for assessing progress toward recovery of endangered species.

FISHERIES MANAGEMENT In fisheries management, historical baselines can be useful for assessing the current status of fish stocks, establishing allowable catch rates, and determining rebuilding targets for overexploited species. Unfortunately, in most cases, fisheries-independent research surveys and catch data used to estimate stock abundance are available only for the past few decades, yet many fish stocks have been exploited for centuries. Here, historical data and analyses can help establish a more accurate view of current stock status and place recent fluctuations in a long-term context, as they have done for the body sizes of cod in the Gulf of Maine over the past 5,000 years (see Figure 8.5C; Jackson et al. 2001) and for cod biomass on the Scotian Shelf in the 1850s (see Figure 8.7B; Rosenberg et al. 2005). Similarly, use of extended time series can demonstrate the degree to which populations fluctuated in the past, which can affect perceptions of modern stock status and rebuilding targets, as they have done for bluefin tuna and Icelandic cod (Figure 8.19C,D; McClenachan et al. 2012). And calculations of pre-exploitation stock size, carrying capacity, or virgin biomass can help to establish more accurate assessments of current compared with past stock abundance (Myers et al. 2001; Myers and Worm 2005), often revealing highly depleted stocks that require management for rebuilding. Data on small-scale fisheries

are often limited, and managers rely on a variety of non-traditional data sources, such as interviews with resource users or logbooks from recreational divers. In these fisheries, marine historical ecology can contribute important knowledge, including changes in fish size (see Figure 8.11; Sáenz-Arroyo et al. 2005; McClenachan 2009) as well as landings and catch-per-unit-effort, stock structure, and ontogenetic shifts (Ames 2004; Lotze and Milewski 2004). Such knowledge can further be used to explore different management outcomes, as discussed for Chilean lobsters (see Figure 8.17B). Historical data can also demonstrate an ecosystem’s capacity to support fisheries. In the Hawaiian Islands, catch reconstructions revealed that high levels of subsistence fisheries were sustained for 400 years prior to European contact, with catch levels higher than today’s (McClenachan and Kittinger 2012).

Fisheries are affected by ecosystem changes, and historical analyses provide essential information on long-term ecological changes that affect fisheries productivity and diversity. In many coastal communities, historical overfishing of marine predators has resulted in the release of their prey, which can create new fisheries beneficial to humans. In the kelp forests of the west coast of North America, for example, sea urchin fisheries developed as the extirpation of sea otters released urchins from predation (Palumbi 2010). Likewise, in the North Atlantic, severe reductions in cod and other large predatory groundfishes resulted in increases in shrimp, lobsters, sea urchins, and other invertebrates that have supported economically valuable fisheries (Worm and Myers 2003; Anderson et al. 2008; Boudreau and Worm 2010). Placing current fisheries in a historical context suggests that the long-term goals of fisheries management should include the restoration of ecosystems with intact food webs, which in time will support more diversified options for fisheries (Steneck et al. 2011).

ECOSYSTEM RESTORATION Restoration ecology has often relied on historical baselines, as it seeks “the return of an ecosystem to a close approximation of its condition prior to disturbance” (National Research Council 1992).

Scaling up from managing species to restoring ecosystems requires a different suite of historical information, including species diversity, community structure, water quality, habitat availability, and the ecosystem functions provided in the past (Lotze et al. 2011b). For example, pollution control in the Thames River in the United Kingdom has allowed the return of more than 110 fish species since the 1960s, and reductions of nutrient loading in the 1970s in Tampa Bay, Florida, increased water clarity, reduced algal blooms, and allowed the recovery of seagrass ecosystems (Cloern 2001). An understanding of historical ecosystem function can provide quantifiable and ecologically meaningful restoration targets. For example, oyster reefs provide up to \$99,000 per hectare per year in value to society through water filtration and denitrification in the surrounding sediments (Grabowski et al. 2012), but over the last 120 years, nearly 90% of oyster reef biomass in U.S. estuaries has been lost (zu Ermgassen et al. 2012). Restoration efforts that aim to return oyster reefs to densities that support their former filtration capacity provide a clear ecological and monetary value that can be used to balance the costs of the restoration effort.

LEARNING FROM PAST MANAGEMENT SUCCESSES AND FAILURES Finally, historical studies demonstrate ways in which societies have managed marine resources in the past and the degree to which those past management measures have been successful (Lotze et al. in press). Throughout history and around the world, people have responded to declines in valuable marine resources by implementing certain management measures, including fishing effort and gear controls, quotas on catches or sizes, temporal or spatial closures, and privatization or governance. Often, however, the chosen measures were overruled, insufficient, or not fully implemented or enforced on the water, and as a result were not able to halt further declines. In many such cases, fisheries instead expanded to other species or less fished regions or enhanced their efficiency through technological advances (Lotze et al. in press). Yet there are also successes in past fisheries management. For example, in ancient Hawaii, high levels of subsistence fishing for coral reef fish and invertebrates were sustained with multiple management measures, including time–area closures, limited entry into the fishery, community-based management, and restrictions on the consumption of vulnerable species such as sharks and turtles. An analysis of sustainability over several centuries demonstrated that social factors strongly influenced the trajectory of fisheries over time and that high levels of fishing can be sustainable if there are multiple management measures in place (McClenachan and Kittinger 2012). Similarly, a combination of multiple management measures has been successful in rebuilding some modern fish stocks in both large- and small-scale fisheries (Worm et al. 2009). Thus past management successes and failures can inform future sustainable management and rebuilding.

Over the last century, many conservation actions have been successful at protecting and restoring populations of a wide range of marine animals (Lotze et al. 2011b). For example, the 1900 U.S. Lacey Act prohibited trade in highly valued bird feathers, and the 1916 Migratory Bird Treaty protected a range of bird species from hunting, egg collection, and nest destruction. These laws were responsible for saving several species of wading birds and seabirds from extinction and allowed widespread increases of bird populations that had been targeted by nineteenth-century hunters. The 1911 International Fur Seal Treaty did the same for populations of fur seals and sea otters that were near extinction at the turn of the twentieth century due to hunting along the west coast of North America. Right, bowhead, and gray whales, whose populations had been hunted for their meat, oil, and baleen since the sixteenth century, received protection from the League of Nations, which banned commercial whaling for these species in the 1930s (Lotze et al. 2011b). These successes can provide a model for specific conservation actions in the future.

Teaching and communication

We began this chapter with a quotation: “Those who cannot remember the past are condemned to repeat it” (Santayana 1905). This quote, of course, does not pertain only to science and scientists, but to all of us. Marine historical ecology provides critical information that can be taught in schools and universities and communicated to the general public with the goal of building a better future for society and the ocean. Public awareness of how people’s choices, cultural values, and actions have influenced the ocean in the past can alter their behavior in the future. Moreover, public engagement is a critical element of translating marine historical ecology into actual conservation and management action.

One important driver of our collective unawareness about the ocean’s history is the loss of living memory. What we perceive as natural today may in fact be very different from what our parents, grandparents, and great-grandparents perceived as natural, as shown by the different sizes of the largest fish and the best day’s catch recalled by young, middle-aged, and old fishermen (see Figures 8.2 and 8.11). This “shifting baselines syndrome” (Pauly 1995) can be used to describe declines in fisheries, species diversity, and other metrics of ocean health, like water quality and ecosystem services, in a way that is accessible to the public. This concept has been communicated to the public across a variety of media types and levels of engagement with issues of marine environmental change. For example, images of trophy fish caught in the Florida Keys over the last 50 were used to help communicate to a scientific audience the results of an analysis that quantified declines in the average size of the largest reef fish (**Figure 8.20A**). These same images were integrated into in a display at the National Museum of Natural History’s Sant Ocean Hall in Washington, DC to illustrate the



(B)

SIZE MATTERS!

BIG FISH were a common catch just 50 years ago. *Not anymore!*

1950s

Where have all the BIG FISH gone?

Overfishing has left few big fish in the ocean.

As the fish we catch have gotten smaller, we've forgotten what a really **big** fish should look like! Our **baseline** of comparison for how big a "big fish" should be has **shifted**.

In the late 1950s, Florida Key's fishermen caught big trophy fish—some over **6 feet long**. Today, despite improvements in fishing technology, most trophy fish caught there measure just a **foot** in length.

Trophy fish—the "big" winners—are getting smaller and smaller.

1957 Early 1980s 2007

Photos like these provide a **historical perspective** to help us understand **changes** in fish size and numbers and what they **should** look like...
 ...so that we know how to restore and sustain them.

YOU CAN HELP!

- **Speak Up!**
 —At the supermarket or restaurant, **ask** if the seafood is sustainable.
Tell them you support fishing practices that ensure there will be plenty of fish for future generations.
- If you go fishing, **follow** the regulated size and catch limits.
- Support **marine protected areas** that help conserve, protect, and enhance biodiversity.

See how scientists use old logs, journals, photos to solve ecological mysteries at <http://sanctuaries.noaa.gov/oceanhall/>.

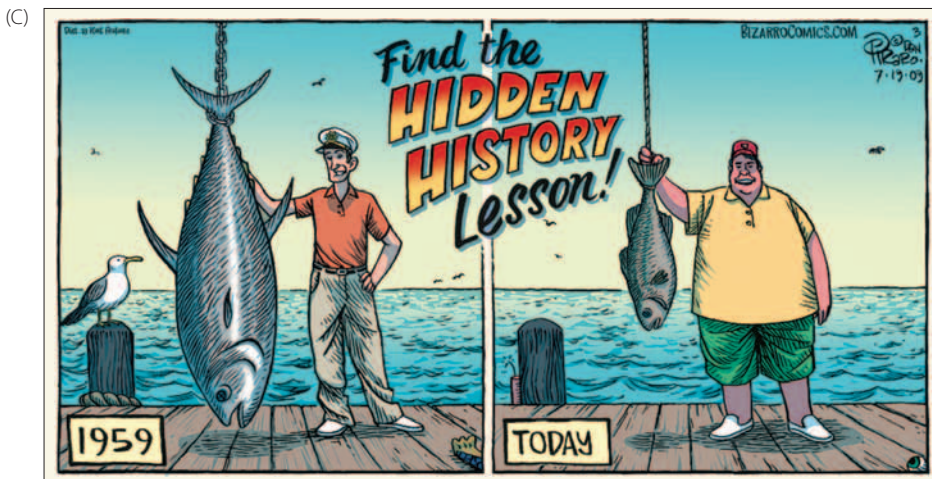


Figure 8.20 Three different ways of communicating the effects of overfishing on the size of fish to the general public: (A) through the scientific literature (from McClenachan 2009); (B) museum exhibit in the National Museum of Natural History's Sant Ocean Hall in Washington DC; (C) through humor, as revealed in a Bizarro comic. (A from McClenachan 2009; B Sant Ocean Hall/ Smithsonian Institution/ NOAA; C © Dan Piraro, bizarrocomics.com).

concept of shifting baselines and the loss of large fish from coral reef environments for a public interested in marine science (Figure 8.20B). And a cartoon with a similar message shows two sizes of “trophy fish” in the 1950s and in 2009 (Figure 8.20C), communicating the idea of shifting baselines (and waistlines) to a wider online audience who might not otherwise be engaged with marine issues. The online site named after Pauly’s concept, “www.shifting-baselines.org,” has also used humor to bring attention to long-term changes in marine ecosystems. For example, a public service announcement communicated concepts of fishing down the food web and the impacts of size selective fisheries on fish populations by parodying anglers who are proud of catching minuscule fish.

Historical ecology can also help communicate tangible conservation and management goals and engage the public to translate them into action. Whereas goals like “improving current conditions” are difficult to envision, information from the past provides the public with a clear vision of the possible states of marine animals and ecosystems that could be achieved if conservation and management actions are successful. For example, the images of large reef fish caught in the 1950s in the Florida Keys (see Figure 8.20A) provide a positive and tangible goal that anglers, scuba divers, and others can use to envision the future that may be achieved if depleted fish populations are allowed to recover. As well, historical data can contradict public opinion and demonstrate to the public what full ecosystem recovery might look like: recovering harbor seals in the Wadden Sea were seen by many as being unnaturally abundant, since nobody had ever seen that many seals in their lifetime. Moreover, the population was affected twice by disease outbreaks, creating the impression that it might have reached its carrying capacity. Yet historical reconstruction revealed that the late-twentieth-century population numbered only half as many seals as were probably present 100 years earlier (see Figure 8.9B), suggesting that current seal populations are natural and still recovering.

Finally, highlighting the link between the well-being of coastal communities and the health of ocean ecosystems over time helps to illustrate the many ways in which humans and their environment are connected. The U.S. National Marine Sanctuaries Program provides a model for such communication, as its goal is to preserve both the ecological integrity and the cultural history of its sanctuaries. For example, the public outreach materials for the Stellwagen Bank National Marine Sanctuary (NOAA 2013) describe the maritime history of the region and the ways in which seafarers interacted with the marine life in the sanctuary’s waters over time. Such integration of history and ecology illustrates the influence of people’s choices and cultural values in the past, today, and for the future. Overall, educating a new generation of children and students and informing the public about the long-term history of change in marine populations, ecosystems,

and coastal societies will contribute to a better future for the ocean and the blue planet.

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LITERATURE CITED

- Ainsworth, C., J. J. S. Heymans, T. J. Pitcher, and M. Vasconcellos. 2002. Ecosystem models of Northern British Columbia for the time periods 2000, 1950, 1900 and 1750. *FCRR* 10: 43.
- Airoldi, L. and M. W. Beck. 2007. Loss, status, and trends for coastal marine habitats of Europe. *Oceanogr. Mar. Biol. Annu. Rev.* 45: 345–405.
- Al-Abdulrazzak, D., R. Naidoo, M. L. D. Palomares, and D. Pauly. 2012. Gaining perspective on what we’ve lost: The reliability of encoded anecdotes in historical ecology. *PLoS ONE* 7: e43386.
- Alter, S. E., E. Rynes, and S. R. Palumbi. 2007. DNA evidence for historic population size and past ecosystem impacts of gray whales. *Proc. Natl. Acad. Sci. (USA)* 104: 15162–15167.
- Altieri, A. H., M. D. Bertness, T. C. Coverdale, et al. 2012. A trophic cascade triggers collapse of a salt-marsh ecosystem with intensive recreational fishing. *Ecology* 93: 1402–1410.
- Ames, E. P. 2004. Atlantic cod stock structure in the Gulf of Maine. *Fisheries* 29: 10–28.
- Anderson, S. C., J. M. Flemming, R. Watson, and H. K. Lotze. 2011. Serial exploitation of global sea cucumber fisheries. *Fish. Fish.* 12: 317–339.
- Baker, C. S. and P. J. Clapham. 2004. Modelling the past and future of whales and whaling. *Trends Ecol. Evol.* 19: 365–371.
- Barrett, J. H., A. M. Locker, and C. M. Roberts. 2004. The origins of intensive marine fishing in medieval Europe: the English evidence. *Proc. Biol. Sci.* 271: 2417–2421.
- Baum, J. K. and B. Worm. 2009. Cascading top-down effects of changing oceanic predator abundances. *J. Anim. Ecol.* 78: 699–714.
- Baum, J. K. and R. A. Myers. 2004. Shifting baselines and the decline of pelagic sharks in the Gulf of Mexico. *Ecol. Lett.* 7: 135–145.
- Beck, M. W., K. L. Heck, K. W. Able, et al. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* 51: 633–641.
- Berkes, F., T. P. Hughes, R. S. Steneck, et al. 2006. Globalization, roving bandits, and marine resources. *Science* 311: 1557–1558.
- Bolster, W. J. 2012. *The Mortal Sea: Fishing the Atlantic in the Age of Sail*. Cambridge, MA: Belknap Press of Harvard University Press.
- Boudreau, S. A. and B. Worm. 2010. Top-down control of lobster in the Gulf of Maine: insights from local ecological knowledge and research surveys. *Mar. Ecol. Progr. Ser.* 403: 181–191.
- Bourque, B. J., B. J. Johnson, and R. S. Steneck. 2008. Possible prehistoric fishing effects on coastal marine food webs in the Gulf of Maine. In, *Human Impacts on Ancient Marine Ecosystems: A Global Perspective* (T. C. Rick and J. M. Erlandson, eds.), pp. 165–185. Berkeley, CA: University of California Press.
- Branch, T. A., K. Matsuoka, and T. Miyashita. 2004. Evidence for increases in Antarctic blue whales based on Bayesian modelling. *Mar. Mamm. Sci.* 20: 726–754.
- Broughton, J. M. 1997. Widening diet breadth, declining foraging efficiency, and prehistoric harvest pressure: Ichthyofaunal evidence from the Emeryville Shellmound, California. *Antiquity* 71: 845–862.

- Broughton, J. M. 2002. Prey spatial structure and behavior affect archaeological tests of optimal foraging models: Examples from the Emeryville Shellmound vertebrate fauna. *World Archaeol.* 34: 60–83.
- Bunce, M., L. D. Rodwell, R. Gibb, and L. Mee. 2008. Shifting baselines in fishers' perceptions of island reef fishery degradation. *Ocean Coast. Manag.* 51: 285–302.
- Byrnes, J. E., P. L. Reynolds, and J. J. Stachowicz. 2007. Invasions and extinctions reshape coastal food webs. *PLoS ONE* 2: e295.
- Carlton, J. T. 1998. Apostrophe to the ocean. *Conserv. Biol.* 12: 1165–1167.
- Carlton, J. T., J. B. Geller, M. L. Reaka-Kudla, and E. A. Norse. 1999. Historical extinctions in the sea. *Annu. Rev. Ecol. Syst.* 30: 515–538.
- Causey, D., D. G. Corbett, C. Lefevre, et al. 2005. The palaeoenvironment of humans and marine birds of the Aleutian Islands: Three millennia of change. *Fish. Oceanogr.* 14: 259–276.
- Cavanagh, R. and C. Gibson. 2007. *Overview of the Conservation Status of Cartilaginous Fishes (Chondrichthyan) in the Mediterranean Sea*. Gland, Switzerland, and Malaga, Spain: World Conservation Union.
- Chaloupka, M., K. A. Bjørndal, G. H. Balazs, et al. 2008. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. *Global Ecol. Biogeogr.* 17: 297–304.
- Christensen, L. B. 2006. Marine mammal populations: Reconstructing historical abundances at the global scale. *UBC Fish. Cent. Res. Rep.* 14: 161.
- Christensen, V., S. Guenette, J. J. Heymans, et al. 2003. Hundred-year decline of North Atlantic predatory fishes. *Fish Fish.* 4: 1–24.
- Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Mar. Ecol. Progr. Ser.* 210: 223–253.
- Cohen, A. N. and J. T. Carlton. 1998. Accelerating invasion rate in a highly invaded estuary. *Science* 279: 555–558.
- Coll, M., H. K. Lotze, and T. N. Romanuk. 2008. Structural degradation in Mediterranean Sea food webs: Testing ecological hypotheses using stochastic and mass-balance modelling. *Ecosystems* 11: 939–960.
- Cooper, S. R. and G. S. Brush. 1993. A 2,500-year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries* 16: 617–626.
- Costanza, R., R. d'Arge, R. de Groot, et al. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387: 253–260.
- Cramer, K. L., J. B. C. Jackson, C. V. Angioletti, et al. 2012. Anthropogenic mortality on coral reefs in Caribbean Panama predates coral disease and bleaching. *Ecol. Lett.* 15: 561–567.
- Dayton, P. K., M. J. Tegner, P. B. Edwards, and K. L. Raiser. 1998. Sliding baselines, ghosts and reduced expectations in kelp forest communities. *Ecol. Appl.* 8: 309–322.
- del Monte-Luna, P., D. Lluch-Belda, E. Serviere-Zaragoza, et al. 2007. Marine extinctions revisited. *Fish Fish.* 8: 107–122.
- Devine, J. A., K. D. Baker, and R. L. Haedrich. 2006. Deep-sea fishes qualify as endangered. *Nature* 439: 29.
- Diaz, R. J. and R. Rosenberg. 2008. Spreading dead zones and consequences for marine ecosystems. *Science* 321: 926–929.
- Dulvy, N. K., J. K. Pinnegar, and J. D. Reynolds. 2009. Holocene extinctions in the sea. In *Holocene Extinctions* (S. T. Turvey, ed.), pp. 129–150. Oxford, UK: Oxford University Press.
- Dulvy, N. K., Y. Sadovy, and J. D. Reynolds. 2003. Extinction vulnerability in marine populations. *Fish Fish.* 4: 25–64.
- Dunne, J. A., R. J. Williams, N. D. Martinez, et al. 2008. Compilation and network analyses of Cambrian food webs. *PLoS Biol.* 6: e102.
- Eddy, T. D., J. P. A. Gardner, and A. Perez-Matus. 2010. Applying fishers' ecological knowledge to construct past and future lobster stocks in the Juan Fernandez Archipelago, Chile. *PLoS ONE* 5: e13670.
- Enghoff, I. B., B. R. MacKenzie, and E. E. Nielsen. 2007. The Danish fish fauna during the warm Atlantic period (ca. 7000–3900 BC): Forerunner of future changes? *Fish. Res.* 87: 167–180.
- Erlanson, J. M., T. C. Rick, and R. Vellanoweth. 2004. Human impacts on ancient environments: A case study from California's Northern Channel Islands. In *Voyages of Discovery: Examining the Past in Island Environments* (S. Fitzpatrick, ed.), pp. 51–83. New York, NY: Praeger Press.
- Essington, T. E., A. H. Beaudreau, and J. Wiedenmann. 2006. Fishing through marine food webs. *Proc. Natl. Acad. Sci. (USA)* 103: 3171–3175.
- Ferretti, F., R. A. Myers, F. Serena, and H. K. Lotze. 2008. Loss of large predatory sharks from the Mediterranean Sea. *Conserv. Biol.* 22: 952–964.
- Ferretti, F., G. C. Osio, C. J. Jenkins, et al. 2013. Long-term change in a meso-predator community in response to prolonged and heterogeneous human impact. *Sci. Rep.* 3: 1057.
- Ferretti, F., B. Worm, G. L. Britten, et al. 2010. Patterns and ecosystem consequences of shark declines in the ocean. *Ecol. Lett.* 13: 1055–1071.
- Finney, B. P., I. Gregory-Eaves, M. S. V. Douglas, and J. P. Smol. 2002. Fisheries productivity in the northeastern Pacific Ocean over the past 2,200 years. *Nature* 416: 729–733.
- Fofonoff, P. W., G. M. Ruiz, B. Steves, and J. T. Carlton. 2003. National Exotic Marine and Estuarine Species Information System. invasions.si.edu/nemesis
- Frank, K. T., B. Petrie, J. S. Choi, and W. C. Leggett. 2005. Trophic cascades in a formerly cod-dominated ecosystem. *Science* 308: 1621–1623.
- Frank, K. T., B. Petrie, J. A. D. Fisher, and W. C. Leggett. 2011. Transient dynamics of an altered large marine ecosystem. *Nature* 477: 86–89.
- Friedlander, A. M. and E. E. DeMartini. 2002. Contrasts in density, size, and biomass of reef fishes between the northwestern and the main Hawaiian Islands: The effects of fishing down apex predators. *Mar. Ecol. Progr. Ser.* 230: 253–264.
- Gerber, L. R., J. Estes, T. Grancos Crawford, et al. 2011. Managing for extinction? Conflicting conservation objectives in a large marine reserve. *Conserv. Lett.* 4: 417–422.
- Gilbert, P. M., D. M. Anderson, P. Gentien, et al. 2005. The global, complex phenomena of harmful algal blooms. *Oceanography* 18: 130–141.
- Goode, G. B. 1884–1887. *The Fisheries and Fishery Industries of the United States*. Washington, DC: Government Printing Office.
- Grabowski, J. H., R. D. Brumbaugh, R. F. Conrad, et al. 2012. Economic valuation of ecosystem services provided by oyster reefs. *BioScience* 62: 900–909.
- Greenstein, B. J., H. A. Curran, and J. M. Pandolfi. 1998. Shifting ecological baselines and the demise of *Acropora cervicornis* in the western North Atlantic and Caribbean Province: a Pleistocene perspective. *Coral Reefs* 17: 249–261.
- Griffiths, C. L., L. van Sittert, P. B. Best, et al. 2004. Impacts of human activities on marine animal life in the Benguela: A historical overview. *Oceanogr. Mar. Biol., Annu. Rev.* 42: 303–392.
- Hallam, A. 2004. *Catastrophes and Lesser Calamities*. Oxford, UK: Oxford University Press.
- Harnik, P. G., H. K. Lotze, S. C. Anderson, et al. 2012. Extinctions in ancient and modern seas. *Trends Ecol. Evol.* 27: 608–617.
- Haug, G. H. and R. Tiedemann. 1998. Effect of the formation of the Isthmus of Panama on Atlantic Ocean thermohaline circulation. *Nature* 393: 325–341.
- Haug, G. H., D. Gunther, L. C. Peterson, et al. 2003. Climate and the collapse of Maya civilization. *Science* 299: 1731–1735.
- Hoffmann, R. C. 1996. Economic development and aquatic ecosystems in Medieval Europe. *Am. Hist. Rev.* 101: 631–669.
- Hoffmann, R. C. 2001. Frontier foods for late Medieval consumers: Culture, economy, ecology. *Environ. Hist. Camb.* 7: 131–167.
- Holm, P. 2005. Human impacts on fisheries resources and abundance in the Danish Wadden Sea, c1520 to the present. *Helgol. Mar. Res.* 59: 39–44.
- Holm, P., A. H. Marboe, B. Poulsen, and B. MacKenzie. 2010. Marine animal populations: A new look back in time. In *Life in the World's Oceans: Diversity, Distribution, and Abundance* (A. D. McIntyre, ed.), pp. 3–23. West Sussex, UK: Wiley-Blackwell.
- Hughes, J. D. 2001. *An Environmental History of the World: Humankind's Changing Role in the Community of Life*. London, UK: Routledge.
- Hutchings, J. A. and R. A. Myers. 1995. The biological collapse of Atlantic cod off Newfoundland and Labrador: An exploration of historical changes in exploitation, harvesting, technology and management. In *The North Atlantic Fisheries: Successes, Failures and Challenges* (R. Arnason and L. Felt, eds.), pp. 38–93. Charlottetown, Canada: The Institute of Island Studies.
- Hutchings, J. A. and J. D. Reynolds. 2004. Marine fish population collapses: Consequences for recovery and extinction risk. *BioScience* 54: 297–309.
- IUCN (International Union for the Conservation of Nature) 2011. IUCN Red List of Threatened Species (Version 2011.2). IUCN. www.iucnredlist.org

- Jackson, J. B. C. 1997. Reefs since Columbus. *Coral Reefs* 16: S23–S32.
- Jackson, J. B. C., M. X. Kirby, W. H. Berger, et al. 2001. Historical overfishing and the recent collapse of coastal ecosystems. *Science* 293: 629–638.
- Jackson, J. B. C., K. E. Alexander, and E. Sala. 2011. *Shifting Baselines: The Past and the Future of Ocean Fisheries*. Washington, DC: Island Press.
- Jennings, S. and J. L. Blanchard. 2004. Fish abundance with no fishing: predictions based on macroecological theory. *J. Anim. Ecol.* 73: 632–642.
- Josephson, E., T. D. Smith, and R. R. Reeves. 2008. Depletion within a decade: The American 19th-century North Pacific right whale fishery. In, *Oceans Past: Management Insights from the History of Marine Animal Populations*. (D. J. Starkey, P. Holm, and M. Barnard, eds.), pp. 133–147. London, UK: Earthscan Research Edition.
- Kappel, C. V. 2005. Losing pieces of the puzzle: threats to marine, estuarine, and diadromous species. *Front. Ecol. Environ.* 3: 275–282.
- Kennett, D. J., B. Voorhies, T. A. Wake, and N. Martinez. 2008. Long-term effects of human predation on marine ecosystems in Guerrero, Mexico. In, *Human Impacts on Ancient Marine Ecosystems: A Global Perspective* (T. C. Rick and J. M. Erlandson, eds.), pp. 103–124. Berkeley, CA: University of California Press.
- Kidwell, S. M. and K. W. Flessa. 1995. The quality of the fossil record: Populations, species and communities. *Annu. Rev. Ecol. Syst.* 26: 269–299.
- Kirby, M. X. 2004. Fishing down the coast: Historical expansion and collapse of oyster fisheries along continental margins. *Proc. Natl. Acad. Sci. (USA)* 101: 13096–13099.
- Kittinger, J. N., J. M. Pandolfi, J. H. Blodgett, et al. 2011. Historical reconstruction reveals recovery in Hawaiian coral reefs. *PLoS ONE* 6(10): e25460.
- Kittinger, J. N., K. Van Houtan, and L. McClenachan. 2013. Using historical data to assess the biogeography of population recovery. *Ecography* doi: 10.1111/j.1600-0587.2013.00245.x
- Kuang-Ti, L. 2001. Prehistoric marine fishing adaptation in Southern Taiwan. *J. East Asian Archaeol.* 3: 47–74.
- Lear, W. H. 1998. History of fisheries in the Northwest Atlantic: The 500-year perspective. *J. Northwest Atl. Fish. Sci.* 23: 41–73.
- Lotze, H. K. 2005. Radical changes in the Wadden Sea fauna and flora over the last 2000 years. *Helgol. Mar. Res.* 59: 71–83.
- Lotze, H. K. 2007. Rise and fall of fishing and marine resource use in the Wadden Sea, southern North Sea. *Fish. Res.* 87: 208–218.
- Lotze, H. K. 2010. Historical reconstruction of human-induced changes in U.S. estuaries. *Oceanogr. Mar. Biol. Annu. Rev.* 48: 267–338.
- Lotze, H. K. and I. Milewski. 2002. *Two Hundred Years of Ecosystem and Food Web Changes in the Quoddy Region, Outer Bay of Fundy*. Fredericton, Canada: Conservation Council of New Brunswick.
- Lotze, H. K. and I. Milewski. 2004. Two centuries of multiple human impacts and successive changes in a North Atlantic food web. *Ecol. Appl.* 14: 1428–1447.
- Lotze, H. K. and B. Worm. 2009. Historical baselines for large marine animals. *Trends Ecol. Evol.* 24: 254–262.
- Lotze, H. K., M. Coll, and J. Dunne. 2011a. Historical changes in marine resources, food-web structure and ecosystem functioning in the Adriatic Sea, Mediterranean. *Ecosystems* 14: 198–222.
- Lotze, H. K., R. C. Hoffmann, and J. M. Erlandson. In press. Lessons from historical ecology and management. In, *The Sea Volume 16: Marine Ecosystem-Based Management* (M. J. Fogarty, and J. J. McCarthy, eds.). Cambridge, MA: Harvard University Press.
- Lotze, H. K., M. Coll, A. M. Magera, et al. 2011b. Recovery of marine animal populations and ecosystems. *Trends Ecol. Evol.* 26: 595–605.
- Lotze, H. K., J. M. Erlandson, M. J. Newman, et al. 2011c. Uncovering the ocean's past. In, *Shifting Baselines: The Past and the Future of Ocean Fisheries* (J. B. C. Jackson, K. E. Alexander, and E. Sala, eds.), pp. 137–161. Washington, DC: Island Press.
- Lotze, H. K., H. S. Lenihan, B. J. Bourque, et al. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312: 1806–1809.
- Lotze, H. K., K. Reise, B. Worm, et al. 2005. Human transformations of the Wadden Sea ecosystem through time: A synthesis. *Helgol. Mar. Res.* 59: 84–95.
- MacKenzie, B. R., H. Mosegaard, and A. A. Rosenberg. 2009. Impending collapse of bluefin tuna in the northeast Atlantic and Mediterranean. *Conserv. Lett.* 2: 25–34.
- McClenachan, L. 2009. Documenting loss of large trophy fish from the Florida Keys with historical photographs. *Conserv. Biol.* 23: 636–643.
- McClenachan, L. and A. B. Cooper. 2008. Extinction rate, historical population structure and ecological role of the Caribbean monk seal. *Proc. Biol. Sci.* 275: 1351–1358.
- McClenachan, L. and J. N. Kittinger. 2012. Multicentury trends and the sustainability of coral reef fisheries in Hawaii and Florida. *Fish. Fish.* doi: 10.1111/j.1467-2979.2012.00465.x
- McClenachan, L., F. Ferretti, and J. K. Baum. 2012. From archives to conservation: Why historical data are needed to set baselines for marine animals and ecosystems. *Conserv. Lett.* 5: 349–359.
- McClenachan, L., J. B. Jackson, and M. J. Newman. 2006. Conservation implications of historic sea turtle nesting beach loss. *Front. Ecol. Environ.* 4: 290–296.
- McCulloch, M., S. Fallon, T. Wyndham, et al. 2003. Coral record of increased sediment flux to the inner Great Barrier Reef since European settlement. *Nature* 421: 727–730.
- Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being: Synthesis*. Washington, DC: Island Press.
- Mortimer, J. A. and M. Donnelly. 2008. Hawksbill turtle (*Eretmochelys imbricate*). Marine Turtle Specialist Group 2008 IUCN Red List status assessment.
- Myers, R. A. and B. Worm. 2003. Rapid worldwide depletion of predatory fish communities. *Nature* 423: 280–283.
- Myers, R. A. and B. Worm. 2005. Extinction, survival or recovery of large predatory fishes. *Phil. Trans. R. Soc. B.* 360: 13–20.
- Myers, R. A., B. R. MacKenzie, K. G. Bowen, and N. J. Barrowman. 2001. What is the carrying capacity of fish in the ocean? A meta-analysis of population dynamics of North Atlantic cod. *Can. J. Fish. Aquat. Sci.* 58: 1464–1476.
- Myers, R. A., J. K. Baum, T. D. Shepherd, et al. 2007. Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science* 315: 1846–1850.
- National Research Council. 1992. *Restoration of Aquatic Ecosystems: Science, Technology and Public Policy*. Washington, DC: National Academy Press.
- Newman, M. J. H., G. A. Paredes, E. Sala, and J. B. C. Jackson. 2006. Structure of Caribbean coral reef communities across a large gradient of fish biomass. *Ecol. Lett.* 9: 1216–1227.
- NOAA (National Oceanic and Atmospheric Administration). 2013. Stellwagen Bank National Marine Sanctuary maritime heritage. stellwagen.noaa.gov/maritime/maritimehistory.html
- Norris, R. D. 1999. Hydrographic and tectonic control of plankton distribution and evolution. In, *Reconstructing Ocean History: A Window into the Future* (F. Abrantes and A. C. Mix, eds.) pp. 173–194. New York, NY: Kluwer Academic/Plenum Publishers.
- Palumbi, S. 2010. *The Death and Life of Monterey Bay: A Story of Revival*. Washington, DC: Island Press.
- Palumbi, S. R. 2011. Whales, logbooks, and DNA. In, *Shifting Baselines: The Past and the Future of Ocean Fisheries* (J. B. C. Jackson, K. E. Alexander, and E. Sala, eds.) pp. 163–173 Washington, DC: Island Press.
- Pandolfi, J. and J. B. C. Jackson. 2001. Community structure of Pleistocene coral reefs of Curacao, Netherlands Antilles. *Ecol. Monogr.* 71: 49–67.
- Pandolfi, J. M. 1999. Response of Pleistocene coral reefs to environmental change over long time scales. *Am. Zool.* 39: 113–130.
- Pandolfi, J. M., R. H. Bradbury, E. Sala, et al. 2003. Global trajectories of the long-term decline of coral reef ecosystems. *Science* 301: 955–958.
- Pandolfi, J. M., J. B. C. Jackson, N. Baron, et al. 2005. Are U.S. coral reefs on the slippery slope to slime? *Science* 307: 1725–1726.
- Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends Ecol. Evol.* 10: 430.
- Pauly, D., J. Alder, E. Bennett, et al. 2003. The future for fisheries. *Science* 302: 1359–1361.
- Pauly, D., V. Christensen, J. Dalsgaard, et al. 1998. Fishing down marine food webs. *Science* 279: 860–863.

- Perdikaris, S. and T. H. McGovern. 2008. Codfish and kings, seals and subsistence: Norse marine resource use in the North Atlantic. In *Human Impacts on Ancient Marine Ecosystems: A Global Perspective* (T. C. Rick and J. M. Erlandson, eds.), pp. 187–214. Berkeley, CA: University of California Press.
- Petersen, K. S., K. L. Rasmussen, J. Heinemeier, and N. Rud. 1992. Clams before Columbus. *Nature* 359: 679.
- Pitcher, T. J., J. J. S. Heymans, and M. Vasconcellos. 2002. Ecosystem models of Newfoundland for the time periods 1995, 1985, 1900 and 1450. *UBC Fish. Cent. Res. Rep.* 10(5): 76.
- Pysonson, N. D. and D. R. Lindberg. 2011. What happened to gray whales during the Pleistocene? The ecological impact of sea-level change on benthic feeding areas in the North Pacific Ocean. *PLoS ONE* 6: e21295.
- Ravier, C. and J.-M. Fromentin. 2004. Are the long-term fluctuations in Atlantic bluefin tuna (*Thunnus thynnus*) population related to environmental changes? *Fish. Oceanogr.* 13: 145–160.
- Reeves, R. R. 2001. Overview of catch history, historic abundance and distribution of right whales in the western North Atlantic and in Cintra Bay, West Africa. *J. Cetacean Res. Manag.* 2: 187–192.
- Reeves, R. R., T. D. Smith, E. Josephson, et al. 2004. Historical observations of humpback and blue whales in the North Atlantic Ocean: Clues to migratory routes and possibly additional feeding grounds. *Marine Mammal Science* 20: 774–786.
- Reijnders, P. J. H. 1992. Retrospective population analysis and related future management perspectives for the harbour seal *Phoca vitulina* in the Wadden Sea. *Neth. Inst. for Sea Res. Pub. Ser.* 20: 193–197.
- Reise, K., E. Herre, and M. Sturm. 1989. Historical changes in the benthos of the Wadden Sea around the island of Sylt in the North Sea. *Helgol. Meeresunters.* 43: 417–433.
- Rick, T. C. and J. M. Erlandson. 2008. *Human Impacts on Ancient Marine Ecosystems: A Global Perspective*. Berkeley, CA: University of California Press.
- Rick, T. C., J. M. Erlandson, T. J. Braje, et al. 2008. Historical ecology and human impacts on coastal ecosystems of the Santa Barbara Channel region, California. In *Human Impacts on Ancient Marine Ecosystems: A Global Perspective* (T. C. Rick and J. M. Erlandson, eds.), pp. 77–101. Berkeley, CA: University of California Press.
- Roman, J. and S. R. Palumbi. 2003. Whales before whaling in the North Atlantic. *Science* 301: 508–510.
- Roberts, C. M. 2007. *An Unnatural History of the Sea*. Washington, DC: Island Press.
- Rose, G. A. 2004. Reconciling overfishing and climate change with stock dynamics of Atlantic cod (*Gadus morhua*) over 500 years. *Can. J. Fish. Aquat. Sci.* 61: 1553–1557.
- Rosenberg, A. A., W. J. Bolster, K. E. Alexander, et al. 2005. The history of ocean resources: Modeling cod biomass using historical records. *Front. Ecol. Environ.* 3: 84–90.
- Rothschild, B. J., J. S. Ault, P. Gouletquer, and M. Héral. 1994. Decline of the Chesapeake Bay oyster population: A century of habitat destruction and overfishing. *Mar. Ecol. Progr. Ser.* 111: 29–39.
- Rugh, D., R. C. Hobbs, J. A. Lerczak, and J. M. Breiwick. 2005. Estimates of abundance of the eastern North Pacific stock of gray whales (*Eschrichtius robustus*) 1997–2002. *J. Cetacean Res. Manag.* 7: 1–12.
- Ruiz, G. M., J. T. Carlton, E. D. Grosholz, and A. H. Hines. 1997. Global invasions of marine and estuarine habitats by non-indigenous species: Mechanisms, extent, and consequences. *Am. Zool.* 37: 621–632.
- Sáenz-Arroyo, A., C. M. Roberts, J. Torre, et al. 2005. Rapidly shifting environmental baselines among fishers of the Gulf of California. *Proc. Biol. Sci.* 272: 1957–1962.
- Sáenz-Arroyo, A., C. M. Roberts, J. Torre, et al. 2006. The value of evidence about past abundance: Marine fauna of the Gulf of California through the eyes of 16th to 19th century travellers. *Fish. Fish.* 7: 128–146.
- Sandin, S. A., J. E. Smith, E. E. DeMartini, et al. 2008. Baselines and degradation of coral reefs in the Northern Line Islands. *PLoS ONE* 3: e1548.
- Santayana, G. 1905. *Reason in Common Sense, Volume 1, The Life of Reason*. New York, NY: Charles Scribner's Sons.
- Saunders, R., M. A. Hachey, and C. W. Fay. 2006. Maine's diadromous fish community: Past, present, and implications for Atlantic salmon recovery. *Fisheries* 31: 537–547.
- Sexton, P. and R. D. Norris. 2008. Dispersal and biogeography of marine plankton: Long-distance dispersal of the foraminifer *Truncorotalia truncatulinoides*. *Geology* 36: 899–902.
- Smith, I. 2005. Retreat and resilience: Fur seals and human settlement in New Zealand. In *The Exploitation and Cultural Importance of Sea Mammals* (G. Monks, ed.), pp. 6–18. Oakville, CT: Oxbow Books.
- Smith, T. D. 1994. *Scaling Fisheries, the Science of Measuring the Effects of Fishing, 1855–1955*. Cambridge, UK: Cambridge University Press.
- Starkey, D. J., P. Holm, and M. Barnard. 2008. *Oceans Past: Management Insights from the History of Marine Animal Populations*. London, UK: Earthscan.
- Steneck, R. S. and J. T. Carlton. 2001. Human alterations of marine communities: Students beware! In *Marine Community Ecology* (M. D. Bertness, S. D. Gaines, and M. E. Hay, eds.), pp. 445–468. Sunderland, MA: Sinauer Associates.
- Steneck, R. S., M. H. Graham, B. J. Bourque, et al. 2002. Kelp forest ecosystems: Biodiversity, stability, resilience and future. *Environ. Conserv.* 29: 436–459.
- Steneck, R. S., T. P. Hughes, J. E. Cinner, et al. 2011. Creation of a gilded trap by the high economic value of the Maine lobster fishery. *Conserv. Biol.* 25: 904–912.
- Swart, P. K. and A. Grotoli. 2003. Proxy indicators of climate in coral skeletons: A perspective. *Coral Reefs* 22: 313–315.
- Trilateral Seal Expert Group (TSEG). 2011. *Aerial Surveys of Harbour Seals in the Wadden Sea in 2011*. Wilhelmshaven, Germany: Common Wadden Sea Secretariat. <http://www.waddensea-secretariat.org/news/news/Seals/Annual-reports/seals2011.html>
- Valentine, J. W., D. Jablonski, S. M. Kidwell, and K. Roy. 2006. Assessing the fidelity of the fossil record by using marine bivalves. *Proc. Natl. Acad. Sci. (USA)* 103: 6599–6604.
- Ward, P. and R. A. Myers. 2005. Shifts in open-ocean fish communities coinciding with the commencement of commercial fishing. *Ecology* 86: 835–847.
- Worm, B. and E. Duffy. 2003. Biodiversity, productivity and stability in real food webs. *Trends Ecol. Evol.* 18: 628–632.
- Worm, B., H. K. Lotze, H. Hillebrand, and U. Sommer. 2002. Consumer versus resource control of species diversity and ecosystem functioning. *Nature* 417: 848–851.
- Worm, B., E. B. Barbier, N. Beaumont, et al. 2006. Impacts of biodiversity loss on ocean ecosystem services. *Science* 314: 787–790.
- Worm, B. and R. A. Myers. 2003. Meta-analysis of cod-shrimp interactions reveals top-down control in oceanic food webs. *Ecology* 84: 162–173.
- Worm, B., R. Hilborn, J. K. Baum, et al. 2009. Rebuilding global fisheries. *Science* 325: 578–585.
- zu Ermgassen, P. S. E., M. D. Spalding, B. Blake, et al. 2012. Historical ecology with real numbers: Past and present extent and biomass of an imperilled estuarine ecosystem. *Proc. Biol. Soc.* 279: 3393–3400.