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LOSS, STATUS AND TRENDS FOR COASTAL MARINE HABITATS OF EUROPE

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Abstract Over the centuries, land reclamation, coastal development, overfishing and pollution have nearly eliminated European wetlands, seagrass meadows, shellfish beds, biogenic reefs and other productive and diverse coastal habitats. It is estimated that every day between 1960 and 1995, a kilometre of European coastline was developed. Most countries have estimated losses of coastal wetlands and seagrasses exceeding 50% of the original area with peaks above 80% for many regions. Conspicuous declines, sometimes to virtual local disappearance of kelps and other complex macroalgae, have been observed in several countries. A few dominant threats have led to these losses over time. The greatest impacts to wetlands have consistently been land claim and coastal development. The greatest impacts to seagrasses and macroalgae are presently associated with degraded water quality while in the past there have been more effects from destructive fishing and diseases. Coastal development remains an important threat to seagrasses. For biogenic habitats, such as oyster reefs and maerls, some of the greatest impacts have been from destructive fishing and overexploitation with additional impacts of disease, particularly to native oysters. Coastal development and defence have had the greatest known impacts on soft-sediment habitats with a high likelihood that trawling has affected vast areas. The concept of 'shifting baselines', which has been applied mostly to the inadequate historical perspective of fishery losses, is extremely relevant for habitat loss more generally. Most habitat loss estimates refer to a relatively short time span primarily within the last century. However, in some regions, most estuarine and near-shore coastal habitats were already severely degraded or driven to virtual extinction well before 1900. Native oyster reefs were ecologically extinct by the 1950s along most European coastlines and in many bays well before that. These shellfish reefs are among the most endangered coastal habitats, but they receive some of the least protection. Nowadays less than 15% of the European coastline is considered in 'good' condition. Those fragments of native habitats that remain are under continued threat, and their management is not generally informed by adequate knowledge of their distribution and status. There are many policies and directives aimed at reducing and reversing these losses but their overall positive benefits have been low. Further neglecting this long history of habitat loss and transformation may ultimately compromise the successful management and future sustainability of those few fragments of native and semi-native coastal habitats that remain in Europe.

Introduction

Habitat modification, fragmentation and loss are widely considered some of the most serious threats to diversity globally (Sih et al. 2000). In terrestrial environments, understanding and abating the effects of habitat loss and fragmentation are a huge focus in science, conservation and management (Wilcove et al. 1998, Brooks et al. 2002). It has been estimated that, throughout history, humans have severely modified or exploited to complete loss >70% of natural habitats in the habitable portion of the planet (Hannah et al. 1994) and that we are still losing somewhere between 0.5% and 1.5% of wild nature each year (Balmford et al. 2003). Habitat loss is also well recognised as an important threat in the marine environment (Suchanek 1994, Gray 1997, Wolff 2000) but has not been as much a focus of science and conservation as in terrestrial environments.

Habitat loss is particularly severe in coastal marine ecosystems, where human activities have been historically concentrated (Suchanek 1994, Lotze 2004, Knottnerus 2005, Lotze et al. 2006, Valiela 2006). Coastal zones occupy <15% of the Earth's land surface, but they accommodate >60% of the world's population (EEA 1999a). Globally, the number of people living within 100 km of the coast increased from roughly 2 billion in 1990 to 2.2 billion in 1995 (Burke et al. 2001), and the population living on the coast is projected to double in the next 30 yr with an expected 75% of the world's population residing in coastal areas by 2025 (EEA 1999a). As human population has increased in coastal areas, so has the pressure on coastal ecosystems through habitat conversion, increased pollution, and demand for coastal resources. Coastal systems provide many important services to humans such as nutrient cycling, food production, provision of habitat/refugia, disturbance regulation, natural barriers to erosion, control of water quality, and nursery grounds. Indeed the global value of services from seagrasses, estuaries and coastal wetlands is estimated to be 10 times higher than that of any terrestrial ecosystems (Costanza et al. 1997).

Recent reviews have examined the extent of habitat loss and fragmentation in tropical environments across large regions for coral reefs (e.g., Sebens 1994, Spalding et al. 2001, Pandolfi et al. 2003, Wilkinson 2004) and mangroves (e.g., Burke et al. 2001, Valiela et al. 2001, Alongi 2002, Wilkie & Fortuna 2003). These studies have done much to advance our understanding of the status and trends of tropical marine ecosystems at multinational and global levels and have been influential in galvanising support for tropical science, conservation and management.

Our understanding of the status and trends of temperate marine habitats is surprisingly further behind. Few scientific institutions, organisations or agencies have programmes that focus on temperate marine environments beyond a regional level, and almost no non-governmental organisations (NGOs) or agencies have multinational or global programmes that focus particularly on temperate ecosystems such as seagrasses, salt marshes or oyster reefs or the issue of habitat loss. There have been a few broad reviews of the condition of key temperate habitats (e.g., Kennish 2002, Steneck et al. 2002, Thompson et al. 2002, Lotze et al. 2006) and some recent exemplary efforts to pull together global distribution data on seagrasses (Short & Wyllie-Echeverria 1996, Duarte 2002, Green & Short 2003). Nonetheless, huge gaps still remain in our knowledge of habitat loss on temperate coasts and estuaries. This gap is particularly disturbing because these coasts contain some of the most productive, diverse and, at the same time, degraded ecosystems on Earth (Suchanek 1994, Edgar et al. 2000).

In Europe, there has been increasing awareness and concern about the degradation of natural habitats (e.g., Laffoley 2000). Many European coastal habitats have been lost or severely degraded, and it is estimated that only a small percentage of the European coastline (<15%) is in 'good' condition (EEA 1999a). Unfortunately, there are no comprehensive summaries of the current distribution and status of marine habitats along European coastlines and even less information is available about long-term trends of habitat loss or degradation.

Aim of the review

The aim of the present work is to review up-to-date estimates of large-scale trends in habitat extent, status and loss along European coastlines. Some of the major causes of these losses and some of the E.U.-wide policies aimed at slowing these losses are also discussed. Knowledge of the loss of coastal habitats and the drivers of this change is critical for identifying future directions in the sustainable conservation and management of Europe's coastlines (Dayton et al. 1998, Jackson et al. 2001, Lotze 2004).

The review is organised into three major sections: (1) an overview of the historical use of coastal resources and general impacts to European coastlines; (2) a compilation of data on the current distribution, historical loss, threats and protection measures of coastal habitats within bays, estuaries and near-shore shelf environments of Europe; (3) a discussion of gaps in the data, ecological knowledge and protection measures for these coastal habitats and recommendations for how to address these gaps.

The information on the coastal habitats of Europe is widely disparate in its availability and quality. The review of information focuses first and foremost on information that was consistent and comparable at the Europe-wide level. This information was augmented with data from country-wide and within-country surveys and occasionally with information from individual bays, estuaries or sections of country coastlines. The information from these finer levels of resolution mostly provides key and in-depth examples of coastal change and its causes; it was not possible to collect this site-specific information for most areas. When possible, information was collected from the primary literature but much of this information exists in agency reports and online databases and these were often the most common sources of information.

Definitions

'Habitat' and 'ecosystem' are terms that have often been used inconsistently and interchangeably (e.g., Beck et al. 2001). In this review, 'habitat' indicates a focus on the predominant features that create structural complexity in the environment, such as plants (e.g., seagrass meadows, kelp forests), animals (oyster reefs) or other geological features (e.g., rocky reefs, mudflats). 'Ecosystem' indicates a focus on the many other plants, animals, natural processes and services associated with the predominant features. These definitions are consistent with those commonly used in European policy (e.g., E.U. Habitats Directive, EC 2003).

'Habitat loss' and 'habitat conversion' are treated as representations of similar impacts, that is, a reduction in the distribution of natural habitats. Habitat degradation and fragmentation also represent serious impacts but they are not often treated as habitat loss because of difficulties in measurement. Loss clearly occurs when natural habitats such as salt marshes are filled with sediments and blocked from the sea to form agricultural fields. Habitat conversion often occurs when more structurally complex natural habitats are converted to less-complex habitats (e.g., oyster reefs are dredged and mudflats are left). These converted habitats (e.g., dredged mudflats) may still have some natural value, but they exist for artificial reasons, and less structurally complex habitats usually have lower diversity and productivity (e.g., Heck & Crowder 1991, Beck et al. 2001). Areas are rarely converted from less-complex to more complex natural habitats unless there is active habitat restoration, which is treated as habitat gain. Structurally complex habitats are clearly becoming rarer across Europe and the world, and that is recognised in their common treatment in policy, conservation and management.

Habitat degradation, such as the invasion of non-native algae into seagrass meadows or ditching in marshes, is also a serious issue that has ecosystem implications and often is a precursor to loss

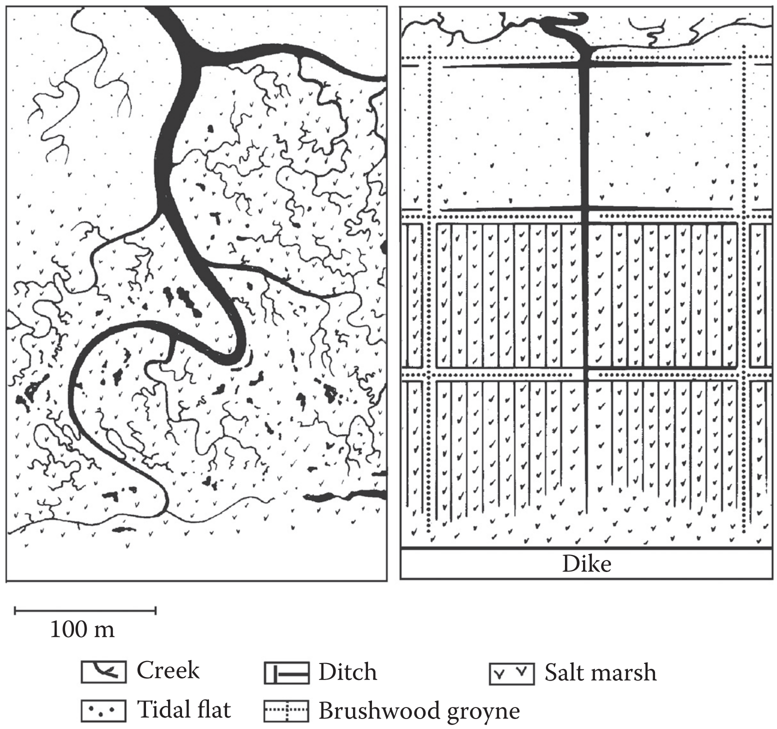

Figure 1 Morphology of a natural salt marsh (left) and an artificial salt marsh (right) functioning as foreland to protect a dyke in the Wadden Sea. (From Reise 2005. With permission.) Compared with natural ones, artificial salt marshes are smaller, fragmented, truncated at the landward side and of a simplified structure, although the halophytic vegetation may be similar.

of natural habitats. Degradation is, however, difficult to measure because it represents a decrease in condition, not a change in distribution (i.e., habitat loss). Degradation is particularly difficult to measure at the regional, national and multinational levels considered in this review. Nonetheless it is clear that present-day salt marshes in Europe, for example, are much different from salt marshes of the past, not only because they are smaller (habitat loss), but also because they are much less complex with fewer channels and straighter, less-fractal edges (Figure 1). This degradation results in much less efficient transfer of nutrients and species at this critical terrestrial/marine interface (Minello et al. 1994). Habitat fragmentation falls between loss and degradation. Fragmentation occurs when previously continuous habitats become patchier (e.g., loss of patches of seagrass within a larger bed). In this review, these changes in the habitat are treated as loss when it can be measured, which is generally an issue associated with monitoring resolution because many large-scale surveys and spatial imagery do not capture increases or decreases in patchiness.

General European context

European estuaries and coastal areas have a long history of intense human impacts and urbanisation and are among the most severely degraded coastal temperate systems worldwide (Lotze et al. 2006). Europeans have exploited near-shore marine resources since at least the Palaeolithic (e.g., Knottnerus 2005), and during the Roman times European coastal landscapes were far from pristine (Rippon 2000). To improve the habitability of coastal areas and exploit their resources humans

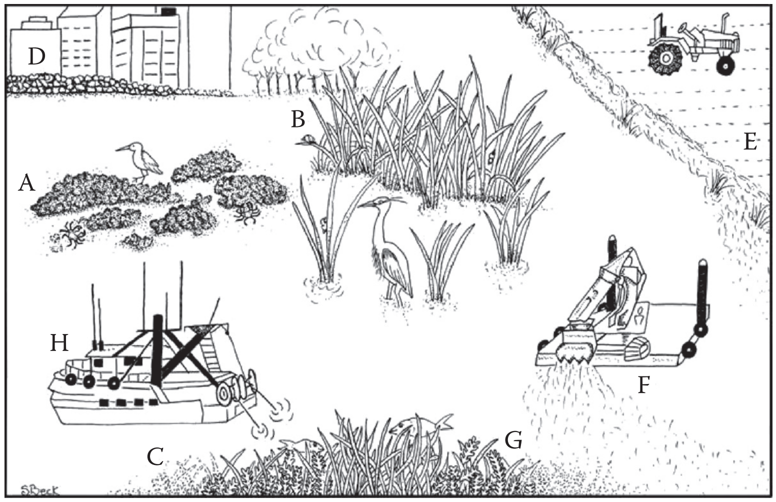

Figure 2 Native coastal habitats such as oyster reefs (A), salt marshes (B), and seagrass meadows (C) are being squeezed out of coastal zones by many factors including coastal development and defence (D), diking and ditching (E), dredging (F), pollution and excess sedimentation (D, E, F), non-native species (G) and destructive fishing and overfishing (H).

have altered the fluvial sedimentation patterns, controlled the river networks, reclaimed marsh lands and developed agriculture and fishing (Cencini 1998). Heavy exploitation, modification and deterioration of coastal habitats increased significantly during the Middle Ages, when coastal areas became more populated and humans started to systematically transform the coastal environment and commercially exploit its resources (Wolff 1992, Hoffmann 2005), and assumed dramatic proportions during the nineteenth and twentieth centuries, when uncontrolled coastal development, industry and tourism destroyed near-shore habitats and assemblages and deeply modified coastal landscapes and seascapes (Cencini 1998, Reise 2005). This section does not intend to give detailed information on the history of human exploitation of coastal resources or provide an extensive review of every type of impact. Rather, it provides baseline information and some key examples of the major past and present human pressures to European coastlines (Figure 2). This information is relevant to an understanding of how the concentration of population, settlements and economic interests in near-shore coastal areas and bays has produced, and still produces, drastic and probably irreversible changes to native habitats and assemblages (e.g., Lotze et al. 2005). The section also provides a broad overview of the main E.U. and trans-national agreements and policies that have been developed to rectify or reduce damage to European natural habitats and associated species.

Threats to European coastlines

The exponential growth of European populations over time (McEvedy & Jones 1978) has driven the historical intensification and multiplication of human impacts on coastal habitats. Since ancient epochs, human densities have peaked along European estuaries and coasts, reflecting their importance for settlement, trade and transport. Many of Europe's capital cities are on or close to the coast, and altogether there are 280 coastal cities with populations above 50,000 (Figure 3). In the 1990s, approximately 200 million Europeans lived within 50 km of coastal waters (Stanners & Bourdeau 1995; see also EEA 1999a, Frid et al. 2003). Today the coasts of many European countries are among the fastest-growing areas in terms of social and economic development and it is expected

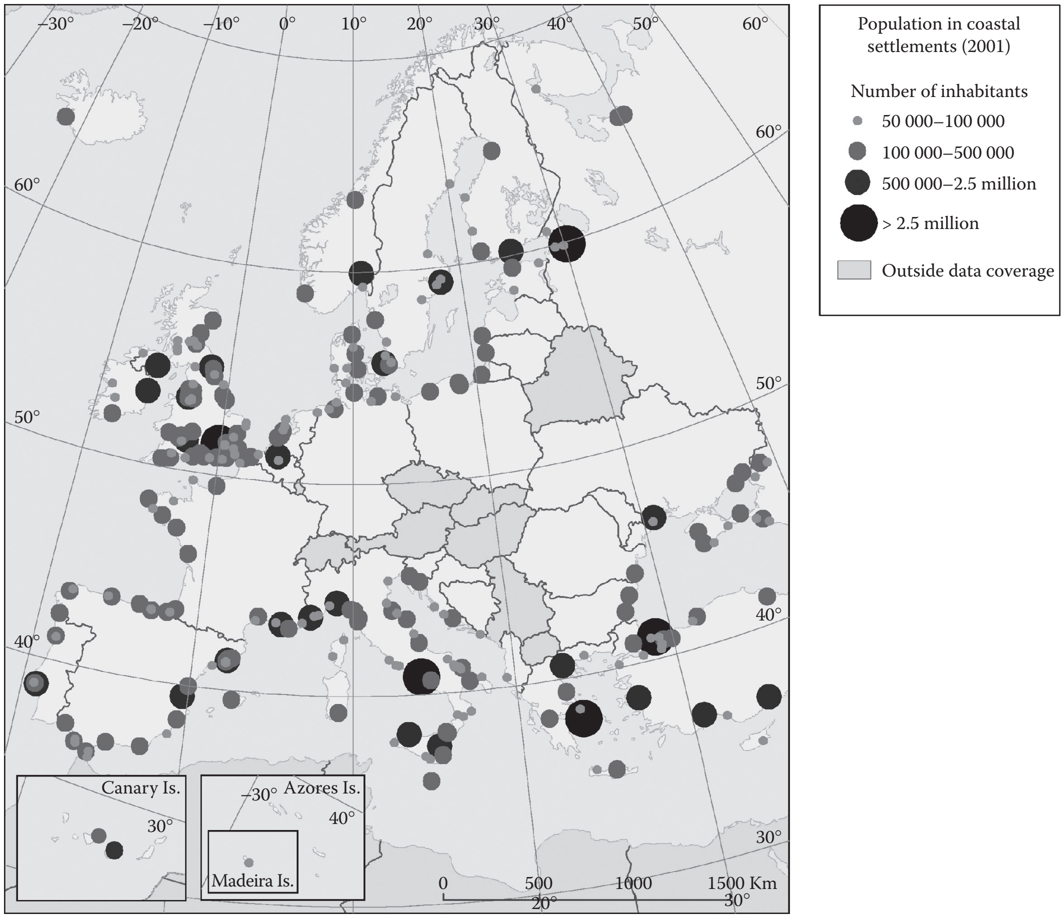

Figure 3 Coastal settlements with more than 50,000 inhabitants along European coasts. (From EEA 2006a. With permission.)

that European coastal areas will face increasing pressures from population growth (EEA 2005, 2006a). The coasts of the Mediterranean Sea, in particular, have always been among the most densely populated regions on Earth, with an estimated 5700–6600 people km^{-1} of coastline in 2000 (UNEP/MAP/PAP 2001). Along Mediterranean coasts, the population increased by 46% between 1980 (84.5 million) and 2000 (123.7 million), and it is projected to nearly double between 2000 and 2025 (UNEP/MAP/PAP 2001).

Increased land use and development of settlements, agriculture, industries, ports, military installations, mines, power plants and other infrastructures has accompanied population growth in European coastal areas. Their development has posed and still poses severe threats to coastal areas (EEA 2006a). Estuarine and coastal landscapes have been deeply modified and transformed in a process that in some regions, such as the western Netherlands, dates as far back as late prehistoric periods, when the first attempts were made to control the flow of water through the construction of dams and sluices (Rippon 2000). During the Roman times, reclamation of coastal marshes became intensive in some regions (e.g., the Severn Estuary; Rippon 1997), and after the tenth to twelfth centuries, large-scale transformations and reclamations took place systematically around Europe (Wolff 1992, Cencini 1998, Rippon 2000, Reise 2005). In the Wadden Sea region, about

15,000 km^2 of wetland, lagoons, coastal lakes and tidal flats have been embanked, drained and converted into arable land and pasture over the centuries (Figure 4; see also Wolff 1992, 1997). In the United Kingdom land reclamation has affected at least 85% of the estuaries since Roman times, with losses of intertidal areas ranging between 25 and up to >80% (Davidson et al. 1991); such widespread claim of estuarine land is continuing at rates of 0.2–0.7% yr^{-1} and affects also estuaries of recognised international wildlife importance included in the Ramsar/Special Protection Area (SPA) network.

Data from the CORINE project indicate that 22,000 km^2 of the coastal zone in Europe are covered in concrete or asphalt (EEA 2005), and that artificial surfaces increased by almost 1900 km^2 between 1990 and 2000 alone (EEA 2006a). The greatest urban developments occur along the Euro-Mediterranean coasts. At present about two thirds of the Mediterranean coastline is urbanised, with this fraction exceeding 75% in the regions with the most developed industries (UNEP/MAP/PAP 2001). More than 50% of the Mediterranean coasts are dominated by concrete with >1500 km of artificial coasts, of which about 1250 km are developed for harbours and ports (EEA 1999c). Growth of cities (particularly tourist developments) and development of industry in some regions (including the French Riviera, Athens, Barcelona, Marseille, Naples, the north Adriatic shorelines) have taken up to 90% of the coastline (Jeftic et al. 1990, Meinesz et al. 1991, Cencini 1998). In Italy, a survey carried out by World Wildlife Fund (WWF) showed that, in 1996, 42.6% of the entire Italian coast was subject to intensive human occupation (areas completely occupied by built-up centres and infrastructures), 13% had extensive occupation (free zones occupied only by extensive building and infrastructures) and only 29% was free from buildings and infrastructures (EEA 1999c). Coastal zone urbanisation will further increase in the near future, with projected increases of 10–20% for most Mediterranean countries (EEA 2006a).

Severe decreases of water quality have generally followed population growth with organic pollution as a major driving factor (Jansson & Dahlberg 1999, Diaz 2001, van Beusekom 2005). Excessive nutrient enrichment has been a problem in European waters historically (Islam & Tanaka 2004). Hoffman (2005) reports that archaeological signs of eutrophication from dense, mainly urban populations were detected on the Bodensee shore at Konstanz (Germany) in late-mediaeval times, and that in 1415 a royal ordinance tried to mitigate the low water quality of the Seine below Paris. Nutrient loads started to rise probably around 1700–1800, increased significantly in the early 1900s and steeply accelerated after the 1950s (Lotze et al. 2006). It is estimated that in the Baltic and North Sea regions nitrogen (N) and phosphorus (P) loads from land and atmosphere have increased about 2–4 and 4–8 times, respectively, since the 1940s (Nehring 1992, EEA 2001, Karlson et al. 2002). Historical reconstructions of the preindustrial trophic status in the Wadden Sea suggest about 5-fold greater organic matter turnovers nowadays compared with preindustrial conditions (van Beusekom 2005). The historical development in nutrient loads to the Mediterranean and Black Seas is unknown, but is probably of the same magnitude (UNEP/FAO/WHO 1996, EEA 1999c). For example, in the north Adriatic Sea nutrient load has been increasing since at least 1900 and it markedly intensified after 1930 (Barmawidjaja et al. 1995, Sangiorgi & Donders 2004), with a doubling of nutrient loads in the Po river between 1968 and 1980 (Marchetti et al. 1989). In the Black Sea, concentrations of nitrate have increased 5 times and phosphate 20 times from the 1960s to 1980s (Gomoiu 1992).

The increased eutrophication has, as a secondary effect, led to increased oxygen consumption on the sea bed and expansion of areas with hypoxia and anoxia (Diaz 2001, Karlson et al. 2002). In the Black Sea up to 90% of the waters are anoxic. The Kattegat has been affected by seasonal hypoxia since the beginning of the 1980s, which has followed a more than 3-fold increase in N input in the 1960s and 1970s (Rosenberg et al. 1990). Similarly, in the north Adriatic Sea the first signs of hypoxia started around 1960 and developed into severe anoxic events over the past 20 yr (Barmawidjaja et al. 1995, Diaz 2001). Since the middle of the 1980s the phosphorus load has

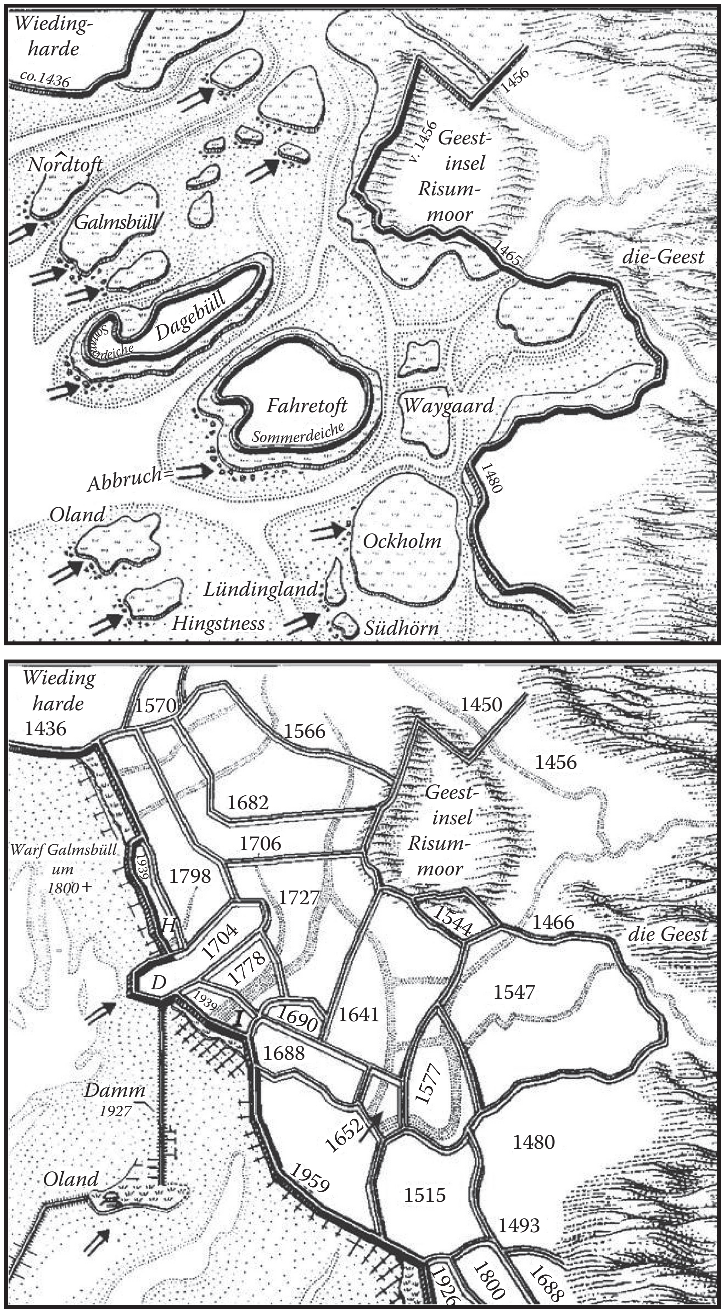

Figure 4 Maps of about 20 km of coasts in Nordfriesland circa 1500 (top) and in 1965 (bottom). (From Reise 2005. With permission.) Shown is the massive loss of coastal habitats due to land claim. In 1500 Dagebüll and Fahretoft islands were surrounded by low summer dykes. All the area was subsequently embanked (years of progressive diking are indicated in the 1965 map), and tidal flats and salt marshes were drained and converted to arable land and pasture. Pleistocene elevations are hatched, salt marshes stippled, tidal flats dotted, former creeks narrowly dotted and arrows point to sites of shore erosion.

generally levelled off or declined locally. In some areas such as the North Sea there have been declines in P up to 50% due to improved sewage treatment, reduced industrial discharges and a change to phosphorus-free detergents (Frid et al. 2003). However, there do not yet seem to be discernible European-scale reduction of nitrogen inputs, marine eutrophication or extent of anoxic areas (Karlson et al. 2002).

Increased loads of sediments have followed changes in land use both inland and along the coasts of Europe, but long-term data on water turbidity and sediment load are limited even at local scales (Lumb 1990). The greatest impacts were felt when forests were extensively cleared for timber, agriculture or urban developments, which together with interferences in the natural course of rivers caused dramatic acceleration of natural soil erosion (Airoldi 2003). In Europe clearing of forested catchments for agriculture commenced several thousand years ago (e.g., see the historical reconstruction by Cencini 1998). Episodes of accelerated erosion following phases of expansion of arable lands were common during mediaeval times (Hoffmann 2005) and became particularly severe during the nineteenth century (Pukaric & Jorissen 1990, Barmawidjaja et al. 1995).

Chemical pollution has also affected European estuaries and coastal waters since ancient times (Islam & Tanaka 2004), particularly in the Mediterranean Sea, where overall 101 pollution 'hot spots' have been identified, generally located in semi-enclosed gulfs and bays near important harbours, big cities and industrial areas (UNEP/MAP/PAP 2001, EEA 2006b). Pollution from shipping, oil spill traffic, drilling activities and related accidents is particularly severe in Europe (EEA 1998, 1999c, 2006a, Thompson et al. 2002). Some of the busiest shipping lanes in the world are found in the Baltic Sea, North Sea and Mediterranean Sea (Frid et al. 2003), and about 22% of the total world petroleum traffic passes through the Mediterranean Sea (Jeftic et al. 1990). Marine pollution has become a major concern in Europe, and many E.U., trans-national and national initiatives (see next section, p. 356) have helped to control the disposal of urban and industrial pollutants in coastal areas. Even so, there are still large pollution loads, and long-term contamination of sediments is a major problem.

Marine food resources have been used by Europeans since prehistory. At some heavily populated localities, particularly along the Mediterranean shores, the most valued species had severely declined in abundance and size by the end of the Roman era (Hoffmann 2005) with detectable effects on coastal systems (Sala 2004). Exploitation increased during late-mediaeval times, when fisheries became subject to market exploitation, and in subsequent centuries growing food demand and technological progress led to almost unrestricted overexploitation of coastal resources (Hoffmann 2005, Wolff 2005, Lotze et al. 2006). The total fish landings in European sea regions peaked at 12 million t in 1997, but have decreased since in both quantity and quality, down to 7.6 million t in 2002 (EEA 2006a).

Disruptive fishing techniques are considered among the major causes of physical destruction of marine coastal habitats at global scales (Watling & Norse 1998, Turner et al. 1999, Thrush & Dayton 2002). In Europe, bottom trawls, bivalve dredging, pneumatic hammering of date mussels, explosives and other disruptive fishing techniques have a long history of use, mainly in estuaries, bays and continental shelf waters (Fanelli et al. 1994, Bavestrello et al. 1997, Lindeboom & de Groot 1998, Cicogna et al. 1999, EEA 1999c, Johnson 2002, Hall-Spencer et al. 2003, Tudela 2004). In Britain, concern about the adverse effects of fishing on marine habitats and wildlife populations dates back to the fourteenth century when it was noted in a petition presented to Parliament in the year 1376–1377 (quoted in Hore & Jex 1880) that "the hard and long iron of the said 'wondyrchoun', [an oyster dredge] … destroys the spawn and brood of the fish beneath the said water, and also destroys the spat of oysters, muscles [sic], and other fish by which large fish are accustomed to live and be supported". The use of trawls and other mobile fishing gears accelerated sharply with the introduction of diesel engines in the 1920s (Watling & Norse 1998). The sea bed in Europe has been trawled to a depth of over 1000 m since the 1970s, affecting extensive areas of benthic habitats.

Aquaculture, which can have some benefits, has had increasingly adverse effects on coastal habitats. World aquaculture production has increased by >300% since 1984, with growth of about 10% a year in the 1990s, making it the fastest-growing food production activity (Mock et al. 1998). In Europe the culture of fish, shrimp, shellfish and seaweeds has been used as an alternative source of marine food at least since Roman times (Hoffmann 2005), and in regions such as the Po delta area salt marshes were transformed centuries ago into artificial fishing lagoons (Cencini 1998). Unprecedented growth in production has occurred in the last decades (Váradi 2001, EEA 2006a), with significant impacts on bottom habitats and assemblages (e.g., Holmer et al. 2001, EEA 2006b). In 1998, total marine aquaculture production in Europe was >1.3 million t, with most production concentrated in Norway, France, Spain and Italy (Váradi 2001). In the Mediterranean region, marine aquaculture production has increased from 19,997 t in 1970 to 339,185 t in 2002 (EEA 2006b), and the total production of salmon in fish farms (mainly in Norway and Scotland) has increased from 70,000 t in 1990 to 148,000 t in 1996 (EEA 2002) up to 540,000 t in Norway alone in 2003 (EEA 2006a).

More recent pressure and threats to European coastlines are from tourism and development of recreational infrastructures, particularly in the Mediterranean region. Before the 1930s, tourism was a relatively minor phenomenon, although it did lead to the beginning of urbanisation in seaside areas (EEA 1999c). From the 1930s onward and especially after World War II, mass tourism started to grow, and the phenomenon was amplified by the development of transport facilities (e.g., Cencini 1998). Nowadays, the Mediterranean coast is the world's leading holiday destination, accounting for 30% of the world's tourism, and in some countries coastal tourism represents up to 90% of all tourism. In 1990 alone, 135 million vacationers flocked to the Mediterranean coast, and by 2025 the annual crowd is projected to increase to 235–350 million tourists (EEA 1999c). Effects on coastal habitats have been devastating. In Spain, tourist developments occupy 42% of the entire coast (Jeftic et al. 1990), with peaks in areas such as the Catalonia coast, where tourist developments make up 337 km of the total 580 km. Similarly, buildings, roads, bathing establishments and other recreational facilities located directly over the beaches and sand dunes almost entirely occupy the Italian coast of the north Adriatic Sea (Cencini 1998). The demand for marinas and yacht harbours has been growing all over the Mediterranean coasts, with an estimated growth for France of 1.5–2.6% yr^{-1} (EEA 2006a).

Increased coastal erosion and flooding (often indirectly related to human activities) also pose serious threats to European coastlines (EC 2004). A recent inventory of coastal evolution in Europe undertaken within the CORINE programme showed 55% of the coastline to be stable, 19% to be suffering from erosion problems and 8% to be depositional (Stanners & Bourdeau 1995). Some coastal regions are also gradually subsiding (Bondesan et al. 1995, EEA 2006a), with subsidence sometimes enhanced by groundwater or petrochemical extraction (Bird 1993), while land lift up to 9 mm yr^{-1} is occurring in areas of the Baltic Sea (HELCOM 1998).

Erosion mitigation schemes have been put in place to respond to the problem of coastal erosion, which affected about 7600 km of coasts in 2001 (EC 2004). Defence measures include a variety of hard defence structures (e.g., breakwaters, groynes, seawalls, dykes or other rock-armoured structures), which have proliferated in the second half of the twentieth century, leading to severe hardening of coastal areas and changes in sediment structure (Airoldi et al. 2005). In the north Adriatic Sea, >190 km of artificial structures, mainly groynes and breakwaters, seawalls and jetties (Figure 5), have been built along 300 km of naturally low sedimentary shores (Bondesan et al. 1995, Cencini 1998). This hardening has caused severe losses and alterations of shallow sedimentary habitats (e.g., Martin et al. 2005) and has introduced new artificial habitats, with dramatic effects on native habitats and assemblages (Bacchiocchi & Airoldi 2003, Bulleri & Airoldi 2005). Similar

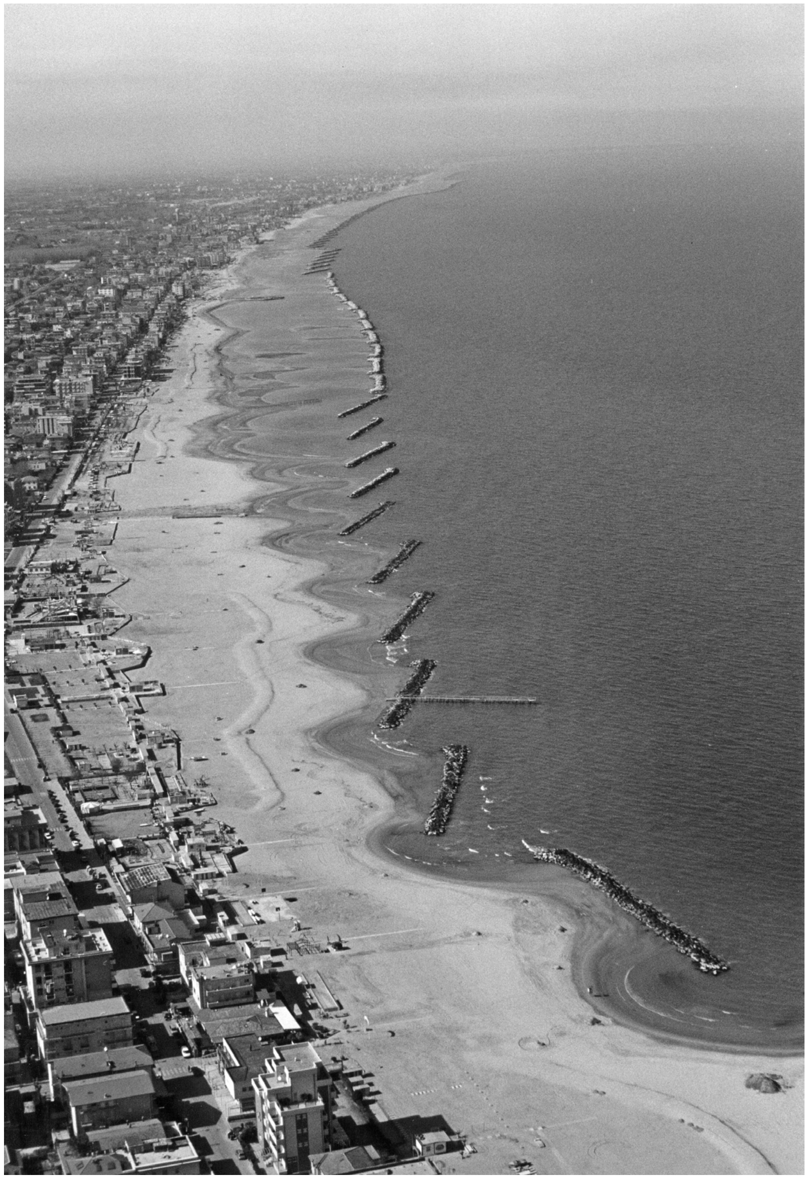

Figure 5 Coastal defence structures along the highly urbanised Italian shores of the north Adriatic Sea. (Photo by Giorgio Benelli. With permission.)

examples occur in many other European coasts (e.g., Davidson et al. 1991, Anthony 1994, Reise 2005), presumably affecting an overlooked enormous amount of benthic habitats.

Overall >15,000 km of coasts in Europe are now actively retreating, some of them in spite of coastal protection works (2900 km), and another 4700 km are artificially stabilised (EC 2004). Globally the problem of erosion and flooding is becoming much more serious because of rising sea levels and an increased storm frequency as a result of global climate change (Bray & Hooke 1997, Valiela 2006). During the past century, the mean global sea level has risen between 10 and 25 cm (Burke et al. 2001). The Intergovernmental Panel on Climate Change (IPCC) Working Group has projected a global sea-level rise of 15–95 cm by the year 2100. The recession of coastlines is expected to continue even in the absence of new human activities (Bondesan et al. 1995).

Approximately 450–600 non-indigenous marine species have been added to European coastal fauna and flora, often facilitated by human-mediated processes such as shipping, aquaculture and aquaria (Reise et al. 2006 and references therein). Some introductions occurred hundreds of years ago, as is the case of the sand-gaper, *Mya arenaria*, which was probably transported as food from North America to Europe by the Vikings (Petersen et al. 1992). Rates of introduction rose dramatically in the past century, probably in relation to increased shipping and aquaculture. In the Mediterranean Sea, the number of introduced species has nearly doubled every 20 yr since the beginning of the twentieth century (Boudouresque & Verlaque 2002). The number of introduced species is often greatest in estuaries, lagoons, embayments, closed seas, canals and harbours, probably because of low species richness combined with strong anthropogenic change (Occhipinti-Ambrogi 2001, Reise et al. 2006). They have profoundly altered European coastal ecosystems and are displacing some native species. One notorious example is the invasion of *Caulerpa taxifolia* along the coastlines of Spain, France, Monaco, Italy, Croatia and Tunisia (Meinesz et al. 2001) although there is debate about the area of benthos affected (Jaubert et al. 2003). Nevertheless, these invasions do not seem to have caused large-scale extinctions in recipient biota and losses of native coastal habitats (Wolff 2000).

E.U. coastal policies and trans-national agreements

The European Union has been involved in efforts to protect the European natural heritage for the past 30 yr (Table 1). At the international level, the European Union has signed a number of important conventions aimed at nature protection, including the Ramsar Convention on the Conservation of Wetlands, the Bonn Convention on Migratory Species, and the Rio Convention on Biological Diversity (CBD), and shares the international commitment of the World Summit for Sustainable Development to establish a globally representative system of marine and coastal protected areas by 2012 (Kelleher et al. 1995, Green & Paine 1997).

At the European level, the Bern Convention has led the development of policy and action in nature conservation in Europe. It lists protected species, including a number of marine plants and invertebrates, and requires its parties to prevent the disappearance of endangered natural habitats. The Sixth Environmental Action Plan, setting the European Union's environmental policy agenda until 2012, highlighted nature and biodiversity as a top priority, and the E.U. leaders in Gothenburg in 2001 launched the E.U. Sustainable Development Strategy to halt the loss of biodiversity in the European Union by 2010. The European Union has adopted a Biodiversity Strategy and Action Plans (currently under review), and Member States have developed — or are developing — their own national strategies and action plans (e.g., U.K. Biodiversity Group 1999).

Other policies, in particular the Birds Directive and the Habitats Directive, have been promoted to rectify or reduce damage to European natural habitats and associated species. Following the criteria set out in the directives, each Member State must draw up a list of sites hosting wild fauna and natural habitats and put in place a special management plan to protect them, combining long-term preservation with economic and social activities, as part of a sustainable development strategy. The final aims are the creation of a European Ecological Network of Special Protection Areas (SPAs) and Special Areas of Conservation (SACs), called NATURA 2000, and the integration of nature protection into other E.U. policy areas, such as agriculture, fishery, industry, regional development and transport.

Indirect protection to a variety of habitats also comes from a number of E.U. Directives that regulate water quality, including the Dangerous Substances, Shellfish Waters, Integrated Pollution

Table 1 Summary of main protection initiatives adopted by the European Union and State Members that directly or indirectly address issues related to the protection of European marine coastal habitats and associated assemblages

Initiative	Description	Web site
Ramsar Convention	Ramsar Convention on Wetlands, Ramsar (1971). Provides the framework for the conservation and wise use of wetlands of international importance especially as waterfowl habitat. Includes salt marshes and some lagoon systems and marine waters to a depth of 6 m.	www.ramsar.org
Bonn Convention	Convention on the Conservation of Migratory Species of Wild Animals, Bonn (1979). Intergovernmental treaty, aiming to conserve terrestrial, marine and avian migratory species throughout their range.	www.cms.int
Rio Convention	Convention on Biological Diversity, Rio de Janeiro (1992). Provides legal framework for biodiversity conservation and sustainable development. The Jakarta Mandate (1995) leads activity in marine biodiversity management and conservation.	www.biodiv.org/convention/default.shtml
Bern Convention	Convention on the Conservation of European Wildlife and Natural Habitats, Bern (1979). Aims at preserving wild flora and fauna (including some marine species) and their natural habitats through national programmes using the co-operation between European States.	www.coe.int/T/E/Cultural_Co-operation/Environment/Nature_and_biological_diversity/Nature_protection/
ICES Convention	Convention for the International Council for the Exploration of the Sea, Copenhagen (1964). Co-ordinates and promotes marine research in the North Atlantic, including the Baltic Sea and North Sea, and the Common Fisheries Policy on the protection of the marine environment and the regulation of fisheries.	www.ices.dk/aboutus/convention.asp
OSPAR Convention	Convention for the Protection of the Marine Environment of the northeast Atlantic, Paris (1992). Merged the 1972 Oslo Convention on dumping waste at sea and the 1974 Paris Convention on land-based sources of marine pollution. It guides the protection of the marine environment of the northeast Atlantic and the identification of priority habitats and species.	www.ospar.org
North Sea Conference Declarations	Six declarations produced at as many International Conferences on the Protection of the North Sea (first in Bremen, 1984). Political commitments to the protection of the North Sea environment, addressing, e.g., species and habitats issues, pollution and fisheries.	www.sweden.gov.se/sb/d/6363/a/57475;jsessionid=abPgelqjfxJ8
Helsinki Convention	Convention on the Protection of the Marine Environment of the Baltic Sea Area, Helsinki (1992). Guides the protection of the marine environment of the Baltic Sea from pollution and the identification of priority habitats and species for protection.	www.helcom.fi
Trilateral Wadden Sea Cooperation	Joint Declaration of The Netherlands, Denmark and Germany, Copenhagen (1982). Aimed at co-ordinating the protection of the Wadden Sea National Park. In 1997, a Trilateral Wadden Sea Plan was adopted.	www.waddensea-secretariat.org

(continued on next page)

Table 1 (continued) Summary of main protection initiatives adopted by the European Union and State Members that directly or indirectly address issues related to the protection of European marine coastal habitats and associated assemblages

Initiative	Description	Web site
Barcelona Convention/ Mediterranean Action Plan (MAP)	Amended in 1995 as the Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean, Barcelona (1976). Provides legal framework to MAP (1975), under UNEP Regional Seas Programme. Aims to control human impacts (e.g., marine pollution, tourism) and protect the marine and coastal Mediterranean environments.	www.unepmap.org
Bucharest Convention	Convention on the Protection of the Black Sea against Pollution, Bucharest (1992). Aims to control and prevent pollution and preserve biodiversity in the Black Sea.	www.blacksea-commission.org
Birds Directive (79/409/EEC)	Council Directive on the Conservation of Wild Birds. Identifies 194 endangered species and subspecies of birds for which the E.U. Member States are required to designate Special Protection Areas (SPAs). Over 4000 SPAs have been designated to date, covering 8% of E.U. territory.	www.ec.europa.eu/environment/nature/ nature_conservation/eu_nature_legislation/ birds_directive/index_en.htm
Habitats Directive (92/43/EEC)	Council Directive on the Conservation of Natural Habitats and of Wild Fauna and Flora. Aims to protect wildlife species and habitats which have conservation that requires the designation of Special Areas of Conservation (SACs). These sites, together with the SPAs of the Birds Directive, make up the NATURA 2000 network, currently covering about 15% of E.U. coasts. Marine habitats broadly defined, and few marine species listed.	www.ec.europa.eu/environment/nature/ nature_conservation/eu_nature_legislation/ habitats_directive/index_en.htm
Shellfish Waters Directive (79/923/EEC)	Council Directive on the Quality Required of Shellfish Waters. Aims to ensure a suitable environment for shellfish harvest. Member States are required to designate coastal and brackish waters that need improvement to support shellfish fisheries.	www.europa.eu.int/eur-lex/en/consleg/pdf/ 1979/en_1979L0923_do_001.pdf
Water Framework Directive (2000/60/EC)	Integrates and updates existing E.U. water legislations (e.g., Discharges of Dangerous Substances, Urban Waste Water Treatment, Nitrates Directive) and provides for water management. Complemented by the recently revised Bathing Water Directive (2006/7/EC).	www.europa.eu.int/comm/environment/ water
Marine Strategy Directive	The proposed directive aims to define common objectives and principles at E.U. level to achieve good environmental status of the European marine environments by 2021. It will establish European Marine Regions as management units for implementation.	www.ec.europa.eu/environment/water/ marine.htm

Note: All web sites last accessed 8 August 2006.

Control, Urban Waste Waters, Bathing Waters and the Water Framework (Table 1). These commitments provide for the regulation of discharges to the sea and have set targets and quality standards covering many metals, pesticides and other toxic substances.

In addition to E.U. initiatives, a number of trans-national agreements have been developed to address some conservation issues within the main European seas, including the OSPAR Convention, the North Sea Conference Declarations, the Helsinki Convention, the Trilateral Cooperation on the Protection of the Wadden Sea, the Mediterranean Action Plan and the Black Sea Environmental Programme (Table 1). These programmes generally address water quality and fishery concerns and are not specifically focused on habitat loss and protection, although initiatives have also included commitments toward establishing an integrated network of Marine Protected Areas (MPAs). A more focused initiative for the Atlantic Ocean and Baltic Sea is the commitment of the Joint Ministerial Meeting of the Helsinki and OSPAR Commissions to complete by 2010 a joint network of MPAs that, together with the NATURA 2000 network, would be ecologically coherent.

In recent years there has been increasing awareness that past efforts to protect European marine coastal habitats and associated species have been marginal relative to terrestrial environments and that there is limited co-ordination of national and transnational initiatives at a European level. The global MPA database indicates that there are 1129 MPAs in Europe (Figure 6) covering 236,000 km^2 (MPA Global 2006). Most of these MPAs are small, and they often lack adequate political and

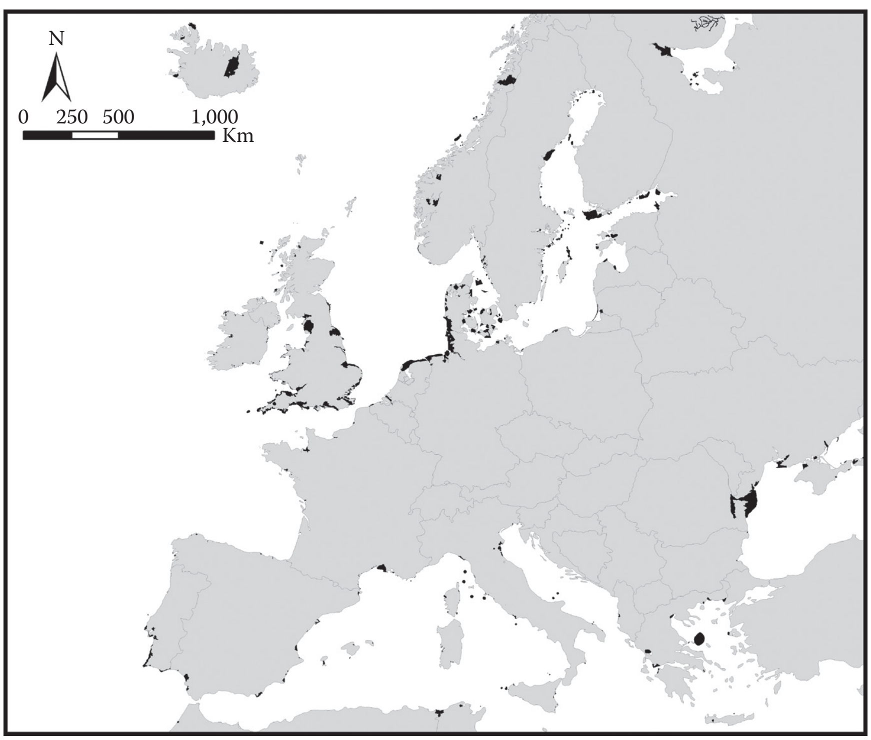

Figure 6 Marine Protected Areas in Europe (UNEP/WCMC 2006; data extracted August 2006). These MPAs include, for example, nature reserves, national parks, habitat/species management areas, RAMSAR Wetlands of International Importance and World Heritage Sites. The map should be considered indicative of general distribution not areal extent of MPAs.

financial support and effective enforcement (an extensive review of MPAs in Europe is offered in Kelleher et al. 1995, and specifically for the Mediterranean Sea in Badalamenti et al. 2000). The application of the Birds and Habitats Directives to the marine environment is also presenting major challenges, with significant delays in the selection and designation of the marine sites of the NATURA 2000 network. In response the European Union has set the protection of the marine environment as a major priority and has launched the Marine Strategy Directive, specifically aimed at protecting and conserving marine ecosystems and promoting the sustainable use of marine resources through the development of an integrated, coherent policy for the marine environment. A 2004 conference in Malahide (Ireland) has also set the necessary steps to complete the selection of the marine NATURA 2000 sites by the end of 2008.

Coastal habitats in Europe

Despite its relatively small geographic size, Europe has a very long coastline, approximating 325,892 km, including islands (Pruett & Cimino 2000). It comprises the main marine regions of the northeast Atlantic, part of the Arctic, the Baltic Sea, the North Sea, the Mediterranean Sea and the Black Sea (Frid et al. 2003, EEA 2006a). It includes primarily temperate environments as well as some Arctic and subtropical climate environments and covers a variety of geomorphological features (EEA 2002).

There are a large variety of extremely productive, diverse and valuable natural coastal habitats, including sea-bed communities of macroalgae and seagrasses, tidal mudflats, salt marshes and biogenic reefs (Stanners & Bourdeau 1995). However, there does not seem to be any comprehensive summary of the current distribution and status of native habitats along European coastlines as a whole. Databases are available or are being prepared for some habitats (e.g., wetlands (Nivet & Frazier 2004) and seagrasses (Green & Short 2003)), countries (e.g., the United Kingdom; Hiscock & Tyler-Walters 2006), and regions (e.g., northwest Europe, the OSPAR regions and the Baltic Sea; ICES 2006a). However, for most European coasts information, if any, is scattered, fragmented and not easily accessible. Even less information is available about long-term trends of habitat loss or degradation. With some notable exceptions (e.g., the Wadden Sea; Lotze & Reise 2005), there is little comprehensive historical information on coastal habitats prior to about 1900 (e.g., Hiscock & Kimmance 2003).

In this section published information, from both the scientific and grey (reports, web sites) literature is critically documented for major coastal habitats. The aim was not to reconstruct trends from local historical sources, maps or databases but to cover the most pertinent literature that reported data and information at regional, national, trans-national or European levels. The data summarised in this section are of variable quality and it should not be inferred that they provide a complete picture of the status of European coastal habitats.

Coastal wetlands and salt marshes

Current distribution and status

Much of the European coastline consists of a chain of extensive estuaries, lagoons and intertidal bays interspersed through stretches of rocky shore and sandy beaches. These coastal wetlands represent some of the most productive and biologically diverse components of near-shore ecosystems (Dugan 1993, Keddy 2000). They provide nursery grounds for commercially important fishes, habitat for shellfish, birds and a variety of biota and play a fundamental role in flood control, nutrient cycling and sediment dynamics. These coastal wetlands are patchworks of sand, mud flats and salt marshes. Salt marshes, with their vegetated complex surfaces, form one of the most important components of these wetlands (Adam 1990).

Coastal salt marshes are distinctive habitats that can be delineated relatively easily on remotely sensed images (e.g., Ekebom & Erkkilä 2003). In Europe, they have been the target of studies for a long time (see, among others, Dijkema 1984, Allen & Pye 1992, Jones & Hughes 1993, Rippon 2000, Adam 2002). However, quantitative information on their distribution and status is limited even at local scales. Information is often available for the more general category of 'coastal wetlands' and here the focus is mainly on this broader habitat type. Accounts exist at regional and local scales but most European countries lack comprehensive inventories of the extent and status of coastal wetlands, with numerous countries lacking almost any organised information on these conspicuous habitats (Table 2). One of the greatest difficulties in completing such inventories is in identifying intact wetlands, that is those wetlands that are not so severely transformed or deteriorated as to be functionally extinct (Allen 2000, Nivet & Frazier 2004). In the Severn Estuary, England, for example, there are a total of about 14 km^2 of intact marshes and about 840 km^2 of enclosed marshes, and this ratio of intact to enclosed coastal marshes may be common across Ireland, France, the Netherlands, northwest Germany and Denmark (Allen 2000). Other difficulties in estimating coverage include incompatible information (e.g., inconsistent classifications and methodologies) and the lack of co-ordination between different studies or for different wetland types (Nivet & Frazier 2004).

A European-scale review of the current distribution and coverage of coastal wetlands has been recently completed by Nivet & Frazier (2004) that integrates and updates previous inventories by Jones & Hughes (1993) and Finlayson & Spiers (1999). According to this inventory, the total cover of marine/coastal wetlands in Europe is around 51,910 km^2, and detailed information for individual countries is summarised in Table 2. An inventory of the distribution of European wetlands, including coastal wetlands, is also available in the CORINE database (EEA 1999b). A broad map of the distribution of salt marshes in Europe is given in Figure 7.

There is little comprehensive information on the status of coastal wetlands and salt marshes in Europe but there are clear indications that the historical concentration of human activities in European coastal wetlands has deeply modified their structure and function (Dijkema 1984, U.K. Biodiversity Group 1999, Allen & Pye 1992). Adam (2002) points out that nowadays minerogenic sedimentation prevails over autogenic (organic matter) sedimentation in the majority of European marshes. Most wetlands have deeply altered flow regimes (e.g., Cencini 1998, Reise 2005) with associated important effects on sediment dynamics as well as nutrient and salinity regimes (e.g., Allen 2003) and often are heavily polluted (e.g., Trombini et al. 2003). Their vegetation composition is the product of centuries of use and management (e.g., Wolff 2000) and their fauna and flora have been deeply transformed by introduced species (Reise et al. 2006).

Historical losses and causes

Coastal wetlands have suffered some of the most serious habitat loss rates (Dugan 1993, Suchanek 1994, Rippon 2000, Valiela 2006) and some estimates suggest that over time temperate estuaries and coastal areas may have lost approximately 67% of the wetlands that existed (Lotze et al. 2006). Even when wetlands have not been completely lost, significant degradation of their environment has often occurred, impairing their functions (Dugan 1993, Wolff 2000).

Exploitation of coastal wetlands and salt marshes in Europe dates back to at least the Neolithic, when salt marshes were used for salt production (e.g., Rippon 2000). Since then, these habitats have been increasingly exploited, providing location for settlement, agriculture and harbours; source of food, water and raw materials; and a focus for transport, trade and exploration (Rippon 1997,

Table 2 Estimates of actual cover and historical losses of coastal wetlands (and when possible salt marshes) for European countries (and eventual additional regional information), main attributed causes of loss and known history of exploitation

Country	Cover (km^2)	Loss	Cause of loss	Exploitation history (yr)	Regional data/additional information	Additional references
Albania	150	>550–600 km^2 since mid-1900s	D			
Belgium	8.3	n.a.		2000	Limited settlement during Iron Age	Rippon 2000
Bosnia-Herzegovina	n.a.	n.a.				
Bulgaria	n.a.	n.a.				
Croatia	n.a.	n.a.				
Cyprus	n.a	n.a.				
Denmark	8851*	60% ^ since 1870	D		Country's coastline shortened by 1168 km (14%)	
Estonia	n.a.	n.a				
Finland	501	50% ^	D		Losses of 22.8% ^ between 1950 and 1985	
France	3 813	86% in twentieth century	D, U		In Bretagne, 40% of coastal wetlands lost since 1960, and 66% of the remaining seriously affected by drainage; ongoing losses in the Languedoc Roussillon wetlands	EEA 2006a
Georgia	n.a.	n.a.				
Germany	6809*	56.6% ^ in 1950s to 1984	D, U	2000	In the Wadden Sea, 200 km^2 of salt marshes lost during 1950–1984, and only 400 km^2 remain today; in the whole Wadden Sea (not only Germany) 14,650 km^2 of coastal wetlands lost since the eleventh century (33% of salt marshes in 1930–1987)	Dugan 1993, Rippon 2000, Reise 2005
Greece	1011	73% salt marshes in twentieth century	WQ, D			EEA 2006a
Iceland	n.a.	n.a.				
Ireland	n.a.	n.a.	D		>250 salt marshes are located/mapped; 65 km^2 of estuarine area reclaimed in the Shannon estuary and 20 km^2 in the Wexford Harbour mainly in nineteenth century	Curtis & Skeffington 1998, Healy & Hickey 2002
Italy	n.a. <1000 of salt marshes	25,000 km^2 ^ since Roman times 6000 km^2 of salt marshes in twentieth century	D, S, U, A	2500	60% of the estimated losses occurred in 1938–1984; on the Po river delta, >70% of salt marshes reclaimed in 1870s–1960s; in Friuli Venezia Giulia 631 km^2 lost in 1877–1990	Stanners & Bourdeau 1995, Cencini 1998

Latvia	1426	n.a.				
Lithuania	413	70% ^ during the past 30 yr	D			
Malta	n.a.	n.a.				
Monaco	n.a.	n.a.				
Montenegro	n.a.	n.a.				
Netherlands	4043 (85 of salt marshes)	4000 km^2 of salt marshes between twelfth century and second half twentieth century	D, WQ, U	>2500	Humans affected salt marshes since at least the Iron Age; earliest dam dates 175 B.C.	Wolff 1992, 1997, 2000, Rippon 2000
Norway	n.a.	n.a.				
Poland	n.a.	89.8% ^	D			
Portugal	795	n.a.	D, S, E		70% of salt marshes in the western Algarve lost before 1988	Allen 2003
Romania	5297	4000–6600 km^2 ^				
Russia[a]	5786*	n.a.				
Slovenia	n.a.	n.a.				
Spain	1041	60%	D, U		Most of the losses occurred in 1965–1990	EEA 2006a
Sweden	6000	23% ^ since early 1800s	D		>50% degraded since early 1800s; in some areas losses up to 80–90%	
Turkey	n.a.	n.a.				
Ukraine	n.a.	n.a.				
United Kingdom	5966 (450 of salt marshes)	>50% of salt marshes since Roman times; 38 lagoons in the latter half of 1980s; >913 km^2 of estuary area	D, U, E	2500	Coastal/marine wetlands comprise (in km^2) 14 sand dune slack, 2658 estuarine waters, 2793 intertidal flats, 450 salt marsh and 50 saline lagoons; salt marsh loss most significant in the 1800s but still ongoing (100 ha yr^{-1}).	Rippon 1997, 2000, Davidson et al. 1991, Boorman 2003

Note: If not otherwise specified in the reference column, information is derived from Nivet & Frazier (2004) and references therein, while information about history of exploitation is generally derived from Rippon (2000). A = aquaculture, D = drainage/embankment/land claim/conversion (e.g., to agriculture), E = erosion/sea-level rise/coastal squeeze, n.a. = not available, S = altered sediment loads (e.g., from inland deforestation), U = urban/harbour developments, WQ = water quality degradation. Note that the definition of coastal wetland is vague for most countries, and in many cases (indicated as *) only important or large marine wetlands were included. Concerning habitat loss, in many cases (indicated as ^) estimates refer to total wetlands because no distinction was made between coastal and inland wetlands, and often no time span is indicated. Overall, estimates should be considered as broad indications.

[a] Including Asian Russia.

Figure 7 Distribution of salt marshes along European coastlines. (From Boorman 2003. With permission.)

2000, Knottnerus 2005, Wolff 2005). European estuaries support major cities and harbours and have an enormous economical and social importance. This long history of human exploitation has deeply altered these habitats in extent and ecological characteristics. Although historical information exists for some regions or localities (Table 2), knowledge of the extent of loss of coastal wetlands is generally limited.

A first overview of E.U. wetlands and their loss was provided by Jones & Hughes (1993), and more recently by Nivet & Frazier (2004), but they could not produce a European-wide estimate because of the scarcity and poor quality of most data available. Denmark, the Netherlands, Germany, Finland, Lithuania, the United Kingdom, Spain, Greece, Italy, France, Poland, Romania and parts of Portugal and Sweden have reported losses of wetland exceeding 50% of original area, with peaks above 80% for some regions (Table 2). These estimates sometimes refer to whole wetlands (without distinction between coastal and inland) and most often cover a relatively small time span (over the last century). However, when historical and archaeological documentation are available, it is evident that significant losses of coastal wetlands started in Roman times and locally even earlier than that (e.g., Wolff 1992, Allen 1997, Cencini 1998, Rippon 2000). Overall, it has been suggested that in the Mediterranean Sea 28,000 km^2 (>90%) of coastal wetlands have been lost since Roman times (UNEP/MAP/PAP 2001). Recent estimates have also suggested that approximately two thirds of all European coastal wetlands that existed at the beginning of the twentieth century have now been lost (EEA 2006a).

Some of the best historical records of the loss of coastal wetlands are from the estuaries around the United Kingdom, the Wadden Sea and the Po river delta in the north Adriatic Sea. In the United Kingdom, coastal wetlands were already used in prehistoric times for salt production, pasture and collection of wild resources (Rippon 2000). During Roman times a more systematic modification of coastal marshes began, with documented reclamation of some areas around the Severn estuary (Rippon 1997). Since mediaeval times, the use of coastal wetlands (e.g., for settlement, agriculture and food) has increased continuously in intensity; salt marshes were systematically enclosed, drained, settled and used for agriculture and pasture, leading to large-scale claims throughout much of British estuaries (detailed historical reconstructions of these developments are given by Rippon 2000). Overall, in the United Kingdom it is estimated that about 913 km^2 of estuary area and 550 km^2 of salt marshes have been claimed since Roman times for agricultural, urban, harbour and industrial developments (Davidson et al. 1991, Nivet & Frazier 2004). Decline was most significant in the 1800s but it has continued until today. For example, in the Wash, 858 ha of salt marshes were converted to agricultural use between 1970 and 1980 (U.K. Biodiversity Group 1999). Similarly, average losses of 20–25% of salt marshes, with individual sites suffering losses of 10–44%, occurred between 1973 and 1998 in Kent and Essex (Cooper et al. 2001, Adam 2002), where many salt marshes are ‘squeezed’ between an eroding seaward edge and fixed flood defence walls (U.K. Biodiversity Group 1999, but see Wolters et al. 2005 for debate about the causes of salt marsh erosion in southeast England). Overall, in England and Wales it is estimated that 15% of salt marshes were lost between the 1940s and the 1970s (Nivet & Frazier 2004).

The history of human transformation of the Wadden Sea ecosystems has been recently reviewed (Lotze et al. 2005). Similar to estuaries in the United Kingdom, exploitation of the wetlands in this region started in prehistory, when people collected wild fauna and flora for subsistence, grazed cattle and sheep and made the first attempts to transform the coastal landscape and control the flow of water (Rippon 2000, Knottnerus 2005). Although these activities did not result in the disappearance of salt marshes, they changed the composition of their vegetation considerably (Wolff 2000). The conversion of coastal wetlands into arable land and freshwater lakes became systematic before the eleventh century (Reise 2005), and after the introduction of windmill technology, in the sixteenth century, it also became possible to drain shallow lakes (Wolff 2000). Embankments continued until the second half of the twentieth century (e.g., Figure 4), supported by improved building technologies, and it was only after the 1960s that exploitation slowed and there was a growing focus on conservation (Wolff 1997). Overall about 15,000 km^2 of coastal wetlands have been lost during this long history of progressive embankments (Reise 2005) and the whole Wadden Sea has been reduced to nearly half of its primordial size.

Massive loss of coastal wetlands has also occurred in Italy. It is estimated that there were about 7000 km^2 of coastal marshes remaining at the end of the nineteenth century, no more than 1920 km^2 in 1972 and <1000 km^2 today (Stanners & Bourdeau 1995). Compiled historical information is limited to a few areas (e.g., the river Tevere; Keddy 2000). Cencini (1998) recently reconstructed the evolution of the Po river delta in the north Adriatic Sea and the effects of human transformation on coastal wetlands. Ancient sources and maps (e.g., from Pliny the Elder, Polybius and Strabo) described the deltaic coastland as a continuous, almost impassable sequence of lagoon, marshes and rivers. Since the Greco-Etruscan times, and more so after the consolidation of the Roman Empire, the area was heavily inhabited and exploited, and important urban settlements, roads and commercial ports were developed. Over the centuries, the main causes of transformation of the delta were altered sedimentation patterns and direct land claim. Sediment loads were enhanced by extensive inland deforestation carried out already by the Celts and Romans (Bondesan et al. 1995, Barmawidjaja et al. 1995). Since the seventeenth and eighteenth centuries, hydraulic works of river diversions, embankment and drainage took place, and after 1870 wetland drainage occurred systematically over large scales not only to improve hygiene conditions and eradicate malaria but also

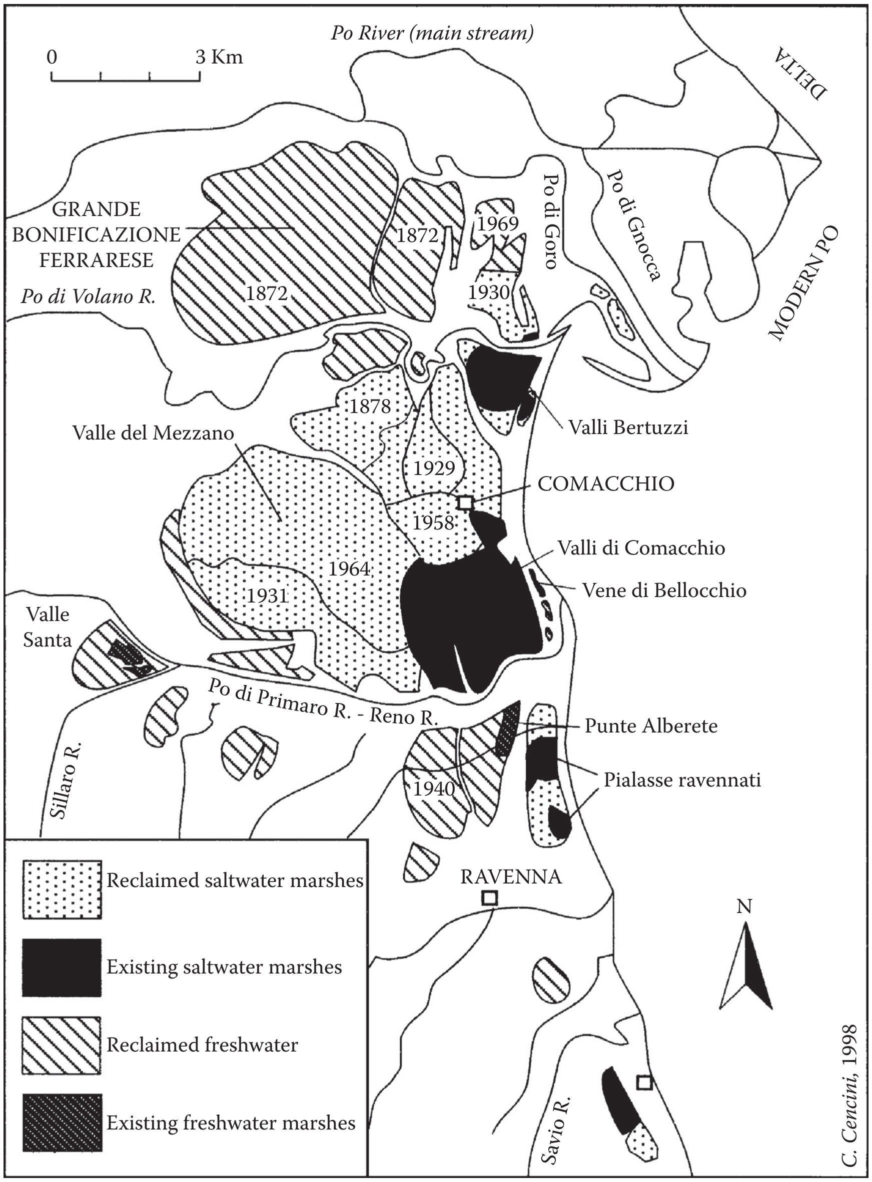

Figure 8 Reclaimed (with indication of year of reclamation) and existing fresh- and saltwater marshlands in the southern part of the Po river delta. (From Cencini 1998. With permission.) Overall 98% of the freshwater marshes and >70% of the salt marshes were reclaimed between the 1870s and 1960s.

to claim new farmland. Reclamation ended by the 1960s with an almost-complete elimination of marshes and irreversible changes to the hydrological and agricultural asset of the coastal plains. Overall in the Po delta, 98% of the freshwater marshes and more than 70% of the salt marsh that existed at the beginning of the twentieth century have been claimed (Figure 8). The ancient delta (until the twelfth century A.D.) covered about 1300 km^2, while the modern delta covers only 730 km^2.

Trends and threats

Pollution, eutrophication, drainage, conversion to agriculture and aquaculture, changes in sedimentation and hydraulic regimes, changes in sea levels, global warming, invasive species, fishery overexploitation and disruptive fishing techniques, urban development, shipping, tourism and water sports are generally considered the most serious threats to present-day coastal wetlands (e.g., Dugan 1993, Cencini 1998, Jansson & Dahlberg 1999, Wolff 2000, Kennish 2002, Hiscock et al. 2005). However, limited information is available on the extent to which these activities affect and impair specific coastal wetlands and thus the general importance in causing further habitat loss is difficult to rate.

Although in some regions embankments and drainage have been stopped or even reversed (e.g., Wolff 1997, Cencini 1998), land loss still continues to be the most pervasive threat to coastal wetlands and salt marshes. Between 1990 and 2000 there has been a total net loss of 390 km^2 of coastal wetlands around European coastlines, a significant proportion of which was lost as a result of drainage to reclaim land for development and afforestation (EEA 2006a). Expanding aquaculture and tourist infrastructures (e.g., marinas) are also making intensive use of estuaries and wetlands, with limited consideration of the consequences on these systems (EEA 2006a).

Global changes in sea levels are beginning to pose severe threats to salt marshes, coastal wetlands and river deltas because of the strong dependence of these habitats on water-level fluctuations and tidal regimes (Adam 2002, Morris et al. 2002). It has been suggested that the projected sea-level rise could cause the loss of up to half of European current coastal wetlands (EC 2004), with some of the largest losses expected to occur around the Mediterranean and Baltic Seas (Nicholls et al. 1999). When combined with other losses directly or indirectly due to human action up to 70% of the world's remaining coastal wetlands could be lost within the next 100 yr (Nicholls et al. 1999), although there is considerable uncertainty. In the United Kingdom, for example, it has been predicted that there could be a loss of freshwater and brackish habitats of around 4000 ha as a consequence of the combined effects of sea-level rise and a temperature increase of 3–4°C (Lee 2001).

Such habitat losses from sea-level rise could naturally be compensated for by the habitat regressing or developing inland. However, these inland areas are often developed and this coastal infrastructure is defended by building barriers; marshes are thus caught in a 'coastal squeeze' between rising seas and expanding development (Doody 2004). In the United Kingdom, for example, current ongoing losses of 100 ha yr^{-1} of coastal salt marsh have been estimated, due to erosion/reduced sediment inputs/land subsidence and coastal defence measures to counteract erosion (U.K. Biodiversity Group 1999, Hughes & Paramor 2004). Fortunately, since the 1990s a new approach has been developed to address coastal erosion in Europe, and increasingly the sea is being allowed to break through barriers and to re-create fringing habitats including salt marshes and lagoons (e.g., EC 2004, Hughes & Paramor 2004, Kabat et al. 2005).

Protection measures

The importance of preserving coastal wetlands is being increasingly recognised in Europe (Jones & Hughes 1993), where wetlands are now the target of numerous international and national initiatives and regulations for conservation, wise use and restoration. Government commitments have been encouraged though various initiatives, such as the Ramsar Convention and the Rio CBD (Table 1). Atlantic, Mediterranean and Baltic salt marshes are specifically listed as habitats in Annex I of the Habitats Directive, with some types of marshes identified as priority habitats for conservation, and wetlands in general fall into several broad habitats of the directive, such as 'Coastal lagoons', 'Estuaries' and 'Large shallow inlets and bays'. Many coastal wetlands are also recognised as SPAs according to the Birds Directive.

As a response to these initiatives, European states have developed their own national strategies and action plans. As an example, in the United Kingdom approximately 80% of present salt marshes (50% in northwest Scotland) are currently designated as Sites of Special Scientific Interest (SSSIs), and in Northern Ireland five out of seven estuaries containing salt marshes have been designated as Areas of Special Scientific Interest (ASSIs) (U.K. Biodiversity Group 1999). Ten areas in the United Kingdom have been proposed as SACs (Habitats Directive), and 27 major salt marshes and many smaller sites are included in SPAs (Birds Directive) and Ramsar sites: the target goal is to stop further net loss of coastal wetlands, and restore 40 ha of salt marsh yr^{-1}, to replace the 600 ha lost between 1992 and 1998 (U.K. Biodiversity Group 1999).

Seagrass meadows

Current distribution and status

Seagrasses are rhizomatous, clonal, marine plants that form some of the most valuable and productive coastal ecosystems in the biosphere (Costanza et al. 1997). They provide food and habitat for a variety of biota and play a fundamental role in carbon and nutrient cycling, control of water quality and sediment dynamics (Duarte 2002). Seagrasses can colonise a variety of coastal habitats from estuarine to marine, subtidal to intertidal, sedimentary to rocky. Global area estimates for seagrasses are beginning to be developed but at present there is no comprehensive dataset of actual seagrass distribution, and even regional datasets can be limited. The *World Atlas of Seagrasses* (Green & Short 2003) provides the most current and comprehensive compilation of information, documenting some 177,000 km^2 of seagrass and suggesting a tentative global acreage estimate of 500,000 km^2.

Several seagrass species occur along the European coastline, including the natives *Zostera marina*, *Z. noltii, Ruppia maritima, R. cirrhosa, Cymodocea nodosa* and *Posidonia oceanica* (endemic to the Mediterranean Sea) plus *Halophila stipulacea*, which was recently introduced in the Mediterranean Sea. Seagrasses around European coastlines are now increasingly well monitored and published accounts exist for many sites and regions (Table 3). National inventories of seagrass distribution are available (e.g., Davison & Hughes 1998 for the United Kingdom) or are being prepared (e.g., REBENT programme for France). There is not, however, an organised comprehensive inventory of the distribution and coverage of seagrasses in Europe, and records for some regions are very incomplete. Furthermore, although in some regions or countries most seagrass beds are known and located, their actual coverage has not been determined. The *World Atlas of Seagrasses* (Green & Short 2003) documents a coverage of seagrasses of 1850 km^2 in Scandinavia and the Baltic Sea, 338 km^2 in western Europe (United Kingdom, Wadden Sea, Portugal and Atlantic France and Spain), 4152 km^2 in western Mediterranean countries (Italy, France and Spain), and 950 km^2 in the northwest Black Sea (Figure 9). It has been conjectured that seagrasses could be much more abundant in the Mediterranean, covering from 25,000 to 45,000 km^2 (Pasqualini et al. 1998).

Present-day seagrasses along European coasts are often described as in a degraded state (Green & Short 2003), with low shoot densities, high mortality rates, and high fragmentation. This is the case, for example, of seagrasses along the Mediterranean coasts of Spain (e.g., Table 3; Duarte 2002, Marbà et al. 1996, Delgado et al. 1997), including some deeper ones along the southeastern coasts, where trawling damages up to 40% of the total *Posidonia oceanica* surface (Tudela 2004). Along the Ligurian coast of Italy, *P. oceanica* meadows (50 beds covering 48 km^2) have been severely degraded due to coastal modification and town developments (Bianchi & Peirano 1995). Some of these beds were severely damaged in the early 1990s by the wreck of the oil tanker *Haven*

Table 3 Estimates of actual cover and historical losses of seagrasses for European countries (and eventual additional regional information) and main attributed causes of loss

Country	Cover (km^2)	Loss	Cause of loss	Regional data/additional information	Additional references
Albania	n.a.	n.a			
Belgium	Not present				
Bosnia-Herzegovina	n.a.	n.a.			
Bulgaria	n.a.	n.a.			
Croatia	n.a.	n.a.	I	Considered as widespread; severe declines of *Posidonia oceanica* in Istria between 1938–1998. Local invasions by *Caulerpa taxifolia.*	Zavodnik & Jaklin 1990, Meinesz et al. 2001
Cyprus	n.a.	n.a.		Considered as widespread.	
Denmark	1380–1710	75–80% in 1900–1990s	WD, S[a], EU, MG	Total cover is the sum of 30 km^2 in the Wadden Sea and 1350–1680 km^2 on the eastern coasts; vertical depth distribution reduced by 50% between 1900 and 1996–1997 (from 5–6 to 2–3 m in estuaries and from 7–8 to 4–5 m in open waters); 261 km^2 lost in Limfjiorden between 1900 and 1994 and 559 km^2 lost in Oresund between 1900 and 1996–2000; local losses continue nowadays (e.g., in 1994 eelgrass temporarily lost at Funen Island due to summer anoxia).	Reise 1994, Rask et al. 1999
Estonia	n.a.	n.a.			
Finland	<10	No losses since 1968		Seagrasses presumably not affected by the wasting disease.	
France	n.a. 1150 km^2 of *P. oceanica*	n.a. 30–40% of *P. oceanica* in recent decades	WD, LC, P, I, F, T	Along the Atlantic coasts >70 sites mapped and coverage of many known (most beds are 1–5 ha, but at least 10 beds cover 10 to >100 ha). South of Arcachon there is the largest *Zostera nolti* bed in Europe (70 km^2 in 1984) and a large bed of *Z. marina* (4 km^2 in 1984). The Glenan Archipelago lost 6 km^2 of seagrasses during 1932–1992. In the Mediterranean many local documented losses of *P. oceanica* (e.g., close to Marseille, 5–6% per decade between 1900 and 1970, nowadays 90% is deteriorated; along the French Riviera, 30.57 km^2 lost since 1800; seagrasses virtually extinct from the region of Toulon). 3184 ha affected by the invasion of *C. taxifolia.*	Meinesz et al. 1991, 2001, Glemarèc et al. 1997, Duarte 2002, EEA 2002

(continued on next page)

Table 3 (continued) Estimates of actual cover and historical losses of seagrasses for European countries (and eventual additional regional information) and main attributed causes of loss

Country	Cover (km^2)	Loss	Cause of loss	Regional data/additional information	Additional references
Georgia	n.a.	n.a.			
Germany	170	n.a.	WD, EU, S, MG[b], LC	Most seagrasses occur in Wadden Sea; only present on eastern coasts. Baltic Sea: in Kiel Bight decreased from 6 to 2 m in depth between 1960 and end of 1980. No losses in Greisvald Lagoon between 1930 and 1988. North Sea: *Z. marina* extinct at Helgoland island as early as 1928.	Reise 1994, Bartsch & Tittley 2004
Greece	n.a.	n.a.		Considered as widespread.	
Iceland	n.a.	n.a.			
Ireland	n.a.	n.a.			
Italy	n.a.	30–40% of *P. oceanica* in recent decades	S, EU, P, I, T	Most beds are mapped, and coverage of many is known (estimated 2350 km^2 in Liguria, Lazio, Sardinia, Veneto and Friuli). Estimate of loss is derived from estimates from France. *P. oceanica* virtually extinct in the north Adriatic Sea, in some areas since earlier than 1850s. *Z. marina* disappeared from areas of the Venice lagoon. 9414 ha affected by the invasion of *C. taxifolia.*	Barmawidjaja et al. 1995, Piazzi et al. 2000a,b, Guidetti 2001, Meinesz et al. 2001, Milazzo et al. 2004
Latvia	n.a.	n.a.			
Lithuania	Extinct?	100%?	MG[c]	Considered abundant over thousands of hectares in the past.	
Malta	n.a.	n.a.			
Monaco	n.a.	n.a.	I	Severe invasion by *C. taxifolia.*	Meinesz et al. 2001
Montenegro	n.a.	n.a.			
Netherlands	200	n.a. Virtually extinct in Wadden Sea	WD, E[c], S, EU, F[d]	Most seagrasses are in southwest estuaries; in the Grevelingen estuary, the construction of two dams facilitated the growth of *Z marina* (from 12 km^2 in 1964 to 34 km^2 in 1985) and the disappearance of *Z. noltii*, which before was the most common seagrass (large-scale, unexplained die-off of *Z. marina* since 1986–1987). In the Wadden Sea 145 km^2 lost in 1919–1971 and 3 km^2 since) and only 1–2 km^2 are left nowadays.	Reise 1994, Short & Wyllie-Echeverria 1996, Wolff 2000, 2005
Norway	n.a.	No losses since 1930			
Poland	n.a.	n.a.	MG[c], EU	Virtually extinct in Puck Lagoon in 1957–1987, with subsequent recolonisation in some areas.	
Portugal	n.a.	n.a.			

Romania	n.a.	n.a.			
Russia	n.a.	n.a.			
Slovenia	n.a.	n.a.			
Spain	n.a. >1000 in Mediterranean	30–40% of *P. oceanica* in recent decades	S, F, I, A, T	Estimate of loss is derived from estimates from France. In the Mediterranean, 57% of *P. oceanica* beds were under regression in 1996 along 1000 km of coasts, and some are now extinct, with losses concentrated over 400 km of coasts. 58% of *P. oceanica* beds and 52% of seagrasses are degraded along the Catalan coasts and near Alicante, respectively. Local invasions by *C. taxifolia*.	Marbà et al 1996, Meinesz et al. 2001, Duarte 2002, EEA 2002
Sweden	n.a.	n.a.	EU, MG	At least 60–130 km^2 along the southern Baltic coasts. Losses of 58% (10.61 km^2) of seagrasses in the Skagerrak in 1980s–2000.	Baden et al. 2003
Turkey	n.a.	n.a.		Considered as widespread.	
Ukraine	n.a.	n.a.		950 km^2 in the northwestern bays.	
United Kingdom	n.a.	n.a.	WD	Most beds mapped, and coverage of many known. About 140 sites of *Z. marina* and about 70 sites of *Z. noltii*, covering from 12 km^2 (Cromarty Firth) to 20–40 ha[e]. The Maplin Sands hosts one of the largest surviving population of *Z. noltii* in Europe (325 ha). Before 1900s, seagrasses were common; they severely declined in 1920s–1930s and have not recovered yet, particularly in southern and eastern England.	Davison & Hughes 1998, U.K. Biodiversity Group 1999

Note: If not otherwise specified in the reference column, information is derived from the *World Atlas of Seagrasses* (Green & Short 2003) and references therein. A = aquaculture, E = engineering works and embankments, EU = eutrophication, F = fisheries, I = invasive species, MG = growth of ephemeral macroalgae (often a consequence of eutrophication), LC = land claim/waterfront development, n.a. = no comprehensive estimate available, P = urban and/or industrial pollution, S = increased water turbidity/load of sediment, beach replenishments, T = tourism, WD = wasting disease. Note that the estimates of covers should be regarded as minimal representation of the actual coverage in most cases. Also note that most often time span is not indicated.

[a] Decrease of maximum Secchi depth from 12 m in 1900 to 6 m in the 1990s.

[b] Seagrasses replaced by filamentous algae. This may not necessarily be the cause of seagrass loss.

[c] Including the closure of the Zuidersee in 1932.

[d] Eelgrass harvesting until 1930, shell fisheries after 1970.

[e] Other important sites are the Exe Estuary, the Solents marshes and the Isles of Scilly, Morfa Nefyn, Milford Haven, the Moray Firth, Carlingford Lough, Dundrum Bay, Strangford Lough and Lough Foyle.

Figure 9 Map of the distribution of seagrasses in Europe. (Data courtesy of UNEP World Conservation Monitoring Centre.) The map should be considered indicative of general distribution not areal extent.

(Sandulli et al. 1994). The extent of seagrass degradation is, however, difficult to quantify even locally.

Historical losses and causes

There is increasing awareness about the severe degradation of seagrass meadows (e.g., see, among others, the reviews by Short & Wyllie-Echeverria 1996, Hemminga & Duarte 2000, Duarte 2002). Reports consistently identify a long-term trend of worldwide seagrass decline, about 70% of which can be probably attributed to direct human-induced disturbance (Short & Wyllie-Echeverria 1996). Less information is available concerning the degradation caused by indirect impacts (Duarte 2002). It has been estimated that a global loss of 12,000 km^2 occurred during the 1990s alone (Short & Wyllie-Echeverria 1996), which represents about 7% of the known distribution of seagrasses (Green & Short 2003). Data covering longer time spans are rare. Based on data from 12 temperate estuaries around the world, it has been estimated that over time these systems may have lost approximately 65% of their seagrasses (Lotze et al. 2006). No comprehensive organised historical information seems to be available for Europe and information is limited to restricted areas (Table 3).

There are different trends for seagrass losses in northern and Mediterranean European countries (Green & Short 2003). In northern Europe, before the early 1900s, several seagrass species, including *Zostera marina*, were common. They were harvested for a variety of uses, including use for fuel, packing, upholstery, insulation, roof material, filling of mattresses and cushions, feeding and bedding for domestic livestock, fertiliser and as resource to obtain salt. Their abundance was, however, severely reduced in the 1930s by a ‘wasting disease’, caused by the pathogenic slime mould *Labyrinthula zosterae* (e.g., Den Hartog 1987). The disease led to the catastrophic die-back of eelgrass (*Zostera marina*) meadows along the north Atlantic coasts, with loss of almost 90% of the *Zostera* populations in north Atlantic western Europe. Some beds progressively recovered, but substantial areas remain lost from most beds.

There is uncertainty about the causes of this disease outbreak and there has been much debate about whether concomitant human impacts that already weakened the plants contributed to the outbreak (Den Hartog 1987). The decline particularly affected sublitoral beds, while intertidal populations were less affected. Probably the best records of the disruptive effects of the wasting disease are from Denmark, where records of eelgrass distribution date back to 1900 (Boström et al. 2003). In 1900 eelgrass covered about 6726 km^2 (Figure 10); by 1940, 93% of the distribution of vegetated areas was lost. Since 1960, there has been a slow recolonisation. Currently, there is approximately 20–25% of the distribution recorded in 1900 (i.e., a total loss of 75–80%). The greatest loss has been in deep *Zostera* populations and in Denmark the vertical depth distribution of *Z. marina* was reduced by about 50% during the twentieth century, from a historical depth limit of 5.6–11 m to a recent limit of 2.5–8 m in sheltered and exposed areas, respectively (Hemminga & Duarte 2000, Baden et al. 2003).

Similar dramatic losses are described for the United Kingdom (Davison & Hughes 1998 and references therein) and the Wadden Sea (Reise 1994, Wolff 2000, Lotze 2005). The distribution of seagrasses in the United Kingdom was only systematically described in the 1930s after the outbreak of wasting disease, when *Z. marina* was already scarce and restricted to few sheltered lagoons but there are indications that seagrasses were widespread until 1917 (Davison & Hughes 1998). There is some uncertainty about when recovery started. According to some, recovery began in 1933 and was quite rapid, with some beds fully recovering within a few years of the 1930s epidemic, while according to others the disease continued to affect *Zostera* populations until the mid-1940s and recovery did not really begin until the 1950s. Nowadays, most *Zostera* beds have not fully recovered, particularly in southern and eastern England where the species was once abundant, and only 20 of Britain's 155 estuaries have eelgrass meadows >1 ha in extent (Davison & Hughes 1998).

Before the 1930s the Wadden Sea also contained large subtidal, seagrass beds. These have been exploited since historical times for construction and insulating material and to fill mattresses and cushions. In the Dutch Wadden Sea, from the thirteenth century to 1825, eelgrass was used to build dykes. The construction of 1 m of seawall required about 8–20 m^3 of compacted eelgrass, equivalent to about 40–100 m^3 of fresh eelgrass, which in some years corresponded to about 1–10% of the annual production (Wolff 2005). Decline of seagrasses appears to have occurred over two phases (Reise 1994): one acute in the 1930s, caused by the wasting disease, after which most subtidal eelgrass beds did not recover, and one more gradual that began in the 1960s, mostly driven by eutrophication. These declines first affected subtidal eelgrass beds, and subsequently also intertidal ones, leading to the almost extinction of seagrasses in some regions of the Wadden Sea (e.g., the Dutch Wadden Sea, where cover dropped from 150 to 1–2 km^2; see Table 3), and to the disappearance of numerous species associated with seagrasses (Wolff 2000).

Along Mediterranean coasts, reliable estimates made by direct observation of the area of seagrass lost or degraded are limited (Green & Short 2003). It is estimated that in the past *Posidonia oceanica* meadows may have covered 50,000 km^2 in the whole basin (Duarte 2002), which considering present estimated covers of seagrasses in the Mediterranean and Euro-Asian Seas (Green & Short 2003) would make an overall loss >85% (but probably many existing seagrass meadows are not presently documented). Rapid local regression (up to complete disappearance) of *P. oceanica* meadows is known to have occurred at numerous localities in France, Italy and Spain (Table 3). It is estimated that shoot density of *P. oceanica* in the western Mediterranean has decreased by up to 50% over a few decades, with major losses between 10 and 20 m depth (EUCC 1998). For the French mainland coast, overall habitat loss is estimated as about 10–15%, which would increase up to 30–40% if the decline in shoot density is also taken into account. Overall, these figures are

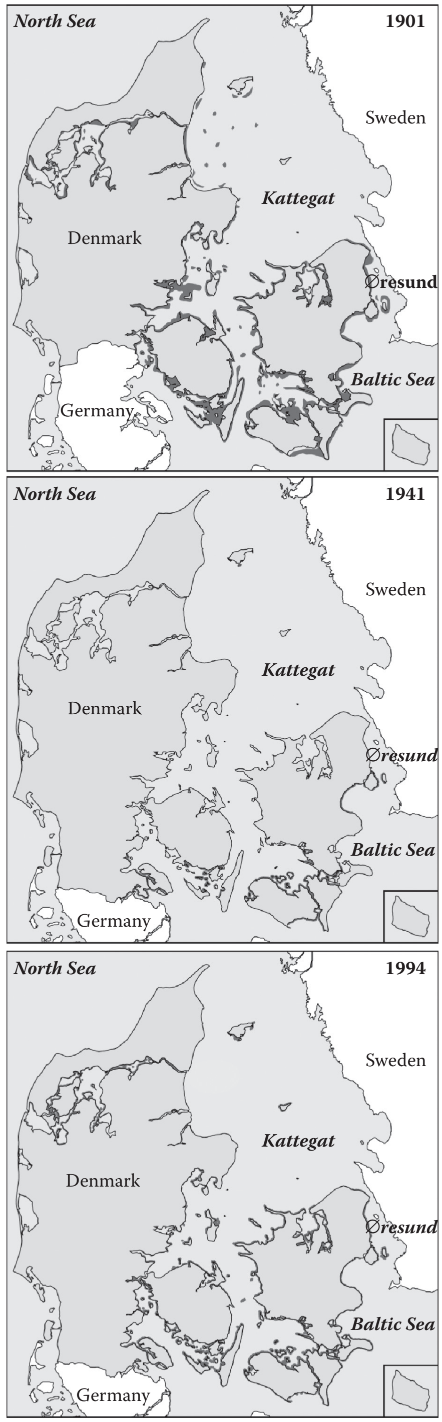

Figure 10 Area distribution of eelgrass *Zostera marina* (in dark grey) along Danish coasts in 1901, 1941 and 1994. (Modified after Boström et al. 2003. With permission.) In the 1930s, eelgrass populations were severely affected by a wasting disease, and in 1941 they covered only 7% of the areas occupied in 1901. Recolonisation took place after the 1960s, but in 1994 cover was still only 20–25% of that in 1901.

considered a good estimate for most western Mediterranean coastlines, with notable exceptions around the islands and in the eastern Mediterranean (EUCC 1998).

There have been clear losses of seagrasses on the Italian coast of the north Adriatic Sea. Geological data have shown that seagrass beds were probably common before the 1800s and experienced dramatic regressions to virtual extinction in the last two centuries (Barmawidjaja et al. 1995, Caressa et al. 1995, Rismondo et al. 1997). For example, faunal changes and the sudden disappearance of epiphytic foraminiferans in a sediment core in front of the Po river delta suggest that seagrass beds were present up to 1840 in this area, and that increased load of fine sediment and nutrients between 1840 and 1870, due to substantial changes to the main outflow canals of the Po river, was probably the main cause of their sudden disappearance (Barmawidjaja et al. 1995). Similarly, dead beds of *P. oceanica* have been found at several sites about 8 miles offshore from the Venice lagoon, indicating the likely past presence of *P. oceanica* (Rismondo et al. 1997). In the Gulf of Trieste the regression has been more recent and there has been some direct documentation. *P. oceanica* was reported as common in the Gulf of Trieste at the beginning of the 1900s (Caressa et al. 1995). In 1938 the species had declined but was still present all over the coastlines of the Istrian peninsula, while just about 30 yr later only the Koper meadow was detected. Nowadays, *P. oceanica* is present only in a fragmented meadow along the coastline of Koper (Slovenia, at the southern side of the Gulf of Trieste) and in a very small area along the coasts of Grado (on the Italian side of the Gulf of Trieste). This drastic reduction has been attributed to a steep increase in water pollution during the last 50 yr as a consequence of industrial and harbour development in the Gulf of Trieste (Caressa et al. 1995).

Another case of rapid regression of *P. oceanica* meadows is in the French Riviera, from Menton to the Rhône delta (Meinesz et al. 1991). Intensive waterfront development started around 1800 covering natural coastal habitats with recreational harbours, artificial beaches, landfills (for the tourist industry) and large commercial and military complexes, ports and airports. A total of 185 reclamation projects 'occupied' 106 km (16.2%) of the coastline, directly removing 30.57 km^2 of bottom substrata and greatly affecting surrounding areas (by modifying water movements and sedimentation patterns). Overall a total of 9.7% of the shallow water zone between 0 and −20 m and 14.5% of the zone between 0 and −10 m were irreversibly destroyed. The vast majority of this area was occupied by *P. oceanica* meadows, which were estimated to originally cover a total of 200 km^2.

Trends and threats

Many anthropogenic factors are considered responsible for the ongoing degradation and decline of seagrasses in Europe as well as globally (e.g., see, among others, the reviews by Short & Wyllie-Echeverria 1996, Davison & Hughes 1998, Hemminga & Duarte 2000, Duarte 2002, Green & Short 2003). The most important of these threats is likely to be poor water quality from pollution, eutrophication and excess sedimentation. These impacts are associated with, and enhanced by, urban and tourist waterfront developments, port constructions, beach replenishments and other interventions for shoreline stabilisation.

Severe seagrass loss is still in progress in Europe, as evident along 200 km of the Skagerrak Swedish coast (Baden et al. 2003). Here, 50 of 69 mapped meadows of *Zostera marina* have shown average declines of 58% between the 1980s and 2000, corresponding to a lost surface of about 1061 ha. Most declines are related to a reduction of the upper and the lower depth distribution of seagrasses, resulting in narrower meadows, but in some areas seagrass meadows have disappeared completely, with dramatic effects on the fish assemblages (Pihl et al. 2006). The reasons for this continuing loss of seagrasses are not known but could be related to an excess growth of phytoplankton and filamentous or other ephemeral macroalgae as a consequence of eutrophication. These micro- and macroalgae outcompete seagrasses (Hauxwell et al. 2001) and decompose on the bottom, favouring anoxia events like those that caused the recent temporary disappearance of *Z. marina* in 1994 around the island of Funen in Denmark (Rask et al. 1999).

There is also current seagrass loss in the Mediterranean Sea as a consequence of the invasion of *Caulerpa taxifolia* (Meinesz et al. 2001, but see Jaubert et al. 2003). This species competes for space and resources with the seagrass *Cymodocea nodosa* (Ceccherelli & Cinelli 1997) and is thought to be able to damage *Posidonia oceanica* beds, particularly when these are already under stress (e.g., de Villèle & Verlaque 1995).

Protection measures

Green & Short (2003) report that worldwide there are only some 247 MPAs that are known to include seagrasses, spread in 72 countries and territories. This estimate may be conservative but these authors note that this is far smaller than the number of MPAs with coral reefs (more than 660) or mangrove forests (over 1800). Even for these 247 MPAs, there are questions about their effectiveness in protecting seagrass ecosystems particularly from threats such as poor water quality (e.g., Marbà et al. 2002, Milazzo et al. 2004).

In Europe, numerous initiatives arising from the Rio CBD, the Habitats Directive and the Birds Directive (Table 1) have led to seagrass meadows being specifically targeted for conservation and restoration. Seagrasses are a named component of several habitats in the Habitats Directive, including 'Coastal lagoons' (a priority habitat), 'Sandbanks slightly covered by seawater all of the time', 'Large shallow inlets and bays', 'Estuaries' and 'Mudflats and sandflats not covered by seawater at low tide' (EC 2003). Furthermore, *Posidonia oceanica* beds are a priority habitat in Annex I of the Habitats Directive. International concern about the conservation of seagrass beds has also led to the banning of trawling on seagrasses in EC waters (Tudela 2004).

As a response to these initiatives, European States have also developed national strategies and initiatives. The U.K. Biodiversity Action Plan, for example, includes a Habitat Action Plan for seagrass beds (U.K. Biodiversity Group 1999). Accordingly, areas of seagrass are included in some coastal ASSIs/SSSIs, Ramsar sites, SPAs and voluntary MPAs. Two of the three U.K. Marine Nature Reserves have seagrass beds and the habitat occurs in a number of areas proposed as SACs.

Macroalgal beds

Current distribution and status

For the purpose of this review 'macroalgal beds' refer to kelps, fucoids and other complex, erect brown and red macroalgae that produce relatively large biogenic habitats. Macroalgal beds form diverse, productive and valuable temperate coastal ecosystems (Steneck et al. 2002). They are widespread on shallow hard substrata around Europe (Birkett et al. 1998b, Steneck et al. 2002, Thibaut et al. 2005 and references therein), including rock, boulders, cobbles and human-made structures from the intertidal down to more than 30 m in depth. *Laminaria* and *Fucus* are the main genera along the coasts of northwest Europe, while *Cystoseira* and *Sargassum* are the main genera in the Mediterranean Sea.

The macroalgal flora of the European coasts are among the best known and studied. Furthermore, large macroalgae have been the target of ecological studies for over a century and there is extensive literature on physical and biological factors operating in these habitats (see, among many, Lüning 1990, Ballesteros 1992). Surprisingly, however, quantitative information on the distribution of macroalgal beds is limited and their extent is unknown. At present, there do not appear to be comprehensive inventories for the macroalgae of any European country and it is difficult to estimate the areal coverage even at regional or local scales. Distribution maps of rocky coast biotopes, including kelps, are available for the United Kingdom (Connor et al. 2004), and ongoing inventories by the Joint Nature Conservation Committee (JNCC) are likely to result in estimates of the cover of macroalgal beds for this country, at least for intertidal habitats, while little information is available for the subtidal. Intensive research on kelp distribution, ecology and effects of human harvesting has also been carried out along Norwegian coasts, where it is estimated that the dominant kelp *Laminaria hyperborea* might cover between 5,000 and 10,000 km^2 (Jensen 1998). Limited quantitative information on the distribution of macroalgal beds is available at a few regional or local scales (e.g., the Albères coast, Thibaut et al. 2005; Kiel Bay, Vogt & Schramm 1991; the Öregrund archipelago, Eriksson et al. 1998; the Gullmar Fjord, Johansson et al. 1998).

Historical losses and causes

A perceived worldwide decline of macroalgal beds has been reported during the last decades (Steneck et al. 2002, Airoldi 2003). Such decrease most often appears to be parallelled by a trend of increasing abundance of turf-forming, filamentous or other ephemeral algae (e.g., Munda 1993, Eriksson et al. 1998, Benedetti-Cecchi et al. 2001, Lotze 2005) that, once established, often inhibit recolonisation of canopy-forming algae and other organisms (Airoldi 1998). Canopy-forming algae and turfs have been suggested to represent alternative states in shallow temperate rocky reefs under different disturbance and stress regimes (Worm et al. 1999, Airoldi 2003, Connell 2005). Macroalgal beds have also been replaced by coralline dominated 'urchin barrens', where outbreaks of urchins may have been the primary cause of macraolgal extirpation (Hagen 1995, Steneck et al. 2002, Guidetti et al. 2003), or by mussel beds (Thibaut et al. 2005).

In Europe, almost no information on trends in abundance of macroalgae is available before the 1900s. In the twentieth century, conspicuous losses, sometimes to virtual local disappearance, of complex macroalgae have been documented for coastal areas in several countries, including Iceland, Norway, Britain and Ireland (Hagen 1995, Steneck et al. 2002 and references therein); Sweden (Lundälv et al. 1986, Eriksson et al. 1998, 2002, Johansson et al. 1998, Nilsson et al. 2004); Denmark (Middelboe & Sand-Jensen 2000); Finland and Germany (Kangas et al. 1982, Messner & von Oertzen 1991, Vogt & Schramm 1991, Schories et al. 1997); Lithuania (Olenin & Klovaitė 1998); Italy (Sfriso 1987, Cormaci & Furnari 1999, Benedetti-Cecchi et al. 2001, Guidetti et al. 2003, L. Airoldi unpublished data); France and Spain (Rodríguez-Prieto & Polo 1996, Thibaut et al. 2005 and references therein); Croatia (Munda 1993, 2000) and Romania (Zaitsev 2006). The causes of these losses are various but mainly include outbreaks of sea urchins and decreased water quality as a consequence of pollution and/or enhanced sediment loads. These trends have been generally traced through comparisons with historic floristic records, which are difficult to translate into an estimate of the extent of habitat loss. Furthermore, complex native species were often replaced by simpler macroalgae or non-native species, possibly affecting the status and functioning of the systems but not their extent.

Frequently, the declines of macroalgae have been greater at depth so that their depth range has become shallower (Lumb 1990, Messner & von Oertzen 1991, Bokn et al. 1992, Munda 1993, Eriksson et al. 1998, 2002, Pedersén & Snoeijs 2001, Thibaut et al. 2005). In Kiel Bay (Germany, western Baltic), for example, there has been a >90% decline in the biomass of *Fucus* spp. between 1950 and 1988, from about 40,000–45,000 t wet wt down to only 2400 t wet wt (Vogt & Schramm 1991). In the 1950s, *Fucus* spp. were the dominant macrophytes down to 6 m in depth and were still frequent in the 1970s, whereas at the end of the 1980s, the species were not found at depths >2 m. In the Öregrund archipelago (Sweden) the average depth penetration of *F. vesiculosus* decreased significantly by 2 m between 1943 and 1996 and this species had completely disappeared at depths >8 m (Figure 11). The amount of macroalgal habitat lost as a consequence of this depth reduction has not been estimated (B.K. Eriksson personal communication) but it must have been

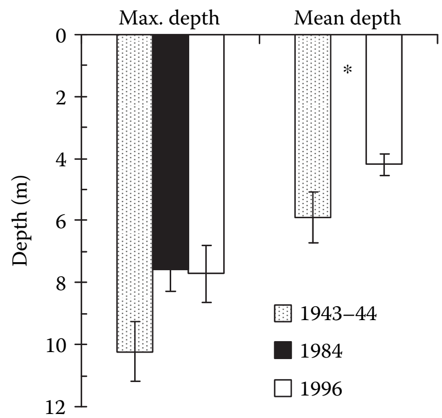

Figure 11 Changes in the depth (maximum and mean) distribution of *Fucus vesiculosus* in 1943–1944, 1984 and 1996 in the Öregrund archipelago. (From Eriksson 2002, based on Kautsky et al. 1986, Eriksson et al. 1998 and Eriksson & Bergstrom 2005; courtesy B.K. Eriksson.) *No mean depth data were available for 1984. Data are averages ± 1 standard error. Maximum and mean depth penetration of *F. vesiculosus* decreased significantly both between 1943–1944 and 1984 and between 1943–1944 and 1996 (t-test, $n = 5$ sites, $p < .05$).

considerable because quantitative measures of the distribution of *F. vesiculosus* indicated that this species covered on average 20–50% of the substrata. Similarly, at Stora Bornö Island, on the Swedish Skaggerak coast, the depth distribution of macroalgae declined on average by 2.8 m between 1941 and 1998 (Eriksson et al. 2002). This loss was particularly severe for large, complex macroalgae (>50 cm), which showed up to 8-m reductions in their depth distribution (Figure 12). These complex macroalgae were replaced by simpler, thin filamentous and sheet-like forms.

Losses to virtual extinction of macroalgal species have been reported from other regions in the Wadden Sea, southern France, the Venice lagoon and the Black Sea, for example. In the Wadden Sea, at least 10 species of macroalgae have become extinct during the past 2000 yr, probably because of the transformation of brackish waters into fresh waters and the destruction of native eelgrass beds and oyster reefs that provided solid surfaces for attachment (Wolff 2000). Along the

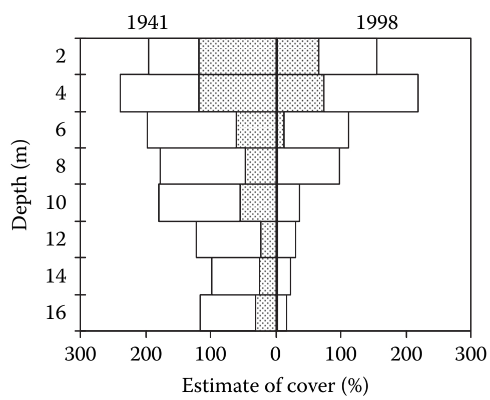

Figure 12 Depth distributions of macroalgae with different thallus shape (thin filamentous and sheet-like = white bars; coarse filamentous and thick, leathery algae = dotted bars) in 1941 and 1998 at Stora Bornö Island. (Based on data from Eriksson et al. 2002. Courtesy B.K. Eriksson.) Bars are mean percentage cover in each depth interval pooled for two vertical profiles (error lines are not shown for clarity).

Albéres coasts (southern France) dramatic reductions in abundance to extinction of populations of Fucales (*Cystoseira* spp. and *Sargassum* spp.) occurred between the early 1900s and 2003 (Thibaut et al. 2005). Only 9 of 14 species of Fucales documented in 1912 were present in 1978, with the genus *Sargassum* entirely lost, and only 5 species were found in 2003. Seven of the extinct species were considered frequent to abundant in 1937. Of the 5 species remaining in 2003, only 1 did not show signs of regression. In the Venice lagoon, some 141 algal species were documented in 1938, 116 in 1962, 107 in 1987 and 96 in 1991 (Sfriso 1987, Sfriso & La Rocca 2005). Some of the most damaged species in the lagoon were previously dominant large brown algae, including species of *Cystoseira, Fucus virsoides*, and *Sargassum hornschuchii*. These algae were adversely affected when channel excavations limited water exchange, leading to an increase of nutrient levels and eutrophication within the lagoon, and in the 1980s their abundance dramatically decreased, sometimes to complete and permanent disappearance. These complex macroalgae were replaced by ephemeral species, mainly green algae, and invasive species. In the last 10 yr, water quality has improved in the Venice lagoon, and recent sampling detected an increase in the number of macroalgae (Sfriso & La Rocca 2005), in part also linked to the introduction of several invasive species, including the now-abundant large brown algae *Undaria pinnatifida* and *Sargassum muticum*. Along the coasts of the Black Sea, several complex macroalgae (including brown algae of the genera *Cystoseira* and *Phyllophora*) have virtually disappeared during the past 30 yr (Zaitsev 2006). Estimates at some Romanian localities indicate that in 1971 these species attained considerable biomasses (about >10,000 t fresh wt).

Trends and threats

Several factors are thought to be responsible for the continuing decline of kelps and canopy-forming macroalgae and to pose serious threats to the future of rocky reefs in general (Steneck et al. 2002, Thompson et al. 2002). Urbanisation is thought to have the most disrupting effects on kelps and other canopy-forming algae, particularly by affecting water clarity and quality as well as other habitat-related changes (e.g., Vogt & Schramm 1991, Munda 1993, Eriksson et al. 1998, 2002, Benedetti-Cecchi et al. 2001).

In some northern European regions, including the west coast of Norway, the French channel coast and parts of the U.K. coast, harvesting is also an issue (e.g., Christie et al. 1998). Kelps washed up on the shore have been traditionally collected for centuries in some regions, for use as an agricultural fertiliser and to improve the soil structure. Nowadays kelps and fucoids are harvested from living beds to be used as basic resource in the alginate industry to produce emulsifying and gelling agents. The most commonly harvested species include *Laminaria hyperborea, L. digitata, Ascophyllum nodosum* and *Fucus* spp. with 70,000–80,000 t of seaweeds collected each year around the coasts of both Brittany (Birkett et al. 1998b) and Norway (EEA 2002). Modern methods of kelp harvesting (e.g., by trawling) seem to have a significant direct influence on kelp biotopes (Birkett et al. 1998b, Christie et al. 1998).

Reef habitats and associated macroalgal beds are also severely damaged by disruptive fishing techniques. For example, the collection of the date mussel *Lithophaga lithophaga* by use of hammers and chisels, pneumatic hammers and explosives is still a widespread practice in most Mediterranean countries, despite its legal ban. This practice directly and irreversibly destroys the rocky environment, causing the loss of canopy-forming seaweeds and the formation of barrens (Fanelli et al. 1994, Guidetti et al. 2003).

The introduction of harbour piers, jetties, dykes, seawalls, coastal defences and other armoured artificial structures has in some regions led to an expansion in the distribution of native and non-native macroalgae and other rocky-bottom species (Moschella et al. 2005). In the Wadden Sea, for example, where hard substrata were naturally scarce, about 730 km of artificial structures have introduced about 2–4 km^2 of hard substrata, providing new habitats for a variety of rocky-bottom species, including *Laminaria saccharina* (Reise 2005). Along the north Adriatic shores, which are naturally devoid of rocky substrata, >190 km of rock-armoured structures (Figure 5), built mainly in the past 40 yr (Bondesan et al. 1995), have introduced about 1 km^2 of artificial hard substrata within natural sandy depositional environments, which are now extensively colonised by the non-indigenous, invasive canopy-forming macroalga *Codium fragile* ssp. *tomentosoides* (Bulleri & Airoldi 2005). The extent of hard coastal structures is expected to increase in the future, with profound but overlooked ecological consequences on native coastal environments (Airoldi et al. 2005).

Protection measures

Although kelp beds and other macroalgal habitats are not specifically targeted in the Habitats Directive, species of the genus *Fucus*, *Laminaria* and *Cystoseira* and other macroalgae are named components of 'Reefs' habitat (EC 2003). Other European initiatives also include the protection of some species of complex macroalgae. For example, six Mediterranean species of the genus *Cystoseira* and two species of *Laminaria* are listed in Annex I of the Bern Convention. The Action Plan for the Conservation of Marine Vegetation in the Mediterranean Sea, adopted within the framework of the Barcelona Convention, identifies the conservation of *Cystoseira* belts as a priority. Several complex brown macroalgae are listed in the *Red Books* of Mediterranean and Black Seas as endangered (e.g., Boudouresque et al. 1990, Zaitsev 2006). Furthermore, *Lithophaga lithophaga* is included in Annex IV of the Habitats Directive and its collection is banned in most Mediterranean countries to protect rocky reefs and associated macroalgal beds from the destructive consequences of the fishery for this rather abundant date mussel (Russo & Cicogna 1991).

There are also national initiatives. For example, the commercial harvesting of kelps is strictly regulated in Norway and in Brittany (Birkett et al. 1998b), including a system of rotation of harvested areas introduced by the Norwegian government to ensure that each area of kelp forest is harvested only once every 4 yr.

Biogenic reefs: oyster reefs

Current distribution and status

The native European flat oyster (*Ostrea edulis*) is a sessile, filter-feeding, bivalve mollusc that used to be very abundant throughout its range (Korringa 1952). It is associated with highly productive estuarine and shallow coastal water habitats with sediments ranging from mud to gravel. The natural distribution of *O. edulis* is along the European Atlantic coasts from Norway to Morocco and across the coasts of the Mediterranean and Black Seas. Their abundance declined significantly during the nineteenth and twentieth centuries and wild native beds were considered scarce in Europe as early as the 1950s (Korringa 1952, Yonge 1966, Mackenzie et al. 1997).

Remains of wild native oyster beds still occur in various regions, including the rivers and flats bordering the Thames Estuary, the Solent, River Fal, the west coasts of Scotland and Ireland (Kennedy & Roberts 1999, U.K. Biodiversity Group 1999, Tyler-Walters 2001, Jackson 2003), the western part of the Swedish Kattegat region of the Baltic (Lozan 1996), the Limfjord region of Denmark (Korringa 1952), the Adriatic Sea, where *O. edulis* is still captured in the wild (Barnabe & Doumenge 2001), the Mar Menor (Spain), where a large flat oyster population, estimated at over 100 million individuals, still produces large amounts of spat (Cano & Rocamora 1996), and areas of the Black Sea, where the species was still valuable commercially until the 1970s (Zaitsev 2006). Limited information, however, is available about the current status of these oyster reefs and there is debate about whether the fragmented patches of wild oyster habitats are self-sustaining or owe their survival to the inputs of larvae from cultivated oysters (Korringa 1952).

Nowadays, aquaculture provides the main supply of native oysters in most European countries (Mackenzie et al. 1997, Ocean Studies Board 2004). This industry has also been seriously affected by epidemic diseases in recent decades, with documented losses of commercial stocks above 80% in France (Kennedy & Roberts 1999, Ocean Studies Board 2004), and most Mediterranean native oyster beds are in such poor conditions that they are unable to support intensive culture (Barnabe & Doumenge 2001). Although marketplace demand for native oysters remains strong, the introduced Pacific oyster *Crassostrea gigas*, which is easier to cultivate than the native oyster, now provides the major share of oyster production in Europe (Cano & Rocamora 1996, Kennedy & Roberts 1999, Lotze 2005; see Figure 13C).

Historical losses and causes

There is some documentation on the declines and loss of native oyster reefs, mostly from fishery landing records; direct quantitative data are uncommon. In Europe oysters have been an extremely popular food for centuries (Jackson 2003). Both ancient Greeks and Romans highly valued oysters. Romans fished and imported them from all over European and Mediterranean coastlines and extensively cultivated them (Günther 1897), and in some British estuaries there are archaeological signs of overexploitation of native oyster beds since the first century (Rippon 2000). For centuries, *Ostrea edulis* reefs supported a productive commercial fishery (Mackenzie et al. 1997). In the eighteenth and nineteenth centuries, large offshore oyster grounds in the southern North Sea and the English Channel produced up to 100 times more than today's 100–200 t (U.K. Biodiversity Group 1999, Berghahn & Ruth 2005). The richest natural oyster beds in Europe until the nineteenth century were probably around Britain, from Stornoway to the Solway in the west and from the Orkney Islands to the Firth of Forth in the east (Berghahn & Ruth 2005). In the mid-nineteenth century these were heavily exploited; dredging of the oyster beds was one of the largest fisheries, employing about 120,000 men around the coast in the 1880s (Tyler-Walters 2001), with an annual yield of >50 million oysters (Berghahn & Ruth 2005). Oyster reefs at Strangford Lough, in Ireland, once supported up to 20 boats employed in oyster dredging (Kennedy & Roberts 1999). In the Wadden Sea, the commercial fishery for oysters started in the eleventh century and flourished in the eighteenth century (Figure 13): in 1765 large oyster beds between Texel and Wieringen supported profitable fishery by 145 vessels, with catches over 100,000 oysters yr^{-1} $vessel^{-1}$ (Wolff 2005).

By the late nineteenth century, beds of *O. edulis* were already severely depleted or physically destroyed around most European coasts (Ocean Studies Board 2004). Regulations and fishery closures were imposed in some regions. In the Wadden Sea, for example, management strategies including minimum landing size, fishery closures, a licence system and maximum yield per year have been applied since the seventeenth century (Berghahn & Ruth 2005). Similar initiatives were taken in France. The decline could not, however, be halted and in the twentieth century catches collapsed (e.g., Figure 13A,B). Overfishing and wasteful exploitation, combined with outbreaks of diseases, habitat loss and change or destruction, reduction in water quality and other large-scale environmental alterations, adverse weather conditions, and the introduction of non-native oysters (and associated parasites and diseases, such as the protozoan *Bonamia ostreae*) for aquaculture and other non-native species (e.g., the invasive gastropod *Crepidula fornicata*) were blamed for the decline (Korringa 1952, Mackenzie et al. 1997, Wolff 2000, Jackson 2003, Berghahn & Ruth 2005). Virtual extinction of native oyster beds has been documented in the Wadden Sea, where wild oysters largely disappeared by 1950 (Wolff 2000, 2005, Lotze 2005); in Helgoland (Germany), where beds largely disappeared by the mid-1900s (Korringa 1952, Franke & Gutow 2004, OSPAR Commission 2005); in the Dutch Easter Scheldt (van den Berg et al. 2005); in Belgium (OSPAR Commission

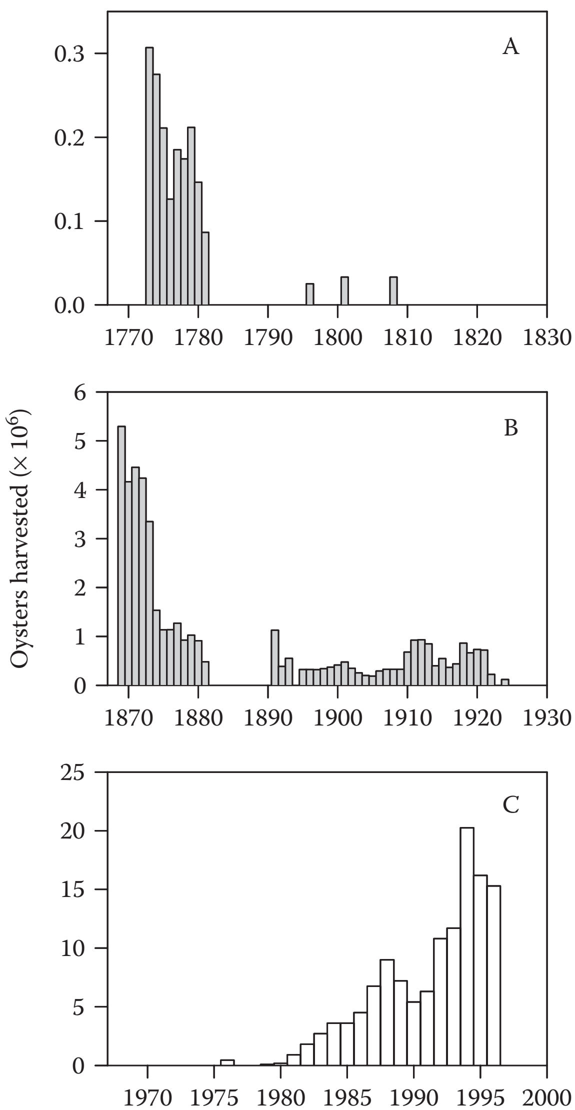

Figure 13 Annual landings of native European oysters, *Ostrea edulis*, (A) in the east Frisian Wadden Sea during 1770–1830, (B) in the north Frisian Wadden Sea during 1868–1930 and (C) landings of cultured Pacific oysters, *Crassostrea gigas*, in the Netherlands during 1970–2000. (From Lotze 2005. With permission.)

2005); in all deeper waters of the southern North Sea, such as in the Oyster Grounds (OSPAR Commission 2005); in most areas of Galicia (Cano & Rocamora 1996) and in some bays in the Black Sea (Zaitsev 2006). In the Firth of Forth (Scotland), which in past centuries had hosted one of the most famous oyster banks, no oysters were found in 1957 (Dodd 2005). Dramatic stock decreases have been reported as well on the Atlantic coasts in French Brittany, the Netherlands, Denmark, Norway, Ireland and England and in the Mediterranean Sea (Korringa 1952, Mackenzie et al. 1997, U.K. Biodiversity Group 1999, Barnabe & Doumenge 2001). In the United Kingdom, where 700 million oysters were consumed in London alone in 1864, the catch fell from 40 million in 1920 to 3 million in the 1960s and has not recovered (Tyler-Walters 2001). In Archachon Basin (France), wild oysters, which had been exploited for ages, were nearly commercially exhausted in the mid-1800s, leading to the introduction first of the Portuguese hollow oyster (*Crassostrea angulata*) and then of the Pacific oyster (*Crassostrea gigas*).

Trends and threats

There is limited information about current trends and threats to remaining native oyster reefs in Europe. *Ostrea edulis* is a relatively long-lived species and reproduces sporadically (Korringa 1952). Thus, presumably, times of recovery from overexploitation or other causes of damage are very long and are estimated to take up to 20 yr (Jackson 2003). *O. edulis* is considered to be highly sensitive to substratum loss, smothering, contamination by synthetic compounds (particularly tributyltin (TBT) antifouling paints used on ships and leisure craft, which, in the early 1980s, caused stunted growth of oysters and probably affected reproductive capacity), oxygen depletion, reduced freshwater inputs, introduction of microbial pathogens/parasites, introduction of non-native species and direct extraction (U.K. Biodiversity Group 1999, Jackson 2003, Hiscock et al. 2005). All these factors impair recovery as well as restoration efforts. The main factors that probably threaten native oyster reefs nowadays include illegal fishing as well as by-catch in trawling targeting other species, poor water quality and pollution, changes to the environment (e.g., habitat loss due to coastal development) and the introduction of non-native competitors, predators and diseases (Jackson 2003, OSPAR Commission 2005).

Protection measures

Nowadays, the sparse remains of wild native oyster beds are probably one of the most endangered marine habitats in Europe. *Ostrea edulis*, however, does not seem to be the target of any specific protection measure, conservation legislation or convention at a European level. ‘Reefs’, including biogenic reefs, are listed as a conservation feature in Annex I of the Habitats Directive; however, native oyster reefs are not mentioned as a component (EC 2003). Since 2003, *O. edulis* beds are included in the OSPAR list of threatened and/or declining species and habitats for all OSPAR areas (OSPAR Commission 2005). *Ostrea edulis* is also included in the ‘Red’ lists of some regions (e.g., Wadden Sea, Black Sea). Indirect protection to native oyster reefs may also come from a number of EC Directives related to shellfish, such as the 95/70/EC, which sets community-wide rules to prevent the introduction and spread of the most serious diseases affecting bivalve molluscs, and the Shellfish Waters Directive (Table 1).

Fisheries for native oysters are regulated (sometimes prohibited) at a national level (e.g., Hiscock et al. 2005, Zaitsev 2006) but other national or regional conservation initiatives seem to be rare. There is little evidence that this management is leading to recovery of stocks. In the United Kingdom, *O. edulis* is included in a Species Action Plan under the U.K. Biodiversity Action Plan (U.K. Biodiversity Group 1999) and naturally occurring native oyster beds are considered a nationally scarce habitat (Jackson 2003), although complex regulations still allow some harvesting.

Biogenic habitats: maerls

Current distribution and status

Maerls (also known as rhodolith beds) comprise several species of crust-forming, free-living (i.e., unattached), calcareous red algae (Donnan & Moore 2003). Over time, they can become abundant enough to form substantial banks of live and dead material, with some European beds dated as older than 5500 yr (Grall & Hall-Spencer 2003). The major maerl-forming species in European waters are *Phymatolithon calcareum*, *Lithothamnion corrallioides* and *L. glaciale.* They can occur in exposed and sheltered environments, from the surface down to 100 m in depth (e.g., near Corsica and Malta), but most typically occur at 20–30 m (OSPAR Commission 2005).

Maerl beds are structurally and functionally complex habitats that support rich and diverse assemblages and host many species unique to those habitats. There is also growing evidence that maerl beds have considerable value as nursery grounds for species of commercial interest (Barbera et al. 2003).

Records of the presence or absence of maerl biotopes on European coasts are patchy (Birkett et al. 1998a). Detailed studies of maerl habitats have been undertaken only in the past 40 yr and at only a few locations, mainly in France, Norway and Ireland. Large, historically accessible maerl banks are relatively well recorded as a result of commercial interests. The locations of other maerl sites are known from the results of grab-and-dredge sampling during scientific research cruises. In more recent times, scuba divers have reported maerl banks. However, the extent of a maerl bed at any given location, its species composition and the species associated with it remain largely unknown (Birkett et al. 1998a). The most recent and comprehensive overview of maerl beds in Europe was compiled under the EC MAST-funded BIOMAERL project (Donnan & Moore 2003). Unpublished databases are expanding for the United Kingdom and France but an overall inventory has not been attempted.

In Europe maerl beds are patchily distributed and relatively restricted in size (Donnan & Moore 2003). They are found throughout the Mediterranean Sea, with important beds in Algeria, at Marseilles, in Corsica and Sardinia and in the Aegean (Birkett et al. 1998a). Maerl beds are also common on the Atlantic coasts, from Norway to Portugal. Spanish maerl deposits are confined mainly to the Ria de Vigo and Ria de Arosa (Galicia, northwest Spain). Maerl beds are relatively rare in the eastern English Channel, Irish Sea, North Sea and Baltic Sea (Barbera et al. 2003), whereas they are particularly abundant in Brittany, with more that 70 beds >1 km^2 and some of the largest and thickest beds in Europe and the world (Grall & Hall-Spencer 2003). In Ireland, maerl is widely distributed in the south and southwest (e.g., Galway Bay, Bantry Bay, Roaringwater Bay; De Grave & Whitaker 1999), and Scotland is home to some of the most extensive maerl beds in Europe (Birkett et al. 1998a).

Information on the status of present maerl beds in Europe is limited. Most Breton maerl beds are affected by human activities and the only pristine grounds remaining are small compared with the extensive maerl beds that covered several square kilometres in the 1960s (Grall & Hall-Spencer 2003). In 1999, surveys at one of the largest maerl beds in Brittany (Glenan), which was covered with living maerl until extraction started some 35 yr ago, showed that live maerl was rare over most of this bank while species-poor assemblages on muddy bottoms prevailed (Grall & Hall-Spencer 2003). Even maerl beds included in Breton NATURA 2000 sites are far from pristine and many are severely degraded.

Historical losses and causes

Information on historical losses of maerl beds in Europe as a consequence of human activities is virtually absent. Maerl has been harvested on a small scale in Europe for thousands of years for use in animal food additives, water filtration systems, acid lake and pond treatment, biological denitrification, toxin elimination, surfacing garden paths and in the pharmaceutical, cosmetics, medical and nuclear industries, but mostly as a cost-effective source of calcium/magnesium soil additive in agriculture and horticulture (Barbera et al. 2003, Grall & Hall-Spencer 2003). Initially, the quantities extracted were small, being dug by hand from intertidal banks, but in the 1970s about 600,000 t of maerl were extracted per year in France alone (Birkett et al. 1998a). Maerl extraction still forms a major part of the French seaweed industry, both in terms of tonnage and value of harvest, although amounts have declined to about 500,000 t yr^{-1} (Grall & Hall-Spencer 2003). In the United Kingdom and Ireland maerl has been harvested since the seventeenth century (De Grave

& Whitaker 1999) with up to 30,000 t $yr^{-1}$of maerl harvested commercially in the River Fal from 1975 to 1991 (Birkett et al. 1998a). Currently only limited extraction of maerl takes place in the United Kingdom and Ireland (De Grave & Whitaker 1999).

The extent to which this historical extraction has affected maerl beds is not known. Comparisons between museum collections made in 1885–1891 and again in 1995–1997 at maerl beds in the Firth of Clyde (Scotland) showed extensive changes, with substantial reduction in size and number of living thalli of *Phymatolithon calcareum* (Hall-Spencer & Moore 2000). Such changes have been attributed to mechanical impacts of scallop dredging, which started in that area in the 1930s and became particularly intensive in the 1960s through the advent of more powerful boats, more efficient dredges and better processing facilities. The wholesale removal of maerl habitats and significant reductions in diversity and abundance to adjacent areas at five sites around the coasts of Brittany have also been reported (Barbera et al. 2003) and attributed to commercial extraction (Grall & Hall-Spencer 2003).

Trends and threats

Maerl habitats are considered highly sensitive to overexploitation and other human activities that result in physical disturbance or deterioration in water quality (Barbera et al. 2003), particularly smothering by fine sediments (Wilson et al. 2004). This sensitivity is compounded by long recovery times due to the slow growth (approximately 1 mm yr^{-1}) and accumulation characteristics of maerl beds. The coralline algae that form the maerl are among the slowest-growing species, and substantial deposits take centuries to millennia to accumulate (Hall-Spencer et al. 2003) so that any effects of habitat removal are irreversible over timescales relevant to humans.

The major threats to maerl habitats have been recently reviewed (Barbera et al. 2003). The most obvious threats are from the ongoing commercial extraction. The three main areas of commercial exploitation in Europe have been Brittany, Cornwall and the west of Ireland. Nowadays, only limited extraction takes place in Ireland and the United Kingdom (De Grave & Whitaker 1999) but maerl extraction still forms a major part of the French seaweed industry (Grall & Hall-Spencer 2003). It has been predicted that if extraction rates persist at current levels the large Glenan deposit (Britttany) could be exhausted within 50–100 yr (Grall & Hall-Spencer 2003).

In addition to the direct effects of harvesting, other direct and indirect impacts on maerl beds have been noted. Damage to the surface of the beds is caused by towed demersal fishing gear, such as scallop dredges, which significantly reduce bed complexity, biodiversity and long-term viability (Hall-Spencer & Moore 2000, Barbera et al. 2003, Hall-Spencer et al. 2003). Hall-Spencer & Moore (2000) found that scallop (*Pecten maximus*) dredging in the Clyde Sea led to a 70% reduction of live maerl on a previously unexploited bed with no signs of recovery over the subsequent 4 yr.

Permanent moorings for pleasure boats can have similar, more localised, effects (Birkett et al. 1998a). The negative effects of increased eutrophication and turbidity in coastal waters both from silt loads and nutrient runoff from agricultural land and aquaculture have been documented in Galicia and in the Bay of Brest (Birkett et al. 1998a, Barbera et al. 2003). Smothering of maerl beds as a consequence of the invasion of the gastropod *Crepidula fornicata* has been observed in Breton bays (Grall & Hall-Spencer 2003). Maerl beds are also threatened by land reclamation and proliferation of coastal structures that alter circulation patterns (Birkett et al. 1998a, Barbera et al. 2003).

Protection measures

Although maerl is confined to a relatively small proportion of European shallow sublittoral waters, their conservation importance is being increasingly recognised (Birkett et al. 1998a, Donnan &

Moore 2003). Two of the most common maerl-forming species, *Phymatolithon calcareum* and *Lithothamnion corallioide*s, are now the only algal species specified as requiring appropriate management measures under the Habitats Directive (Annex V). Free-living Corallinaceae are also a named component of the habitat 'Sandbanks which are slightly covered by sea water all of the time' (EC 2003). Maerl beds are also included in the OSPAR list and Mediterranean *Red Book* of threatened habitats (Boudouresque et al. 1990, OSPAR Commission 2005).

In the United Kingdom, maerl is the subject of a Habitat Action Plan (U.K. Biodiversity Group 1999) and both *L. corallioides* and *Phymatolithon calcareum* are on the long list of species in the U.K. Biodiversity Steering Group Report (U.K. Steering Group 1995). Furthermore, in the JNCC interpretation of the EC Habitats Directive, maerl is identified as a key habitat within the Annex I category 'Sand banks which are slightly covered by seawater at all times'. This means that a number of SACs being designated under the directive will provide protection to the maerl that they contain. Maerl beds occur in three of 12 demonstration SACs within the United Kingdom, while the Fal and Helford (Cornwall) candidate SAC includes the largest maerl bed in England. Recently, the Board of Falmouth Harbour Commissioners in Cornwall has decided to cease licensing maerl extraction (Hall-Spencer 2005).

France has also recognised biogenic reefs, such as maerls, as vulnerable habitats, and some maerl grounds in Brittany lie within SACs (Grall & Hall-Spencer 2003). However, many of these are already severely degraded and are affected by dredge fishing, eutrophication and the spread of *Crepidula fornicata*. Maerl extraction in Brittany is under the control of the French mining management scheme, with quota schemes (80,000 t in 2001 on the Glenan bank) and regular environmental surveys. However, such quotas are considered not compatible with regeneration of the resource (Grall & Hall-Spencer 2003).

Biogenic formations

There are other examples of biogenic habitats that are severely affected and presumably have been subject to massive losses over the centuries. However, most of these losses have probably passed unnoticed and the information is scattered and mainly anecdotal. For example, off-shore rocky formations in the north Adriatic Sea, both organogenic and fossil in nature, have been flattened and reduced in size by trawling or other destructive forms of fisheries (Bombace 2001), sometimes to virtual extinction. Mediterranean 'coralligenous' reefs, which are considered one of the most valuable and diverse habitats in the Mediterranean Sea, are degraded and highly threatened by a variety of human activities (Ballesteros 2006) but how much coralligenous habitat might have been lost in the past as a consequence of these activities is not known. Significant declines in the extent of wild intertidal mussel beds have been reported from large coastal areas of Germany, the Netherlands and Denmark, and *Mytilus edulis* beds are now rare in the Wadden Sea (OSPAR Commission 2005, Wolff 2005) and are considered under threat in the United Kingdom (Hiscock et al. 2005). Large subtidal *Sabellaria spinulosa* reefs in the German Wadden Sea, which provided an important habitat for a wide range of associated species, have been completely lost since the 1920s (U.K. Biodiversity Group 1999, Wolff 2000) and similar losses have been reported also from areas of the northeast Atlantic and the United Kingdom (OSPAR Commission 2005). A significant contraction in the range of *S. alveolata* reefs on the south coast of England has occurred over a period of at least 20 yr until 1984. Declines have also been reported in the western part of the north Cornish coast, the upper parts of the Bristol Channel and in North Wales and the Dee Estuary but the causes of this regression are not known (U.K. Biodiversity Group 1999).

The scarcity of information probably underlies the lack of adequate policies and protection measures for these biogenic habitats. Biogenic reefs and concretions are all broadly covered as 'Reef' by the Habitats Directive but most are not specifically mentioned (EC 2003) and even national and regional initiatives are limited. Coralligenous assemblages are listed in the *Red Book* of Mediterranean assemblages (Boudouresque et al. 1990) and *S. spinulosa* reefs are included in the *Red List of Macrofaunal Benthic Invertebrates of the Wadden Sea* and the OSPAR list of threatened habitats (OSPAR Commission 2005). In the United Kingdom, both *S. spinulosa* and *S. alveolata* are the subject of Habitat Action Plans (U.K. Biodiversity Group 1999).

Sedimentary habitats (mudflats, sandflats and subtidal soft bottoms)

Current distribution and status

Coastal areas are dominated by soft-sediment habitats. The marine biotope classification for Britain and Ireland identifies a number of littoral and sublittoral sediment biotopes (U.K. Biodiversity Group 1999). These are grouped into several major categories (gravels and sands, muddy sands, muds and mixed sediments) and subdivided further according to depth (littoral, infralittoral or circalittoral) and sediment size. Muddy habitats usually occur in sheltered areas, such as sea lochs, enclosed bays and estuaries, whereas sandflats and coarser sediments tend to develop in more exposed situations on the open coast. Distinctions are also sometimes made between estuarine and marine habitats (e.g., OSPAR Commission 2005).

Despite the importance of these highly productive soft-sediment habitats, which support large numbers of predatory birds and fishes, providing nursery, feeding and resting areas, and the long history of studies on many aspects of their ecology, no comprehensive inventory of their extent and status is available at a European level, and even regional or local initiatives are rare. The only habitat for which there are some rough estimates of the amount and distribution are intertidal mudflats. In the OSPAR region, the largest continuous area of intertidal mudflats borders the north coasts of Denmark, Germany and the Netherlands in the Wadden Sea, covering around 4990 km^2 (OSPAR Commission 2005). In the United Kingdom, intertidal mudflats are widespread, with significant examples in the Wash, the Solway Firth, Mersey Estuary, Bridgwater Bay and Strangford Lough; overall they are estimated to cover about 2700 km^2 (U.K. Biodiversity Group 1999).

Historical losses and causes

The extent of historical losses of soft-bottom habitats is virtually unknown for any country and even regional or local information is scarce. Available information suggests that loss or deep alteration of these types of habitats may have been extremely high, particularly in estuaries and enclosed bays but there is also the possibility that unvegetated soft bottoms have increased their area at the expense of losses in other more structurally complex habitats (e.g., seagrass beds). Past losses are likely to have been related mainly to land claim for agriculture, ports and industrial and urban developments. In the United Kingdom, for example, it is estimated that at least 88% of estuaries have lost intertidal habitats and about 25% of overall estuarine intertidal flats have been removed with peaks of up to 80% in some estuaries such as the Tees (U.K. Biodiversity Group 1999, OSPAR Commission 2005). The amount of intertidal mudflats and sandflats may have also changed significantly over time in relation to coastal erosion, changes in sea levels and human interventions to control these factors (e.g., Lee 2001).

Most often, however, these habitats have probably been deeply altered in their fundamental characteristics, including sediment structure and composition, accretion or erosion rates, and inhabiting fauna. As an example, in Europe the massive use of hard defence structures and beach replenishment schemes has deeply changed the structure of shallow surf-zone sediments along whole coastlines in past decades (Airoldi et al. 2005, Martin et al. 2005), as is the case of the north Adriatic Sea (e.g., Figure 5), presumably affecting an enormous and overlooked amount of shallow soft-bottom habitats. Similarly, most sedimentary benthic systems on the continental shelf of Europe have been modified by fishing activities, particularly bottom trawls and dredging, in the last 100 yr (Ball et al. 2000, Frid et al. 2000). In the southern North Sea fishing is thought to have long been the main ecological structuring force on the benthos, to an extent that makes it difficult even to design robust field experiments due to the virtual lack of control areas (Hall-Spencer & Moore 2000). How much of this transformation should be considered as habitat loss or degradation is difficult to quantify.

Trends and threats

There is limited organised information on the current trends and threats to sedimentary environments (Brown & Mclachlan 2002). Today the major threats to sedimentary habitats are more likely to be linked to further land claim, construction of marinas and slip ways, the widening and dredging of channels for navigation, pipe and cable laying, oil and gas extraction, tourist developments and infrastructures and the construction of sea defences. Some of these threats have slowed considerably in recent years, at least in some European countries, but they have not stopped. In the United Kingdom, many coastal areas, including estuaries, are now either licensed or available for exploration and development (U.K. Biodiversity Group 1999). Pollution from sewage discharge, aquaculture activities, industries and shipping are also important threats to sedimentary environments and associated fauna, leading to anoxic conditions particularly in estuaries and enclosed basins, as observed in Scandinavian and Baltic waters (Karlson et al. 2002), and to long-term accumulation of contaminants (Islam & Tanaka 2004).

Physical disturbance by fishing and aggregate dredging activities also represents a major threat to Europe's sedimentary habitats and associated biota (Lindeboom & de Groot 1998, Tudela 2004), although nowadays highly disturbed seabeds may appear to be relatively unaffected by fishing activities or other physical disturbances (e.g., Hall-Spencer & Moore 2000). On the Dutch continental shelf, the fisheries are now so intensive that every square metre is trawled, on an average, once to twice a year (Lindeboom 1995), and this broadly applies to the entire sea bed of the North Sea (Gray 1997).

Direct extraction of sands and gravels for coastal developments, use in the construction industry and beach nourishments are also major, increasing threats for sedimentary habitats (Newell et al. 1998, van Dalfsen et al. 2000). Extraction of sands has steadily increased in most north European countries during the past few decades (ICES 2006b). In the Netherlands, for example, extraction of sands has increased from <5 million $m^3\ yr^{-1}$ in 1974 to >35 million $m^3\ yr^{-1}$ in 2001, and in the coming decades an average request of 19–43 million $m^3\ yr^{-1}$ is expected. Much of the extracted sand is used for beach recharges and coastal defence. Beach nourishments are being increasingly used along European coasts as a 'soft' measure to counteract erosion (Hamm et al. 2002), but the consequences of both the extraction and the disposal of sands on sedimentary habitats and biota have received limited attention (Desprez 2000, van Dalfsen et al. 2000, Simonini et al. 2005).

Some projections of loss are available for intertidal areas in relation to possible future changes in sea level, recession of coastlines and coastal 'squeeze'. For example, sea-level rise is projected to cause a loss of 80–100 km^2 of intertidal flats in England between 1993 and 2013 (U.K. Biodiversity Group 1999), particularly in southern and southeast regions; the major firths in Scotland will probably also be affected.

Protection measures

Protection for intertidal and shallow mudflats and sandflats is provided by various international and E.U. agreements, including the Ramsar Convention, the Bonn Convention, the Bern Convention, and the Birds and Habitats Directives (Table 1). In particular, 'Mudflats and sandflats not covered by seawater at low tide' and 'Sandbanks which are slightly covered by sea water all the time' are listed in Annex I of the Habitats Directive. Mudflats are also included within several other designated Annex I Habitats: 'Estuaries', 'Lagoons' and 'Large shallow inlets and bays'. Mudflats are also in the list of OSPAR threatened habitats (OSPAR Commission 2005). Some countries also have national protection measures. For example, in the United Kingdom mudflats are the subject of a Habitat Action Plan (U.K. Biodiversity Group 1999); furthermore, over 300 SSSIs including mudflats have been designated in estuaries and 10 coastal ASSIs in Northern Ireland contain significant areas of mudflats.

Soft bottoms deeper than 30 m do not seem to be the target of any specific protection measure at a European level (Hiscock et al. 2005), although a number of EC Directives that regulate water quality provide indirect protection from some types of impacts (Table 1). There are, however, national initiatives. For example, in the United Kingdom 'Sublitoral sands and gravels' and 'Mud habitats in deep waters' are the subjects of Habitat Action Plans (U.K. Biodiversity Group 1999). Commercial fishing activities are excluded from a number of estuaries and bays around the coast of the United Kingdom, which are important nursery areas for juvenile commercial species (e.g., River Exe, River Conwy and Filey Bay). Fishing activities are prohibited within 500 m of gas and oil platforms, from firing ranges and in close proximity to certain military installations (U.K. Biodiversity Group 1999).

Discussion

Population density along European coasts has been growing since ancient times. There are increasingly greater demands and impacts on the habitats and resources in coastal environments. The present review has shown that such intensive exploitation has caused dramatic losses and severe deterioration of native coastal habitats (e.g., Tables 2–4). There are many policies and directives

Table 4 Summary of main characteristics of European coastlines and habitats based on reviewed sources

Characteristic	Value	Main references
Coastline length[a]	325,892 km	Pruett & Cimino 2000
Population within 50 km[b]	200×10^6	Stanners & Bourdeau 1995
Degraded coastlines	85%	EEA 1999a
Years of impact[c]	2500 yr	Rippon 2006, Lotze et al. 2006
Artificial coastlines	22,000 km^2	EEA 2005
Defended/eroding coastlines	7,600/20,000 km	EC 2004
Increase in N/P loads 1940s–1980s	2- to 4-/4- to 8-fold	Nehring 1992, EEA 2001, Karlson et al. 2002
Number of invasive species	450–600	Reise et al. 2006
MPAs (Number/Total surface)	1,129/ 236,000 km^2	UNEP/WCMC 2006, MPA Global 2006
Present coastal wetlands/loss since 1900s	51,910 km^2/>65%	Nivet & Frazier 2004, EEA 2006a
Present seagrasses/historical losses[d]	7290 km^2/>65%	Duarte 2002, Green & Short 2003
Present wild native oyster reefs/historical losses[d]	Scarce/>90%	Mackenzie et al. 1997
Present macroalgal beds/historical losses[d]	Unknown/2–4 m in depth	Vogt & Schramm 1991, Eriksson 2002

[a] Including islands.
[b] In the 1990s.
[c] Since beginning of modification and transformation of coastal landscapes.
[d] Estimate based on reviewed local to regional sources.

Table 5 Past (earlier than 1900 = P), recent (twentieth century = R) and present (N) main drivers of habitat loss along European coasts based on reviewed sources

	Wetlands			Seagrasses			Macroalgae			Biogenic habitats			Sediments		
Impact	P	R	N	P	R	N	P	R	N	P	R	N	P	R	N
Claim/conversion	***	***	**	**	**	*		*		*	*		***	***	**
Coastal development	***	***	***	***	***	***	?	**	**				***	***	***
Coastal defence		**	***		*	**		g	*g[a]					**	***
Exploitation				**				*	*	***	***	***		*	**
Water quality	**	**	**	**	***	***	?	***	***	**	**	**			
Diseases/pests/predators		*	*		***	**	?	**	**	**	***	***			
Destructive fishing				*	***	*	?	**	*	**	***	***		?[b]	?[b]
Aquaculture		*	**		*	**				*	**	***			

Note: Habitats are coastal wetlands (including salt marshes); seagrass meadows; macroalgal beds (kelps, fucoids and other complex macroalgae); biogenic habitats (including oyster reefs and maerls); and sedimentary habitats (mudflats, sandflats and subtidal soft bottoms). Impacts are drainage, embankment, land claim and habitat conversion (e.g., into agriculture land or into freshwater lake); coastal development including urban and industrial developments, ports and infrastructures, marinas and tourist and recreational developments; coastal defence which includes hard structures, beach nourishments and other measures associated with impacts from erosion, storms and sea-level rise; direct exploitation or harvesting; water quality including organic and chemical pollution and altered sedimentation regimes; outbreaks of diseases, introduced species, competitors or predators; destructive fishing techniques (e.g., trawling); aquaculture. Blank cells = nil or modest, * = low, ** = moderate/locally high, *** = high and widespread, g = habitat gain, ? = not known.

[a] Negative effects of beach nourishments, enhancement of substratum availability from hard structures.

[b] Fishing affects soft bottoms extensively, but it is difficult to evaluate how many soft sediment habitats are lost.

aimed at reducing and reversing these losses (e.g., Table 1) but their overall positive benefits have been low.

Coastal habitats are affected by many threats with a few dominant threats that vary somewhat by habitat and over time (Table 5). The greatest impacts to wetlands have been land claim and coastal development with the latter rising in importance over time. The greatest impacts to seagrasses and macroalgae are presently associated with degraded water quality whereas in the past there have been more effects from destructive fishing and diseases. Coastal development remains an important threat to seagrasses in particular. For biogenic habitats, some of the greatest impacts have been from destructive fishing and overexploitation with additional impacts of disease, particularly to native oysters. Coastal development and defence have had the greatest known impacts on soft-sediment habitats with a high likelihood that trawling has impacted vast areas while not causing loss of soft-sediment habitats per se.

Shellfish, oyster reefs in particular, have been among the most severely affected of all coastal habitats by overexploitation and other human-driven changes to the environment. By the late nineteenth century, overfishing combined with outbreaks of diseases, habitat transformation, and the introduction of non-native competitors and parasites, had already wiped out wild *Ostrea edulis* reefs around much of the European coastline. Where documentation is available, it is evident that the loss of shellfish habitat is a major cause of species extirpation and declines in biodiversity (Wolff 2000, Lotze et al. 2005). Currently most native oyster reefs are functionally or entirely extinct in most coastal areas of Europe and they are probably one of the most endangered marine habitats. However, loss and threats to these habitats are largely overlooked and shellfish beds do not seem to be covered by adequate protection measures, conservation legislation or convention at a European level. Oyster reefs are not mentioned specifically in the Habitats Directive, which sets the present framework for habitat protection in Europe. Although less documented and more localised, similar losses seem to have occurred also to other types of shellfish reefs and biogenic formations, such as maerl beds, intertidal mussels beds, *Sabellaria* spp. reefs and coralligenous formations.

The current losses in European coastal habitats are alarming; even worse is the fact that these losses are only measured against recent distributions with little recognition of the compounding impact of centuries and millennia of habitat loss. This historical short-sightedness has been referred to as 'shifting baselines', where we only recognise declines in the natural environment relative to the baseline of recent memory in each generation (*sensu* Dayton et al. 1998, Jackson et al. 2001). At present, there seems to be limited public, political and even scientific awareness of the extent, importance and consequences of such a long history of coastal habitat loss (Lotze 2004).

The evidence reviewed in the present work clearly indicates that in some European regions most estuarine and near-shore coastal habitats were probably already severely degraded or driven to virtual extinction well before 1900 and sometimes much earlier than that. Ecological descriptions of coastal marine habitats are rather recent (mid-1900s), and long-term documentation of habitat declines and losses are virtually missing for most systems. Even when documentation is available, evidence tends to be overlooked. For example, Wolff (2000) points out how most Dutch people are inclined to ignore the profound habitat changes and losses that occurred during the past 2000 yr in the Wadden Sea, and they tend to consider present-day systems to be in a 'natural' state.

Nowadays only a small percentage of the European coastline is considered in 'good' condition (EEA 1999c). The degradation has continued at high rates over the past few decades. In France, for example, it is estimated that 15% of natural areas on the coast have disappeared since 1976 and are continuing to do so at the rate of 1% a year (Stanners & Bourdeau 1995). In Italy, around 7000 km^2 of coastal marshes were present at the beginning of the 1900s, no more than 1920 km^2 in 1972 and fewer than 1000 km^2 today (Stanners & Bourdeau 1995). Losses of coastal wetlands and seagrasses exceeding 50% of original area have been documented for most countries where long-term data were available, with peaks above 80% for many regions. Beds of complex macroalgae have been under severe recession since at least the early 1900s. An impressive number of local to regional extinctions of habitats have been documented. Overall, it is estimated that every day between 1960 and 1995, a kilometre of 'unspoilt' European coastline has been developed (EUCC 1998). Those fragments of native habitats that remain are under continued threat.

Recommendations for conservation and management

Fortunately there is recognition at the European level that the current management and conservation of marine diversity and habitats is insufficient. In particular, there have been recent and significant E.U. policies to identify the coastal and marine habitats of Europe and to develop networks of MPAs around them. These efforts have been inspired in part by the international commitments developed as part of the Rio CBD and the World Summit for Sustainable Development. The development of E.U. MPAs is included in the Habitats Directive and the Birds Directive and embodied in particular in the development of protected areas in the NATURA 2000 network. There are also other regional (e.g., OSPAR) and national initiatives with similar aims for protection of endangered habitats and species (Table 1). These policies are necessary but not sufficient. Indeed MPAs alone have limited use for addressing threats such as poor water quality, disease or even coastal development. Some key needs and opportunities for enhancing the overall conservation and management of coastal and marine habitats in Europe are suggested below.

Currently, there is no comprehensive summary of the distribution of habitats along European coastlines and their management is not well informed by adequate knowledge of their distribution and status. Despite millennia of reliance on the resources from coastal and marine ecosystems, we cannot accurately describe the distribution of even the shallowest habitats. Detailed habitat mapping should therefore be given high priority to promote conservation and sustainable management practices. Some databases are available or are being prepared for some countries or habitats but most information is scattered, fragmented or limited to few case studies. At least there has been good progress in developing a consistent habitat classification as part of the European Union Nature Information System (EUNIS) (Davies et al. 2004), which is a necessary precursor to developing a consistent database of habitat distribution.

These habitat distributions then need to be compared with the existing protected areas to identify gaps in protection. To fill these gaps, there should be systematic planning for placement of new protected areas and other management measures. Increasingly scientists, agencies and organisations are using systematic planning approaches to target the placement of their protection and management efforts particularly at regional levels (e.g., Beck & Odaya 2001, Possingham et al. 2002, Airamè et al. 2003, Leslie et al. 2003, Beck et al. 2004). These approaches enable decision makers to develop a range of solutions for protection and management and to examine how changes in decisions can affect solutions.

The values of these coastal habitats also need to be better assessed to provide real estimates of the ecosystem services that they provide such as pollution regulation, storm hazard reduction, productivity of nurseries for fisheries and recreation (Agardy & Alder 2005). Better valuations of these services will illustrate for communities and governments the real costs of this habitat loss and should provide impetus and economic incentives for their protection and restoration.

There are still reasonable opportunities for conservation of coastal habitats in key areas throughout Europe. This protection should be put in place quickly because conservation is cheaper than restoration. The current EC policies are mostly aimed at these protections but they do need to be implemented.

Given the extent of damage to coastal habitats, restoration will be required in many places to meet any reasonable goals for conservation and management. Information on historical distributions and loss is extremely important because management goals for these habitats should be based on historical estimates of the distributions of these habitats, not the vastly reduced current distributions (Beck 2003). Even extremely modest goals of 10% protection of the historical distributions of European coastal habitats will require some restoration for many habitats.

To meet these goals for conservation and restoration there should be greater involvement by nongovernmental organisations and community groups and there are tools that they can use to contribute directly to conservation and management. These groups have often been involved in efforts to develop MPAs and restore coastal habitats and these efforts should be encouraged and expanded. There are also new tools, such as the private leasing and ownership of marine lands and resources that can be employed more often by private groups to help protect and restore these coastal habitats (Beck et al. 2004). For example, the National Trust in the United Kingdom leases intertidal lands and sea beds from the government along some 180 km of the coast for conservation and restoration. Recently there has been new policy adopted by the French government that allows private groups also to lease subtidal lands as is commonly done by many business interests (e.g., aquaculture industry).

Most coastal habitats lie within the exclusive economic zones of individual countries and thus the individual coastal zone management of these countries must be strengthened to protect and manage these habitats. The European Union and member nations have been dedicated to developing better Integrated Coastal Zone Management (ICZM) for some time. The development of strong and effective ICZM programmes has been slow for most nations. These programmes need to advance further to slow and reverse coastal habitat loss.

There is much academic and agency interest in developing a more ecosystem-based management (E-BM) approach for managing the many marine resources and the overlapping stakeholder needs for access to these resources (e.g., Browman & Stergiou 2005). E-BM has been incorporated as a central goal of the European Union's emerging 'Marine Strategy' (Table 1). While this approach is needed and sensible, it will take many years to develop and its development should not be allowed to slow efforts to protect and restore habitats now.

Conclusions

Europe has seen decades, centuries, even millennia of coastal habitat loss and it continues today. Estuaries, enclosed bays and near-shore shelf habitats around the European coasts are some of the most degraded environments on Earth as they have been used intensively for thousands of years. These functionally valuable coastal ecosystems are still a focal point for human colonisation and use. Before all the services from these ecosystems are lost, efforts should be redoubled to protect the remaining habitats, slow and stop future losses and restore some of these habitats.

Recent efforts toward a more sustainable use of these coastal resources have not reversed the trend. There is no single best strategy for addressing these losses and there are many approaches that can help. Many of these are commonsense recommendations that are common in policies within and between nations, but the progress toward them has been slow with a few exceptions. The United Kingdom clearly has made more progress than most nations in the European Union and internationally. If the long history of physical destruction, fragmentation and transformation is further neglected, and if significant, irreversible thresholds are passed, the future sustainability of those few fragments of native or semi-native habitats that remain may ultimately and finally be compromised.

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