

Chapter 36A. North Atlantic Ocean

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1. Introduction

The North Atlantic is characterized by relatively wide continental shelves, particularly in its northerly portions, with steep slopes to the abyssal plain¹. The width of the shelf decreases towards the south, with typical boundary current systems, characterized by strong seasonal upwelling, off the Iberian Peninsula and northwest Africa. Two chains of volcanic islands, the Azores and the Canaries, are located in the east central North Atlantic, and a large number of islands of volcanic origin, many with associated warm water coral reefs, are found in the southwest portion of the North Atlantic. In the far north of the region is the world's largest island, Greenland, primarily of Precambrian origin, whereas Iceland and the Faroe Islands are of more recent volcanic origin. All have rugged coastlines with rich faunas.

The biota of the North Atlantic is strongly influenced by both the warm Gulf Stream flowing north-eastward from the Gulf of Mexico and the Caribbean to northwest Europe, and the cold, fresh Labrador Current flowing south from the Canadian Archipelago and Greenland to the northeast coast of the United States. Major oceanographic and associated biotic regime shifts have been documented in the North Atlantic, but not with the frequency or scale of the North Pacific.

Around the coasts of the North Atlantic are a number of semi-enclosed seas. These seas have distinct oceanographic and bathymetric regimes, and ecosystems with many characteristics determined by local-scale processes and pressures. Hence each of these semi-enclosed seas, including the Black Sea, Mediterranean Sea, Baltic Sea (and similar coastal estuaries of the United States), the Caribbean Sea, and the Gulf of Mexico, receive some individual consideration in this assessment. Within the North Atlantic, there are several habitat types of special importance for biodiversity, such as seagrass beds and cold- and warm-water corals. Since these are important where they are found on the globe, they are treated in an integrated manner, respectively, in chapters 47, 42-43, rather than separately.

Coastal areas of the Northeast Atlantic have been settled in and used for several millennia. Commercial fisheries, both coastal and, as technology developed, offshore, have exploited fish and shellfish resources for centuries as well (Garcia et al., 2014), with

¹ Biodiversity of the abyssal plain and mid-Atlantic Ridge that divides the eastern and western Atlantic is dealt with in Chapter 36F.

periods of widespread overfishing in the twentieth century. Industrialization developed first in northwest Europe and eastern North America, and land-based pollution and coastal infrastructure have been significant pressures on coastal biodiversity of the North Atlantic for nearly two centuries. Large urban centres developed on the coasts of the North Atlantic at the time of industrialization, and below the boreal latitudes most of the North Atlantic coastal areas have been altered by various combinations of urban or municipal development, industry, agriculture, and tourism. Although some tens of kilometres of coasts and a few hundreds of square kilometres of coastal seabed are now protected in various ways, almost all biotic communities have been altered by centuries of pressures from human uses.

2. Coastal and Shelf Holoplankton

2.1 Status²

2.1.1 Phytoplankton

Diatoms and dinoflagellates account for most phytoplankton species ($> 2 \mu\text{m}$) in coastal and shelf waters of the North Atlantic (Tables 36A.1 and 36.2)³, with diatom blooms typically peaking during spring and dinoflagellate blooms during summer (cf., McQuatters-Gollop et al., 2007). The ubiquitous prokaryotic species *Synechococcus* spp. ($< 2 \mu\text{m}$) also peaks in abundance ($> 10^7$ cells liter⁻¹) during summer (cf., Wang et al., 2011). On a decadal time scale (1960-2009), dinoflagellate species richness has increased while abundance has decreased relative to diatoms in the North-East North Atlantic and North Sea, a trend that has been attributed to the combined effect of increases in sea surface temperature (SST) and wind shear during the summer (Hinder et al., 2012).

2.1.2 Mesozooplankton (200 - 2000 μm)

Calanoid copepods dominate the holoplankton in coastal and shelf waters throughout the North Atlantic (Table 36A.3). Many of these species are cosmopolitan, e.g., > 60 per cent of the species described from the Caribbean Sea and Gulf of Mexico are also found in the North-East Atlantic (Park, 1970). A decadal scale (1958 – 2005) progressive increase in the abundance of warm-temperate calanoid species (e.g., *Calanus helgolandicus*, *Centropages typicus*) and a decline of cold-temperate calanoid species (e.g., *Calanus finmarchicus*, *Euchaeta norvegica*) has been documented in the North-East North Atlantic (Beaugrand et al., 2002, 2009; Chust et al., 2014). Coincident with

² Sampling for species identification employs different techniques (e.g., water samples, net samples, continuous plankton recorder) and, therefore, are not comparable from region to region in terms of species richness.

³ Note that the lists of abundant taxa in Tables 36A.1 and 36A.2 are not meant to be comprehensive. For example, a phytoplankton check list for the Baltic Sea describes over 1,500 species (Hällfors, 2004).

this trend, the mean size of copepods decreased as their species diversity increased and SST increased (Beaugrand et al., 2010).

2.2 Long-Term (multi-decadal) Trends and Pressures in Holoplankton

2.2.1 Regime Shifts: Overfishing and Climate Change

The primary pressures responsible for regime shifts in shelf ecosystems are overfishing and climate-driven changes (hydro-climate pressures including Arctic ice melting, ocean warming, and mode variability) in the marine environment (Steele, 2004; Edwards et al., 2006; Kane, 2011). Synchronous, system-wide regime shifts in plankton communities were initiated during the late 1980s and early 1990s in the Baltic Sea, North Sea, Scotian Shelf and Gulf of Maine (Reid et al., 2001; Edwards et al., 2002; Alheit et al., 2005; Record et al., 2010; Kane, 2011; Möllmann, 2011). In each case, synergies between trophic cascades triggered by overfishing and changes in hydro-climate were the primary pressures with overfishing reducing resiliency and hydro-climate forcing initiating the regime shift (Drinkwater, 2005; Beaugrand et al., 2008; Fogarty et al., 2008; Hilborn and Litzinger, 2009; Möllmann, 2011).

Gulf of Maine – During 1961-2008, the annual cycle of species richness and abundance was characterized by a seasonal peak during spring when diatoms dominated (Kane, 2011). The most abundant taxa were *Thalassiosira* spp., *Rhizosolenia hebetate*, *Phaeoceros* spp., *Thalassiothrix longissima*, and *Thalassionema nitzschioides*. On a decadal time scale, Kane (2011) documented three consecutive multi-year periods of varying species richness and abundance: below average (1961 – 1989), above average (1990 – 2001), and average (2002 – 2008). Decadal changes were more pronounced for diatoms than dinoflagellates (Möllmann, 2011), and the most striking feature in the time-series was the persistent positive anomaly of the 1990s.

Zooplankton species richness and abundance also increased sharply during the 1990s as the abundance of smaller copepod species increased and larger species declined (Pershing et al., 2005; Kane, 2007; Record et al., 2010). Increases in zooplankton and phytoplankton stocks were also reported during this period over the Newfoundland and Scotian shelves as Arctic species originating from the Gulf of St. Lawrence and the Labrador Current (e.g., *Calanus glacialis*, *Calanus hyperboreus*) became more abundant and warmer water species (e.g., *Calanus finmarchicus*, *Centropages typicus*, *Metridia lucenss*, *Temora stylifera*) became less abundant (Head and Sameoto, 2007; Möllmann, 2011). This inter-decadal zooplankton biodiversity signal was significantly correlated with phytoplankton biomass (Record et al., 2010).

North Sea – As indicated by rapid changes in plankton biomass and species diversity, a regime shift was initiated between 1983 and 1988, apparently as a consequence of increases in SST, a positive phase of the North Atlantic Oscillation (NAO), and increases in advection from the North-East North Atlantic (Beaugrand, 2004;

McQuatters-Gollop et al., 2007). Mean phytoplankton chlorophyll levels peaked in 1989 (Reid et al., 1998), and the new regime (1990 - 2003) maintained 13 per cent and 21 per cent higher chlorophyll concentrations in open and coastal waters, respectively (McQuatters-Gollop et al., 2007). The regime shift was also marked by increases in the abundance and diversity of dinoflagellates relative to diatoms, decreases in the abundance of *Ceratium* spp. (e.g., *C. furca*, *C. fusus*, *C. horridum*, *C. tripos*, *C. lineatum*), increases in diversity and abundance of warm water calanoid copepod species (e.g., *Rhincalanus nasutus*, *Eucalanus crassus*, *Centropages typicus*, *Candacia armata*, *Calanus helgolandicus*), decreases in cold water species (e.g., *Heterorhabdus norvegicus*, *Euchaeta norvegica*, *Calanus finmarchicus*), and increases in the frequency of jellyfish outbreaks (most notably the hydrozoan *Aglantha digitale* and the scyphozoan *Pelagia noctiluca*) (Attrill et al., 2007; Beaugrand et al., 2009, 2010; Edwards et al., 2009; Richardson et al., 2009; Licandro et al., 2010).

2.1.2 Toxic Phytoplankton Blooms: Invasions, Eutrophication and Climate Change

The frequency of toxic phytoplankton blooms has increased over the last three decades in coastal waters of both the western and eastern North Atlantic (Anderson et al., 2012). Toxic taxa include dinoflagellates (*Alexandrium* spp., *Gymnodinium catenatum*, *Karenia mikimotoi*, *Karenia brevis*, *Dinophysis* spp., *Prorocentrum crassipes*, *Prorocentrum* spp.), diatoms (*Pseudo-nitzschia* spp.), and microflagellates (*Chrysochromulina polylepis*, *Chattonella* spp., *Fabrocapsa japonica*).⁴ Increases in toxic events associated with these species have been attributed to more frequent and comprehensive observations, the dispersal of invasive toxic species via the ballast water of ships, coastal eutrophication, and climate change (Skjoldal and Dundas, 1991; Hallegraeff and Bolch, 1992; Belgrano et al., 1999; Anderson et al., 2002; Sellner et al., 2003; Glibert et al., 2005; Dale et al., 2006; Edwards et al., 2006; Moore et al., 2008; Fu et al., 2012). The risk of harmful phytoplankton blooms in the future has increased due to synergy between climate-driven changes and anthropogenic nutrient inputs (Hallegraeff, 2010).

2.1.3 Invasive Species: Ballast Water, Aquaculture and Climate Change

Large numbers of non-indigenous species have been introduced to coastal marine and estuarine ecosystems, largely due to transoceanic shipping (ballast water) and aquaculture (cf., Reise et al., 1999; Gollasch et al., 2009). Many of these species become invasive. Four ecologically significant invasions have been unequivocally documented (Birnbaum, 2006; Javidpour et al., 2006; Reid et al., 2007; Riisgård, 2007). The nuisance diatom (produces mucilage), *Coscinodiscus wailesii*, was introduced in the 1970s. The increase in abundance and geographic expansion of this species from the English Channel into the North Sea over the last three decades is a prototypical example of a planktonic species invasion. Likewise, by the 1990s the cladoceran *Cercopagis pengoi* and comb jelly *Mnemiopsis leidyi*, both voracious planktivores with the potential to disrupt trophic dynamics, had spread from the Gulf of Riga into the Baltic Sea and Gulf

⁴ http://www.marbef.org/wiki/OSPAR_eutrophication_assessment;
<http://www.whoi.edu/redtide/species/by-syndrome>

of Finland. These species appear to have been introduced via ballast water. The Pacific diatom *Neodenticula seminae* was transported into the Labrador Sea via the Canadian Arctic Archipelago in the late 1990s. It has since spread south to Georges Bank and further east, south of Iceland. The geographic expansion of this species portends of more trans-Arctic invasions from the Pacific Ocean as climate-driven Arctic ice melt continues.

2.1.4 Jellyfish: Overfishing, Eutrophication and Climate Change

Jellyfish blooms have increased in frequency in the northeast North Atlantic since 2002, e.g., *Pelagia noctiluca*, *Aglantha digitale*, and *Physalia physalis* outbreaks in the North-East North Atlantic (Attrill et al., 2007; Doyle et al., 2007; Richardson et al., 2009; Licandro et al., 2010), and mounting evidence suggests that these outbreaks may lead to trophic shifts from fish to jellyfish and other gelatinous zooplankton as the dominant consumers (Richardson et al., 2009). Primary pressures leading to a more gelatinous state include overfishing (Lynam et al., 2006; Bakun and Weeks, 2006; Roux et al., 2013), coastal eutrophication and hypoxia (Purcell et al., 2001; Condon et al., 2001), and climate-driven ocean warming and mode variability, e.g., the NAO (Purcell et al., 2001; Attrill et al., 2007; Gibbons and Richardson, 2008; Doyle et al., 2007), although decline in predation by sea turtles has also been implicated as a factor in the outbreaks.

2.1.5 Calcifying Plankton: Climate Change

A multi-decadal time series (1960-2009) for the North-East North Atlantic region shows that changes in the abundance and distribution of foraminifera (*Globerigina* spp.), coccolithophores (*Emiliana huxleyi*), pteropods (*Clione limacine*, *Limacina helicina*), non-pteropod molluscs and echinoderm larvae were positively correlated with decadal changes in annual SST and the Atlantic Multidecadal Oscillation and negatively correlated with pH, i.e., abundance increased as pH decreased (Beaugrand et al., 2013). Beare et al. (2013) found no statistical relationship between the abundance of calcifying plankton in the North Sea and pH, although *Globerigina* spp, *Emiliana huxleyi* and echinoderm larvae increased during the period, and pteropods and bivalve larvae decreased. Thus, although acidification may become a serious threat to calcifying plankton, observations to date suggest that the primary driver of calcifying plankton abundance has been ocean warming.

3. Benthos

Most studies dealing with “benthos” in the North Atlantic focussed on macrobenthos (i.e., metazoan animals living in the seafloor and retained on a sieve of 500-1000 µm). Studies on meiobenthos are more scattered, and often targeted towards the dominant meiobenthic taxon: the nematodes.

Much of the available information has been compiled at the beginning of this century, in the framework of large-scale projects including the Census of Marine Life (CoML, 2000-

2010), and the European Network of Excellence MarBEF (2004-2009). MarBEF was highly successful in compiling data on coastal benthos of the North-East Atlantic, and CoML captured many data of the deeper parts of the Atlantic in general.

3.1 *Coastal Benthos*

Within MarBEF, a large database (MacroBen), consisting of more than 460,000 distribution records on the distribution of 7,203 taxa from almost 23,000 stations was compiled (Gage et al., 2004; Vanden Berghe et al., 2009). Data were collected between 1972-2005 in the North-East Atlantic and adjacent semi-enclosed seas (North Sea, Baltic Sea, Mediterranean Sea) in a depth range of 0-450 m. Highest sampling density was in the North Sea and North-East Atlantic. As the analysis of this entire database reduces the potential problem of a disproportionally large effect of site-specific features on large-scale patterns (Renaud et al., 2009), and as the analyses of this database took into account methodological issues related to sampling and treatment of samples, the findings from these analyses are taken to reflect the general trends in macrobenthic communities in the North Atlantic. A first macro-ecological analysis showed that most macrobenthic communities follow the same right-skewed frequency distribution known from terrestrial ecosystems, revealing that most species are rare, and only few species have a wide distribution (Webb et al., 2009). Except for polychaetes, macrofaunal communities on a geographical scale of tens of km² or less are not random subsets of a species list at a wider geographical scale: species belonging to the same community tend to be more closely related to each other than would be expected when communities were randomly assembled from a regional species pool (Sommerfield et al. 2009). Hence, community composition for most taxa is determined by regional processes, whereas random assembly, followed by local environmental and ecological processes, is more important for polychaete communities.

After removal of the confounding effect of depth, sampling effort and the low diversity in the Baltic, Renaud et al. (2009) demonstrated a modest increase in diversity with increased latitude. Much stronger effects were observed for the diversity-depth gradient where a unimodal trend with water depth (within the 0-450 m depth range) was observed: species richness peaked at 100-150 m depth, and maximum values for the expected number of species in a sample with 50 individuals (ES (50)) were observed at 200-350 m (Escaravage et al., 2009). Diversity was negatively related to the fraction of primary production reaching the seafloor, corresponding with the decreasing part of the unimodal productivity-diversity curve (Escaravage et al., 2009).

3.2 *Offshore Benthos*

General patterns on the offshore benthos from the eastern and western part of the North Atlantic, here defined as the macrobenthos from continental slopes and the deep sea, have been investigated in relation to both latitudinal and depth-related patterns. However, data are scarce in comparison with the more shallow areas, and trends have

been deduced from a limited number of studies, focusing on a limited number of taxa. A poleward decrease in diversity was observed for deep-water molluscs and crustaceans (Rex et al., 1993) and cumaceans (Gage et al., 2004). Trends for the cumaceans were stronger in the eastern part of the basin. Data seem to be too scarce at the moment to make statements about the mechanisms behind the patterns (Narayanaswamy et al., 2010). Trends in the relationship between diversity and depth differ between sites for the same taxon: bivalve diversity (measured as ES (50)) is a significant unimodal function of depth, with diversity peaking at mid-bathyal depths in the western part of the North Atlantic, whereas the diversity-depth relationship can be described as a significant, linear function of depth (Brault et al., 2013) in the eastern part. The absence of the unimodal relationship between depth and diversity has been described for other taxa in other areas as well (Narayanaswamy et al., 2010). However, “depth” itself is not the explanatory variable, as it covaries with a variety of environmental characteristics and not always in the same manner (Narayanaswamy et al., 2010). Strong differences exist in the flux of organic carbon to the seafloor between the eastern and the western part of the Atlantic at depths > 3800m, where the flux of organic carbon is 56 per cent higher in the eastern basin, which is probably reflected in the macrofaunal densities and the feeding mode composition of bivalves (Brault et al., 2013). Although the flux of organic matter is indeed important, processes regulating diversity at local, regional and global scales in the deep sea are multivariate, and smaller-scale processes are hierarchically embedded in larger-scale processes and tend to occur at faster rates (Levin et al., 2001). Processes at the local scale involve biological interactions (competition, facilitation, and predation), patch type characteristics (biogenic structures, nutrient concentrations, topography) and disturbance and recruitment. All of these are hierarchically embedded in environmental gradients at the regional scale, dispersal, metapopulation dynamics and gradients in habitat heterogeneity. Processes at the global scale include, amongst others, speciation/extinction, large-scale disturbances and large-scale environmental gradients.

Apart from patterns in diversity, the relationship between biomass and depth received considerable attention as well. The decrease in total biomass with increasing depth has been observed at both sites of the Atlantic Ocean (i.e., Heip et al., 2001; Rex et al., 2006), and is explained by a decreased flux of organic matter when depth increases (Johnson et al., 2007). A global-scale analysis (Wei et al., 2010) showed that this decrease in total biomass is related to a decrease in individual size of the organisms.

3.3 *Dominant Pressures*

Many human activities have been documented to have impacts on benthic communities (Rice et al., 2010). Effects of mobile bottom-contacting fishing gear on coastal and shelf benthic communities have been documented essentially everywhere that such gear has been used. However, the nature of those impacts and their duration have been shown in many reviews to depend on the type of substrate and frequency of trawling (Collie et al., 1997; FAO, 2009; Hiddink et al., 2006; Kenchington et al., 2007; National Research

Council, 2001), with the longest-lasting impacts on hard-bodied biogenic structures, such as corals and glass sponges (see chapters 42 and 43). Recovery of benthic communities following cessation of bottom trawling has also been documented in several studies (Grizzle et al., 2009; Kaiser et al., 2006). Similar effects have been documented for other physical disturbances, such as aggregate extraction (Barrio Froján et al., 2011), with moderate recovery rates from local extraction events (Boyd et al., 2005), although such disturbances are usually concentrated in coastal areas (ICES, 2009).

Coastal benthic communities are also documented to be affected by pollution from land-based and coastal sources (see Chapter 20), such as nutrient runoff from the land, and shoreline alteration for human recreation and infrastructure. These impacts are nearly universal where pollution and nutrient inputs occur, and where coastlines are urbanized or adapted for tourism. However, their nature depends on the type, intensity, and duration of the pollution or nutrient input and extent of alteration, although persistent pressures of this type can greatly alter the species composition and biomass of benthos directly and indirectly through processes such as hypoxia (Borja et al., 2008; Gagné et al., 2006; HELCOM, 2009a; Middelburg and Levin, 2009). These effects can be specific enough that benthic community composition and/or productivity is often used as an indicator of ecosystem stress from pollutants or nutrient inputs (Borja et al., 2009; Quintaneiro et al., 2006; Solimini et al., 2006).

Climate change, including multi-year climate variability, is another major and growing pressure on the benthos. Not only are there effects of coastal warming, but the dependency of deep-sea benthos on the export of organic matter from the ocean surface seems to be at the basis of possible large-scale changes in biomass distribution under future climate scenarios (Jones et al., 2013). Surface ocean warming can result in increased stratification and a less efficient nutrient supply for primary production, leading to a projected decrease in upper ocean biomass (Joos et al., 1999; Steinacher et al., 2010), and a subsequent decrease in organic matter flux to the open ocean seafloor communities (McClain et al., 2012). Jones et al. (2012) modelled particulate organic carbon (POC) fluxes to the seafloor and the resulting macrofaunal biomass distribution under future climate scenarios (RCP 8.5 and RCP 4.5). Under the more severe RCP 8.5 scenario, global macrofaunal biomass is predicted to decrease by 3.771 per cent as a result of decreased organic matter fluxes. Both the greatest negative (-49.7 per cent) and positive (+36.79 per cent) change are predicted for the North Atlantic, with positive changes mainly located on the western side and major negative changes to the east. These shifts can result in range changes of species and facilitate colonization by invasive species in certain areas (Thatje, 2005). The reduction in total biomass of macrofauna will coincide with a size-shift toward smaller organisms (Jones et al., 2012, 2013), which has important biological consequences, including increasing respiration rates and reduction in overall biomass production efficiency (Brown et al., 2004; McClain et al., 2012; Smith et al., 2008) and a reduced energy transfer to higher trophic levels (Brown et al., 2004). Ocean acidification is an increasing threat to coastal marine benthos. Species having calcareous exoskeletons may have particularly high vulnerability to acidification, but it

has been found to affect many species of benthos (Andersson et al., 2011). It can be concluded that climate change, through its effect on primary production, is likely to have an important impact on the deep-water benthos, as well as coastal and shelf benthic communities.

4. Fish Communities

Much has been written about the status, trends and drivers of North Atlantic fish communities. These fish communities have been exploited for centuries, and have experienced additional pressures from coastal development, land-based inputs to the seas, and climate variation and change. Much effort has been devoted to teasing apart the influences of exploitation and environmental conditions as drivers of change in fish populations and communities, but definitive answers are elusive. Although long time-series of survey data and commercial catches exist, most effort has gone into assessing the status of commercially exploited populations.

Studies of fish biodiversity in the wider sense are much less numerous, and usually restricted to portions of the North Atlantic only. Thus general patterns have to be assembled from multiple studies, with many differences among them. Consolidating results of multiple studies as a basis for inferences on status, trends and pressures on fish communities has risks, because of the need to account for differences in the catchability of various species (Fraser et al., 2008), the underrepresentation of smaller fish in surveys (Cook and Bundy, 2012), the large sampling effort needed to document fish diversity (Greenstreet and Piet, 2008), and the dependence of inferred patterns of community change on the spatial scale of the analyses (Gaertner et al., 2007). Hence there will be many exceptions to generalizations and many gaps in knowledge. Some species groups, such as tuna and other large pelagics, and elasmobranchs, are addressed in separate chapters of this Assessment, and should be reviewed there. Likewise the parts of this Chapter on the several semi-enclosed seas have information on fish communities at those scales, and should also be consulted. The rest of this section looks at general patterns of near-coastal and shelf fish communities more generally.

4.1 Coastal Fish Communities

The need to piece together emergent messages from numerous separate studies is particularly true for near-coastal fish communities, where time series are usually much shorter and of localized spatial coverage. Most studies devote more attention to explaining variation among coastal fish community properties relative to features of the physical and chemical habitats, including temperature, salinity, oxygen and nutrient levels, clarity of and pollutants in the water column, and to depth, sediment types, benthic communities, contaminant levels, oxygen levels, and disturbance regimes of the

seafloor. All of these factors have been shown to influence fish community composition and structure in at least some coastal areas around the North Atlantic.

A few generalizations can be drawn from the diversity of results, but many are common sense. Studies have documented that when any of a large number of environmental factors are altered substantially from values historically characteristic of a coastal area, fish communities are highly likely to be altered as well. Large effects have been documented for factors like oxygen depletion (Oguz and Gilbert, 2007; Stramma et al., 2010), eutrophication (Buchheister et al., 2013; Olsen et al., 2012), sedimentation (França et al., 2012; Jordan et al., 2010; Kopp et al., 2013), and contaminants (Bergek et al., 2012; McKinley et al., 2011; Pato et al., 2008). Emergent macroalgae are an important determinant of coastal fish community diversity and fish abundance, and correspondingly many studies have documented that altering aquatic vegetation can have large effects on fish community status (Pihl et al., 2007; Waycott et al., 2009; York et al., 2012).

However, effects are often situation-specific, and multiple factors interact. For example, coastal flow regimes can dominate over nutrient loading, possibly through ensuring reoxygenation of water and sediments (Kotta et al., 2009; Reiss et al., 2010). Scales of wave energy can also influence how strongly local perturbations of habitat conditions are reflected in changes in fish communities (Jordaan, 2011). The local scale at which fish community structure is determined and variation is documented (e.g., Bonaca and Lipej, 2005) can be amplified, because many drivers of change in coastal fish communities are either both local in scale, such as coastal infrastructure development (e.g., Bulleri et al., 2012) or episodic, such as major oil spills (e.g., Mendelssohn et al., 2012).

Unfortunately the local scales at which coastal fish communities are structured, and where many impacts are experienced, means that it is not possible to present a quantitative accounting of trends in coastal fish communities on regional or North Atlantic scales. Long-term or large-scale studies do document that effects of major oceanographic drivers, such as warming or cooling trends, can be seen in coastal fish communities, documenting that large-scale as well as local factors affect community status (Henderson et al., 2011; Hurst et al., 2004). Species composition generally will react to such large-scale drivers at greater rates than more integrative community metrics (Bui et al., 2010; Hurst et al., 2004).

Given the intensification of use of coastal areas for aquaculture (Chapter 12), infrastructure (several chapters in part V), and land-based inputs to coastal areas (Chapter 20), it is likely that overall the status of coastal fish communities around the North Atlantic has been altered, and in areas with high human use and large habitat changes, the alterations could be large, with a reduction in species diversity and simplification of community structure (Lotze et al., 2006; Waycott et al., 2009). In addition, invasive species can have a large impact in coastal fish communities, and in cases such as the lionfish invasion of Caribbean and coastal southern northern American waters, may spread rapidly from multiple points of initial establishment, seriously

disrupting native fish communities (Whitfield et al., 2007; Muñoz et al., 2011). Such changes would, in turn, have consequences much wider than the local scale of the impacts, given the important role of coastal systems as nursery habitats (Beck et al., 2001; Persson et al., 2012).

Consequently, even if the overall trends in coastal fish communities cannot be quantified on the scale of the North Atlantic, the impacts of many pressures on these communities have been documented, as have the effects of larger-scale oceanographic and climatic drivers. With the increase in intensity of human activities causing many of these pressures (Sections IV and V) and a background of a changing ocean climate, there is ample justification for attention to the conservation of these systems. The evidence also indicates that appropriate management regimes need to be designed and implemented on local scales, to accommodate local communities and pressures, even if the overarching policies are developed at larger scales (*sensu* FAO Ecosystem Approach to Fishing (Staples and Funge-Smith, 2009)).

4.2 *Shelf Fish Communities*

A few studies have reconstructed fish communities and their variation over centuries into the past, albeit usually for just a few selected species and using catch records, sediment layers, or middens for local areas. These studies have consistently shown major changes in the composition of the fish community over the full time series, sometimes in regime-like ways. Likely impacts of overfishing were already evident early in the second half of the previous millennium (Mackenzie et al., 2007; Poulsen et al., 2007), but changes to the fish community associated with warmer and cooler periods of the North Atlantic are documented for the last several centuries (Enghof et al., 2007).

The current status of shelf fish stocks is best evaluated by the assessments done by the major fisheries management authorities around the North Atlantic. When data are sufficient, assessments provide estimates of fishing mortality and biomass, and interpret these relative to sustainability benchmarks. The biomass benchmarks reflecting that a stock is not overfished vary among jurisdictions and often are based on data series that do not extend back to a time when the stocks were unexploited (Lotze and Worm, 2009; Greenstreet et al., 2012). Nevertheless, Table 36A.4 presents the evaluations for most of the major assessment jurisdictions. The general messages are clear: many stocks are overfished and/or experiencing current overfishing, based on their current status relative to their management benchmarks, and the status of a number of other exploited stocks is not known. However, that only reflects part of the picture. For a large fraction of these stocks, the severe overfishing occurred in the 1990s and 2000s, and their status is improving. The improvement is consistently attributed to reductions in fishing effort (ICES and NOAA websites).

Of course the status of exploited stocks is only part of the fish diversity of the North Atlantic shelf systems. Many studies have analysed trends in the properties of fish communities, but these studies have varied greatly in the time intervals used, the parts

of the North Atlantic examined, the metrics of community status quantified, and the species included in the metrics. Given that the results of community analyses are scale-dependent (e.g., Gaertner et al., 2007), metrics are often partially redundant but not interchangeable (e.g., Greenstreet et al., 2012), and both fishing pressure and environmental conditions have changed substantially over the past several decades (ter Hofstede et al., 2012), only a few broad generalizations can be drawn from the diversity of results reported (Table 36A.5).

Based on Table 36A.5 and consistent with other overviews (ICES annual reports, NOAA annual reports), it is without question that in nearly every area of the North Atlantic examined, even moderate fishing pressure has been associated with a decrease in the proportion and absolute number of large fish in the community. Everywhere that heavy fishing was reported, not only does the size composition of the community continue to be truncated, but dominance usually declines as the most common species are reduced in abundance and species of lower or no commercial importance increase in at least relative and often absolute abundance. Whether this changes the diversity metrics depends on case-specific properties of the fish community (how dominant were the most dominant species) and fishing pressure (how intense, how sustained, and how selective).

It is also without question that for every time series of even moderate (decadal) length, effects sought of changing oceanographic conditions on fish community have been documented (e.g., Perry et al., 2005; Lucey and Nye, 2010; Pinsky et al., 2013). Warming is usually associated with increases in richness and diversity metrics, as the pool of warm-water species that can move into an area from the south during warm periods is almost always larger than the pool of cold-water species that can move in from the north during cold periods. Occasionally one of these environmentally sensitive species becomes very abundant (e.g., the Snake Pipefish outbreak in the North Sea around 2005, Harris et al., 2007), affecting diversity and evenness metrics. However, a number of studies report a negligible or even no trend in community metrics over moderate periods of varying environmental conditions, yet report large changes in the species composition of the community underlying the aggregate metrics. This highlights the strong buffering capacity that is increasingly being argued, where functional redundancy (Schindler et al., 2010; Widemann et al., 2012) gives resilience to fish communities, even if the species composition is changing substantially and without strong structuring processes (Rice et al., 2012).

In a few cases, time series of several decades are available. In all these reports the effects of both changes in fishing effort and in oceanographic conditions are apparent. These can be seen in individual species (e.g., herring, Harma et al., 2012; Larsson et al., 2010; cod, Drinkwater, 2010; Eero et al., 2011; Lilly et al., 2013; sole, Horwood, 2010) and such metrics of community as can be assembled from samples over long time periods (Foch et al., 2014; Greenstreet et al., 1999; ter Hofstede and Rijnsdorp, 2011; Shackell et al., 2012).

Bringing together the results of studies that look at how environmental drivers and fisheries have affected North Atlantic fish communities, the key messages include:

(i) Essentially every shelf fish community in the North Atlantic has been altered by decades to centuries of fishing. For many areas, excessive fishing persisted long enough for target species to be depleted to states where recovery has been slow, and whole communities have had their diversity reduced, with size metrics showing the greatest effects at community scales. However, in the twenty-first century, fishing effort has been reduced in most parts of the North Atlantic shelves, particularly where stocks and communities were most stressed, and there is evidence of recovery in most of these areas, albeit at different rates for different species, with some species having recovered to target levels.

(ii) Where data have been examined, every shelf fish community has had its species composition change as oceanographic conditions have changed. Responses to warming and cooling trends seem to be most prevalent, but these also have been looked for most often. Regime-like changes in fish community composition have been documented often, but they are not universal. Aggregate community metrics have often changed much less than the abundance of the species contributing to them.

(iii) On case-by-case examples it is often hard to definitively untangle the effects of fishing and of environment on fish communities although some at least partial successes are being reported (Bell et al., 2014). However, unless fishing is kept at a sustainable level, community-scale effects of depletion of target species are highly likely, and may reduce resilience to environmental drivers (Shackell et al., 2012).

5. Seabirds⁵

5.1 North Atlantic Overview

Overall, populations of breeding seabirds in the North Atlantic appear to be decreasing. This contention is the outcome of an integration of negative trends in both the North Atlantic Fisheries Organization (NAFO) and the International Council for the Exploration of the Sea (ICES) Regions. Most of the uncertainty about the Iceland population centres on estimates of the very abundant auk species which drive the overall population patterns. Further resolution of these estimates is essential.

Trends are considered for all species and for diving and surface-feeding taxa, which often have different sensitivities to climatic and anthropogenic environmental changes. Considerations of marine bird biodiversity are swamped by these most abundant species, although some aspects of species trends and community changes are addressed in the appended regional accounts.

⁵ Information in this subchapter is based on the appended spreadsheets of regional trends, numbers and sources, and can be modified as gaps are filled and new information obtained. Trends in breeding seabird populations from the 1970s/80s through the 2000s are reported where possible, although more often only the most recent decade[s] is available.

The overall picture here indicates that surface-feeders (storm-petrels, gulls, terns) drive the negative NAFO trend, and diving auks (Dovekie *Alle alle*, Thick-billed Murre *Uria lomvia*, Common Murre *Uria aalge*) in Iceland drive the negative ICES trend, with the ICES decrease being six times greater than that reported for NAFO.

Within regional trends considerable variation is observed (Table 36A.6), with different areas exhibiting increasing trends (E Baffin Island, Newfoundland/Labrador, E Canada + US, Faroes) or decreasing trends (W Greenland, Gulf of St. Lawrence, Caribbean, E Greenland, Iceland, Norwegian and North Seas).

5.2 NAFO Area

The negative trend in the NAFO Region is driven by surface-feeding species (gulls, terns, petrels) that are decreasing in eastern Canada (Cotter et al., 2012) and in the Caribbean (Bradley and Norton, 2009). The decline is also driven by an inferred decreasing trend in a diving planktivore (Dovekie) in Western Greenland based on North American Christmas Bird Counts (BirdLife International, 2014). Otherwise, divers are increasing in all regions, with the exception of the Caribbean, where a small population of Brown Pelicans (*Pelecanus occidentalis*) is declining (Bradley and Norton, 2009). Decreasing trends in surface-feeders and increasing trends in diving species are associated with fisheries closures in eastern Canada and the concurrent cessation of discards and gill-net removals (Bicknell et al., 2013; Regular et al., 2013). Surface-feeders are vulnerable to sea-surface temperature perturbations (Schreiber and Schreiber, 1984) and long-line fishing (Zydalis et al., 2009). Some ocean regions, notably the Gulf of Mexico, are data-deficient.

5.3 ICES Area

The decreasing trend in the ICES Regions is overridden by the uncertain negative Icelandic estimates. Positive trends are reported for the Faroes Island (Denmark)/Western United Kingdom and for the Barents Sea (which is excluded from consideration as it is in the Arctic rather than in the North Atlantic region).

Decreasing trends in auks in the Norwegian Sea (Anker-Nilsen et al., 2007) are associated with warming ocean trends and the consumption of forage prey by warm-water predatory fishes (e.g., Atlantic mackerel) moving into the region (T. Anker-Nilsen, pers. comm.).

6. Marine mammals

Many marine mammals primarily inhabit the margins of the North Atlantic Ocean, especially the continental shelf and within the many semi-enclosed regional seas. Other species primarily occupy the North Atlantic gyre, bounded by clockwise flowing currents

most famously defined by the Gulf Stream in the north and the Canary Current in the east (Figure 36A.1). Many of these latter species also utilize habitats in areas further north and south of the gyre, at least seasonally.

The gyre species as identified here include many historically subjected to whaling, including slower whales such as right whales, humpback whales and sperm whales, all targeted by open boat whalers through the nineteenth century. Faster whales, such as blue whales and fin whales, were targeted beginning in the late nineteenth century. Several medium-sized whales were also subject to whaling, with some continuing to be so down to the present, including long-finned pilot whales (targeted, for example, for a millennium in a drive fishery in the Faeroe Islands), the northern bottlenose whale (targeted from ships from the mid-nineteenth to the mid-twentieth century), and minke whales (targeted from ships by twentieth- and twenty-first-century whalers). The effects of whaling range from slight for species such as long-finned pilot whales (Taylor et al., 2008) to near extinction for right whales (Reeves et al., 2007). In recent decades, species such as humpback whales that migrate across the North Atlantic gyre from breeding grounds near islands in the Caribbean have recovered from earlier effects of whaling. However, those humpbacks breeding near the Cabo Verde islands apparently have not (Reilly et al., 2008).

Other gyre cetaceans include short-finned pilot whales, killer whales, pygmy killer whales, various other bottlenose whales and common dolphins. Generally these have not been subject to intense whaling.

In addition to the effects of historical and ongoing whaling, cetaceans in the North Atlantic are subject to various forms of disturbance (see Chapter 37). For example, disturbance and sometimes injury of individual animals by noise, including sound generated from military operations and from seismic operations has been demonstrated for some species of beaked whales (Cox et al., 2006; Whitehead, 2013). Similarly, mortality from ship strikes and entanglement in fishing gear have been demonstrated for humpback whales and right whales in the North Atlantic, and, especially for the small population of right whales that survived whaling, such mortality can be significant (Laist et al., 2001).

Harbour porpoise and common dolphin occupy continental shelf regions in the North Atlantic, and also occur further north. These have been subject to entanglement in fishing gear in many areas, especially bottom-tending gill nets (NOAA, 2014). Pinniped species, such as harbour seals, gray seals and harp seals, generally occur further north, but also occupy northern continental shelf regions around New England in the west and the United Kingdom of Great Britain and Northern Ireland in the east. They are dependent on haul-out areas, beaches and ice cover, and have often been thought to compete with fishermen for prey. Gray and harp seals have been subject to predator control programmes and, especially in the western North Atlantic, commercial harvests. However, both have increased in abundance in recent decades to relatively high levels. For example, in the northwest North Atlantic, gray seals have increased, reaching roughly half a million animals in 2014 (Figure 36A.2).

7. Specific areas of the North Atlantic

The predominance of semi-enclosed seas with characteristic biota around the North Atlantic, particularly the more southern and central portions of the region, and the concentration of human pressures around these seas, result in many important trends in biodiversity being observed most clearly at the scale of these seas. Hence this chapter includes brief summaries of the main patterns of and pressures on biodiversity for a number of these regional seas.

7.1 Black Sea

The Black Sea is a very deep inland sea with an area of 432,000 km². The thin upper layer of marine water (up to 150 m) supports the unique biological life in the Black Sea ecosystem. The deeper and more dense water layers are saturated with hydrogen sulphide, that over thousands years, accumulated from decaying organic matter in the Black Sea. Due to the unique geomorphological structure and specific hydrochemical conditions, specific organisms, basically on the level of protozoa, bacteria, and some multi-cellular invertebrates inhabit the deep-sea waters. Knowledge about biological forms of life in the deep waters of the Black Sea is very limited. The disturbance of the natural balance between the two layers could trigger irreversible damage to the people and ecosystem of the Black Sea⁶.

The recently published evidence raises the number of species, including supra-specific taxa, inhabiting the Black Sea to 5,000 (Gomoiu, 2012).

The distribution diagram of different physiological types of species from the Black Sea fauna shows the coexistence of four categories of species, according to a salinity gradient: (1) marine species, (2) freshwater species, (3) brackish water species, and (4) Ponto-Caspian relic species (Skolka and Gomoiu, 2004). The Black Sea biota consist of 80 per cent of Atlantic-Mediterranean origin species, and 10.4 per cent and 9.6 per cent of species of freshwater and Ponto-Caspian origin, respectively (Shiganova and Ozturk, 2010). The eastern sector is one of the biologically richest regions on Earth and is recognized as a biodiversity hotspot, along with other parts of the Caucasus Biodiversity Hotspot Region (Kazanci et al., 2011).

Genetic studies confirm the recent origin of many Black Sea marine taxa from the Mediterranean. The majority of these taxa most probably entered the Black Sea through the Marmara Sea and the Straits linking the Black Sea and the Marmara Sea after the last glacial maximum, when a connection between the Mediterranean and the Black Sea was re-established (Ryan et al., 1997). For this reason, these Black Sea populations are genetically similar to the Mediterranean ones, although in some cases they have already diverged, implying reduced genetic connectivity (e.g., Durand et al., 2013). There are also cases of Black Sea taxa, such as the copepod *Calanus euxinus*, to which the species

⁶ http://www.blacksea-commission.org/_geography.asp

status has been attributed, although they have only recently diverged (Papadopoulos et al., 2005). On the other hand, genetic studies have confirmed the ancient origin of the Ponto-Caspian species and have even revealed an additional diversity in the form of cryptic species (Audzijonyte et al., 2006).

The migration of marine species from the Mediterranean is hampered by a number of ecological barriers: (1) low salinity and ionic composition and the difference in the thermal conditions, (2) the presence of hydrogen sulphide in the Black Sea bottom areas, and (3) the lack of tides (Skolka, 2004; Gomoiu, 2004).

Our current knowledge on the biodiversity of certain taxa in the region shows a well-defined zoogeocline from the Marmara Sea and Bosphorus Strait to the inner parts of the region (Azov Sea), depicted both as a pattern in overall species composition and species (or taxa) numbers. As a general trend, species numbers decrease along with the decrease in salinity towards the inner parts of the basin. The trend is homologous to that seen in the benthic invertebrate inventories of all the major European semi-enclosed regional seas. Salinity and food availability appear to be the dominant abiotic factors correlated, though weakly, with the various patterns deriving from the taxonomic/zoogeographic categories (Surugiu et al., 2010).

The invasion of the basin by alien species began in the Middle Ages, with the bivalve *Teredo navalis*, as the first one recorded. The anthropogenic disturbance gradually increased and reached an unprecedented amplitude, becoming evident during the 1950s, concurrently with the penetration of the Indo-Pacific predator gastropod *Rapana thomasi*, which has severely reduced stocks of native oysters - *Ostrea sublamellosa* (Gomoiu, 1998; Skolka, 1998).

The invasive species may affect not only the ecosystem but also various sectors of the economy, with devastating effects for some of these sectors. This is the case of the American comb jelly, *Mnemiopsis leidyi*, accidentally introduced into the Black Sea by ship ballast water in the early 1980s. The introduced comb jelly nearly led to the collapse of pelagic fish populations (over 26 commercial Black Sea fish stocks), and finally caused a major shift in the marine ecosystem. Only after the penetration of a new warm-water ctenophore, *Beroe ovata*, ten years later, did the *M. leidyi* population diminish, allowing the ecosystem to recover its entire trophic web.

Another particularly important case of accidental immigration is that of toxic microscopic algae, whose outbreaks produce the so-called toxic blooms. The biotoxins produced inside the algal cells (i.e., domoic, okadaic, yessatoxine or azaspiracids acids), can have toxic effects on the other taxa and even on humans. The vast majority of these species, such as *Noctiluca scintillans*, also can reduce or deplete the oxygen concentration in the water column and sediments, leading to hypoxia or anoxia.

During recent decades, the temperature increased both in the surface mixed and in the cold intermediate water layers. This has been shown to be another factor accelerating the establishment of more thermophilic species populations, promoting their northward expansion from the Mediterranean (Shiganova and Ozturk, 2010).

In the conditions present during periods after major eutrophication outbreaks have returned towards more typical states, two antagonistic and synergistic processes have taken place: (a) the penetration of some opportunistic species, and (b) the disappearance of some economically valuable native species. In 1999, the first Black Sea Red Book was issued, which includes 160 endangered and rare plant and animal species. Sturgeons are the most endangered species, along with those that inhabit shallow coastal waters (turbot, sharks), seals, shrimp and oyster species.

The same critical status is also attributed to most of the coastal margin neritic and open sea habitats, and to 13 out of 37 benthic habitats. Among the habitats at risk are included the neritic water column, coastal lagoons, estuaries/deltas and wetlands/saltmarshes (Goriup, P., 2009).

In the Black Sea Region, almost a quarter of the habitat types listed in the European Union (EU) Habitats Directive can be found. Many are located in the intertidal zone and are consequently heavily influenced by the presence of salt water and continuous wave action. They include extensive areas of mud and sand flats, salt meadows and marshes, and long stretches of white sandy beaches (Sundseth and Barova, 2009).

Coastal forests are also well represented, especially on the low-lying hills in the south of Bulgaria and within the Danube Delta. They include a variety of rare habitat types listed in the Habitats Directive, such as the Western Pontic beech forests, as well as floodplain forests, alluvial and mixed riparian forests, all of which are important roosting and resting habitats for bats and birds (Sundseth and Barova, 2009).

7.2 *Mediterranean Sea*

The Mediterranean Sea is a marine biodiversity hot spot. Approximately 17,000 marine species have been reported from the Mediterranean Sea (Coll et al., 2010). Of these, at least 26 per cent are prokaryotic (Bacteria and Archaea) and eukaryotic (Protists) marine microbes. Phytoplankton includes more than 1,500 species. Macrophytes include approximately 850 species. Among microzooplankton, the foraminifera are the main group with more than 600 species, and about 100 species are commonly present in Mediterranean waters (Dolan, 2000). However, it is within the Animalia group that there is published evidence for the majority of the species so far reported (~11,500) with the greatest contribution coming from the Crustacea (13.2 per cent) and the Mollusca (12.4 per cent), followed by the Annelida (6.6 per cent), the Platyhelminthes (5.9 per cent), the Cnidaria (4.5 per cent), the subphylum Vertebrata (4.1 per cent), the Porifera (4.0 per cent), the Bryozoa (2.3 per cent), the subphylum Tunicata (1.3 per cent), and the Echinodermata (0.9 per cent). With regard to the Vertebrata, there are ~650 marine species of fish, of which approximately 80 are elasmobranchs and the rest are mainly from the Actinopterygii class (86 per cent). Nine species of marine mammals (five belong to the dolphins, Delphinidae, and one each to the Ziphiidae, Physteridae, Balaenopteridae, and Phocidae (seals)) and three species of sea turtles (the green *Chelonia mydas*, the loggerhead *Caretta caretta* and the leatherback *Dermochelys*

coriacea) are regularly recorded in the Mediterranean Sea. A total of 15 species of seabirds frequently occur in the Mediterranean Sea (Coll et al., 2010).

However, estimates of marine diversity are still incomplete as yet-undescribed species will be added in the future (Coll et al., 2010; Danovaro et al., 2010). In many cases, several cryptic species, mainly invertebrates, have been revealed through molecular approaches (Calvo et al., 2009), thus increasing the number of reported species. Moreover, diversity for microbes is substantially underestimated, and the deep-sea areas and portions of the southern and eastern regions are still poorly known (Coll et al., 2010). The next generation of sequencing data are already producing a wealth of unique sequences, in unprecedented rates, many of which are new operational taxonomic units (OTUs). Tens of thousands of OTUs are produced in the context of minimal monitoring projects (Pavloudi et al., in press), a fact which will soon alter the numbers so far reported. In addition, the invasion of alien species is a crucial factor that will continue to change the biodiversity of the Mediterranean (Zenetos et al., 2010), mainly in its eastern basin, where invading species can spread rapidly northwards and westwards due to the warming of the Mediterranean Sea (Lejeusne et al., 2010).

Genetic diversity, especially the presence of genetically distinct populations within a species, is another important component of biodiversity. Although the lack of strong physical barriers in the marine environment and the high dispersal ability of many marine taxa tend to diminish genetic differentiation, several marine species within the Mediterranean exhibit a strong genetic structure as a result of their life history traits and of the complex geography and hydrography of the Mediterranean. Diverse taxa, from small pelagic fish, like anchovy (Magoulas et al., 2006), to molluscs (Cordero et al., 2014) have been shown to consist of genetically distinct populations, with low connectivity, which calls for more local-scale management and conservation actions, especially for commercially exploited or vulnerable species. This differentiation often occurs across well-known biogeographic barriers, like the Almeria-Oran front (Patarnello et al., 2007) and the Siculo-Tunisian strait (Mejri et al., 2009).

Spatial patterns of species diversity show a general gradient characterized by the decreasing number of species from the west to the east (Arvanitidis et al., 2002, 2009; Coll et al., 2010). For certain taxa such as polychaetes, indications of habitat diversification, such as the average island distance from the nearest coast, number of islands and island surface area, have been reported to be best correlated with their multivariate community patterns (Surugiu et al., 2010). The decrease in species richness gradient has also been attributed to a synergy of variables, such as food availability, salinity (Surugiu et al., 2010) and current knowledge gaps with regard to the biota over large sectors, such as the northern African and eastern coasts (Coll et al., 2010; Coll et al., 2012). Lower productivity rates (oligotrophism) are related to the significantly lower size of the species in the eastern basin, a phenomenon known as “Levantine nanism” (dwarfism) (Por, 1989). Biodiversity is also generally higher in coastal areas and continental shelves, and with some exceptions it decreases with depth (Coll et al., 2010; Danovaro et al., 2010). However, fish biodiversity components, measured as species

richness of total, endemic and threatened coastal fish assemblages, as well as their functional and phylogenetic diversity, have been mapped and described as spatially mismatched between regions of the Mediterranean Sea (Mouillot et al., 2011).

The Mediterranean Sea is also diverse in terms of habitats and ecosystem types, due to its unique biogeography (Bianchi et al., 2012). Although empirical data are insufficient to have a full representation of habitat types (Danovaro et al., 2010; Levin et al., in press) and are only fully available for some coastal habitats (Giakoumi et al., 2013), a series of surrogates or modelling techniques are used to characterize marine habitats in the whole Mediterranean basin (Micheli et al., 2013; Martin et al., 2014).

Temporal trends have indicated that overexploitation of some fish and macro-invertebrates and habitat loss have been the main human drivers of historical changes in biodiversity (Coll et al., 2010; Lotze et al., 2011; Coll et al., 2012). At present, habitat loss and degradation, followed by fishing, climate change, pollution, eutrophication, and the establishment of invasive species, are the most important factors that affect most of the taxonomic groups and habitats (Claudet and Fraschetti, 2010; Coll et al., 2010; Abdul Malak et al., 2011; Lotze et al., 2011; Bianchi et al., 2012; Coll et al., 2012; Micheli et al., 2013). All these impacts are expected to grow in importance in the future, especially climate change and habitat degradation.

7.3 *Baltic Sea*

The Baltic Sea is a small sea on a global scale, but as one of the world's largest and most isolated bodies of brackish water, it is ecologically unique. Eutrophication, caused by nutrient pollution, is a major concern in most areas of the Baltic Sea. The biodiversity status was classified as being unfavourable in most of the Baltic Sea, as only the Bothnia Sea and some coastal areas in the Bothnian Bay were classified as having an acceptable biodiversity status. The results indicate that changes in biodiversity are not restricted to individual species or habitats; the structure of the ecosystem has also been severely disturbed (Helcom, 2010).

Baltic Sea biodiversity and human pressures on it have been summarized for all components except bacteria in Helcom (2009a) integrated thematic assessment. Alongside the general deterioration of the Baltic Sea biodiversity positive signs were also found for grey seals and some fish and bird species. A recent expert evaluation of endangered species in the Baltic Sea by Helcom (2013) shows the risk for extinction among a number of plant and animal species still existing.

The Baltic Sea is characterized by large areas (ca 30 per cent) that are less than 25 m deep, interspersed by a number of deeper basins with a maximum depth of 459 m. The western and northern parts of the Baltic have rocky seabeds and extended archipelagos; the seafloor in the central, southern, and eastern parts consists mostly of sandy or muddy sediment (ICES, 2008b).

The Baltic Sea phytoplankton community is a diverse mixture of microscopic algae representing several taxonomic groups, with more than 1,700 species recorded. The species composition of the phytoplankton depends on local nutrients and salinity levels and changes gradually from the southwest to the northeast. Primary production exhibits large seasonal and interannual variability (Helcom, 2002). In the southern and western parts, the spring bloom is dominated by diatoms, and by dinoflagellates in the central and northern parts (Helcom, 2002, 2009a).

Cyanobacteria are a natural component of the phytoplankton community in most parts of the Baltic Sea area. They usually dominate in summer in the coastal and open areas of most sub-basins of the Baltic Sea, with the exception of the Belt Sea and the Kattegat. Cyanobacterial blooms in the Baltic Proper are typically formed by the diazotrophic species *Aphanizomenon flos-aquae*, *Anabaena* spp. and *Nodularia spumigena* that can fix molecular nitrogen. *N. spumigena* blooms are potentially toxic, whereas no toxic blooms of *A. flos-aquae* have been recorded in the Baltic Sea. The blooms of N₂-fixing cyanobacteria as such do not necessarily indicate strengthened eutrophication (Helcom, 2009b).

The zooplankton of the Baltic Sea is dominated by calanoid copepods and cladocerans. The species composition is influenced by the salinity gradient. Generally marine species (e.g., *Pseudocalanus* spp.) prevail in the southern more saline part, while brackish species (e.g., *Eurytemora affinis* and *Bosmina longispina maritima*) dominate in the northern areas (ICES, 2008b). The latitudinal distribution of marine macrozoobenthos in the Baltic Sea is limited by the gradient of decreasing salinity towards the north. The decreasing salinity reduces macrozoobenthic diversity, affecting both the structure and function of benthic communities. In addition, the distribution of benthic communities is driven by strong vertical gradients. Generally, the more species-rich and abundant communities in shallow-water habitats (with higher habitat diversity) differ from the deep-water communities, which are dominated by only a few species (Helcom, 2009a).

The composition of the benthos depends both on the sediment type and salinity level, with suspension-feeding mussels important on hard substrata and deposit feeders and burrowing forms dominating on soft bottoms. The species richness of the zoobenthos is generally poor and declines from the southwest towards the north due to the drop in salinity. However, species-poor areas and low benthos biomass are also found in the deep basins in the central Baltic due to the low oxygen content of the bottom water (ICES, 2008b).

The distribution of the roughly 100 fish species inhabiting the Baltic is largely governed by salinity levels. Marine species (some 70 species) dominate in the Baltic proper, and fresh-water species (some 30–40 species) occur in coastal areas and in the innermost parts (Nellen and Thiel, 1996, cited in Helcom, 2002). Cod, herring, and sprat comprise the large majority of the fish community in both biomass and numbers. Commercially important marine species are sprat, herring, cod, various flatfish, and salmon. Sea trout and eel, once abundant, are now in very low populations. Sturgeon was a very important component of local exploited fish fauna for centuries, especially in the

southern Baltic. Currently, sturgeon is a red-listed fish in the Baltic Sea and a reintroduction programme has been initiated (Helcom, 2009a). The seabirds in the Baltic Sea comprise pelagic species like divers, gulls, and auks, and benthic-feeding species like dabbling ducks, sea ducks, mergansers, and coots (ICES, 2003). The Baltic Sea is more important for wintering (ca 10 million) than for breeding (ca 0.5 million) seabirds and sea ducks. The common eider exploits marine waters throughout the annual cycle, but ranges from being highly migratory (e.g., in Finland) to being more sedentary (e.g., in Denmark).

The marine mammals in the Baltic consist of gray (*Halichoerus grypus*), ringed (*Phoca hispida*), and harbour seals (*Phoca vitulina*), and a small population of harbour porpoise (*Phocoena phocoena*). Seals and harbour porpoise were much more abundant in the early 1900s than they are today (Elmgren, 1989; Harding and Härkönen, 1999) where their fish consumption may have been an important regulating factor for the abundance of fish (MacKenzie et al., 2002). Baltic seal populations – harbour seals, gray seals, and ringed seals – are generally increasing. The recent abundance of the harbour porpoise in the Baltic Proper is low (Helcom, 2009a).

7.4 North Sea

The North Sea is a large semi-enclosed sea on the continental shelf of northwest Europe, formed by flooding in the Holocene period. The sea is shallow, deepening towards the north. The seabed is predominantly sandy, but muddy in deeper parts and in southern coastal areas with extensive river influence.

The strong coupling between benthic and pelagic communities in the shallow parts of the North Sea makes it one of the most productive marine areas in the world, with a wide range of plankton, fish, seabirds and benthic communities.

The most commonly found zooplankton genus in the North Sea is of the copepod *Calanus*. Hays et al. (2005) observed between 1960 and 2003 a clear decrease in the abundance of *Calanus finmarchicus*, and an increase in *C. helgolandicus*, with a marked overall decrease in both species combined. Beaugrand et al. (2002) also found a decrease in the abundance of cold water and Arctic zooplankton species and an increase in warmer water ones.

The 50-m, 100-m, and 200-m depth contours broadly define the boundaries between the main benthic communities in the North Sea (Künitzer et al., 1992; Callaway et al., 2002). Bottom temperature, sediment type, and trawling intensity have been identified as the main environmental variables affecting local community structure. Epifaunal communities are dominated by free-living species in the south and by sessile species in the north.

Throughout the year, the pelagic fish component is dominated by the herring *Clupea harengus*. The mackerel *Scomber scombrus* and the horse mackerel *Trachurus trachurus* are mainly present in summer, when they enter the area from the south and from the northwest. Dominant gadoid species are the cod *Gadus morhua*, the haddock

Melanogrammus aeglefinus, the whiting *Merlangius merlangus*, and the saithe *Pollachius virens*; the main flatfish species are the common dab *Limanda limanda*, the plaice *Pleuronectes platessa*, the long rough dab *Hippoglossoides platessoides*, the lemon sole *Microstomus kitt*, and the sole *Solea vulgaris*. The major forage fish species are the sandeel *Ammodytes marinus*, the Norway pout *Trisopterus esmarki*, and the sprat *Sprattus sprattus*, but juvenile herring and gadoids also represent an important part of the forage stock. However, large annual variations in species composition occur as a consequence of natural fluctuations in the recruitment success of individual species. Fish species richness is highest around the edges of the North Sea (particularly along the coast of Scotland, in the Southern Bight, and in the Kattegat) and lowest in the central North Sea (ICES, 2008).

Certain highly migratory species that historically were fairly common in the North Sea have become very rare (e.g., tunas and the halibut *Hippoglossus hippoglossus*). The stocks of most elasmobranchs are at low levels (ICES, 2008). The spurdog (*Squalus acanthias*) was the most common shark species, but is now considered to be depleted to approximately 5 per cent of its virgin biomass in the whole Northeast Atlantic (Hammond and Ellis, 2005). Decades of intensive fishing have been shown to have altered both the species (ICES, 2008; Piet et al., 2009; Greenstreet et al., 2010) and size composition (Daan et al., 2005) in the North Sea, with greatest effects where fishing has been most intense. There is some evidence that these effects are being reversed since fishing pressure was reduced in the late 2000s, but community metrics are still far from their values observed prior to the 1970s (Greenstreet et al., 2011).

About 2.5 million pairs of seabirds, belonging to some 28 species (ICES, 2008) breed around the coasts of the North Sea. Although most species breed in dense colonies along the coast, they make very different use of the marine ecosystem. During the breeding season, some species depend on local feeding conditions within tens of kilometres around their colony, whereas others may cover several hundreds of kilometres during their foraging trips. Outside the breeding season, some species stay quite close to their breeding grounds, and others migrate across the North Sea or elsewhere, even as far as the Antarctic. Feeding habits also diverge. Auks and cormorants dive from the surface, gannets and terns use plunge diving, and gulls feed mostly from the surface. A few (especially skuas) are kleptoparasites (Dunnet et al., 1990). Their food resources vary accordingly, ranging from plankton to small schooling fish and discards. Because of all these differences, seabirds do not represent a single homogeneous group that responds to fisheries in a uniform way. A few species profit directly from human consumption fisheries, either discards or offal, e.g., fulmars and gulls. Current seasonal distributions, status, and trends of these species have shown an increasing trend over the last century. Auks and cormorants are now protected in some areas (e.g., the southern North Sea and Kattegat). Gull numbers have been controlled in many areas. Fulmars may have benefited from the expansion in fishing. Skuas may have profited directly from the increase in population size of seabirds in general. On a shorter time scale, 12 out of 28 species show an increasing trend during the last decade and

four a decreasing trend; four appear to be stable and for another four the situation is unknown (ICES, 2008).

Many cetacean and pinniped species have been observed within the North Sea, but most of these must be considered vagrants and only a few are resident representatives of the North Sea ecosystem. Harbour *Phoca vitulina* and gray *Halichoerus grypus* seals have undergone large population changes over the past century. Populations of harbour seals along the continental coast reached an all-time low in the 1970s. Subsequently, these populations increased steadily at an annual rate of 4 per cent, with two major interruptions in 1988 and 2002, when the populations were hit by outbreaks of the phocine distemper virus (ICES, 2008).

Although several cetacean species visit the North Sea, the dominant species are minke whales *Balaenoptera acutorostrata*, harbour porpoises *Phocoena phocoena*, and whitebeaked dolphins *Lagenorhynchus albirostris*. Preliminary abundance estimates from a survey conducted in 2005 indicate the *status quo* for all these species. Harbour porpoises, however, have shifted their focal distribution from the northern part of the North Sea to the southern part. The reasons for this southward shift of harbour porpoise distribution are unknown; however, a change in distribution and availability of prey species is considered the most likely explanation, although other explanations are possible (ICES, 2008).

A number of sand banks across the North Sea qualify for protection under the European Union Habitats Directive, mainly along the UK coast, the eastern Channel, the approaches to the Skagerrak, and the Dogger Bank. Extensive biogenic reefs of *Lophelia* have recently been mapped along the Norwegian coastline in the eastern Skagerrak, and *Sabellaria* reefs have been reported in the south, although their distribution and extent are not known. Gravel deposits also qualify for protection, but comprehensive maps at a total North Sea scale are not readily available (ICES, 2008).

7.5 Gulf of St. Lawrence

The Gulf of St. Lawrence (hereafter the Gulf) is a relatively small (236,000 km²) sea located along the southern Canadian Atlantic in the Northwest Atlantic connected to the ocean by the shallow (60 m) and narrow (17 km) Strait of Belle Isle in the northeast, where dense Labrador Shelf water enters the Gulf and the deep (480 m) and wide (104 km) Cabot Strait in the south (Figure 36A.2bis). The Gulf receives large quantities of freshwater from the St. Lawrence River system, more than half of the freshwater inputs from the east coast of North America, which gives the Gulf an estuarine-like circulation. The Gulf has a complex geomorphology, with a broad shallow shelf (<100 m) in the south and a northern region characterized by narrow coastal shelves and deep (>300 m) channels. The environment is influenced by climate variability in the Northwest Atlantic. The seasonality is characterized by severe winters with low temperatures and ice covering a large surface from December to April, and by warm surface water in summer. The sea surface temperature differences between winter (less than 0°C) and summer (close to 20°C in the south) are among the largest recorded around the Atlantic. The

Gulf's multilevel complexity provides the conditions for a highly diverse fauna, with a mixture of boreal, temperate and sub-tropical species, and productive biological communities (Benoît et al., 2012).

Global climate change, including ocean warming affects the Gulf as indicated by a clear decreasing trend in winter ice coverage and volume and increasing sea surface temperatures in recent years (DFO, 2013). The characteristics of the deep layer are influenced by the quality and relative proportions of the deep Atlantic water masses from which it originated. Low levels of dissolved oxygen (and associated low pH levels) are found in the deep channels and some areas (e. g., the head of the Laurentian Channel) have been hypoxic (oxygen concentration < 20 per cent) since the mid-1980s (Benoît et al., 2012). Appropriate oxygen and pH levels are important conditions for the health of the organisms; the observed levels have been attributed in part to local biological oxygen consumption in the deep layer, but also to changes in the source water masses from the North Atlantic. About 1 M people live in periphery of the Gulf and depend (82 per cent of regional employment, 79 per cent of regional GDP) directly on marine activities (DFO, 2013). The St. Lawrence River upstream and other tributaries from the urban and industrial zones are sources of contaminants that, combined with human activities in the coastal zone (e.g., agriculture/aquaculture, habitat destruction/modification, nutrient loading), can have disturbing effects on the ecosystem. The Gulf is also an important maritime route and the heavy commercial shipping connecting the Atlantic (and other oceans of the world) with the Great Lakes industrialized region in mid-North America represent another pressure (e.g., noise, potentially invasive species from ballast water) on the ecosystem. Human activity may have directly contributed to the establishment of 20 non-indigenous (alien) aquatic species in the Gulf; half of these since 1994 (SOTO, 2012). For decades, if not centuries, the principal human activity in the Gulf has been fishing, both commercial and recreational. That may change in the near future, however, with the development of oil and gas exploitation projects.

The timing of the phytoplankton bloom is affected by the timing of the winter ice retreat and water stratification. The bloom may have been earlier in the late 1990s than today, but no clear trend in timing and intensity is detected (Dufour et al., 2010). However, systematic monitoring of the lower level of the ecosystem (phytoplankton, zooplankton) is recent, less than 20 years, for the Gulf, and temporal trends are difficult to detect from the important interannual variability. Phytoplankton community structures vary regionally but, generally, diatoms (the group of phytoplankton typically associated with intense spring blooms in productive northern seas) and small-celled planktonic organisms (e.g., dinoflagellates) are found in equal proportions (Bérard-Therriault et al., 1999). Recently, decreasing diatom/dinoflagellate and flagellate ratios have been observed due to an increase in dinoflagellates and flagellates that may reflect increasing stratification, temperature and nutrient loading in the system (Dufour et al., 2010). Ecosystems dominated by dinoflagellates and flagellates are less productive. The Crustacea are the most important group (60 per cent) in the zooplankton, and the Copepoda account for 70 per cent of all species (Brunel et al., 1998). There are close

associations between the abundance of some small copepod species, the young stages of larger species and feeding by young fish. In the Gulf, all copepods seem to have increased around 2005, but the abundance of the key species *Calanus finmarchicus* has been below average since 2009 (DFO, 2013). Although there are important uncertainties about total biomass and trend, high concentrations of krill (Euphausiids) are found at specific sites and large numbers of blue whales from the Northwest Atlantic population migrate to the Gulf to feed (Gagné et al., 2013). At least 12 species of whales migrate to the Gulf every year, which makes the Gulf (and the Estuary) one of the best whale-watching sites in the World. In addition, the head of the Laurentian Channel in the Estuary is the refuge of an endangered Beluga whale population. Despite conservation efforts (control on contaminants, noise, and a protected areas project), the Beluga whale population shows no sign of increase; at present it is, at best, stable.

The total number of fish and invertebrates species is approximate, due in part to the poor sampling of the shallow or coastal zones. The current inventory lists some 2200 marine invertebrate species (Brunel et al., 1998). An analysis of scientific bottom trawl survey records indicates that ~130 species of fish may be present in the northern Gulf alone (Dutil et al., 2006, 2009). The total fish biomass is dominated by a small number of large species, most of commercial interest; hence many species are relatively rare or occasional visitors. Nonetheless, the importance of the habitat is shown by at least 48 species of ichthyoplankton (eggs and larval fish) recorded in the Gulf. As for the Canadian shelf regions, dramatic shifts to both the northern and southern Gulf ecosystems occurred in the late 1980s, particularly in response to fishing and, to a lesser extent, to changes in environmental conditions. These shifts include changes in species abundance and/or biomass and food web structure and functioning (Savenkoff et al., 2008) (Figure 36A.3). The ecosystems that were dominated by large demersal fish predators (e.g., Atlantic cod, redfish and white hake) are today dominated by small-bodied forage species and invertebrates. The biomass of shrimps (close to 20 species) has increased since the early 1980s and at an accelerated rate beginning in the early 1990s. Despite a 15-year moratorium on harvesting, the ecosystem structure has not returned to its previous state. In the southern Gulf, the natural mortality of cod remained high, causing a decline in their abundance. Evidence is mounting that predation by top predators (e.g., seals, whose abundance has been increasing) is the cause of this mortality (Benoît et al., 2012).

Fishing can affect population structure, entire communities, and the habitat itself. The only important bottom trawl use remaining in the northern Gulf is by the northern shrimp fishery. The fishery is well regulated and the use of a separator grate is mandatory to reduce the catch of large fishes. Nonetheless, an analysis in 2012 showed small specimens from 69 taxa (the majority in low numbers) as remaining in the catch (Savard, 2013).

7.6 *Chesapeake and other coastal estuaries and bays of the United States*

The Chesapeake Bay estuarine system⁷ supports more than 3,000 species of plants and animals (Table 36A.7). A subset of species has been identified as being *ecologically valuable*⁸ (Table 36A.8), based on their importance in (1) regulating the flow of carbon through the food web, (2) providing habitat, and/or (3) supporting ecosystem services (Baird and Ulanowicz, 1989; Costanza et al., 1997; Jordán, 2001; Martínez et al., 2007; Koch et al., 2009). Marshall et al. (2005) documented 1454 phytoplankton species in the Bay during 1984-2004; diatoms were the most abundant taxon (Table 36A.6). The ratio of planktonic centric diatom species to benthic pennate diatom species increased from ~ 1 prior to European settlement to ~ 5 today (Cooper and Brush, 1991), a trend that coincided with a decrease in species richness and an increase in phytoplankton biomass due to increases in anthropogenic nutrient loading and a rapid decline in the abundance of filter feeders (oysters) during the twentieth century (Newell, 1988; Cooper and Brush, 1991; Marshall et al., 2003; Kemp et al., 2005). Projected increases in water temperature and winter-spring precipitation associated with climate change are likely to enhance these trends and promote the growth of toxic dinoflagellates (Pyke et al., 2008).

Species richness is correlated with fisheries productivity (Worm et al., 2006). Historically important fisheries in the Bay included striped bass (*Morone saxatilis*), Atlantic sturgeon (*Acipenser oxyrinchus*), American shad (*Alosa sapidissima*), Atlantic menhaden (*Brevoortia tyrannus*), blueback herring (*Alosa aestivalis*), alewife (*Alosa pseudoharengus*), soft-shelled clam (*Mya arenaria*), eastern oyster (*Crassostrea virginica*), and blue crab (*Callinectes sapidus*). Of these, landings of sturgeon,⁹ shad,¹⁰ soft-shelled clams¹¹ and eastern oysters (Rothschild et al., 1984; Wilberg et al., 2011) experienced dramatic declines (> 98 per cent) during the twentieth century due to overfishing and habitat loss. Herring¹² and menhaden¹³ landings have declined by ~80 per cent.

Habitat loss has been expressed primarily in terms of increases in the spatial extent of summer hypoxia and decreases in the spatial extent of tidal marshes, submerged vascular plant beds, and oyster reefs. Oyster reefs, submerged vascular plant beds and

⁷ Chesapeake Bay (including tidal waters of its tributaries) is the largest estuary in the U.S. (~ 320 km long, 11,700 km²). It is a partially stratified, coastal plain estuary (drowned river valley) with a mean depth of 8.4 m and three salinity zones: oligohaline, 0 – 10 psu; mesohaline, 11 – 18 psu; and polyhaline, 19 – 36 psu (Schubel and Pritchard, 1987). Eleven major rivers flow into the Bay through six states and a drainage basin of 172,000 km². River flows are typically highest during spring and lowest during summer. The climate is moderate with mean water temperatures ranging from a winter low of ~ 5°C to a summer high of ~25°C. With a large ratio of watershed to estuarine area (14:1), the Bay is closely connected to the landscape. With a relatively long residence time of water (~ 6 months), the Bay is susceptible to impacts from land-based inputs of nutrients and toxic contaminants (Kemp et al., 2005).

⁸ The Ecologically Valuable Species Workgroup of the Living Resources Subcommittee for the Chesapeake Bay Program (<http://nepis.epa.gov/Exe/ZyPDF.cgi/P1001WSJ.PDF?Dockey=P1001WSJ.PDF>).

⁹ <http://www.dnr.state.md.us/fisheries/fishfacts/atlanticsturgeon.asp>

¹⁰ <http://www.fws.gov/chesapeakebay/SHAD.HTM>

¹¹ http://www.dnr.state.md.us/irc/docs/00000260_04.pdf

¹² <http://www.dnr.state.md.us/fisheries/fishfacts/herring.asp>

¹³ <http://www.asmfc.org/species/atlantic-menhaden>

tidal marshes support high species diversity (Heck and Orth, 1980; Orth et al., 1985; Newell, 1988; Chambers et al., 1999; Coen et al., 1999; Jackson et al., 2001; Wyda et al., 2002; USACE, 2009; Philine et al., 2012). During the course of the twentieth century, the spatial extent of these habitats declined significantly: oyster reefs by 92 per cent (USACE, 2009; Wilberg et al., 2011), submerged vascular plants by 65 per cent (Kemp et al., 1983, 2005; Orth and Moore, 1983, 1984) and marshes by 60 per cent.¹⁴

Sea level rise is expected to result in even greater losses of marshes, putting hundreds of species of fish, invertebrates and birds at risk (Titus and Strange, 2008), and estuarine acidification poses a significant threat to oyster restoration efforts in the Bay (USACE, 2009; Waldbusser et al., 2011; Sanford et al., 2014). Loss of these habitats exacerbates the impacts of overfishing and is one of the main pressures on species richness, often leading to species extirpation (Orth and Moore, 1983, 1984; Ruiz et al., 1993; Duarte et al., 2008; Heck et al., 2008; Keppel et al., 2012).

Recurring deep-water hypoxia (Cooper and Brush, 1991; Malone, 1991)¹⁵ represents a major loss of pelagic and benthic habitat during a critical period for reproduction and growth of benthic macrofauna and fish, resulting in declines in their abundance (Llansó, 1992; Ruiz et al., 1993; Baird et al., 2004; Kemp et al., 2005; Buchheister et al., 2013) and the threat of extirpation of commercially valuable species that have been overfished, e.g., *Acipenser oxyrinchus* (Secor et al., 2000) and *C. virginica* (Wilberg et al., 2011). Projected increases in climate-driven water temperature and winter-spring precipitation over the twenty-first century may increase the pressure of summer hypoxia on species richness (Pyke et al., 2008), an impact that may be exacerbated by the direct effects of rising water temperatures: e.g., *M. arenaria* is near its southern distribution limit and may be extirpated if summer temperatures approach and remain near 32°C, and temperate fish species such as white perch (*Morone americana*), striped bass (*Morone saxatilis*), and summer flounder (*Paralichthys dentatus*) may experience similar fates (Najjar et al., 2010).

More than 170 known or possible non-native species have invaded the Bay.¹⁶ Of these, at least eight are potentially major threats to species richness in the Chesapeake Bay estuarine system (Table 36A.7).¹⁷

Given the collective importance of vegetated habitats and oyster reefs as refugia for a broad diversity of species and the impacts of fishing, seasonal hypoxia and invasive species, it is highly likely that the species diversity of the Chesapeake Bay estuarine system significantly declined during the 20th C. Continued declines in habitat extent, combined with the impacts of seasonal hypoxia and climate-driven sea level rise, estuarine warming, and acidification, portend increases in the rate of extirpations and declines in species diversity during the twenty-first century. Given similarities in

¹⁴ <http://chesapeakebay.noaa.gov/wetlands>

¹⁵ http://mddnr.chesapeakebay.net/eyesonthebay/documents/DeadZoneStatus_Summer2013.pdf

¹⁶ <http://invasions.si.edu/nemesis/chesapeake.html>

¹⁷ <http://www.mdsg.umd.edu/topics/aquatic-invasive-species/aquatic-invasive-species>

pressures, all of the estuaries of the Virginian Province¹⁸ (Hale et al., 2002) are likely to exhibit similar trends in their capacity to support species diversity.

7.7 Caribbean Sea

The Caribbean is the most biologically diverse area of the Atlantic Ocean, hosting approximately 10 per cent of the world's coral reefs, including the Mesoamerican Barrier Reef System; extensive coastal mangroves and shallow banks with seagrass communities; as well as sandy beaches, rocky shores and many bays, lagoons and brackish estuaries. The Caribbean also has open-ocean and lesser-known deep-sea environments, and has been listed as a global-scale hotspot of marine biodiversity (Roberts et al., 2002).

The Caribbean Sea receives primarily oligotrophic, high-salinity North Atlantic water from the North Equatorial Current, but it also receives South Atlantic water entrained in North Brazil Current rings which transport water from the Amazon into the Caribbean basin via the Guiana Current (Cowen et al., 2003). The persistent through-flow of the warm Caribbean Current is modulated by a highly complex and variable pattern of mesoscale eddies (Lin et al., 2012) and upwelling along the South American coastline. Two significant South American rivers, the Orinoco and the Magdalena, also discharge directly into the southern Caribbean. The considerable spatial heterogeneity of physical environments and habitats across the Caribbean Sea influences the distribution, population connectivity and biodiversity of marine organisms found there. Several significant barriers to gene flow in Caribbean reef populations have been recognized (Cowen et al., 2006). This has led to relatively high levels of endemism. Miloslavich et al. (2010) estimate a value of 25.6 per cent regional endemism across 21 of 78 marine taxa examined in the Caribbean, with values ranging from 45 per cent for fish, 26 per cent for molluscs and 2 per cent for copepods. They also summarize the diversity, distribution and key threats to marine biota in the Caribbean and conclude that the 12,046 species currently reported is a gross underestimation, considering that the marine biota is far from well known in this area.

The significant drivers of declines in Caribbean marine biodiversity are overexploitation and environmental degradation. These are being exacerbated by external drivers, including climate variability and change, and alien species invasions. Iconic Caribbean mega-vertebrates have suffered from historical overexploitation (Jackson, 1997), including the now extinct West Indian monk seal (*Monachus tropicalis*); the endangered Caribbean manatee (*Trichechus manatus manatus*); the North Atlantic humpback whale (*Megaptera novaeangliae*); and marine turtles, of which five species are found here, all endangered. The number of fishery stocks that are fully exploited or overexploited has grown over the last few decades and total landings have declined significantly since the late 1980s, driven by increasing market demand, inadequate fisheries management, and

¹⁸ Chesapeake Bay, Delaware Bay, Hudson-Raritan system, and Long Island Sound
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exacerbated by habitat degradation (Salas et al., 2011; Sea Around Us Project, www.seaaroundus.org).

The abundance of reef fishes has decreased region-wide (Paddack et al., 2009). Most fishable reef species are fully exploited or overexploited; those most vulnerable to fishing are now rare (Roberts, 2012). Notable is the Nassau grouper (*Epinephelus striatus*), once of great commercial importance and now endangered (Sadovy, 1999). There is concern regarding the decline of key functional groups on reefs, especially herbivores, such as parrotfishes, that are vital to reef resilience (Mumby et al., 2006). Large highly migratory pelagic species, such as the billfishes (swordfish and marlins), have suffered significant population declines from fishing by foreign fleets operating in the Atlantic.

Caribbean coral reefs are considered globally unique (UNEP, 2005). Overexploitation and deterioration of coastal water quality (primarily due to high nutrient, sediment and bacterial loads, and toxins from domestic, agricultural and industrial activities in coastal watersheds) have altered reef communities and resilience, leading to region-wide decreases in live coral cover (Gardner et al., 2003; Jackson et al., 2014) and reef structural complexity (Alvarez-Filip et al., 2009) over the last three decades. Concomitant increases in disease epidemics (Rogers, 2009) and in macro-algae (Bruno et al., 2009) have resulted in ecosystem shifts from coral-dominated to algal-dominated reefs (Hughes et al., 2007). The once dominant *Acropora palmata* was severely reduced by coral disease from the late 1970s through the 1980s, and this genus is now listed as endangered in the United States Caribbean (<http://www.nmfs.noaa.gov/pr/laws/esa/>). The 1982-1984 mass die-off of the *Diadema antillarum* sea urchins left Caribbean reefs without a keystone herbivore (Jackson et al., 2014). Pioneering coral species, such as *Porites astreoides*, are becoming more prevalent (Green et al., 2008). The degradation is exacerbated by global climate change resulting in warming causing mass coral bleaching and associated coral mortality (Eakin et al., 2010), physical destruction from more intensive storms (Gardner et al., 2005; Wilkinson and Souter, 2008) and the threat of ocean acidification. Caribbean reef biodiversity is being further affected by the alien invasive Pacific red lionfish (*Pterios volitans*), which has spread across the Caribbean in the last decade (Schofield, 2010).

Mangroves and their associated biodiversity occur throughout the insular and continental Caribbean coastlines (Bossi and Cintron, 1990). The Caribbean has nine mangrove tree species (Polidoro et al., 2010), but is reported to host the world's richest mangrove-associated invertebrate fauna (Ellison and Farnsworth, 1996). Mangrove area has declined by about 1 per cent annually over the last three decades, representing the second highest loss rate globally (FAO, 2007). *Pelliciera rhizophorae*, endemic to Central America, is now listed as vulnerable (Polidoro et al., 2010). Mangrove declines are driven by exploitation (of timber); deteriorating water quality (especially petroleum pollution); and coastal development (aquaculture ponds, marinas, reclamation for coastal construction and agriculture); and climate change (Ellison and Farnsworth, 1996; Polidoro et al., 2010).

Seagrass communities occur throughout the Caribbean and support a high diversity of epiphytic and other species (van Tussenbroek et al., 2010). Seven native seagrass species are known from the region; two (*Halophila engelmanni* and *H. baillonii*) are considered to be near-threatened and vulnerable, respectively. A recently introduced species, *Halophila stipulacea*, is spreading rapidly through the Lesser Antilles (Willette et al., 2014). CARICOMP found that most study sites showed a decline in seagrass health between 1993 and 2007 (van Tussenbroek et al., 2014).

Caribbean seabirds comprise 25 breeding species, of which seven are regionally endemic species or subspecies: the abundant laughing gull subspecies *Larus atricilla atricilla*; the rare and declining white tailed tropic bird subspecies *Phaethon lepturus catesbyii*; the near threatened Audubon's shearwater *Puffinus lherminieri lherminieri*; the endangered brown pelican subspecies *Pelicanus occidentalis occidentalis*; the critically endangered black capped petrel *Pterodroma hasitata* and black noddy subspecies *Anous minutus minutus*; and the probably extinct Jamaica petrel *Pterodroma caribbaea* (Lee and Mackin, 2012). Most of the 25 species including both surface feeders and divers are declining, threatened by human disturbance and nest predation by introduced species; pollution of ocean waters; and fishery by-catch impacts (Schreiber and Lee, 2000; Lee and Mackin, 2012).

Caribbean economies are the most tourism-dependent in the world (CLME, 2011). Declining marine biodiversity will have enormous social and economic consequences, through loss of goods and critical ecosystem services. Caribbean-wide degradation of coral reef, mangrove and seagrass ecosystems, ecosystems that are fundamental to the Caribbean tourism product and at the core of the region's ability to cope with climate change sea level rise, will mean annual losses amounting to billions of United States dollars (CARSEA, 2007; Burke et al., 2011).

7.8 Gulf of Mexico

The Gulf of Mexico is a semi-enclosed sea and one of the most economically and ecologically productive bodies of water in the world (Tunnell, 2009). The Gulf is connected to the Caribbean Sea through the Yucatan Channel between the Yucatan Peninsula and Cuba, where warm, tropical water flows into the Gulf and forms the Loop Current, the dominant Gulf current, and then exits via the Florida Straits between Florida and Cuba into the Atlantic Ocean, where it forms the Gulf Stream, one of the world's strongest and most important currents.

As a large receiving basin, the Gulf of Mexico receives extensive drainage from five countries (the United States, Canada, Mexico, Guatemala, and Cuba). The Mississippi River dominates this drainage, which includes over two-thirds of the U.S. watershed in the north, and the Grijava-Usumacinta River System dominates in the south. Thirty-three major rivers and 207 estuaries and lagoons are found along the Gulf coastline (Darnell and Defenbaugh, 1990; Tunnell, 2009).

Biologically, the shallow waters of the northern Gulf are warm-temperate (Carolinian Province) and those in the southern Gulf are tropical (Caribbean Province) (Briggs, 1974). Oyster reefs and salt marshes are the dominant estuarine habitat type in northern, low-salinity estuaries, and seagrass beds are common in clearer, more saline bays. In the tropical southern Gulf, mangroves line bay and lagoon shorelines; some oyster reefs, salt marshes, and seagrasses are distributed in similar salinity conditions as in the northern Gulf. Offshore, coral reefs are common in the Florida Keys, Cuba, and the southern Gulf off the state of Veracruz and on the Campeche Bank (Tunnell et al., 2007; other topographic highs or hard bottoms are sporadic on the normally smooth, soft substratum of the continental shelves (Rezak and Edwards, 1972; Rezak et al., 1985). Unique, recently discovered, and highly diverse habitats in deeper Gulf waters include chemosynthetic communities and deepwater coral communities (*Lophelia* reefs) (CSA International Inc., 2007; Brooks et al., 2008; Cordes et al., 2008).

Regarding the biodiversity of the Gulf of Mexico, the Harte Research Institute for Gulf of Mexico Studies at Texas A&M University-Corpus Christi recently led a multi-year, multi-international effort (Biodiversity of the Gulf of Mexico Project) involving 140 taxonomists from 80 institutions in 15 countries to prepare a comprehensive list of all species (Felder and Camp, 2009). This list of 15,419 species with range, distribution, depth, habitat-biology, and updated taxonomy was subsequently added to GulfBase in 2011 at <http://www.gulfbase.org/biogomx/biosearch.php>, where it is openly accessible and completely searchable by any topic or species. The database has since been used in two major papers comparing biodiversity of other United States regions (Fautin et al., 2010) and four other global case studies in marine biodiversity (Ellis et al., 2011).

Several Gulf of Mexico iconic or well-known species are of historical, social, and economic importance (Davis et al., 2000). The West Indian monk seal (*Monachus tropicalis*) was probably the first large animal to become extinct because of human activity in the Gulf and Caribbean region. It was last seen on the Campeche Bank islands in the Gulf in 1948 and in the Caribbean in the early 1950s (Wursig et al., 2000). Other species that have become endangered include the Kemp's Ridley sea turtle (*Lepidochelys kempii*), brown pelican (*Pelecanus occidentalis*), and whooping crane (*Grus americana*). Restoration programs for each of these have increased their populations in recent decades. West Indian manatees (*Trichechus manatus*) are greatly reduced, and they only exist now in certain drainage areas along the west coast of Florida. The largest commercial fishery by weight in the Gulf is for menhaden (*Brevoortia patronus*), and the penaeid shrimp fishery is the largest by value (the white shrimp *Litopenaeus setiferus*, the pink shrimp *Farfantepenaeus duorarum*, and the brown shrimp *Farfantepenaeus aztecus*).

Predominant commercial estuarine shellfish in the northern Gulf include the eastern oyster (*Crassostres virginica*) and blue crab (*Callinectes sapidus*) (Nelson, 1992; Patillo et al., 1997). In the tropical southern Gulf, spiny lobster (*Panulirus argus*) and queen conch (*Eustrombus gigas*) are taken. However, these are now commercially extinct in many

areas and are taken only by recreational fishers, sometimes under strict regulations (Tunnell et al., 2007).

The Gulf-wide bottlenose dolphin (*Tursiops truncatus*) is probably the single most recognizable Gulf species by the public, as it is abundant in coastal bays and estuaries, as well as offshore in the northern Gulf (Wursig et al., 2000).

Gulf-wide biodiversity patterns cannot be completely explained, for lack of complete information, although we do know that the Gulf of Mexico exhibits great habitat complexity that probably supports high levels of biodiversity due to both endemic and cosmopolitan species (Rabalais et al., 1999). Linkage to the Caribbean Sea with large-scale circulation provides the southern and eastern Gulf with a distinct Caribbean biota. However, strong regional endemism appears to exist, as demonstrated in large-scale studies across the entire northern Gulf (Rabalais et al., 1999; Harper, 1991; Carney et al., 1993). Eventual analysis of databases from the Biodiversity of the Gulf of Mexico Project on GulfBase will provide considerable insight into the spatial distribution of species. Of the 15,419 species found, 1,511 (10 per cent) are endemic to the Gulf of Mexico and 341 (2 per cent) are non-indigenous (Felder and Camp, 2009). The most diverse taxa include crustaceans (2,579 species), mollusks (2,455), and vertebrates (1,975), and the least diverse include kinorhynchs (2 species), entoprocts (2), priapulids (1), hemichordates (5), and cephalochordates (5). In addition, other taxa are known to exist in the Gulf of Mexico (placozoans, orthonectids, loriciferans, and pogonophorans), but representatives have not yet been identified (Felder and Camp, 2009; Fautin et al., 2010).

A recent ecosystem status report for the Gulf of Mexico, utilizing the DPSER (Drivers, Pressures, States, Ecosystem Services, Responses) conceptual modelling framework, gives a high-level overview of the state of the Gulf (Karnauskas et al., 2013). Major, large-scale climatic drivers include the Atlantic Multidecadal Oscillation, Atlantic Warm Pool, sea surface temperature, Loop Current, and geostrophic transport in the Yucatan Channel and Florida Current. Long-term trends or changes in these drivers in turn cause fluctuations or changes in selected pressures, such as hurricanes or hypoxic zones. Other pressures include contamination by pollution (e.g., mercury, cadmium), oil and gas exploration and production (including major oil spills, such as Ixtoc I in 1979 and Deepwater Horizon in 2010), bacterial water quality problems, and habitat destruction, mainly caused by coastal development. Harmful algal blooms (HABs), such as red tide and brown tide, are well documented in the Gulf, as are some invasive species (Tunnell, 2009; Fautin et al., 2010; Karnauskas et al., 2013).

The recent Deepwater Horizon oil spill prompted a study of the ecosystem services of the Gulf of Mexico by a leadership committee of the United States National Research Council (NRC). This comprehensive report utilizes the ecosystem services approach and frames for the first time the goods and services provided by the Gulf for an economically and ecologically healthy ecosystem (NRC, 2013).

8. Factors of Sustainability

The biodiversity of the North Atlantic has supported harvesting and trade by bordering cultures for millennia. Pressures from human uses both diversified and intensified with industrialization and associated coastal development already more than two centuries ago. Every form of direct use of North Atlantic biodiversity and every indirect effect of human activities on coastal populations and habitats have been unsustainable in at least some times and places. Some of these impacts, such as the depletion of populations of the great whales by overharvesting (section 36A.6) will take centuries to recover, even with effective policies and high compliance.

Efforts towards sustainability have been greatly aided by coordinated international efforts to provide scientific and technical information on the status and trends in biodiversity, and threats to sustainable uses. The Quality Status reports (QSRs) from the OSPAR Commission at the start of each of the past three decades have proven invaluable in assessing status and trends in many marine environmental indicators and the biodiversity they represent, and guiding policies and management measures to address poor or declining marine environmental quality in the northeast Atlantic (OSPAR, 2010, and earlier QSRs). Other examples of such efforts are the coordinated processes carried out within the framework of the Convention on Biological Diversity for identifying Ecologically and Biologically Significant Areas in the Northwest Atlantic (CBD, 2014a), and Mediterranean (CBD, 2014b), and the on-going process in the Northeast Atlantic (ICES, 2013a), and the processes to identify Vulnerable Marine Ecosystems in the NAFO (NAFO, 2013) and North-East Atlantic Fisheries Commission (NEAFC) (FAO, 2015).

Although there are well-documented examples in the sections above of cases where habitats, lower-trophic-level productivity, benthic communities, fish communities, or seabirds or marine mammal populations were severely altered by pressures from a specific activity, such as over-fishing, pollution, nutrient loading, physical disturbance, or transplanted species, many biodiversity impacts, particularly at larger scales, are the result of cumulative and interactive effects of multiple pressures from multiple drivers. It has repeatedly proven difficult to disentangle the effects of the individual pressures, impeding the ability to address the individual causes (Fu et al., 2012; Blanchard et al., 2005; ter Hofstede and Rijnsdorp, 2011). Particularly given that the North Atlantic is surrounded by many of the best marine research centres in the world, has many of the longest and most systematic data sets, and has an international science organization, the International Council for Exploration of the Seas (ICES), that has functioned for over a century to promote and coordinate scientific and technical cooperation among many of the circum-North Atlantic countries with the highest science capacities, this inability to consistently disentangle causation of unsustainable uses of, and impacts on, marine biodiversity may seem initially discouraging.

On the other hand, well-documented examples also exist of the benefits that can follow from actions to address past unsustainable practices. Many of the fish stocks depleted

by overfishing in both the Northeast and Northwest Atlantic have shown increasing trends in abundance and recovery of range when unsustainable levels of fishing effort have been reduced (Table 36A.4). Efforts to control pollution and nutrient inputs, driven by the EU Water Framework Directive and the United States Environmental Protection Act, have led to reduction in these pressures and in many cases to the commencement of the recovery of benthic communities (EEA, 2012). Coastal habitat restoration activities have also shown clear benefits in improved environmental quality and biodiversity measures in many coastal areas around the North Atlantic (Pendleton, 2010). All of these improvements have come with at least short-term costs, which are sometimes large, such as displaced or reduced fishing opportunity (see Part IV), the costs of pollution control and nutrient management in coastal areas and watersheds (costs summarized in the chapters of Part V), and the direct costs of habitat restoration, which may run to the millions of dollars for restoration projects of even moderate scale (Diefenderfer et al., 2011; Kroeger and Guennel, 2014).

In summary, the North Atlantic presents examples of both the extent to which unsustainable actions can adversely affect biodiversity and the benefits that can accrue from policies and programmes that are well developed, adequately resourced, and effectively implemented. The best examples of effective policies and programmes have been designed to address the dominant pressures from the twentieth century: overharvesting of living marine resources and pollution and excessive nutrient inputs from coastal and land-based sources. In the twenty-first century, additional pressures are growing, particularly climate change, invasive species (both responding to changing environmental conditions and transported by shipping), and in many areas, particularly at lower latitudes, ocean-based tourism. Lessons learned from dealing successfully with the earlier pressures, if applied proactively, may help safeguard biodiversity from unsustainable impacts, and result in healthy ecosystems producing many ecosystem services of value to the circum-Atlantic human populations.

Table 1. Abundant phytoplankton species of selected taxa (based on abundance and number of genera represented) in coastal and shelf waters of the western North Atlantic (* produce mucilage and foam, ** potentially toxic species).

Domain	Location	Division	No.	Abundant Species
Coastal & Shelf	Gulf of Maine ¹⁹	Heterokontophyta (Diatoms)	386	<i>Chaetoceros</i> spp., <i>Navicula</i> spp., <i>Nitzschia</i> spp., <i>Rhizosolenia hebetata</i> , <i>Coscinodiscus</i> spp., <i>Pleurosigma</i> spp., <i>Thalassiosira</i> spp., <i>Gyrosigma</i> spp., <i>Phaeoceros</i> spp.
		Alveolata (Dinoflagellates)	151	<i>Peridinium</i> spp., <i>Alexandrium</i> spp.**, <i>Amphidinium</i> spp., <i>Ceratium</i> spp.
		Haptophyta	31	<i>Chrysochromulina</i> spp.**, <i>Diacronema</i> spp., <i>Emiliana huxleyi</i> , <i>Pavlova</i> spp., <i>Prymnesium</i> spp.
		Cyanophyta	22	<i>Synechococcus</i> spp.
		Chlorophyta	20	<i>Halosphaera viridia</i> , <i>Micromonas pusilla</i> , <i>Ostreococcus</i> sp., <i>Pycnococcus provasolii</i> , <i>Tetraselmis</i> spp.
		TOTAL	665	
	Gulf of Maine & New York Bight ²⁰	Heterokontophyta	274	<i>Leptocylindrus danicus</i> , <i>Skeletonema costatum</i> , <i>Asterionella glacialis</i> , <i>Pseudo-nitzschia pungens</i> **, <i>Rhizosolenia delicatula</i>
		Alveolata	332	<i>Procentrum micans</i> **, <i>P. minimum</i>
		Haptophyta	19	<i>Emiliana huxleyi</i>
		Cyanophyta	12	<i>Nostoc commune</i>
		Chlorophyta	13	<i>Nannochloris atomus</i>
TOTAL		678		
Coastal	New York Bight ²¹	Heterokontophyta	-	<i>Skeletonema costatum</i> , <i>Asterionella japonica</i> , <i>Leptocylindrus danicus</i> , <i>Thalassionema nitzschioides</i> , <i>Chaetoceros debilis</i>
		Alveolata	-	<i>Gymnodinium splendens</i> , <i>Prorocentrum micans</i> **, <i>P. triangulatum</i>
		Chlorophyta	-	<i>Nannochloris atomus</i>
Shelf	New York Bight ³	Heterokontophyta	-	<i>Rhizosolenia abta</i> , <i>R. faeroense</i> , <i>Chaetoceros socialis</i> , <i>Cylindrotheca closterium</i>
		Alveolata		<i>Ceratium tripos</i> , <i>C. macroceros</i> , <i>C. furca</i> ,

¹⁹ Li et al., 2011

²⁰ Marshall and Cohn, 1982

²¹ Malone, 1977

			-	<i>Peridinium depressum</i>
Coastal & Shelf	Gulf of Mexice ²²	Heterokontophyta	152	<i>Guinarda spp.</i> , <i>Hemiaulus senensis</i> , <i>Leptocylindrus danicus</i> , <i>Thalassionema spp.</i> , <i>Cylindrotheca closterium</i> , <i>Pseudo-nitzschia delicatissima</i>
		Alveolata	124	<i>Ceratium spp.</i> , <i>Dinophysis caudate**</i> , <i>Gyrodinium fusiforme</i> , <i>Scrippsiella trochoidea</i>
		Cyanophyta	18	<i>Trichodesmium spp.</i>
		TOTAL	306	

Table 2. Abundant phytoplankton species of selected taxa (based on abundance and number of genera represented) in coastal and shelf waters of the eastern North Atlantic (* produce mucilage and foam, ** potentially toxic species).

Domain	Location	Division	No.	Abundant Species
Coastal & Shelf	NE Atlantic ²³	Heterokontophyta (Diatoms)	59	<i>Bacteriastrum spp.</i> , <i>Chaetoceros spp.</i> , <i>Cylindrotheca closterium*</i> , <i>Guinardia delicatula</i> , <i>Odontella aurita</i> , <i>Proboscia alata</i> , <i>Pseudo-nitzschia spp.**</i> , <i>Rhizosolenia spp.</i> , <i>Skeletonema costatum</i> , <i>Thalassionema nitzschioides</i>
		Alveolata (Dinoflagellates)	48	<i>Ceratium furca</i> , <i>C. fusus</i> , <i>C. horridum</i> , <i>C. lineatum</i> , <i>C. longipes</i> , <i>C. macroceros</i> , <i>C. tripos</i> , <i>Dinophysis spp.**</i> , <i>Gonyaulax spp.**</i> , <i>Noctiluca scintillans</i> , <i>Prorocentrum spp.</i> , <i>Protoperidinium spp.</i>
		Haptophyta	1	<i>Phaeocystis pouchetii*</i>
		TOTAL	170	
Coastal	German Bight ²⁴	Heterokontophyta (Diatoms)	109	<i>Chaetoceros curvisetus</i> , <i>Chaetoceros spp.</i> , <i>Coscinodiscus spp.</i> , <i>Coscinodiscus wailesii*</i> , <i>Guinardia flaccida</i> , <i>Odontella sinensis</i> , <i>Pseudo-nitzschia pungens**</i> , <i>Rhizosolenia imbricate</i> , <i>Rhizosolenia styliformis</i>
		Alveolata (Dinoflagellates)	26	<i>Noctiluca scintillans</i> , <i>Ceratium tripos</i> , <i>Ceratium fusus</i> , <i>Ceratium longipes</i> , <i>Gyrodinium spirale</i> , <i>Protoperidinium depressum</i>
		Haptophyta	3	<i>Emiliana huxleyi</i> , <i>Phaeocystis sp.*</i>
		TOTAL	292	
Coastal	English Channel ²⁵	Heterokontophyta (Diatoms)	131	<i>Guinardia spp.</i> , <i>Phaeocystis globosa</i> , <i>Paralia sulcata</i> , <i>Pseudo-nitzschia spp.**</i> , <i>Chaetoceros spp.</i> , <i>Thalassiosira spp.</i>

²² Merino-Virgilio et al., 2013

²³ Barnard et al., 2004; Edwards et al., 2006

²⁴ Wasmund et al., 2012

²⁵ Guilloux et al., 2013

		Alveolata (Dinoflagellates)	28	<i>Prorocentrum</i> spp.**
		Haptophyta		<i>Phaeocystis globosa</i> *
		TOTAL	178	
Coastal	Iberian Peninsula ²⁶	Heterokontophyta (Diatoms)	68	<i>Chaetoceros</i> spp., <i>Leptocylindrus danicus</i> , <i>Pseudonitzschia delicatissima</i> **, <i>Guinardia</i> spp., <i>Rhizosolenia fragile</i> , <i>Thalassiosira</i> spp., <i>Nitzschia longissima</i>
		Alveolata (Dinoflagellates)	69	<i>Prorocentrum micans</i> **, <i>Amphidinium curvatum</i> , <i>Dinophysis</i> spp.** <i>Ceratium lineatum</i> , <i>Gymnodinium</i> sp.**, <i>Scrippsiella trochoidea</i>
		TOTAL	161	

²⁶ Rodríguez et al., 2003; Not et al., 2007; Ospina-Alvarez et al., 2014
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Table 3. Abundant mesozooplankton species for selected taxa (based on abundance, number of genera represented and their importance as indicators of climate-driven changes in hydro-climate) in coastal and shelf regions of the North Atlantic (NE – British Isles, Baltic and North Seas; SE - Bay of Biscay, Iberian coast, west Africa; North – Labrador and Norwegian Seas, Greenland and Iceland; NW – New York Bight, Gulf of Maine, Newfoundland and Scotian Shelves; SW – Caribbean Sea, Gulf of Mexico, South Atlantic Bight).

Location	Taxa	No.	Abundant Species
NE North Atlantic ^{27,28}	Calanoida	286	<i>Acartia</i> spp., <i>Calanus finmarchicus</i> , <i>Calanus helgolandicus</i> , <i>Centropages</i> spp., <i>Clausocalanus</i> spp., <i>Eurytemora affinis</i> , <i>Metridia lucens</i> , <i>Paracalanus</i> spp., <i>Para-Pseudocalanus</i> spp., <i>Pseudocalanus</i> spp., <i>Temora longicornis</i> ,
	Cyclopoida	1	<i>Oithona</i> spp.
	Cladocera	3	<i>Evadne</i> spp., <i>Podon</i> spp., <i>Penilia avirostris</i>
	Thecosomomata	4	<i>Limacina</i> spp.
	Copepod Total	381	
SE North Atlantic ^{1,2,29}	Calanoida	580	<i>Acartia</i> spp., <i>Calanoides carinatus</i> , <i>Calanus helgolandicus</i> , <i>Candacia armata</i> , <i>Centropages typicus</i> , <i>Clausocalanus</i> spp., <i>Ctenocalanus vanus</i> , <i>Euchaeta hebes</i> , <i>Metridia lucens</i> , <i>Paraeuchaeta gracilis</i> , <i>Para-Pseudocalanus</i> spp., <i>Pseudocalanus</i> spp., <i>Temora stylifera</i> , <i>Undeuchaeta</i> spp., <i>Lucicutia flavicornis</i> , <i>Nannocalanus minor</i> , <i>Paracalanus parvus</i> , <i>Ctenocalanus vanus</i> , <i>Neocalanus gracilis</i> , <i>Rhincalanus cornutus</i> , <i>Eucalanus subtenuis</i>
	Cyclopoida	4	<i>Oithona</i> spp.
	Cladocera	7	<i>Evadne</i> spp., <i>Podon</i> spp
	Thecosomomata	4	<i>Limacina inflata</i> , <i>Limacina trochiformis</i> , <i>Creseis acicula</i>
	Copepod Total	747	
North North Atlantic ^{1, 2}	Calanoida	188	<i>Acartia</i> spp., <i>Calanus finmarchicus</i> , <i>Heterorhabdus norvegicus</i> , <i>Paraeuchaeta norvegica</i>
	Cyclopoida	-	<i>Oithona</i> spp.
	Cladocera	-	<i>Evadne</i> spp.
	Thecosomomata	-	<i>Limacina</i> spp.
	Copepod Total	204	

²⁷ Barnard et al., 2004

²⁸ Van Ginderdeuren, 2012; Laakmann et al., 2013; Razouls et al., 2014

²⁹ Thiede, 1975; Poulet et al., 1996; Valdés et al., 2007; Albaine and Irigoien, 2007, Hernández-León et al., 2007

NW North Atlantic ^{1,2,30}	Calanoida	204	<i>Acartia</i> spp., <i>Calanus finmarchicus</i> , <i>Calanus glacialis</i> , <i>Calanus hyperboreus</i> , <i>Centropages</i> spp., <i>Labidocera aestiva</i> , <i>Metridia lucens</i> , <i>Para-Pseudocalanus</i> spp., <i>Pseudocalanus</i> spp., <i>Clausocalanus arcuicornis</i> , <i>Paracalanus</i> spp., <i>Temora longicornis</i> , <i>Tortanus discaudatus</i>
	Cyclopoida	-	<i>Oithona</i> spp.
	Cladocera	-	<i>Evadne</i> spp., <i>Podon</i> spp., <i>Penilia</i> spp.
	Thecosomomata	-	<i>Limacina retroversa</i> , <i>Limacina</i> spp.
	Copepod Total	261	
SW North Atlantic ^{2,31}	Calanoida	553	<i>Acartia</i> spp., <i>Calanus tenuicornis</i> , <i>Centropages</i> spp., <i>Clausocalanus</i> spp., <i>Corycaeus</i> spp., <i>Eucalanus</i> spp., <i>Euchaeta</i> spp., <i>Haloptilus longicornis</i> , <i>Labidocera aestiva</i> , <i>Lucicutia flavicornis</i> , <i>Paracalanus crassirostris</i> , <i>Paracalanus parvus</i> , <i>Parvocalanus</i> spp., <i>Pleuromamma gracilis</i> , <i>Nannocalanus minor</i> , <i>Temora</i> spp.
	Cyclopoida		<i>Oncaea venusta</i> , <i>Corycaeus amazonicus</i> , <i>Oithona brevicornis</i> , <i>Oithona nana</i> , <i>Oithona plumifera</i> , <i>Oithona simplex</i>
	Cladocera		<i>Evadne</i> sp.
	Thecosomomata		<i>Limacina trochiformis</i>
	Copepod Total	715	

Table 4. Tabulation of conclusions of assessment authorities on stock status. Each authority has its own standards for benchmarking status. Where quantitative reference points are not estimated, a stock was counted as “healthy or cautious” if abundance was reported as average or high, or as increasing if below average. Stocks reported as depleted or low and declining were counted as negative status. F = status relative to fishing mortality reference points; B = status relative to biomass reference points (,)

Authority	US-NMFS	Canada-DFO	NAFO	ICES
Positive or Cautious Status	195 (B) 290 (F)	72 Healthy 31 Cautious	5	60
Negative Status	38 overfished 27 overfishing	17	5	42
Unknown	247	35	4	75

³⁰ Malone, 1977, Johns et al., 2001; Durbin et al., 2003; Runge and Jones, 2012

³¹ Grice, 1960; Park, 1970, 1975; Cummings, 1983; Elliott et al., 2012

Table 5. Tabulation of a number of primary publications documenting status and trends in fish community metrics for areas in the North Atlantic.

A large number of community metrics were used, and have been grouped into several categories: “size” includes metrics of body size; “diversity” includes any of the typical indices of species diversity; “richness” and “evenness” include numbers of species recorded and how numbers were distributed among species; “dominance” includes measures of how much the abundance of the few most common species in a community comprised of all the individuals in the community; “N” includes measures of total abundance of individuals in a community, “B” includes measures of community biomass; “slope” and “intercept” are parameters of community size spectra; “species composition” are diverse ways of reporting how the representation of particular species in the community changed. For reporting trends in the metric, “+” means an overall upward trend, “-” means an overall downward trend and “nt” means no overall trend, although there could be substantial interannual variation in the metric.

Location	Indicators and trends	Interval	Comments	Reference
Georges Bank	Size +, diversity+, Biomass/area+	1960s - 2009	Decline in F and effort	Collie et al. 2013
North Sea	Size +, diversity+, B+	1960s - 2009	Decline in F and effort	Collie et al. 2013
Ionian Sea	N+, Richness+, Size+	1998-2008	Decline in F and effort	Tsagarakis et al. 2011
Galicia	Diversity, Richness, N, B	1980-1991	Species composition changed more than indices	Farina et al. 1997
Multiple	Slope +, intercept *	Various; 2 decades+	Increases in F	Bianchi et al. 2000
Scotian Shelf	Slope+, intercept+, diversity nt	1970-1997	Increases in F	“
North Sea	Slope+, intercept+, richness+	1972 - 1998	Increase F and expansion of southern species	“
Portugal	Slope nt, intercept nt, evenness nt	1982 – 1998 with gaps	Large variation in species composition w/o trend	“
NW North Sea	Diversity -, dominance +	1920 – 1990s (with gaps)	Large changes in non-commercial species as commercial ones decreased	Greenstreet et al. 1999

Canary Islands	Diversity nt, richness nt	1990s	Large changes w/o trend in species composition	Uiblein et al. 1996
Baltic and Kattegat	Richness +, N nt	1990 - 2008	Difference in connectance not a factor	Hiddink and Coleby 2012
West of Scotland	Richness -	1997-2008	F stayed high	ter Hofstede et al. 2010
North Sea	Richness+	“	Southern species incursions	“
Celtic Sea	Richness+	“	“	“
Iceland	Richness +, diversity nt	1996-2007	Warming	Steffansdottir et al. 2010
Iceland	Species Composition	1970s-2010s	Warming	Valdimarsson et al. 2012
Dogger Bank	Diversity-, dominance +	1991 - 2010	Warming, and common species increased	Sonnewald and Turkey 2012
NE Shelf	Richness+, Diversity+	1980 - 2008	Abrupt regime like change Species makeup changed most	Simpson et al. 2011
North Sea	Richness+ (eggs & larvae)	1958 - 2005	Warm species entering	Beaugrand et al. 2008
Barents Sea	Functional diversity nt	2004 - 2009	Many changes in species comp.	Wiedmann et al. 2014
Scotian Shelf	B-, Size-, Evenness-	1970 - 2006	Period of heavy fishing	Shackell et al. 2012
W of Scotland	Richness nt, diversity nt	1980s – 2000s	Fishing stable but high	Campbell et al. 2011
North Sea	Slope+, Intercept+, Lmax-	1972 - 2000s	Large species becoming rarer. Greater in high F areas	Daan et al. 2005
Scotian Shelf	Evenness+. Dominance -	1970 - 2000	F high and increasing	Shackell and Frank 2003

Medit. and NE Atlantic	Length nt, N nt	1997 - 2007	Large variability but no persistent trends	Rochet et al. 2010
Portuguese shelf	Richness nt, Diversity nt	1989 - 1999	High spatial patchiness, no overall trends	Sousa et al. 2006
NW Atlantic	Diversity nt	1970s – 2000s	Latitudinal trend but no time trend	Fisher et al. 2008
NW Atlantic	N-, MaxAge-, Size-	1978 various	F high and not declining	Hutchings and Baum
Medit.	Abundance-, Richness-	1948-2005	Just sharks	Ferretti et al. 2013

Table 6. Population trends of breeding seabird taxa in the North Atlantic Ocean. Divers include shearwaters, gannets, cormorants, shags, pelicans, auks; surface-feeders include fulmars, storm-petrels, frigatebirds, skuas, jaegers, gulls, terns.

REGION	ALL SPECIES	DIVERS	SURFACE-FEEDERS
NORTH ATLANTIC	Decrease	Decrease	Decrease
NAFO	Decrease	Increase	Decrease
E Baffin Is	Increase	Increase	?
W Greenland	Decrease	Decrease	No Change?
Newfoundland/Labrador	No Change	Increase	Decrease
Gulf St. Lawrence	Decrease	Increase	Decrease
E Canada/United States	Increase	Increase	Increase
Gulf of Mexico	?	?	?
Caribbean	Decrease	Decrease	Decrease
ICES	Decrease	Decrease	Increase
E Greenland/Iceland	Decrease	Decrease	Increase
Barents Sea ^a	Increase	Increase	Decrease
Norwegian Sea	Decrease	Decrease	Increase
Faroes Island (Denmark), Shetland, Western United Kingdom	Increase	Increase	Decrease
N Sea/English Channel	Decrease	No Change?	Decrease
Baltic Sea, Skagerrak, Kattegat	?	?	?

France, Iberia, Azores	?	?	?
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a – not included

Table 7. Estimates of indigenous species richness and pressures on them in the Chesapeake Bay estuarine system (sans meiofauna, bacteria and microzooplankton). Of the phytoplankton, diatoms accounted for 46%, chlorophytes 19%, dinoflagellates 13%, cyanobacteria 9% and toxic species 2%. Of the benthic (soft bottom) macrofauna, arthropods accounted for 37%, annelids 25%, and mollusks 25%. (Data sources: Orris, 1980; Musick et al., 1985; Brownlee and Jacobs, 1987; Birdsong et al., 1989; Birdsong and Buchanan, 1993; Wagner, 1999; Wagner and Austin, 1999; Nizinski, 2003; and Marshall et al., 2005)

Trophic Level	Category	Number of Species	Major Anthropogenic Pressures
Primary Producers	Marsh grasses	19	Coastal development, Sea level rise, Invasive species
	Submerged vascular plants	15	Nutrient loading, Invasive species
	Macroalgae	25	Nutrient loading, Invasive species
	Phytoplankton	1453	Nutrient loading, Harvest of pelagic & benthic filter feeders
Consumers	Zooplankton (> 200 µm)	400	Nutrient loading, Harvest of pelagic filter feeders & predators, Ocean warming
	Fin fish	348	Fishing, Habitat loss, Invasive species, Ocean warming
	Benthic macrofauna	696	Fishing, Seasonal hypoxia, Ocean warming
	Waterfowl	49	Fishing, Habitat loss
TOTAL		3005	

Table 8. Representatives of ecologically important species drawn from Heck and Orth (1980), Orth et al. (1987), Baird and Ulanowicz (1989), Sellner and Marshall (1993), Birdsong and Buchanan (1993), Houde (1993), Dauer et al. (1993), Newell and Breitburg (1993), McConaugha and Rebach (1993), Jorde et al. (1993), Stevenson and Pendleton (1993), Jordán (2001), Buchanan et al. (2005), Orth et al. (2006), and Chambers et al. (2008).

(*Iconic species, **Toxic species).

Category	Species
Tidal marsh grasses	<i>Pontederia cordata</i> , <i>Zizania aquatica</i> , <i>Scripus olneyii</i> , <i>Spartina cynosuroides</i> , <i>Spartina alterniflora</i> , <i>S. patens</i>
Submerged vascular plants	<i>Vallisneria americana</i> , <i>Stuckenia pectinata</i> , <i>Potamogeton perfoliatus</i> , <i>Ruppia maritime</i> , <i>Zostera marina</i>
Macroalgae	<i>Ulva lactuca</i> , <i>Agardhiella</i> spp., <i>Enteromorpha</i> spp., <i>Cladophora</i> spp.
Phytoplankton	Diatoms – <i>Cerataulina pelagica</i> , <i>Rhizosolenia fragilissima</i> , <i>Leptocylindrus minimus</i> , <i>Skeletonema costatum</i> , <i>Asterionella glacialis</i> ; Dinoflagellates – <i>Gymnodinium</i> spp., <i>Ceratium lineatum</i> , <i>Prorocentrum minimum**</i> , <i>Dinophysis acuminata*</i> ; Cyanobacteria – <i>Microcystis aeruginosa**</i>
Zooplankton	<i>Bosmina longirostris</i> , <i>Leptodora kindtii</i> , <i>Acartia tonsa</i> , <i>Eurytemora affinis</i> , <i>Nemopsis bachei</i> , <i>Mnemiopsis leidyi</i> , <i>Chrysaora quinquecirrha</i>
Benthic infauna	<i>Mya arenaria</i> , <i>Macoma balthica</i> , <i>M. mitchelli</i> , <i>Nereis succinea</i>
Benthic epifauna	<i>Crassostrea virginica*</i> , <i>Callinectes sapidus*</i>
Forage fish	<i>Alosa pseudoharengus</i> , <i>A. aestivalis</i> , <i>A. sapidissima</i> , <i>A. mediocris</i> , <i>Anchoa mitchilli</i> , <i>Brevoortia tyrannus</i> , <i>Gobiosoma boscii</i> , <i>Menidia menidia</i> , <i>Fundulus heteroclitus</i> , <i>Cyprinodon variegatus</i> , <i>Gambusia holbrooki</i>
Intermediate predators	<i>Micropogonias undulatus</i> , <i>Trinectes maculatus</i> , <i>Leiostomus xanthurus*</i> , <i>Morone americana*</i> , <i>Ictalurus punctatus</i>
Top predators	<i>Pomatomus saltatrix</i> , <i>Cynoscion regalis</i> , <i>Paralichthys dentatus</i> , <i>Morone saxatilis*</i>
Waterfowl	<i>Aythya americana</i> , <i>Aythya valisineria</i> , <i>Anas rubripes</i> , <i>A. americana</i> , <i>Phalacrocorax auritus</i> , <i>Ardea herodias</i> , <i>Branta canadensis*</i> , <i>Cygnus columbianus</i> , <i>Haliaeetus leucocephalus</i>
Invasive species	Marsh grass – <i>Phragmites australis</i> , <i>Lythrum salicaria</i> , <i>Trapa natans</i> ; Benthic macrofauna – <i>Dreissena polymorpha</i> Fin Fish – <i>Ictalurus furcatus</i> , <i>Pylodictis olivaris</i> ; Other – <i>Cygnus olor</i> , <i>Myocastor coypus</i>
Threatened & endangered species	Sturgeon – <i>Acipenser oxyrinchus</i> , <i>A. brevirostrum</i> , Mussels – <i>Alasmidonta heterodon</i> Turtles – <i>Caretta caretta</i> , <i>Lepidochelys kempii</i>

Figures



Figure 1. Currents defining the North Atlantic gyre.

From <http://earth.usc.edu/~stott/Catalina/Oceans.html>

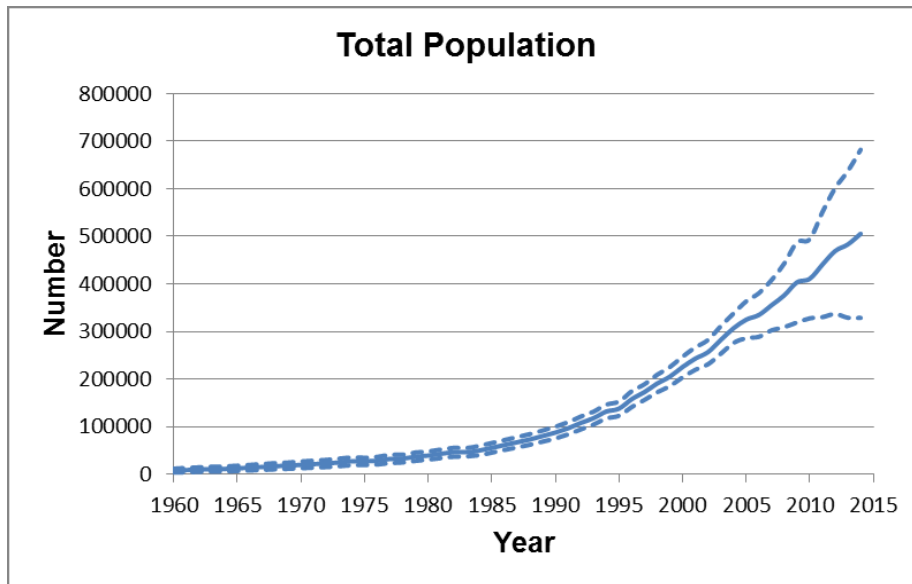


Figure 2. Estimated recent population trends of Northwest Atlantic gray seals.

Taken from Hammill et al., 2014. Estimated based on surveys and population model; dashed lines represent 95% confidence interval of estimates

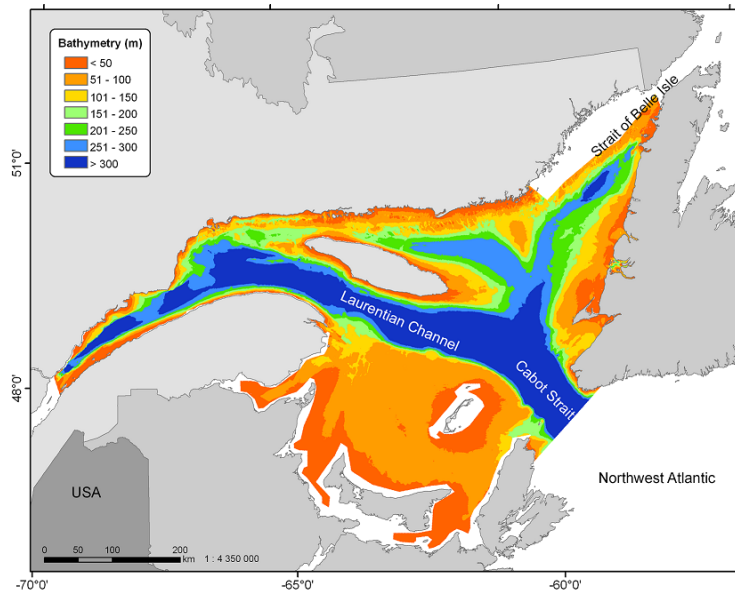
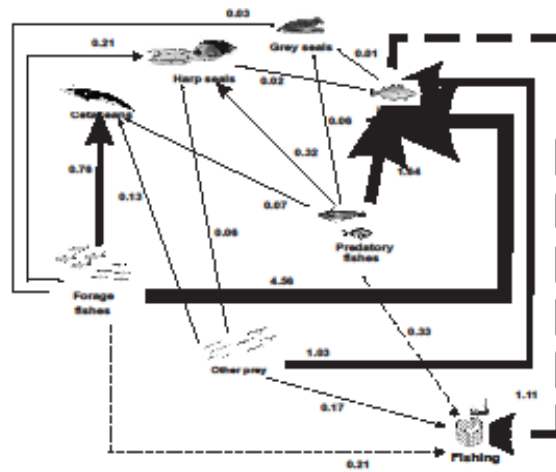
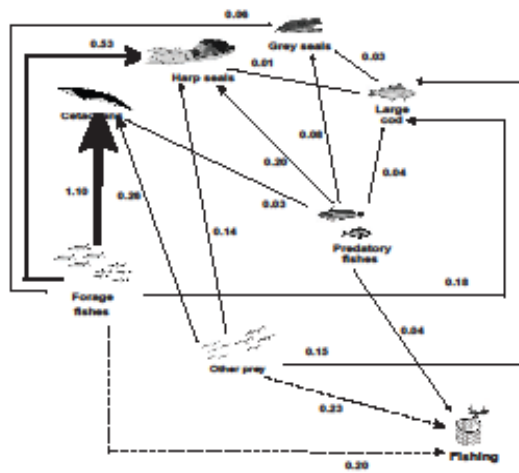


Figure 2bis. Bathymetric chart of the Gulf of St. Lawrence in eastern Canada. Cabot Strait in the southeast and the Strait of Belle Isle in the northeast connect the Gulf to the continental shelf regions of the Northwest Atlantic. (Prepared by Marie-Noëlle Bourassa, DFO, Canada)

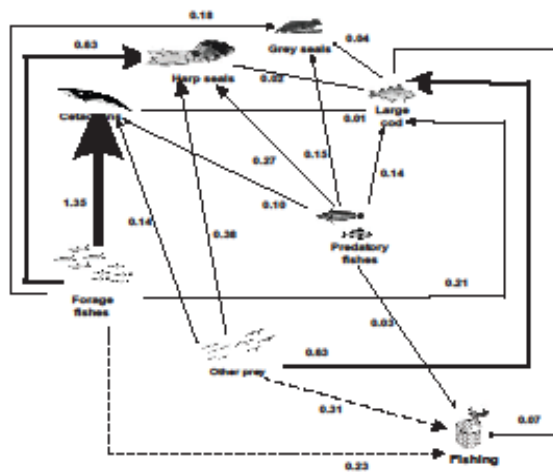
Mid-1980s



Mid-1990s



Early 2000s



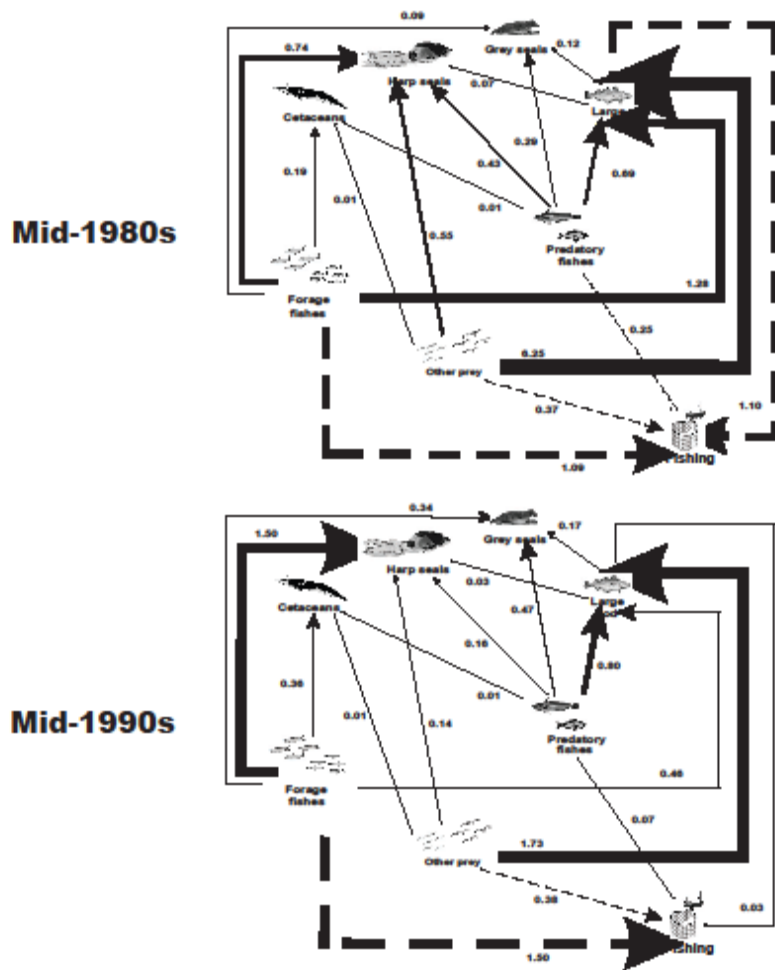


Figure 3. Changes in species abundance and/or biomass and food web structure and functioning

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