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Maritime traffic effects on biodiversity in the Mediterranean Sea

Volume 1 - Review of impacts, priority areas and mitigation measures

Edited by Ameer Abdulla, PhD and Olof Linden, PhD

IUCN Centre for Mediterranean Cooperation / IUCN Global Marine Programme

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Table of contents

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Foreword

Only in the last decade has there been recognition that marine ecosystems worldwide are suffering a massive decline in biodiversity and irreparable alterations to ecosystem functions. The capacity of oceans to recover from global perturbations and, thus, to maintain ecosystem goods and services is rapidly weakening. Climate change, pollution, overfishing, introduced species and habitat degradation have been identified as the principal causes of marine biodiversity loss and thus priorities for conservation intervention. The Mediterranean Sea is representative of such extreme conditions resulting from persistent historical impact sustained over thousands of years of human development, settlement, commerce and resource exploitation. Currently, there are 601 cities with a population of 10,000 or more inhabitants along the Mediterranean coasts and 175 million tourists a year visit these shores.

An enclosed sea such as the Mediterranean is particularly vulnerable to ship-associated impacts due to a high-volume of shipping routes, long history of use, and sensitive shallow and deep-sea habitats. Over the past half century, shipping has greatly expanded in the Mediterranean Sea. Between 1985 and 2001, a 77% increase was recorded in the volume of ship cargo loaded and unloaded in Mediterranean ports. An estimated total of 200,000 commercial ships cross the Mediterranean Sea annually and approximately 30% of international sea-borne volume originates from or is directed towards the 300 ports in the Mediterranean Sea. These values are expected to grow three or four fold in the next 20 years.

It is logical, then, to predict that there will be various maritime-associated impacts on marine biodiversity and that these are also expected to grow at an alarming rate. These impacts are due to ship pollution and emissions, collisions and noise, grounding and anchor damage, and transportation of non-indigenous species. This review is the first attempt, as far as we know, to present and discuss these impacts scientifically, comprehensively and from a multi-disciplinary perspective in an effort to identify potential management and collective mitigation measures. We are optimistic that such a first and important step will initiate international and regional dialogue and pragmatic multi-lateral efforts towards addressing these serious issues in the Mediterranean.

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Chapter 1

Biodiversity impacts of ship movement, noise, grounding and anchoring

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1 Shipping noise, a challenge to the survival and welfare of marine life?

1.1. Introduction

Marine life in the Mediterranean Sea is threatened by intensive human activities such as fisheries, ship traffic, pollution and coastal development. Cetaceans and other vertebrates are affected not only by chemical pollution, but also by noise pollution (Richardson *et al*., 1995; Simmonds *et al*., 2004). Noise has become a ubiquitous form of marine pollution, especially in areas of heavy maritime traffic and along developed coasts. Intense underwater noise is generated by airguns, widely used for geophysical exploration in the oil and gas industry as well as for academic and government research purposes; by high power sonar, either military or civilian; by ship traffic; by shoreline and offshore construction works; and by a number of other commercial, scientific, military and industrial sources. The most powerful noises (from airguns, sonars, and explosions) may directly injure animals in the vicinity of the source. General ship traffic, heavy industries on the coast and a variety of other human activities generally do not generate such intense noise, but the acoustic pollution they produce is constant over time and may affect large areas. It may be a serious hazard not only to individual animals, but also to entire populations. Such increased background noise affects underwater life just as airborne noise affects terrestrial animals, including human beings.

However, since sound travels five times faster in water than in air, and since the density of water transmits acoustic energy very efficiently over much greater distances than in air, the effects of underwater noise may extend throughout very large volumes of water. The awareness that man-made noise can affect marine life, marine mammals in particular, and that a regulatory system is needed to mitigate such effects has grown in recent years, mainly in the context of military sonars and seismic surveys. There is now increasing concern about all types of noise pollution and their impact on other zoological groups, such as fish and invertebrates.

1.2. Marine mammals and noise

The underwater environment has its own acoustic peculiarities and marine mammals are extraordinarily well adapted to them. Acoustic communication and perception are especially well developed in these animals both compared with their other senses and compared with other zoological groups. Marine mammals live in a medium which transmits light poorly but through which sound propagates very well even over long distances, especially when frequencies are low or the sound is channelled by pressure and temperature gradients (Urick, 1983; Richardson *et al*., 1995). Marine mammals rely heavily on sound to communicate, to coordinate their movements, to navigate, to exploit and investigate the environment, to find prey and to avoid obstacles, predators, and other hazards.

Noise can severely interfere with their lives. Noise pollution can cause marine mammals to abandon their habitat (Borsani *et al*., 2007) and/or alter their behaviour by directly disturbing them (Aguilar Soto *et al*., 2006) or by masking their acoustic signals over large areas (Payne & Webb, 1971; Hildebrand, 2005); loud sounds may directly affect their hearing abilities by producing either temporary or permanent hearing loss (Simmonds & Lopez-Jurado, 1991; Richardson *et al*., 1995; NRC, 2000; NRC, 2003; Gordon *et al*., 2004). All these effects may be critical for the survival of marine mammals. Some high-energy sound sources can have immediate impacts and even trigger mortality events, as recently evidenced by several dramatic and well-documented cases of atypical mass strandings[1] of beaked whales, as in Greece in 1996, the Bahamas in 2000 and the Canary Islands in 2002 (D'Amico,

1 Mass strandings are defined as two or more animals stranded in the same area.

1998; Frantzis, 1998; NOAA, 2001; Department of the Environment, 2002; Evans & Miller, 2004; Fernández *et al*., 2005).

In some cases anthropogenic high-power sound sources (up to 250dB re 1μPa at 1m distance) radiate low- to high-frequency sound, and individual animals are exposed to high sound levels (above 160dB re 1μPa) over relatively short periods of time (acute exposure), as in some military sonar operations. In other cases potential exposure to high noise levels can occur for longer periods—weeks or months—as in the case of seismic surveys or some construction works, such as pile driving for port or bridge construction (Borsani *et al*., 2007).

As well as producing high sound levels close to the source, seismic surveys and low-frequency naval sonar may radiate low-frequency sound over very large areas, thereby exposing populations to lower sound levels (below 160dB re 1μPa) over relatively long periods of time (chronic exposure). Continuous exposure to low-frequency sound is also an effect of distant shipping noise, multiple distant seismic surveys or construction works (Tyack, 2003; Nieukirk *et al*., 2004; Borsani *et al*., 2007; Pavan, personal observation).

It is generally accepted that received levels greater than 120dB re 1μPa may cause behavioural changes (Richardson *et al*., 1995; Moore *et al*., 2002) and levels greater than 150dB can lead to effects ranging from severe behavioural disruption to TTS (Temporary Threshold Shift), a temporary lowering of hearing sensitivity; levels greater than 170–180dB are considered enough to cause PTS (Permanent Threshold Shift), which means permanent hearing loss, deafness and physical damage, including death in some circumstances. These numbers are debatable: they may vary according to environmental context, behavioural context and species, as demonstrated by Cuvier's beaked whale strandings that occurred after repeated exposure to levels believed safe.

Although atypical mass strandings are the most dramatic kind of incidents associated with acute sound exposure, at least for beaked whales (Frantzis, 1998; NOAA, 2001; Department of the Environment, 2002; Evans & Miller, 2004; Fernández *et al*., 2005), it should be remembered that the effects of repeated non-lethal exposure and of increased noise levels are generally unknown but may potentially be significant in both the short and the long term. Furthermore, the biology of 'disturbance' and the effect of noise on the survival and fecundity of marine mammals and their prey species are not well understood.

1.3. Impacts on other marine organisms

While most interest in anthropogenic noise and its mitigation has focused on marine mammals (mainly cetaceans and pinnipeds) and a few other vertebrates (such as sea turtles), there is increasing concern regarding the impact of such noise on fish, other vertebrates such as aquatic and diving birds, and marine invertebrates (including crabs and lobsters).

Fish use sounds to communicate and to perceive information from the environment; more than 50 families of fish use sound, generally below 2-3 kHz, in a wide variety of behaviours including aggression, protection of territory, defence and reproduction.

Although much less is known about the effects of anthropogenic sounds on fish than on terrestrial or marine mammals, there is a small but growing body of literature demonstrating that such sounds can mask fish communication (Wahlberg & Westerberg, 2005), generate stress that negatively affects the animals' welfare (Wysocki *et al*., 2006), induce fish to abandon noisy areas (Mitson & Knudsen, 2003), destroy the sensory cells in fish ears and, in the long term, cause temporary and possibly permanent loss of hearing (McCauley *et al*., 2003; Popper, 2003; Smith *et al*., 2004; Popper & al., 2005), and also damage eggs.

In addition, the gas-filled swim bladder in the abdominal cavity, which may serve as a sound amplifier for both hearing and sound production, is a potential receiver for sound energy even at frequencies not used for communication.

Although it is known that noise can deafen fish and otherwise have a serious impact on them (McCauley *et al*., 2003; Popper *et al*., 2004, 2005), little concern has been shown for the ecological implications of such effects and few mitigation procedures involve fish or spawning aggregations. This field has only been addressed on a limited scale and requires further exploration. The effects of noise on the food web and on fisheries also need to be investigated.

It should be emphasised that the reaction of fish to sound has only been studied in a limited number of species, and existing data cover only a few types of noise source. Some data suggest that noise exposure may occur not only in the natural environment but also in locations such as marine aquaria and aquaculture facilities as a result of background noise like that produced by pumps and air-bubblers (Bart *et al*., 2001).

Great care is needed, however, when extrapolating existing data to other species and sound types and to different environmental and behavioural contexts. In addition, few studies have specifically addressed marine invertebrates.

1.4. Shipping noise

Ship traffic has been increasing in the oceans in recent decades, especially in the northern hemisphere, and very likely will increase exponentially in future. Ship traffic produces diffuse and almost continuous noise that may affect very wide areas. Low-frequency (below 1,000Hz) ambient noise levels generated by ship traffic have increased in the northern hemisphere by two orders of magnitude over the last 60 years (3dB/decade: Andrew *et al*., 2002); their masking effect has therefore reduced the potential for long-range communication in mysticetes (Payne & Webb, 1971).

Ship propulsion noise accounts for more than 90% of the acoustic energy that humans put into the sea (Green *et al*., 1994). Commercial shipping is estimated to have raised average ambient noise levels in the 20–200Hz band by about 10dB in the past century. Payne and Webb (1971) point out that this is the dominant frequency band used by baleen whales for communication. Ubiquitous and continuous noise may have chronic effects, degrading the quality of marine habitats; even subtle effects, such as avoidance and signal masking, may have long-term population consequences if exposure is continuous. In addition, some problems such as collisions between whales and vessels may involve acoustic risk factors. In this case, the question is not whether there are adverse reactions to the noise itself, but why whales may sometimes not react to the noise of an oncoming vessel and get out of the way (Tyack, 2003).

Ship noise can include different features or result from a combination of multiple radiating sources. Noise can be of a burst/pulsed type, such as that produced by propeller cavitation, or continuous broadband with tonal components. Low frequencies (<100Hz) may be generated by engines, higher frequencies (<1000Hz) by rotating gears and mechanical resonances, and even higher tonals (1-2kHz) by turbine engines and hydro-jets (such as fast ferries). Other sources can be pumps and auxiliary engines, generators, compressors and other machinery. Sound levels and frequency characteristics caused by propulsion are roughly related to ship size and speed, but there is significant variability among ships of the same class and no accurate prediction models are available (Heitmeyer *et al*., 2004). Large traditional ships may have dominant tones with source spectrum levels near 180dB re 1μPa/Hz^2 at 1m, with broader-band tonal components near 200dB (Richardson *et al*., 1995). Large ships may create louder, lower-frequency sounds with greater potential for long-range propagation because of their greater power, more slowly rotating engines and propellers, and larger surface areas for efficiently transmitting vibrations to water.

1.4.1. Special case of fast ferries

Fast ferries are an important type of shipping in the Mediterranean Sea, and in the Ligurian Sea (Pelagos Sanctuary) in particular. These ferries have different propulsion systems from traditional vessels, and different sources of noise. Fast ferries generate broader-band noise than traditional vessels; they produce high level hydrodynamic noise, up to 10kHz and more, and engine noise often with narrow peaks at high frequencies (1–2kHz). In some cases they are quieter than large cargo ships, but they move so fast that they may pose an increased risk of vessel collision rather than of noise impact.

1.4.2. Recreational boating

Small recreational craft, which can potentially move almost anywhere with very few restrictions, may be an additional cause of disturbance to marine life both in pelagic waters, where they can affect marine mammals, and in shallower waters, where the noise may affect local fish populations. In shallow waters their impact may extend beyond acoustic effects on animals to physical alteration of benthic habitats and communities. Yachts and motor sailers with inboard engines may produce multiple noises like large ships, normally at lower levels but higher frequencies. In contrast, inflatables and other craft with outboard engines and small propellers may produce very loud broadband noise, particularly if pushed at high speed. Although the in-air noise emissions are regulated by EU Recreational Craft Directive 2003/44/EC, no limits are set for underwater noise emission.

Severe restrictions should be applied to recreational boating to safeguard marine animals. In areas where marine mammals are present, in fish breeding grounds, and in particular in Marine Protected Areas (MPAs) and Special Areas of Conservation (SACs), the underwater noise emissions of all vessels should be regulated and monitored.

1.4.3. Whale-watching boats

Whale watching is a rapidly growing activity that may have an impact on marine mammals at the individual, population and stock levels. Rules and permits are already in force in many countries, but the noise issue is seldom taken into consideration. Noise irradiated by engines and propellers is an important component of disturbance to the animals (Erbe, 2002). Beyond complying with national rules and restrictions on approaching marine mammals, whale-watching operators should also comply with noise emission limits to minimize their disturbance.

1.5. Impact of ship noise

Whilst there is little evidence to suggest that ship noise has an immediately acute or lethal effect, the impact of repeated disturbance and increased noise levels is generally unknown but may potentially be significant over the long term at the population or stock level. Shipping noise in high-traffic areas can be louder and more widespread than the levels that have caused Cuvier's beaked whale strandings.

Low-intensity sounds can cause masking and behavioural disruptions; although there is little direct evidence, it is likely that if such disruptions occur frequently, for extended periods of time, or during biologically important activities such as mating, feeding, birth or mother-young bonding, they may affect longevity, growth, and reproduction. Noise may induce animals to abandon areas otherwise beneficial to them, or to deviate from their usual migration routes.

Furthermore, frequent or chronic exposure to low intensity sounds may cause hearing loss and make animals that rely on hearing to locate and capture prey and to detect and avoid predators less able to do so. Frequent or chronic exposure to sounds of variable intensity may cause stress (Wysocki *et al*., 2006), which human and terrestrial animal studies indicate can affect growth, reproduction, and disease resistance.

Masking appears to be the most relevant issue for animals that rely on low frequencies to communicate. Baleen whales do so over long distances; if whales have no mechanisms to compensate for the increased noise, the noise may significantly reduce the range over which they can communicate and investigate the environment. In addition, the fact that commercial whaling has decimated populations of many baleen whale species may mean that whales now need to communicate over even greater ranges than in the environment in which their communication evolved.

Masking is an issue for fish too, as they use low-frequency sound to communicate, but they do so generally over shorter distances with lower-level sounds than whales.

The consequences of masking may be serious, in particular in the case of long-range communication. In a simple 20*log(range) transmission loss scenario—an ideal type of situation in which sound energy spreads out spherically—any 6dB increase in background noise level reduces the communication distance (the range at which a signal can be heard above the background noise) by a factor of two and the area within which the signal can be heard by a factor of four. But when propagation approaches cylindrical spreading and transmission loss is close to 10*log(range), the same 6dB noise increase reduces the communication range by a factor of 4 and the area by a factor of 16. In such a case, a 20dB increase in background noise reduces the communication range by a factor of 100, a dramatic reduction from, for example, 100km to 1km.

The combination of increased distance between signalling and receiving whales in populations that have already been reduced by other impacts and the reduction in their effective range of communication caused by shipping noise may have an adverse impact on endangered whale species if the noise interferes with communication used for reproduction and social behaviour. Furthermore, the negative effect of masking could be further aggravated if whales have lowered hearing sensitivity caused by long exposure to noise.

In spite of significant advances in ship-induced noise research, there remain major limitations to our ability to predict either current levels of ambient noise or future trends in noise levels that might result from changes in the world's shipping fleet. This is a consequence both of deficiencies in the environmental and shipping databases that are used as inputs for the noise models and of limitations in the noise models themselves.

Fundamental research on underwater acoustics, on marine animals, on their habitats and habits,

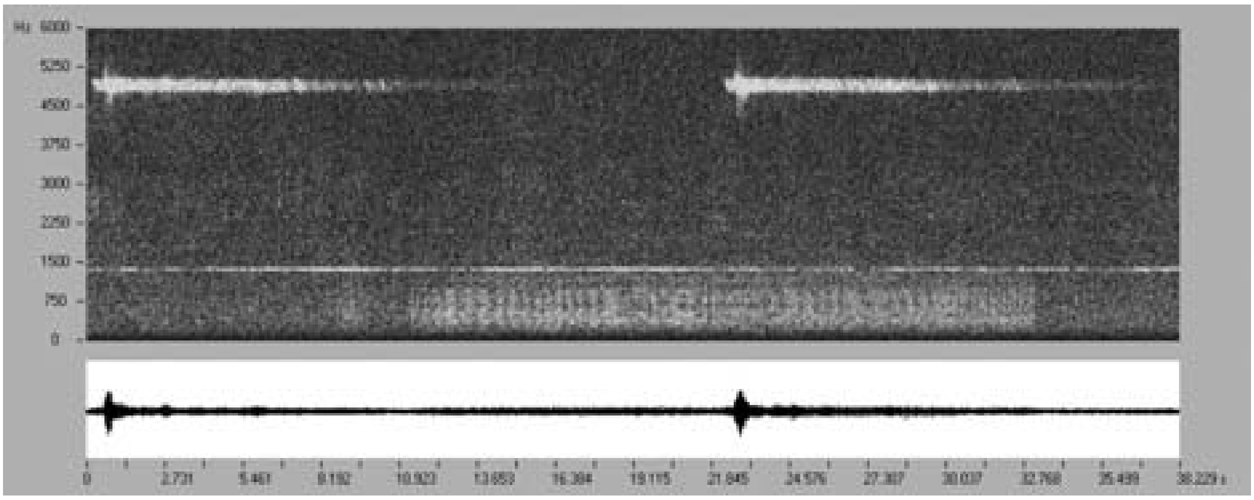


Figure 1.1—Spectrogram showing noises from three different man-made sources recorded in the Ligurian Sea: impulse noises (below 1kHz) from a jack hammer used for construction in Monaco Harbour (about 40km away), a fast ferry passing by (the line at 1.4kHz), and a sonar operating from an unknown location (at 5kHz). (Source: CIBRA; For the color version of this figure please refer to the Annex at the end of the document)

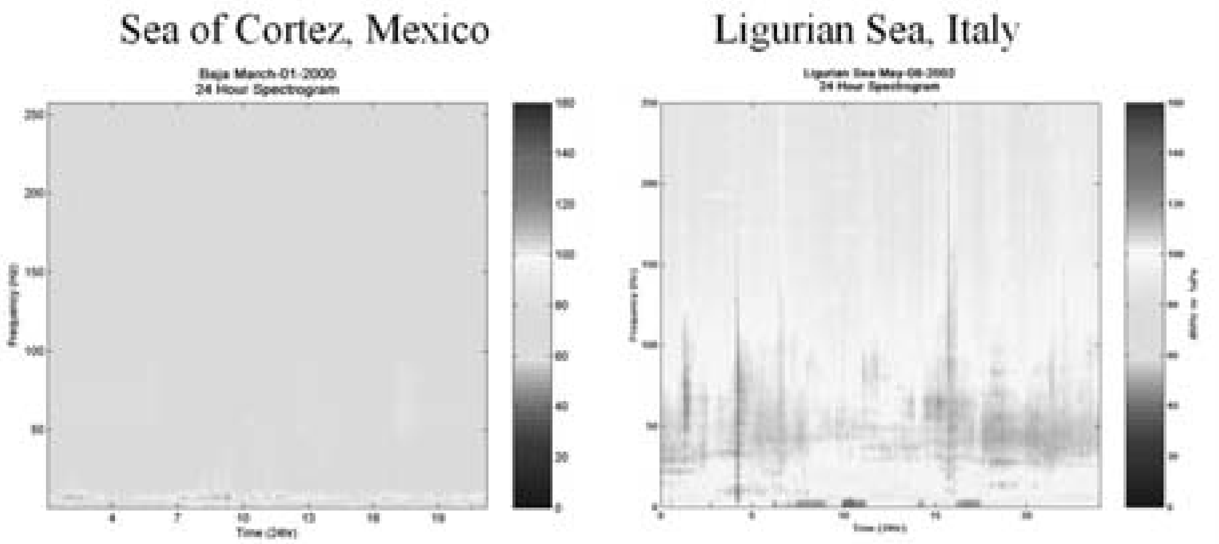


Figure 1.2—24-hour noise map of the Ligurian Sea compared with the Sea of Cortez (Mexico): the noise level in the 0-250Hz band is up to 40dB higher in the Ligurian Sea. Courtesy of C. Clark (Cornell University, USA; For the color version of this figure please refer to the Annex at the end of the document)

and on the biology of disturbance is thus needed to address this very complex issue.

1.6. The Mediterranean Sea case

The Mediterranean Sea in general and the Ligurian Sea in particular are severely affected by many different sources of man-made noise (Figures 1.1 and 1.2); nevertheless, few scientific papers dealing with noise and little basic information on the main sources of noise are available for the purposes of setting up noise management strategies.

The most important sources of anthropogenic noise in the Mediterranean are maritime traffic, seismic surveys, military sonar, drilling operations, coastal construction works and underwater explosions originating from military exercises.

Despite its small area (0.8% of the world's sea area) the Mediterranean Sea probably suffers from heavier maritime traffic than any other sea in the world. According to Frantzis and Notarbartolo di Sciara (2007), about 220,000 vessels larger than 100 tonnes cross the Mediterranean each year. Ten years ago the region's maritime traffic volume was estimated at 30% of the world's total merchant shipping and 20% of its oil shipping. Although most of the traffic is along an east–west axis, there is a complex web of lanes in some areas, including important marine mammal habitats. The total number of large cargo vessels crossing the Mediterranean Sea at any given moment exceeds 2000, indicating that silent areas may no longer exist in the basin. Dobler (2002) provides a more detailed analysis of maritime traffic in the Mediterranean Sea.

The volume of shipping in the Mediterranean Sea results in high background noise levels (Figure 1.2) that are likely to make it harder for whales to communicate with each other or to receive acoustic cues, for example to detect approaching vessels or other hazards. The short- and long-term impacts of this are difficult to evaluate but, despite some controversial facts, there appears to be a link between noise and collisions. Collisions may be related to a number of factors: (a) high-density maritime traffic, (b) increased masking ambient noise, (c) possible hearing impairment in whales, due to long-term exposure to unnaturally high noise levels, (d) the whales' inability to avoid the collision area because of the high density of shipping noise all around.

1.7. Reducing the risk to marine mammals

We know that anthropogenic sound in the ocean is a serious threat, although we do not yet have sufficient information to understand the full extent of the problem. One of the biggest challenges we face in regulating the effects of noise is our ignorance of the characteristics and levels of exposure that may pose risks to marine mammals and fish, particularly in the long term and when multiple exposures act together.

Given the current state of our knowledge it is essential to take a precautionary approach to noise regulation. Efforts to protect and preserve marine mammals must be expanded by instituting and using effective mitigation measures, such as geographic exclusion zones, to keep them at a distance from noise sources that have the potential to harm or kill them.

Because the occurrence and use of sources of potentially harmful anthropogenic noise are likely to increase in the coming years and new sound sources are continually being introduced, the question of how to mitigate their harmful effects is pressing. Acoustic Risk Mitigation procedures have been or are currently being developed by navies, governments and commercial companies. Generally these are concerned with avoiding the exposure of animals to sound pressures that might directly damage their hearing systems or cause other types of physical damage that could lead to impairment of vital functions or to death, or that might disrupt their behaviour and thereby threaten their survival.

Marine mammals are difficult animals to study in the wild and relatively little effort has been directed towards understanding this problem. Consequently large data gaps exist in relation to both marine mammal populations and the effects of noise. This combination results in substantial uncertainty regarding the effects of noise on marine mammal populations, especially in the long term.

Fundamental research on marine mammal acoustics, on their habitats and habits, as well as on their prey, is thus needed to address this very complex subject and to introduce appropriate protection policies and mitigation measures. Very similar considerations also apply to fish. Monitoring of ships' underwater noise is needed, for example, to model noise diffusion and the impact on the underwater environment. Ship noise impact can be diminished by reducing the noise emitted by engines and propellers, and by modifying ship tracks to avoid sensitive areas such as breeding grounds, feeding grounds and migratory corridors.

Acoustic impacts on the marine environment need to be addressed through a comprehensive and transparent management and regulatory system (McCarthy, 2004). This should address chronic and acute anthropogenic noise, long-term and short-term effects, cumulative and synergistic effects (Figure 1.1), and impacts on individuals and populations. The regulatory system should be part of a strategy based on prevention and the precautionary principle. The implementation of such a system will require a series of steps and synergistic actions to promote education, awareness and research. Much effort should be devoted to developing a legal framework within which underwater noise is recognised and regulated as a real threat.

In this context, the creation of SACs and MPAs that take noise pollution into account should ensure the protection of critical and productive habitats, and vulnerable and endangered species in particular. The designation of SACs and MPAs can be used to protect marine mammals and their habitats from environmental stressors, including the cumulative and synergistic effects of noise. In these areas, noise levels should not be allowed to exceed ambient levels by more than a given value, including noise from sources located outside the MPA that crosses the MPA boundaries. This would require additional research to establish baseline noise data and evaluate acceptable thresholds for noise levels, that is, levels that can be tolerated without any significant negative effect.

In other words, in addition to defining which impacts should be avoided or mitigated, we also need to draw up a model of 'acoustic comfort' that we should guarantee to animals, at least over sufficiently extensive protected areas. This is a novel concept. It means we should define the (near to) zero-impact noise level that a habitat should have for each type of marine life.

1.8. Regulating shipping noise

Reduction of shipping noise is a world-wide problem closely connected to the general problem of the impact of underwater noise on marine life (Richardson *et al*., 1995; Gisiner, 1998; NRC, 2000, 2003; Tyack, 2003; McCarthy, 2004; Merrill, 2004; Popper *et al*., 2004; Southall, 2005; Vos & Reeves, 2005; Weilgart, 2006; Nowacek *et al*., 2007, and many others). This issue was discussed at the international symposium 'Shipping Noise and Marine Mammals: A Forum for Science, Management, and Technology,' in 2004; the final report (Southall, 2005) made several recommendations, including raising awareness within the shipping industry concerning marine noise issues, creating alliances across various stakeholder groups, and engaging the industry and other maritime industries in the development of creative and practical solutions to minimize vessel noise. In 2007 the NOAA organized the symposium 'Potential Application of Vessel-Quieting Technology on Large Commercial Vessels' to further explore the problem, in particular to examine the economic and practical issues in the extensive application of those noise reduction solutions already applied to military and research vessels (Mitson, 1995; NOAA, 2007).

To address the problem of increased ambient noise due to shipping, governments and stakeholders should promote the introduction of ship-quieting technologies, such as those reviewed in the NOAA symposium. Until new classes of quiet ship come into operation, alternative measures should be adopted to reduce noise exposure at least in critical areas.

The International Maritime Organization (IMO) should adjust routes, merge existing routes and/or create new routing measures or speed restrictions to minimize exposure of marine mammals sensitive to noise and preserve critical habitats from commercial shipping and other large ocean-going vessel traffic. This approach has been applied in the US Exclusive Economic Zone (EEZ) (shifting of traffic lanes in Massachusetts Bay relative to the distribution of endangered western North Atlantic baleen whale populations), in the Canadian EEZ (mainly shifting of traffic routes relative to the

North Atlantic right whale (*Eubalaena glacialis*) population in the Bay of Fundy), and within the Straits of Gibraltar and the neighbouring Spanish Alboran Sea (shifting of traffic routes and speed restrictions relative to the sperm whale (*Physeter macrocephalus*) population west of the Strait) (Agardy *et al*. 2007).

In areas of heavy traffic and sensitive marine mammal populations, regulatory authorities, marine mammal scientists, shipping industry representatives and NGOs should initiate dialogues in order to identify possible measures and suggest re-routings and/or consolidations that would balance the needs of species protection (from noise and collisions, as well as chemical pollution and overfishing) and commercial needs. This would require further research into the appropriate placement of shipping routes, the evaluation of generated noise fields, and the implementation of basin-wide monitoring networks to control noise levels, especially in critical habitats and where quieting measures are being applied. The Pelagos Sanctuary in the Central Mediterranan Sea could be a laboratory where new rules to balance human activities and nature conservation could be tried out; to date, however, the noise issue has scarcely been considered.

Priority actions to reduce the impact of shipping noise include:

- **Reducing noise**

  a) reducing noise radiated by existing ships and boats by encouraging good maintenance of engines and propellers;

  b) adopting quieting technologies in the design of new ships and boats;

  c) encouraging speed restrictions and alternative routes to avoid sensitive habitats, including marine mammals' key habitats and marine protected areas; defining appropriate buffer zones around them; and considering the impact of long-range sound propagation.

- **Improving research**

  a) developing models of the generated sound field in relation to oceanographic features (depth/temperature profile, sound channels, water depth and seafloor characteristics);

  b) using models to produce predictive maps of noise and to simulate impacts and mitigation measures;

  c) considering cumulative impacts over time and effects modelling; including consideration of seasonal and historical impacts from other activities (shipping, military, industrial and other seismic activities) on marine mammal populations;

  d) determining safe and harmful exposure levels for all zoological groups (e.g. mysticetes, odontocetes, pinnipeds, marine turtles, fish, invertebrates) and for critical species (e.g. beaked whales).

1.9. Legal instruments for management

In general, underwater noise should be expressly classified by states as a pollutant (where it is not already so defined) and managed accordingly.

In the absence of specific laws, and given the fact that underwater noise is a transboundary pollutant, the European Union Habitats Directive (European Economic Community, 1992) is a possible framework for developing a regulatory system for Mediterranean waters. The system would comply with the opinions expressed by ACCOBAMS (2004a, 2004b, 2006, 2007), the International Whaling Commission Scientific Committee (IWC, 2004, 2006), the European Parliament (2004) and the United Nations Convention on the Law of the Sea (UNCLOS), which consider underwater noise a type of pollution of the marine environment.

The EU Habitats Directive states that it is not permissible to deliberately disturb in the wild any creature which is listed in Annex IV(a), which includes all cetaceans and several other marine mammals. In addition to species protection, the Habitats Directive also makes provision for the site-based protection of a range of marine mammal species listed in Annex II. However, the Directive does not cite noise explicitly (Pavan, 2007). With few exceptions relating to high-power acoustic sources (explosions, sonars and seismic surveys) that may have a direct immediate impact on marine mammals, underwater noise is largely unregulated (McCarthy, 2004).

ACCOBAMS Resolution 3.10 (ACCOBAMS, 2007), based on the document prepared by Pavan (2006), presses all the parties to take noise into full consideration and to consider underwater noise levels a quality parameter in habitat assessments, zoning and managing marine areas of special interest. This parameter should also be considered a priority in the protection of critical habitats and areas where noise might affect essential behaviour of marine mammals, such as feeding, reproduction and nursing.

The EU Recreational Craft Directive (European Community, 2003) requires compliance with specific sound emission levels in air, but there is no mention of noise emitted underwater. New directives are required to force maritime industries to take into account the noise emitted underwater as well. An example is set by the ICES recommendation about the noise generated by research vessels, which can introduce a bias into fish abundance estimates (Mitson, 1995; Mitson & Knudsen, 2003).

To preserve the quality of the underwater environment, specific underwater noise emission limits must be introduced for new ships and boats, analogous to the limits already imposed on motor vehicles on land. As emitted noise and vibration often mean a loss of energy and mechanical problems, it might be possible to establish cooperation with shipbuilding industries for the design of quieter and more energy-efficient ships that in the long term will be more economical and more environmentally compliant. In this respect, Members of the IMO should propose an amendment to MARPOL (the International Convention for the Prevention of Pollution from Ships) to include ‘energy’ in its definition of pollution, consistent with Article 1(1)(4) of the UN Convention on the Law of the Sea (UNCLOS).

More immediately, the IMO, as the only organization competent to regulate international shipping, should consider possible options for reducing the impact of ship-generated noise on marine life, such as building upon its Guidelines for the Identification and Designation of Particularly Sensitive Sea Areas (PSSAs), (Paragraph 2.2 of Resolution A. 982(24)), which identify shipping noise as a marine pollutant. This could be done by appropriately using existing navigational measures (such as planning traffic lanes) or developing new ship quieting requirements (Agardy *et al*., 2007).

Existing regional agreements in the Mediterranean region include ACCOBAMS (Agreement on the Conservation of Cetaceans in the Black Sea, Mediterranean Sea and Contiguous Atlantic Area) and the SPA protocol (Specially Protected Areas Protocol of the Barcelona Convention for the protection of the Mediterranean Sea). These should extend their competence over noise and become an effective means of identifying and designating noise-related issues. They should act as interfaces among interested parties (ACCOBAMS 2004a, 2004b, 2006, 2007).

1.10. Noise monitoring programmes and research needs

In order to determine current noise levels in the oceans, set a baseline for future observations and correctly address and monitor noise-control strategies, a widespread network of sensors must be set up to measure underwater noise and its short- and long-term variations. A long-term observation programme is needed to assess any trend in the levels and spatial distribution of background noise.

Priorities should be established in setting up the monitoring network, taking into account existing information about known critical habitats as well as habitat databases, where available. Monitored areas must be carefully chosen to correctly represent both low-noise habitats, for example those that are far from commercial shipping lanes or noisy coasts, and high-noise areas close to shipping lanes and port facilities.

It is then important to separate short-term variations, due to the local or temporary presence of marine mammals or anthropogenic noise sources, from long-term variations. The latter may be due to seasonal changes in oceanographic parameters that may influence long-term propagation, and possibly also to variations in distant shipping traffic (in the number, frequency, type and tonnage of ships).

The monitoring network can also provide data about the presence of marine mammals based on their acoustic signatures; sensors located in specific areas will provide information for marine mammal databases.

The development of a monitoring network could be extremely demanding and expensive, though it might be possible to establish cooperation for interdisciplinary and joint activities, for example by using underwater sensors deployed for other needs such as military and geophysical monitoring. In many cases the bandwidth or the recording capabilities of those installations are not suitable for continuous monitoring of marine mammals, but they could be suitable for sampling ocean noise or for monitoring low-frequency whale calls, as with the US Navy ‘Dual Use’ programme in the Atlantic Ocean.

By establishing cooperation with all parties involved in the study of the marine environment, joint programmes could be developed to provide the data needed. With the rapidly evolving technologies now available, monitoring networks can be set up to serve a wide scientific community. Within the INFN NEMO ONDE (Italian National Institute of Nuclear Physics, Neutrino Mediterranean Observatory, Ocean Noise Detection Experiment) project, for example, a deep underwater station with four wideband hydrophones has been used to collect wideband acoustic data for both noise and biological monitoring (Riccobene *et al*., 2004, 2007).

A network of similar installations is planned for geophysical studies in the Mediterranean Sea; based on the results of the NEMO ONDE project (Pavan *et al*., 2007; Riccobene *et al*., 2007), a new project named LIDO (Listening Into Deep Ocean) has been designed by INGV (Italian National Institute of Geophysics and Volcanology), INFN and CIBRA (Interdisciplinary Centre for Bioacoustics and Environmental Research, University of Pavia) to deploy wideband acoustic monitoring platforms and to improve ESONET (European Seafloor Observatory Network) platforms with wideband sensors. Such a network could be a valuable tool for a long-term research programme on underwater noise and for the monitoring of marine mammals. Based on data provided by such a network, propagation models could be developed to predict the communication range for each marine mammal species and the effect of changes in noise levels and characteristics.

One of the most important goals of a noise monitoring programme will be to determine the spatial and temporal extent of the acoustic energy emitted by all the different noise sources in the Mediterranean, to support the development of predictive noise maps. This will also allow the construction of a noise ‘budget’, determining how much of the noise in the sea is caused by each of these different human activities. A similar project, named ESME (Effects of Sound on the Marine Environment), is being developed by the US Navy and the US Office of Naval Research.

Basic and applied research is required in several areas covering both the biological and the ecological aspects to understand more about the long-term and cumulative effects of sound on marine animals. Technological research is also needed to develop quieter ships and to exploit marine resources in a more balanced fashion. It is also important to develop more effective ways to monitor the presence and behaviour of animals as part of current noise control measures, so that the resulting data can be used to evaluate impacts and the effectiveness of mitigation.

Biological and behavioural research is needed to study how noise interferes with the sensory systems of all marine organisms, even in those species where hearing and acoustic communication appears to be confined to short ranges. Controlled exposure experiments in the field, at present conducted only on a few species of marine mammals, can produce important information about observable behavioural changes (Johnson & Tyack, 2003; Tyack, 2003) driven by exposure to controlled noises. New laboratory tools are also being developed to investigate changes in physiological processes that rarely turn into measurable effects. For example, auditory brainstem response (ABR) techniques can make an important contribution to knowledge of hearing sensitivity and threshold changes due to noise exposure (Scholik & Yan, 2002; Popper *et al*., 2005), and analysis of the levels of the stress-related hormone cortisol may reveal subtle effects of noise exposure (Wysocky *et al*., 2006) that may lead to significant diseases in the long term.

Dedicated funding, possibly based on the 'polluter pays' principle, is required to support research, management and conservation issues, as well as to continuously refine and update noise control rules and tools.

Suggested priority actions include:

a) creating or improving regional and worldwide databases to model and predict marine animals' presence, distribution and density, as well as to map and model seasonal movements and seasonal specific behaviours;

b) improving marine mammal detection tools (passive acoustics, visual detection and new technologies such as active acoustics, infrared and radar detection) to be used for (i) creating distribution databases, (ii) assessing marine mammal presence, distribution and density in sensitive areas, and (iii) detecting and monitoring marine mammal activity and movements;

c) reviewing and evaluating the available information on the impacts of human-generated sound on marine mammals at the individual and population level and on other components of the marine environment, including the prey field; developing research to fill the knowledge gaps;

d) expanding research on the effects of noise on fish and invertebrates;

e) developing new tools to investigate physiological processes and behavioural responses related to noise exposure;

f) investigating acoustic exposure criteria by taking into account signal duration and repetition, energy, frequency, directionality, bandwidth and their relationship to physical, physiological and behavioural effects on vertebrates and invertebrates;

g) developing a 'noise budget' model in which the synergistic and cumulative effects of multiple exposures are taken into consideration;

h) developing an 'acoustic comfort' model for each zoological group, to define a range of noise levels above the natural background that can be tolerated with negligible effects;

i) developing a network of underwater noise monitoring stations to collect baseline noise data, to keep track of changes in underwater noise levels, to monitor cetacean presence and transits, and to monitor for unusual events;

j) developing databases of noise sources and models to produce predictive noise propagation maps to be used for evaluating the impact of new noise sources and the effects of mitigation measures;

k) setting up specialist research teams to examine noise-related problems.

1.11. Summary

Although we know that anthropogenic sound in the ocean is a serious threat, we do not have sufficient information at this time to understand the full extent of the problem. One of the biggest challenges faced in regulating the effects of noise is our ignorance of the characteristics and levels of sound exposure that may pose risks to marine animals in the long term. Given the current state of our knowledge we must therefore take a precautionary approach to the regulation of noise.

We must also expand our efforts to protect and preserve marine life by instituting and using effective mitigation measures—such as geographic noise-exclusion zones—to keep marine animals at a distance from noise sources that have the potential to harm or kill them.

While most interest in the effects of anthropogenic noise has focused on marine mammals (mainly cetaceans and pinnipeds) and a few other vertebrates (sea turtles), there is increasing evidence for the impact of such noise on fish and marine invertebrates. This issue will need further research, which should also take into consideration the ecological effects on the whole food web and on fisheries. In particular, research is needed to better understand the acoustic-mediated effects of noise on the behaviour and biology of all marine creatures.

Acoustic impacts on the marine environment need to be addressed through a comprehensive and transparent research, management and regulatory system that includes all sources of noise, whether continuous and ubiquitous (such as shipping) or localized in space and time (sonars, seismic surveys, offshore and coastal construction works, scientific experiments, etc.). This system should address chronic and acute anthropogenic noise, long-term and short-term effects, cumulative and synergistic effects, and impacts on individuals and populations.

2. Direct physical effects of marine vessels on benthic habitats and species

2.1. Introduction

Marine vessels have several effects on marine habitats and species, several of which constitute adverse impacts. The most documented adverse effects are those resulting from pollution, especially by petroleum hydrocarbons and other chemicals that originally constitute cargo but end up in the sea following collisions, groundings or other accidents. Large-scale pollution events, such as those that have resulted from accidents involving the *Amoco Cadiz*, *Erika*, *Prestige* and *Torrey Canyon* have received worldwide attention and notoriety, while a multitude of studies on the effects of such events on the marine environment have been undertaken (e.g. Labarta *et al*., 2005; Davoodi & Claireaux, 2007). Discharges resulting from operational activities also lead to adverse effects on marine habitats and species, although such pollution events are less often documented. Data are also available for other effects of ship activities on marine habitats (such as seagrass beds and coral reefs) and species, namely the introduction of exotic species (e.g. Gollash, 2002; Niimi, 2004) and physical damage resulting from propeller scarring (e.g. Orth *et al*., 2001; Kenworthy *et al*., 2002), anchoring (e.g. Walker *et al*., 1989; Creed & Amado Filho, 1999; Rogers & Garrison, 2001) and groundings (e.g. Hudson & Goodwin, 2001; Whitfield *et al*., 2002; Olesen *et al*., 2004).

In the Mediterranean region, the best documented adverse effects of vessel activity on the marine environment are those resulting from pollution by petroleum hydrocarbons, antifouling biocides, litter, noise pollution, and introduction of alien species (see review by Galil, 2006). However, insofar as Mediterranean marine habitats and biota are concerned, there appears to be a general lack of information on direct physical effects of vessels, namely anchoring, abrasion by ship hulls in shallow waters, propeller scarring, groundings, disturbance of soft sediment bottoms during navigation, and shading.

2.2. Anchoring

A number of studies have dealt specifically with the effects of recreational boat anchoring on *Posidonia oceanica* seagrass beds (Francour *et al*., 1999; Pasqualini *et al*., 1999; Milazzo *et al*., 2004; Ganteaume *et al*., 2005; Montefalcone *et al*., 2006). Data from these studies indicate that boat anchors lowered onto seagrass beds (Figure 1.3) damage the habitat by uprooting plants, leading to reduced shoot density and bed cover (Garcia-Charton *et al*., 1993; Francour *et al*., 1999). For example, results from a side scan sonar survey undertaken by Pasqualini *et al*. (1999) in the vicinity of Bonifacio harbour in Corsica during 1995–1996 indicated degradation of *P. oceanica* beds within an area representing 33% of the total area surveyed, which was attributed to anchoring by pleasure boats. It is not uncommon to observe large chunks of *P. oceanica* matte on the seabed that have been completely detached from a seagrass bed by anchoring (Figure 1.4), and which end up being washed ashore or transported by currents and water movement to deeper parts where environmental conditions are not suitable for survival of the plants. Eventually, the seagrass shoots on such detached chunks of matte die, leaving slowly decomposing masses of seagrass root-rhizome material. The available data also indicate that anchor chains have adverse effects on seagrass beds (Montefalcone *et al*., 2006). The type and magnitude of alterations to seagrass habitat resulting from anchoring depend on the dimensions and type of the anchor used and on chain size and length, which in turn depend on the size of the vessel (Milazzo *et al*., 2004; Montefalcone *et al*., 2006). Crabbing, which refers to sideways movement of the anchor and chain or rope due to movement of the vessel in response to currents and wind, exacerbates the effect since a larger area of the benthic habitat is affected.

Anchoring on rocky bottoms poses a threat to assemblages of infralittoral algae and sensitive

Figure 1.3—A 'CQR' anchor on a Posidonia oceanica bed at a depth of 10m. (For the color version of this figure please refer to the Annex at the end of the document)

Figure 1.4—A chunk of Posidonia oceanica matte lying on a sandy bottom at a distance of several hundred meters from the seagrass bed from which it was detached. The striped rod visible in the photo is 30cm long. (For the color version of this figure please refer to the Annex at the end of the document)

species that are associated with such habitat types, including algae belonging to the genus *Cystoseira* and the stone coral *Cladocora caespitosa*. In deeper waters, anchoring may have an adverse impact on sensitive circalittoral benthic habitats, including coralligenous assemblages and maerl beds (Table 1.1). As in the case of seagrass beds, besides obliteration of flora and fauna through direct physical damage, anchoring on infralittoral and circalittoral habitats affects the associated fauna, particularly sessile species, through alteration of habitat structure, reduced primary production and changes to trophic relationships (García Charton *et al*., 2000).

Table 1.1—Mediterranean key species and habitats that are susceptible to disturbance by anchoring (adapted from Milazzo et al., 2002).

Key species	**Key habitat**
Posidonia oceanica	*Posidonia oceanica* beds
Cymodocea nodosa	*Cymodocea nodosa* beds
Cystoseira spp.	Infralittoral algae
Cladocora caespitosa	Infralittoral algae
Eunicella spp.	Coralligenous
Lophogorgia ceratophyta	Coralligenous
Paramuricaea clavata	Coralligenous
Pentapora facialis	Coralligenous
Lithothamnion corallioides	Maerl beds
Phymatolithon calcareum	Maerl beds

Unfortunately, published data on the effects of anchoring by large commercial vessels are completely lacking. However, one would assume that anchoring by commercial ships would have a much larger adverse impact on benthic habitats and species, given the relatively larger anchors and heavier chains used by such vessels. Furthermore, the magnitude of adverse impact will be greater in areas that are designated as anchoring grounds, such as bunkering areas, and ports and harbours.

Adverse impacts of anchoring on marine benthic habitats and species can be reduced or eliminated altogether by adopting measures that include: (a) restricting the activity to designated areas; (b) prohibiting it where sensitive habitats are present; (c) providing 'permanent' boat moorings to vessels as an alternative to anchoring; and (d) educating skippers on the potential negative effects of anchoring. Such measures have been adopted in several marine protected areas in the Mediterranean, such as Port Cros, France (Francour *et al*., 1999). More recently, guideline documents for the whole Mediterranean region have been made available (Francour *et al*., 2006).

2.3. Abrasion by hulls, propeller scarring and groundings

Physical adverse effects on benthic habitats and species resulting from abrasion by ship hulls and propeller scarring are mainly restricted to shallow-water areas, namely shoals, the inner reaches of harbours, bays and inlets, and navigation canals and straits. However, data on such effects are completely lacking for the Mediterranean Sea. Apart from spillage of cargo that may have adverse impacts on the marine environment, which will not be dealt with here, vessel groundings result in direct physical damage to benthic habitats and biota. Again, published data on the effects of vessel groundings on the benthos are unavailable, although Oral and Öztürk (2006) emphasise that the adverse impacts of grounding on seagrass habitat and mussel beds in the Turkish straits system are considerable. Groundings appear to make up a considerable percentage of ship accidents in the Mediterranean Sea. For example, Öztürk and Öztürk (1996) indicate that 30.9% of the ship accidents that occurred in the Bosphorus during the period from May 1982 to April 1992 comprised groundings.

2.4. Other direct physical effects

Ships navigating in shallow water areas, such as embayments, canals, straits and the inner reaches of harbours and ports, tend to stir up sediments from soft bottoms (Figure 1.5). While the finer sediment fraction may remain suspended in the water column for some time, the coarser fraction settles rapidly and is dispersed over a wide area of the seabed, smothering benthic habitats and biota in the process. Certain habitats, such as algal forests, coral banks and seagrass beds, are particularly sensitive to such disturbance, as this leads to alteration in the physico-chemical characteristics of the water column and, ultimately, to potential adverse impacts. For example, the capacity of seagrasses to store carbohydrate reserves in their rhizomes allows them to withstand transient periods of reduced light availability, such as those resulting from increased turbidity of the water column. However, over a long period characterized by frequent episodes of reduced light availability, as would occur when soft sediments are disturbed and the fine fraction is suspended in the water column, death of the plant eventually ensues (Gordon *et al*., 1994, Onuf, 1996). Disturbance of sediments also releases nutrients into the water column, leading to increased phytoplankton populations and excessive epiphyte loading on seagrass leaves. Both of these effects reduce the amount of light reaching the plant's photosynthetic tissue (Silberstein *et al*., 1986; Buzzelli & Meyers, 1998), leading to impaired growth and ultimately (if the turbid water conditions and epiphyte loading persist) to death of the benthic vegetation (Hemminga, 1998; den Hartog & Phillips, 2001).

Other, often overlooked effects of marine vessels are those caused by shading when ships spend long periods moored inside ports and harbours. Whatever the reason for the immobility (usually long-term repairs and scrapping), long-term shading by stationary ships results in adverse effects on the benthic biota present underneath the vessels. Again, data on the effects of long-term shading by ships on benthic habitats and biota are completely lacking. However, Struck *et al*. (2004) report significantly lower abundance and diversity of estuarine macroinvertebrates present in areas below low bridges, compared to reference sites without bridges, and attribute the difference to diminished above- and below-ground macrophyte biomass due to the shading. The effects of long-term shading by stationery ships on seagrass present below the vessels would be similar to those resulting from reduced availability of light as described above. In extreme but frequent cases, ships moored for very long periods inside ports and harbours may eventually sink, leading to obliteration of benthic biota present within the footprint occupied by the sunken vessel, and to adverse effects to habitats and biota located in its vicinity.

Figure 1.5—Turbidity (red ellipse) resulting from suspension of soft sediments by a departing merchant vessel navigating in a Maltese harbour. The sediment plume was still visible several hours after the vessel had left the harbour. (Source: Google Earth; For the color version of this figure please refer to the Annex at the end of the document)

2.5. Conclusions and recommendations for future research and management

There is clearly a lack of data on direct physical effects of vessels on Mediterranean benthic habitats and species. Published studies deal almost exclusively with the effects of recreational boat anchoring on seagrass (*Posidonia oceanica*) meadows, while there is a dearth of information on the effects of anchoring by commercial vessels in the vicinity of deep-water habitats, and on other physical effects resulting from recreational boating and commercial shipping. There is, therefore, an urgent need to acquire data that are necessary to (a) understand the magnitude and extent of adverse impacts resulting from direct physical effects, where knowledge is lacking; (b) develop the necessary technical and procedural strategies and management guidelines for shipping activities to eliminate, or at least minimize, adverse impacts; (c) implement habitat restoration programmes in areas that have been affected adversely. Since data are currently only available for the effects of anchoring on seagrass beds, management guidelines can only be drawn up at present for this activity. In this respect, the recommendations made by some workers (e.g. Francour *et al*., 2006) and the initiatives taken in some countries (e.g. France) concerning anchoring in protected and sensitive sites should be noted and transposed to other areas of the Mediterranean. In France, permanent moorings have been introduced in marine protected areas (MPAs) to avoid anchoring, while different areas within such MPAs have been designated where anchoring is controlled or prohibited altogether. On the other hand, management guidelines for other maritime activities that may have adverse effects—anchoring in other habitats, abrasion by hulls, propeller scarring, groundings, disturbance of sediments by ships during navigation in shallow waters, and long-term shading by vessels—can only be formulated once the necessary data have been acquired. Research should be primarily focused in areas which support sensitive habitats and species that are susceptible to direct physical effects of marine vessels. Such areas include pristine sites where anchoring by commercial vessels (e.g. cruise liners) takes place, harbours, popular bays and inlets, straits and canals, and bunkering areas and other anchoring grounds, including those located in deep waters that are known to support marine benthic assemblages of high ecological and conservation importance.

3 Shipping-derived antifouling biocides in the Mediterranean Sea

3.1. Introduction

Antifouling paints are critical for shipping and have been used to improve speed and fuel economy, reduce dry-docking expenses, and prevent the translocation or introduction of the fouling biota into new areas. Organotin-based antifouling paints were introduced in the mid-1960s and their use increased unchecked for two decades (Hoch, 2001). The deleterious impacts of organotin contamination were first noticed in the late 1970s when reproductive failure and shell deformations affected shellfish farms on the Atlantic coast of France (Alzieu *et al*., 1980). Since then, tributyltin (TBT) and its degradation products, mono- (MBT) and dibutyltin (DBT), and triphenyltin (TPT) have been recognised as the most toxic materials intentionally introduced into the sea and confirmed as harming a wide range of organisms: their ecotoxicological impacts have been amply documented. Studies have revealed that organotin compounds degrade slowly: TBT half-life is estimated at 1–3 weeks in shelf seawater (Seligman *et al*., 1986, 1988), and at 1–5 years in the sediment (Adelman *et al*., 1990), and is predicated on microbial degradation, ultraviolet photolysis and temperature. However, it is possible that half-life for TBT in the open sea is considerably longer, at least for the oligotrophic waters of the Mediterranean with their low kinetic biodegradation (Michel & Averty, 1999), and in deep sediments it may be 87±17 years (Viglino *et al*., 2004). The environmental problems associated with TBT have led to its imminent ban (*see below*) and replacement with alternative antifouling coatings.

3.2. Distribution and accumulation of biocidal antifoulants

The first coordinated survey in the Mediterranean Sea of TBT and its degradation derivatives was conducted in 1988. One hundred and thirteen water samples were collected along the French Mediterranean coast, the Tyrrhenian coast of Italy, the southern coast of Turkey and in Alexandria (Egypt). At most sites examined the concentrations of TBT in seawater exceeded $20 ng \cdot l^{-1}$. The harbours of Mersin (Turkey) ($936 ng \cdot l^{-1}$), and Livorno (Italy) ($810 ng \cdot l^{-1}$) displayed the highest levels of contamination among the sampled harbours, but TBT levels inside recreational marinas generally exceeded contamination levels at commercial shipping ports, with particularly high levels at Cecina and Punta Ala (Italy) and the Vieux Port of Marseille (France) (3,930, 960 and $736 ng \cdot l^{-1}$, respectively). All the sediment samples from Alexandria contained TBT; highest concentrations were detected in the western and eastern harbours and in the Bay of Abu Kir (975, 260 and $252 ng \cdot l^{-1}$, respectively) (Gabrielides *et al*., 1990). Subsurface water samples were taken that same year at several additional locations in the western Mediterranean: the Ebro delta, the port of Barcelona and El Masnou marina, all on the Spanish Mediterranean coast; along the Midi coast of France; and along the French and Italian rivieras. Substantial contamination was reported for the entire region, with elevated levels of TBT in all samples, the highest records occurring in Toulon harbour and the Beaulieu and San Remo marinas (Alzieu *et al*., 1991).

Regulations concerning the use of organotin-based antifouling paints in the Mediterranean Sea were introduced in 1991 (*see below*, Figure 1.6). Yet organotin compounds were detected in all subsurface water samples taken in 1995 from ports and marinas along the Côte d'Azur, with high concentrations persisting in the ports of Antibes, Golfe Juan, Cannes and Nice (459, 348, 142 and $138 ng \cdot l^{-1}$, respectively), though levels in recreational marinas were substantially lower than those recorded in 1988 (Tolosa *et al*., 1996). Measurable levels of TBT and DBT were found in samples taken from the bathing beaches of Eze, Nice, Cannes and Villefranche (<0.6–$5.2 ng \cdot l^{-1}$) (Tolosa *et al*., 1996). Subsurface water samples taken in 1999 at 14 sites along the Corsican coast proved that contamination levels in the commercial harbours of Bastia, Porto-Vecchio and Ajaccio (200, 169 and $88 ng \cdot l^{-1}$, respectively), as well as in the marinas of

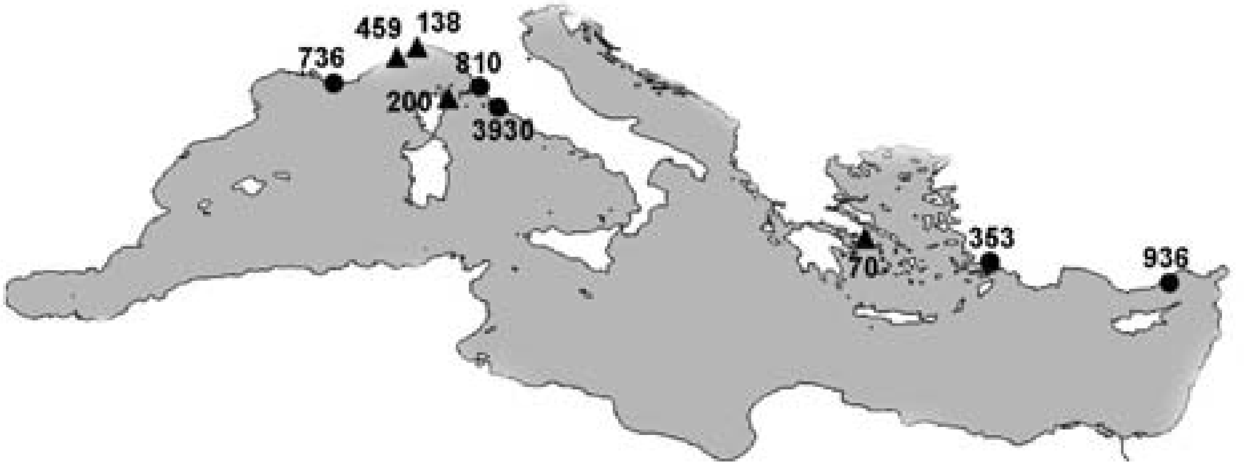


Figure 1.6—Concentrations of TBT in seawater (ng·l-1) before and after the enactment of the ban on organotin-based antifouling paints in the Mediterranean in 1991. Sites sampled in 1988 (●): Mersin port, Turkey; Marmaris port, Turkey; Livorno port, Italy; Cecina marina, Italy; Marseille vieux port (marina), France (Gabrielides et al., 1990). Later samples (▲): Nice, Antibes, France (Tolosa et al., 1996); Bastia port, Corsica (Michel et al., 2001); Piraeus, Greece (Thomaidis et al., 2007; For the color version of this figure please refer to the Annex at the end of the document)

Ajaccio, Porto-Vecchio and Propriano (189, 169 and 161ng·l^{-1}, respectively) were 'quite excessive' (Michel *et al*., 2001). In samples collected in 1999 in the Gulf of Saronikos (Greece), TBT levels reached 70ng^{-1}, and since they were taken in a merchant harbour the contamination was probably due to commercial shipping (Thomaidis *et al*., 2007). More discouraging was the presence of contamination in Corsica in the immediate vicinity of Scandola nature reserve (7.2ng·l^{-1}), and in the Lavezzi Islands nature reserve (2.0ng·l^{-1}), far from maritime shipping, where TBT concentrations of 1-2ng·l^{-1} have been shown to induce deleterious effects (Alzieu, 2000).

Organotin contamination is not limited to port and port-proximate environments. Samples collected in 1998 in the north-western Mediterranean along vertical profiles offshore, between 25m and 2,500m depth, showed that contamination of surface waters was as high as 0.47ng·l^{-1} 20 km offshore and 0.08ng^{-1} midway between Toulon and Corsica; contamination of abyssal water reached a maximum of 0.04ng·l^{-1} at 1,200m (Michel & Averty, 1999). TBT compounds may reach great depth, possibly through the winter cooling and descent of the surface mass, or through chipped and discarded paint fragments (Galil et al., 1995).

Figure 1.7—Concentrations of TBT in sediments (ng·l^{-1} dry weight) after the enactment of the ban on organotin-based antifouling paints in the Mediterranean in 1991. Sites sampled: Barcelona port, Almería port, Sotogrande marina, Sant Carles marina, Spain (Díez et al., 2002); Piraeus, Greece (Tselentis et al., 1999); Haifa port, Ashdod port, Israel (Herut et al., 2004; For the color version of this figure please refer to the Annex at the end of the document)

Deep sea fish collected at depths of 1,000–1,800m in the Gulf of Lion carried as much as 175ng·l^{-1} (wet weight) total butyltin residues in their tissues, comparable to contamination levels in coastal fish collected along the Catalan coast, attesting to exposure of deep sea biota to TBT (Borghi & Porte, 2002).

Sediments from harbours and marinas along the Catalan and Alboran seas were sampled in 1995 and 1999–2000, respectively. The highest TBT concentrations were associated with large-vessel input, as in Barcelona commercial harbour (maximum 18,722, average 4,487ng·l^{-1} dry weight), and Almería commercial harbour (2,135ng·l^{-1} dry weight), although high values were also found in fishing and recreational ports such as the harbour of Sant Carles (maximum 5,226, average 1,617ng·l^{-1} dry weight), and Sotogrande recreational marina (3,868ng·l^{-1} dry weight) (Díez *et al*., 2002). High TBT concentrations, in excess of 10,000ng·l^{-1} dry weight, were also found also in sediment samples taken from Piraeus harbour (Greece) (Tselentis *et al*., 1999). In a study of organotin compounds in the Aegean Sea in 2001–2003 it was found that concentrations in bivalves were higher in summer, '… indicative of a direct continuous exposure to TBT in this area, probably from the increased marine activities during summer and/or desorption from polluted sediments' (Chandrinou *et al*., 2007). The more contaminated of the 12 sites in the Lagoon of Venice examined in 1999–2000 for the presence of organotin compounds in the sediment were those affected by higher boat traffic or boat maintenance activities. High levels of TBT degradation products such as MBT (2,053ng·l^{-1}) near Porto Marghera attest to the occurrence of old butyltin pollution (Bortoli *et al*., 2003). None of the 14 locations sampled along the Israeli coast in 2003 were free of contamination, but the highest concentrations of TBT were recorded in the sediments of the commercial ports of Haifa and Ashdod (770 and 730ng·l^{-1} dry weight, respectively); high levels of contamination (>100ng·l^{-1}) were also detected in the waters of four recreational marinas in addition to Haifa port (Herut *et al*., 2004) (Figure 1.7).

With the restrictions on the use of organotin-based compounds in antifouling paints (see below), they are being replaced by tin-free antifouling paints composed of seawater-soluble matrices containing biologically active compounds, mostly copper and zinc compounds combined with organic booster herbicides such as Irgarol (2-methylthio-4-tertiary-butyl-amino-6-cyclopropylamino-s-triazine), Diuron (3-(3,4-dichlorophenyl)-1,1-dimethylurea), chlorothalonil, dichlofluanid, zinc pyrithione and others (Voulvoulis *et al*., 1999; Evans *et al*., 2000). These paints too may be released directly from the paint surface or accumulate in marine sediments (especially in the vicinity of docks, ports and marinas) in the form of paint fragments which may form important sources of booster biocide contamination (Thomas *et al*., 2002, 2003). Few recent data are available regarding the spread and accumulation of these alternative biocides in the Mediterranean. Substantial levels of Irgarol 1051, the most commonly detected antifouling biocide worldwide (Konstantinou & Albanis, 2004), were present in water samples collected in 1992 and again in 1995 along the French Riviera, with higher concentrations recorded from recreational marinas (Fontvieille, 1,700ng·l^{-1}; St Laurent, 640ng·l^{-1}) than from Antibes harbour (264ng·l^{-1} in 1995) (Readman *et al*., 1993; Tolosa *et al*., 1996), confirming its use primarily on small boats at the time. Irgarol 1051 was the main pollutant (along with another herbicide, Diuron) among the recently introduced antifouling pesticides detected in marinas, fishing ports and harbours along the Mediterranean coast of Spain between 1996 and 2000, with concentrations as high as 330ng·l^{-1} (Martínez *et al*., 2001), 450ng·l^{-1} (Agüera *et al*., 2000) and 1,000ng·l^{-1} (Hernando *et al*., 2001). Similarly, the presence of Irgarol 1051 and two other 'booster biocides' was recently confirmed in sediments collected from Greek harbours and recreational marinas, with the highest concentration (690ng·l^{-1}) in marinas during the summer boating season (Albanis *et al*., 2002). These records suggest that Irgarol has already become a ubiquitous contaminant in areas of high recreational boating activity in the Mediterranean, as commercial shipping has not yet made the transition to organotin substitutes.

3.3. Impact of antifouling biocides on the Mediterranean biota

TBT is '… probably the most toxic substance ever introduced deliberately into the marine environment' (Mee & Fowler, 1991). An effective, long-acting antifoulant, TBT affects non-target biota as well, especially in harbours and marinas with high vessel density and restricted water circulation. Marine molluscs are notably sensitive to the substance, and suffer well-documented sublethal impacts such as the superimposition of male sexual characters in female gonochoristic prosobranch gastropods. This phenomenon, termed ‘imposex’, can result in masculinized females at TBT concentrations as low as $1ng \cdot l^{-1}$ (Smith, 1981). Other morphological changes include shell malformation and reproductive failure in bivalves at concentrations of $20ng \cdot l^{-1}$ (Alzieu, 2000). Imposex has been associated with reduced reproductive potential and altered population structure in several species (Axiak *et al*., 1995, EPA, 2003). In addition, TBT disrupts gastropod reproductive behaviour: according to Straw & Rittschof (2004), at high levels of imposex even the morphologically normal snails were behaviourally and reproductively compromised. As the severity of imposex characteristics in a population has been correlated with concentrations of TBT in the environment and in mollusc tissues, it has served as a widely used and specific biomarker for monitoring TBT contamination. Exposure to TBT may also inhibit growth, impair immune functions and reduce fitness (Leung *et al*., 2006).

In 1988 Martoja & Bouquegneau described a case of ‘pseudohermaphroditism’ in female *Hexaplex trunculus* in Corsica, which they attributed to heavy metals. The first study in the Mediterranean to relate levels of TBT in the sediment to vas deferens and penis development in females of the common muricid gastropod *H. trunculus* was conducted in Malta in 1992 (Axiak *et al*., 1995). All female gastropods collected near major recreational marinas and within the commercial harbours of Marsamxett, Rinella, Marsaxlokk and Marsascala were affected, and the severity of the phenomenon was correlated with the levels of organotins in their digestive glands and gonads, and the amounts of TBT in the superficial sediments. Most females in the highly contaminated harbours exhibited split capsule glands and might have been sterile. Females of *Hexaplex trunculus* sampled in 15 yachting, fishing and commercial harbours along the Italian coast in 1995–96 exhibited nearly 100% sterility in all, except at Linosa and Lampedusa islands, where yachting activity was limited to the summer months; even in heavily impacted populations, however, no evidence was found of any decrease in abundance (Terlizzi *et al*., 1998). Very high levels of imposex were found in Naples harbour (66.7% sterile females, Relative Penis Size Index (RPSI) 77.2, Vas Deferens Sequence Index (VDSI) 4.8). All female *H. trunculus* sampled in 2002 in Laiguelia marina on the Ligurian coast, in Venice outer port, and in Rovinj port (Croatia) showed sign of imposex, though in Italy a partial ban on TBT has been in force for vessels less than 25m in length since 1982. However, ‘differences in the incidence of imposex were detected in relation to the intensity and type of shipping’ (RPSI 12.5, 9.0 and 142.3, respectively; VDSI 4.4, 4.5 and 5.0, respectively) (Garaventa *et al*., 2006). All female *H. trunculus* collected in the canal connecting the lagoon of Bizerte (Tunisia) to the sea showed external male characteristics (Lahbib *et al*., 2004).

Another common muricid, *Bolinus brandaris*, has been used in monitoring TBT along the Catalan coast. At five of the six locations sampled in 1996–97 imposex affected all the female specimens collected (Solé *et al*., 1998), and nearly all the females in the samples collected between 1996 and 2000 displayed advanced imposex characteristics, revealing that ‘frequency and intensity of imposex have increased on the Catalan coast’ (Ramón & Amor, 2001). Similarly, nearly all muricid gastropods off NW Sicily sampled in 1999–2000 were affected, despite low organotin concentration levels in the sediments, save in the marine reserve of Ustica Island where imposex was ‘relatively less severe’ (Chiavarini *et al*., 2003). The authors attributed their results to ‘the existence of a significant number of pleasure crafts. … Evidently, the legislation [forbidding TBT-based antifouling paints] is not rigorously followed” (Chiavarini *et al*., 2003). Imposex was recorded in the *H. trunculus* populations sampled in 13 Italian MPAs in 2002, with a frequency of 100% in eight of the 13 (Terlizzi *et al*., 2004). Imposex was also found offshore at sites with high shipping densities and a year-round vertically mixed water column, which facilitates direct transport of dissolved organotins and organotins adsorbed onto particulate matter to the sea bed (Swennen *et al*., 1997; ten Hallers-Tjabbes *et al*., 1994, 2003). In the eastern Mediterranean paint chips were found in the open sea at depths ranging from 1,017 to 2,411m (Galil *et al*., 1995).

TBT and its degradation products accumulate within tissues of marine organisms and move up the food chain. Very high concentrations have been found in top predators such as the bottlenose dolphin, bluefin tuna and blue shark collected off Italy, with total butyltin in dolphin liver tissues reaching $1,200–2,200ng \cdot g^{-1}$ wet weight (Kannan *et al*., 1996).

Very little work has been performed to assess the accumulation of organic booster biocides (Albanis *et al*., 2002), and there are few published data on the toxicity and possible environmental impacts of many of the tin-free biocidal compounds used in antifoulants. Even so, it is known that Irgarol 1051 inhibits photosynthetic electron transport in chloroplasts, and causes significant growth inhibition of marine algae at concentrations as low as 100ng·l^{-1} (Scarlett *et al*., 1997). It is feared that, if accumulated at high enough levels, it may damage periphyton, algae and seagrasses and thus affect primary productivity (Thomas *et al*., 2001). Manzo *et al*. (2006) found that embryos and sperm of the sea urchin *Paracentrotus lividus* were sensitive to the presence of Irgarol even in minute amounts: it was shown to cause larval malformation in 90% of individuals at 7.5mg·l^{-1}, and significant effects on sperm fertilization and transmissible damage to offspring at just 0.01mg·l^{-1}. Neither the risk associated with booster biocides nor their short- and long-term environmental fate has yet been assessed, but the US Environmental Protection Agency (EPA) has had some concerns over Irgarol due to its ubiquity in coastal waters worldwide, occasionally at levels that could affect primary productivity and even prove acutely toxic to eelgrass (*Zostera marina*) (California Department of Pesticide Regulation, 2007).

3.4. Policy and management of antifoulants

France pioneered regulations restricting the use of organotin-based antifoulants: as early as 1982 the use of organotin paint on boats smaller than 25m was prohibited (with exemption for aluminium hulls). The legislation reduced contamination within shellfish culture areas on the French Atlantic coast, but 'the efficacy of the legislation does not extend to the Mediterranean coast' (Alzieu *et al*., 1991; see also Michel & Averty, 1999).

The Mediterranean countries were the first to propose restrictions on the use of organotins on a region-wide basis. The Protocol of the Barcelona Convention for the Protection of the Mediterranean Sea against Pollution from Land-Based Sources, signed in 1980, listed organotin compounds (Annex I, A.5) among substances for which legal measures should be proposed and adopted. The Mediterranean Action Plan (MAP) of the United Nations Environment Programme (UNEP), with the cooperation of international agencies, conducted a pilot study of organotin contamination in 1988 that recorded 'high and potentially toxic concentrations of TBT … in the vicinity of harbours and marinas' (Gabrielides *et al*., 1990). In 1989 these data led the Contracting Parties to the Barcelona Convention to adopt measures limiting the use of TBT antifouling paints in the Mediterranean. These measures, which entered into effect in 1991, included a ban on organotin-based antifouling paints 'on hulls of boats having an overall length … of less than 25m'. A recommendation was made that 'a code of practice be developed in minimizing the contamination of the marine environment in the vicinity of boat-yards, dry docks, etc., where ships are cleaned of old anti-fouling paint and subsequently repainted.' (UNEP, 1989). However, post-1991 data (*cited above*) show continuing high levels of TBT and raise suspicions that the legislation banning the paints is being ignored.

In 1990 the International Maritime Organization (IMO) adopted a resolution recommending governments to adopt measures to eliminate antifouling paints containing TBT (IMO, 1990). Despite regulations, concentrations of TBT in sediments and water in ports, in coastal regions and offshore, failed to decline and in some cases increased during in the 1990s. In 2001 the IMO, following evidence that the incidence of imposex in open seas is highly correlated with shipping densities, adopted the International Convention on the Control of Harmful Anti-Fouling Systems on Ships (AFS Convention), which called for a global prohibition on the application of organotin compounds. Annex I attached to the Convention and adopted by the Diplomatic Conference states that by 1 January 2003 all ships shall not apply or re-apply organotin compounds, and that by 1 January 2008 ship hulls either shall be free of organotin compounds or shall be coated over to prevent leaching. The AFS Convention was due to enter into force on 17 September 2008.

The EPA has established that for TBT the criterion to 'protect aquatic life from chronic toxic effects is 0.0074 µg/L … [and] from acute toxic effects is 0.42 µg/L' (EPA, 2003). TBT contamination in the Mediterranean, even in the immediate vicinity of nature reserves away from maritime shipping lanes, far exceeds the levels needed to protect the biota from chronic effects, and in or near the numerous ports and marinas these levels are hundreds or thousands of times higher (*see above*). These levels may stem from antifouling paints on large vessels (including hosed and scraped paint fragments), illegal use of organotin-based paints on recreational vessels, and dredging of TBT-laden harbour sediments. Organic booster biocides recently introduced as substitutes for TBT-based antifouling paints are also toxic and, although some are rapidly degraded, the more persistent compounds build up high concentrations especially in sediments within high-density marinas with poor water exchange. At present there is no regulation as to their permitted concentrations in surface water or sediments: '[i]n Spain, Greece, and France, there are very limited registration schemes and, in principle, all [booster biocides] can be used' (Readman, 2006).

Thirty years after the impacts of TBT were identified, and although its compounds have been singled out as priority hazardous substances (European Community, 2001), the continuing high levels of TBT in Mediterranean seawater and sediments point to grave failures at national and regional levels to check the damage done to the marine biota by shipping-derived antifouling paints.

With the imminent elimination of TBT-based paints, booster biocides are increasingly being used in antifouling products. Unfortunately, the signal failure of the industry and the authorities to check on the ecotoxicological risks of TBT prior to its usage is about to repeat itself. Detailed testing of the toxicity, persistence and sorptive behaviour of the alternative biocides have not been completed, nor their impact on the marine environment clarified (Yamada, 2006). Guidelines have been mostly based on information extrapolated from laboratory-scale tests to whole ecosystems. One of the lessons of the TBT debacle is that if biocides are to be used responsibly in a sustainable manner, regulations based on comprehensive testing should be developed prior to licensing them.

Since the AFS Convention will enter into force in September 2008, it is imperative to monitor the expected decline in the levels of TBT and its derivatives in Mediterranean ports and marinas, as well as to monitor the levels of labile copper and organic booster biocides.

4 Significant collisions with marine mammals and turtles

4.1. Marine mammals

4.1.1. Introduction

Collisions between ships and whales, both odontocetes and mysticetes, are regularly reported from all the world's oceans. Even where the fatal strike rate does not threaten the species at the population level, it can be a major cause of human-induced mortality. In certain cases it can be a serious threat to the survival of a species, as in the case of the North Atlantic right whale, *Eubalaena glacialis* (Knowlton & Kraus, 2004; Kraus *et al*., 2005; Knowlton & Brown, 2007). To date, evidence has emerged of ship collisions with at least 11 species of large whales (Laist *et al*., 2001; Jensen & Silber, 2003). Of these, the fin whale (*Balaenoptera physalus*) is the species most commonly recorded as being hit by ships worldwide (Panigada *et al*., 2006).

Ship strikes have also been reported for small cetaceans: Van Waerebeek *et al*. (2007) presented evidence of at least 19 documented cases. The effect of ship strikes may be irrelevant for the survival of marine species, but it may lead to an unsustainable mortality rate for estuarine and river dolphin populations (Van Waerebeek *et al*., 2007).

The Mediterranean Sea is particularly susceptible to ship-associated impacts because of a high volume of shipping routes, a long history of use, and sensitive shallow-water and deep-sea ecosystems. Shipping has greatly expanded in the Mediterranean over the past half century. Between 1985 and 2001, a 77% increase was recorded in the volume of ship cargoes loaded and unloaded in Mediterranean ports. Every year 220,000 ships larger than 100 tons cross the Mediterranean basin, and approximately 30% by volume of international sea-borne trade has its origin or destination in the 300 ports in the region. This volume is expected to grow three- or fourfold in the next 20 years (Dobler, 2002). Furthermore, a total of over 9,000 vessels, including ferries, fast ferries and hydrofoils, as well as military, fishing, pleasure and whale-watching boats, cross the waters of the Western Basin daily (SCOT, 2004). The reported levels of marine traffic and the forecast increase in commercial shipping are not the only threats faced by cetaceans in the Mediterranean Sea. Noise, noxious man-made pollutants in the marine food web, increasing disturbance, interactions with fisheries, depletion of prey, habitat degradation and, more recently, questions about the impact of global climate change all suggest the urgent need for proper protection measures. In addition, the populations of most of the species occurring in the Mediterranean Sea are genetically isolated and have little gene flow with their North Atlantic conspecifics (Bérubé *et al*., 1998; Reeves & Notarbartolo di Sciara, 2006); exposing these species to such high anthropogenic pressures may lead to severe losses at population or sub-population levels.

The reason why cetaceans, and in particular fin whales, do not avoid being struck by ships is not completely evident. In contrast to other baleen whales, fin whales are fast swimmers, achieving short bursts of speed of up to 55.5km/h (Slijper, 1979). Such a speed suggests that they should be able to avoid a ship by moving out of its path, provided it is detected in time. However, specific behaviours like feeding, resting or mating may reduce whales' attentiveness to environmental sounds. In particular, Mediterranean fin whales perform unusually deep foraging dives (Panigada *et al*., 1999, 2003), while some baleen whales (blue, fin and North Atlantic right whales) glide during the final stages of ascent from a dive, thus reducing their ability to abruptly change their trajectory upon arrival of a ship (Williams *et al*., 2000; Nowacek *et al*., 2001). In addition, they may not be able to detect sounds originating from surface vessels until they have reached the end of their ascent and are already in the path of the vessel.

4.1.2. The Pelagos Sanctuary

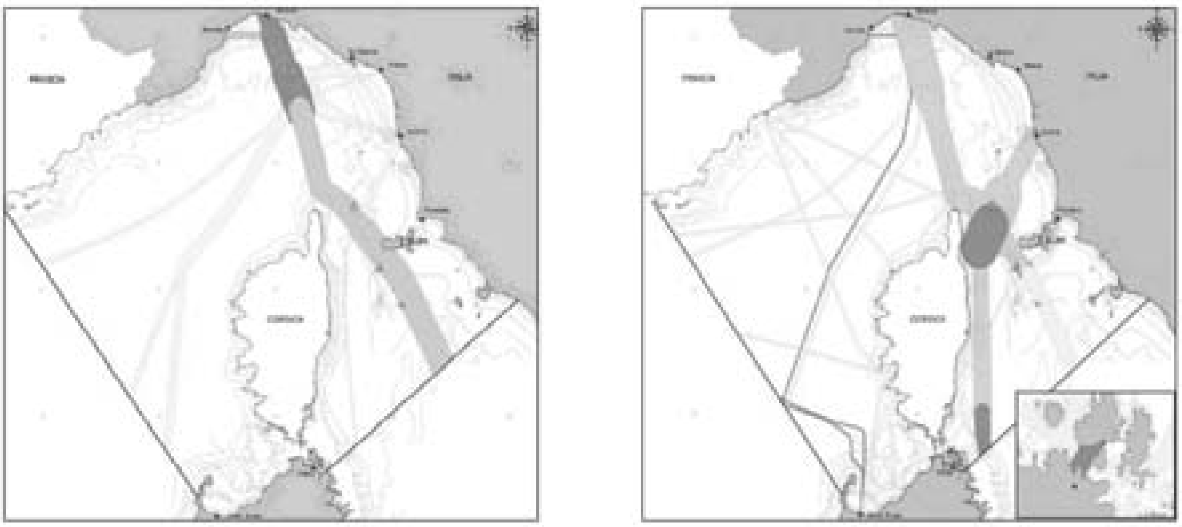
Whales and dolphins often congregate during the summer months to feed precisely in areas where vessel traffic is highest. One such aggregation area for fin whales, striped dolphins and other cetacean species is the Pelagos Sanctuary in the Ligurian Sea, where particular oceanographic features support high levels of prey and consequently a large number of cetaceans (Jacques, 1989; Notarbartolo di Sciara *et al*., 1993; Astraldi *et al*., 1994).

The Pelagos Sanctuary was established on 25 November 1999, in recognition of the local abundance of cetaceans. Italy, France and Monaco signed an agreement to establish an International Sanctuary for the Protection of Mediterranean Marine Mammals (Notarbartolo di Sciara *et al*., 2008), which entered into force in 2002. The Sanctuary was listed as a Specially Protected Area of Mediterranean Importance (SPAMI) in 2001, within the framework of the UNEP Barcelona Convention. The area encompassed by the Sanctuary is bounded by the Côte d'Azur in France, northern Sardinia, and the coasts of Liguria and Tuscany in Italy. The large number of cities and harbours on the surrounding coasts and the density of merchant, passenger and recreational traffic in the Sanctuary means that human pressure on this part of the Mediterranean is extremely high (Figure 1.8).

4.1.3. Estimating the extent of ship strikes in the Mediterranean Sea

To provide a complete picture of the ship strike problem within the Mediterranean Sea, Panigada and colleagues (2006) reviewed all the available records of fin whale collisions, including both dead and photo-identified free-ranging individuals. In addition, in order to place the available information in a conservation context, a Workshop on Large Whale Ship Strikes in the Mediterranean Sea, funded by the Italian Ministry of the Environment, was held in Monaco in November 2005 (ACCOBAMS, 2006).

The objectives of these initiatives were to synthesize knowledge of ship strikes on fin and sperm whales and other cetaceans in the Mediterranean Sea, with particular emphasis on the Pelagos Sanctuary for Mediterranean Marine Mammals, and to place them in a global and local context; to assess the extent of this threat for Mediterranean cetaceans; to determine the types of vessels that hit cetaceans; to determine data gaps vital for a more comprehensive assessment of the issue, in order to suggest further research aimed at reducing the potential for vessel collisions and minimizing mortality rates in the Mediterranean populations; to assess the effectiveness of existing mitigation



(a) (b)

Figure 1.8—(a) Merchant shipping and (b) passenger ferry routes in the Pelagos Sanctuary. Colours indicate traffic density (red high; orange medium; yellow low; green unclassified). The inset in (b) shows passenger routes in NE Sardinia. (From Tunesi et al., 2007; For the color version of this figure please refer to the Annex at the end of the document)

and management measures; and to discuss what further measures might effectively be employed to address the issue.

4.1.3.1. Fatalities

Records concerning 287 fin whales stranded along the Mediterranean coasts, caught on the bows of ships or found floating at sea were examined (Panigada *et al*., 2006). Of these, 46 (16.0%) were confirmed to have died because of a ship strike. Between 1972 and 2001, 43 whales were killed, yielding a mean fatal strike rate of 1.43 animals per year. Seasonal differences were found, with spring and summer having significantly more collisions than autumn and winter; this matches the presumed Mediterranean fin whale feeding season (April–September) versus the presumed breeding months (October–March), with the majority of the accidents (76.7%, 33 versus 10) occurring within the feeding season (Notarbartolo di Sciara *et al*., 2003).

In 24 cases it was possible to ascertain the vessel type involved in a strike: standard ferries were most frequently implicated (15, 62.5%), followed by merchant ships (4, 16.7%), fast ferries (3, 12.5%) and yachts. High-speed ferries were introduced into the area in 1996; in the six years following that period they accounted for almost 50% of the total known collisions.

The majority of reported strikes (82.2%) was recorded in the Pelagos Sanctuary and the Gulf of Lion or adjacent waters, suggesting that these are high-risk areas for whale collisions; the remaining strikes were reported in Spanish and southern Italian waters. Based on approximately 900 fin whales assessed in this area, the estimated minimum fatal collision rate would be 0.0013, three times higher than for the whole Western basin (0.0004). However, this result may be confounded by increased observer effort and more efficient stranding networks in those areas.

4.1.3.2. Injured individuals

Nine out of 383 photo-identified whales (2.4%) had wounds positively attributed to a ship strike (Figure 1.9). No information on the year or the location of the incident was available in any case, as no animal was seen before and after the collision. Healed-over lesions (depressed scars from old wounds) were present on six whales (66.7%), propeller scars were found on two whales (22.2%) and unhealed open wounds were recorded on one whale (11.1%). Six whales had a cut dorsal fin or fluke while four animals had a 'humpbacked' body.

The low reported number of live whales showing evidence of collisions may indicate that few animals survive a ship strike or that collisions with small boats are less frequent; moreover it is likely that the vessels involved were of small enough size and weight to allow the whale to survive the consequences of the collision, which would otherwise be fatal.

Figure 1.9—A fin whale showing propeller scars. (For the color version of this figure please refer to the Annex at the end of the document)

4.1.3.3. Estimating actual fatal strike rates

One of the main problems in estimating the fatal strike rate for cetaceans is that the occurrence and frequency of collisions may be either under-estimated (owing to unnoticed or unreported events, incomplete or non-existent necropsies, masking of fatal ship strikes by advanced carcass decomposition, or inadequate data collection techniques) or over-estimated (as where animals died from other causes, but their floating carcasses were struck after death). Considering all the biases possibly affecting the Mediterranean data set, we believe that these numbers are more likely to be an under-estimate rather than an over-estimate. This conclusion was also supported by Kraus *et al*. (2005), who analysed North Atlantic right whale strandings, relating them to estimated mortality rates, and found that human-caused fatalities were considerably under-estimated.

A fact supporting the under-estimation hypothesis is that almost half of the fin whales that were reported as fatally struck in the Mediterranean Sea were found lodged on the bow of the colliding ship. In the majority of these collisions the whale was discovered only once the vessel was in port, suggesting that in cases where the carcass did not become lodged, or fell off prior to the ship's arrival in port, the strike would have gone unnoticed. In addition, many of these carcasses showed no noticeable external wounds, confirming that other ship-strike fatalities might be missed unless thorough necropsies are performed as a matter of course. Such complete necropsies can also ascertain whether the collision occurred pre- or post-mortem.

4.1.4. Sperm whales along the coast of Greece

According to the research and data of Pelagos Cetacean Research Institute during the last decade (1997–2007), 1.4 sperm whales strand per year along the Greek coasts. At least 70% of the stranded whales have clear propeller marks on their body and their deaths are likely to have been caused by collisions with large ships (Figure 1.10). Most of them are young and immature individuals, which live in social units usually comprising 8–14 members (Frantzis *et al*., in preparation). On 17 and 20 June 2007 two sperm whales were found stranded in remote locations close to Elafonissos Island, off the south-western coast of Crete, an area where both solitary males and social units are present all year round (Frantzis *et al*., 2000, 2003). The two animals had total lengths of 6.5m and 7.0m and were male and female respectively. The first was found in two separate pieces (head to dorsal fin and detached tailstock), and the second bore clear propeller marks on its forehead area. The state of decomposition and the proximity of the two stranding positions (about 300m apart) indicated that these animals died at the same time and very close to each other, apparently when a ship struck them and possibly other members of their social unit. This was the first time that two animals had stranded simultaneously in Greece after a likely collision with a vessel. The incident shows that more than one member of a socializing group of sperm whales may be at risk if they are found on the route of large vessels. Propeller marks have also been observed on at least three photo-identified live animals from the same population unit in the Hellenic Trench. Considering the small number of sperm whales inhabiting the Hellenic Trench (recently estimated at about 180 animals; Frantzis *et al*., in preparation) and the Mediterranean Sea in general (Notarbartolo di Sciara *et al*., 2006), the rate of vessel collisions with sperm whales seems to be unsustainable and clearly threatens the endangered population of this species in the Mediterranean Sea.

4.1.5. Mitigation measures and conservation recommendations

Many different solutions have been proposed to reduce the risk of collisions, ranging from instruments mounted on board ships to detect whales (such as sonar or night vision devices) to acoustic alerting devices to warn whales of approaching boats (Nowacek *et al*., 2004), bottom-anchored passive sonar systems designed to detect whale locations, and specially trained observers on board ferries. None of these solutions alone would seem to be effective or capable of achieving a significant reduction in ship strikes, since each of them either has undesirable side-effects (such as interfering with the whales' communication, or being too unreliable) or is only effective in particular situations (e.g. during day time, during specific weather conditions, only when the whales vocalize, only at short distances, or just within certain angles of the ship's bow). In particular, acoustic alerting

Figure 1.10—A sperm whale with propeller scars. (For the color version of this figure please refer to the Annex at the end of the document)

devices may not be the most appropriate solution since in high-density shipping areas the noise present may disturb or block the animals' acoustic perception of approaching vessels or their timely alertness to warning signals emitted from a ship; moreover, the frequency of a warning signal may benefit one species, whereas another that has a different acoustic window may suffer from the impact of the signal itself.

Mitigation measures, including the training of crew members at the French national merchant shipping school and the development of a real-time network for commercial ships to report the positions of large cetaceans in order to limit collision risks (REPCET), are being developed and tested in the Pelagos Sanctuary by French scientists.

In the absence of a better understanding of why cetaceans, and in particular large whales, are struck by ships, the following mitigation measures may be more effective and realistic.

4.1.5.1. Eliciting cetacean avoidance behaviour

A thorough inventory of potential triggers that may elicit avoidance behaviour in whales is required and the role of each one needs to be assessed. Further research should also be conducted into the possible reasons why cetaceans fail to perceive approaching vessels.

4.1.5.2. Reducing ship speed and re-routeing shipping lanes

Reducing ship speed when crossing areas of high whale density would both allow cetaceans more time to avoid the oncoming vessel and give the operator more time to react to the whales' presence. This could be coupled with the presence of trained observers onboard, to alert the crew of approaching cetaceans. Reducing ship speed may be unpalatable to operators, since it runs counter to the current trend of increasing speed; however, damage resulting from collisions can also be serious for the shipping companies. It can lead to passenger injuries, loss of profit for the days when the vessel is laid up in the shipyard, and bad publicity for the shipping companies in the eyes of the general public.

Several mitigation measures applied to protect the North Atlantic right whale are relevant here. They include the decision by the USA for ships to reduce speed when crossing right whale areas of importance along the eastern coast of the USA; the decision to re-route shipping lanes crossing right whale habitats in the Bay of Fundy (Canada); and the designation of an area to be avoided in the Roseway Basin, which was proposed by Canada and recently approved by the IMO. Similar measures have been approved by the Spanish Government to protect sperm whales in the Strait of Gibraltar and other cetaceans in the Alboran Sea.

The Strait of Gibraltar is a critical place in terms of maritime traffic: every year more than 80,000 vessels cross the Strait on various routes, resulting in very high traffic densities. Large ships, including cargo vessels, tankers and passenger ships, pass through the area along its east–west axis, while ferries and fast ferries cross between its northern and southern shores. The Strait has a notable abundance of cetaceans, mostly sperm whales, which occur in the southern area between March and July every year, with a peak in May. In addition, Morocco is building a new harbour (Tanger-Med Port) in the Strait, just opposite the area mainly used by sperm whales; its first terminal started operating in 2007. This new port will change the ferry and fast ferry lanes in the area, and the new lanes will directly cross the main area of sperm whale distribution.

In light of this evidence, the Spanish Ministry of the Environment has suggested a number of measures to be implemented in the Strait to reduce the impact of ship strikes. A Notice to Mariners was published in January 2007 by the Instituto Hidrográfico de la Marina (Navy Hydrographical Institute, under the Ministry of Defence) establishing a security area characterized by high densities of sperm whales, where ships must limit their maximum speed to 13 knots (following the suggestions of Laist *et al*., 2001) and navigate with particular caution (IHM, 2007). The same Notice will be broadcast regularly by VHF radio from April to August and included on Nautical Charts (Tejedor *et al*., 2007).

A recent European Commission LIFE Nature project, 'Conservation of cetaceans and sea turtles in Murcia and Andalusia' (LIFE02NAT/E/8610), coordinated by the Spanish Cetacean Society with the involvement of the Spanish Ministries of the Environment and Fisheries and the National Oceanographic Institute, developed a Conservation Plan for the loggerhead turtle and bottlenose dolphin and a Management Plan for the Southern Almería Special Area of Conservation (SAC). This area covers an extremely valuable and sensitive coastal habitat for bottlenose dolphins and loggerhead turtles within the framework of the European Union's Habitats Directive (European Economic Community, 1992). The project's central management body and the UNESCO Chair for the Environment suggested relocating the Cabo de Gata Traffic Separation Scheme (TSS) in view of the high risk of ship collisions and oil spills in the area. This action was discussed with the Spanish maritime authorities and the International Maritime Organization (IMO); consequently the Cabo de Gata TSS has been moved from 5 to 20 nautical miles off the coast. The new location has been published in the Notice to Mariners and on International Nautical Charts.

In the Ligurian Sea such solutions will probably be difficult to adopt since, as already stressed, the great majority of ferries connecting the islands with the Italian and the French mainland cross the region where fin whales are most concentrated. Nevertheless, with the inclusion of the Pelagos Sanctuary for Marine Mammals on the List of Specially Protected Areas of Mediterranean Importance (SPAMIs), it would represent an ideal framework to apply similar regulations. Such measures could be limited to particularly risky vessel types or activities, or possible sub-areas characterized by particularly high concentrations of fin whales. An example of what can be done is the ban on motorboat racing within the 12-mile limit of Italian territorial waters, under the Italian law implementing the Pelagos Sanctuary (Law No 391 of 11 October 2001).

Strong similarities in terms of ship collision problems and long-term management philosophy may be seen in the Stellwagen Bank National Marine Sanctuary (SBNMS) in the USA and the Pelagos Sanctuary. Similar management and research strategies should be applied and tested in the two areas, leading to an effective reduction in the risk of vessel strikes on large whales.

4.1.5.3. Data collection, databases and modelling

Recent technology based on the Universal Shipborne Automatic Identification System (AIS)—a VHF tracking system that sends information about a ship's speed, heading and position to other vessels and to shore-based stations—should be coupled with real-time reporting of whale sightings by major whale-watching bodies and ferries regularly crossing the areas, and applied to quantify and assess ship behaviour (Moller *et al*., 2005).

A standardized database needs to be prepared and populated to record vessel collisions with cetaceans, with the ultimate aim of developing a global data repository. A comprehensive database containing both biological and vessel information would be used to model specific probabilities of collision, from which better estimates of true mortality rates could be derived, as well as to point to causative factors and unsuspected global collision hotspots. A Vessel Strike Data Standardization Group has been established within the International Whaling Commission (IWC) to develop a process by which data from a range of sources, including interviews with captains and crews, could be stored in a database in a standardized way that clearly identifies the level of certainty and uncertainty in the data. The most appropriate methods of bringing cetacean issues to the IMO and obtaining relevant information from them should also be explored and implemented. This proposition has already been tested in the Pelagos Sanctuary on shipping companies.

Long-term monitoring of cetacean presence and distribution, including the use of AIS data, should be combined with habitat selection models to investigate the relationship between biological parameters, including prey abundance, and remotely sensed physical parameters, such as sea-surface temperature, ocean colour and wind speed. Consequently, particular areas characterized by high densities of both cetaceans and ships can be identified as critical whale habitats, where speed reductions or the shifting of ferry routes to areas of lower cetacean density may be proposed.

4.2. Sea turtles

Other marine vertebrates, such as sea turtles that need to come to the sea surface at regular intervals to breathe, are also exposed to the risk of ship strikes (Hazel *et al*., 2007). This problem has, in recent years, become a major challenge for marine turtle conservation in the Mediterranean Sea and worldwide.

Several parts of the Mediterranean Basin play a key role in the life cycle of these vertebrates. Nesting grounds are mainly concentrated in the eastern part of the Basin, especially in Greece, Turkey, Libya and Cyprus (Aureggi *et al*., 2005). Feeding grounds for the oceanic phase include oceanic areas in the western and eastern Mediterranean, whereas wide continental shelves, as in the north Adriatic, central Mediterranean, south-east Turkey and Egypt, are important in the neritic phase (Margaritoulis *et al*., 2003).

Sea turtles tend to frequent these areas regularly and show high levels of site fidelity (Casale *et al*., 2007). These regions are, however, exposed to high levels of maritime traffic, including fishing, pleasure and recreational boats, particularly during the summer months when sea turtles spend more time at the surface to breathe, rest and bask and are therefore more susceptible to collision events.

Ship collisions with turtles may cause different degrees of injury (Figure 1.11). In rare cases, turtles show scars along the marginal scutes of the carapace that do not impede their movements or threaten their lives. On most occasions, however, a turtle does not survive an impact or is left seriously injured, with limited movement and diving ability. In such cases, injuries occur most frequently in the anterior area, with severe cuts or amputation of the fore flippers, head and front scutes of the carapace, since most impacts occur when turtles surface to breathe (Vallini, personal communication). The brain and the lungs are the internal organs most commonly affected, in which cases the consequences are always fatal.

Several important factors may lead to a collision. The animal may have difficulty in detecting the direction of sounds underwater because of particular sound propagation characteristics or the presence of several sound sources that tend to mask single, isolated sounds, such as that from an approaching outboard engine. Prolonged engine and propeller sounds may

Figure 1.11—A sea turtle with propeller scars.

merge with ambient, background noises, thus becoming more difficult for individuals to discriminate. Body size also matters in avoiding a ship strike, as smaller animals are more agile in the water than larger ones. As in the case of cetaceans, boat speed is likely to be a major cause of fatal ship strikes on sea turtles. The reason why sea turtles do not avoid approaching boats is probably that they simply do not detect them. Although their underwater visual abilities have been shown to be quite efficient (Oliver *et al*, 2000; Constantino & Salmon, 2003), visual range is limited, especially in turbid waters like the northern Adriatic Sea.

A study on green turtles in shallow waters in Australia reported an avoidance reaction at a distance of less than 12m from the approaching vessel (Hazel *et al*., 2007). The authors speculate that sea turtles are evolutionarily adapted to react to visual clues only and not acoustic ones, due to the kind of natural predators they have (sharks). Hence, ship speed seems to be the key factor: at low speeds (2–3 knots) onboard observers can easily see a turtle surfacing in front of the vessel; low speeds also give the turtle time to avoid crossing the ship's course or to change direction. Even at low speeds, however, Hazel *et al*. (2007) observed effective avoidance reactions only in 60% of turtles. Vessel operators therefore cannot rely on turtles to actively avoid being struck by vessels approaching at speeds of more than 2 knots. As most vessels travel much faster than 2 knots in open waters, Hazel *et al*. (2007) infer that mandatory speed restrictions will be necessary to reduce the cumulative risk of vessel strikes on green turtles in key habitats subject to frequent vessel traffic.

In conclusion, of the various threats at sea—including ghost gear and by-catch, which represent the main causes of human-induced sea turtle mortality in the Mediterranean (Gerosa & Aureggi, 2001)—ship strikes may be a minor cause of death, but they are becoming increasingly common and should not be ignored. In particular, turtles are principally threatened by pleasure craft rather than commercial traffic. They tend to frequent the surface waters of the neritic zones, where they congregate in large numbers for nesting purposes (Casale *et al*., 2007); this is where pleasure traffic is most abundant and where ship strikes occur most frequently (Vallini, unpublished).

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Chapter 2

Ship-generated oil discharges and exhaust emissions in the Mediterranean basin: their distribution and impact

Charles Galdies

1. Background

The amount of oil transported by sea worldwide increased from 1.3 billion tons in 1988 to over 2 billion tons in 2004 (Mitropoulos, 2007). Resulting from this massive transportation, the annual global anthropogenic marine oil discharge has been estimated to be around 4.15 million tons (Berthe-Corti & Hopner, 2004), whilst natural oil seepages into the sea are around 600,000 tons (Committee on Oil in the Sea, 2002; Burgherr, 2007). Adding these two inputs together results in a net estimate of around 5 million tons of oil entering our marine areas each year. It is heterogeneously distributed mainly in the vicinity of oil production platforms, shipping lanes, transhipment facilities and natural marine oil seepage areas.

From the available data, UNEP/IOC (1988) considers the Mediterranean Sea to be more polluted by oil than any other sea. It is a fact that 30% of all international maritime trade by volume has its origin or destination in Mediterranean ports or passes through this semi-enclosed region, and that a quarter of the total oil transported by sea passes through the Mediterranean (REMPEC, 2002). Hence, as might be expected, it has a significant impact on the health of the Mediterranean ecosystems.

In spite of the Mediterranean's importance for global maritime shipping, data on the total discharges resulting from such maritime traffic are particularly scarce. Most of the available data are relevant only to the western Mediterranean area, while significant data gaps occur for the eastern and southern basins, with the exception of certain Algerian and Egyptian coasts (Zaghden *et al*., 2005). This lack of full data coverage is mainly attributed to a lack of financial and technical resources (EEA, 1999; World Bank, 2006).

2. Oil tanker traffic volume in the Mediterranean Sea

The Mediterranean Sea is crossed by thousands of oil tankers sailing along the main routes each year (Figure 2.1), transporting crude oil from the Middle East mostly to ports in Europe and North America via the Suez Canal. According to recent estimates, in 2003 over 14,500 ships passed through this narrow canal between the Mediterranean and the Red Sea. At the other end of the Mediterranean, the Straits of Gibraltar are traversed by over 80,000 vessels every year (REMPEC, 2002). The Istanbul Straits are another point of maritime congestion: a total of 49,304

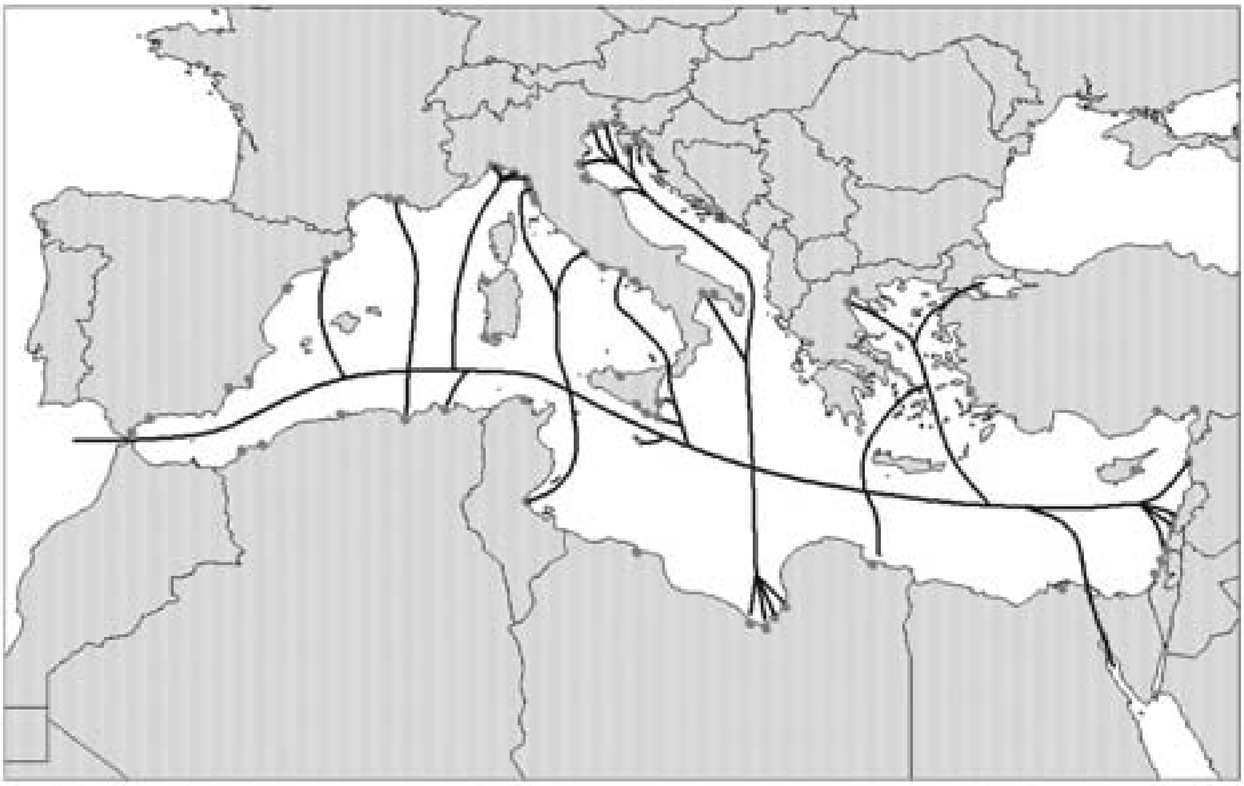


Figure 2.1—Transit routes through the Mediterranean and to its main ports and oil terminals (Source: EEA, 1999; Base map: ESRI; For the color version of this figure please refer to the Annex at the end of the document)

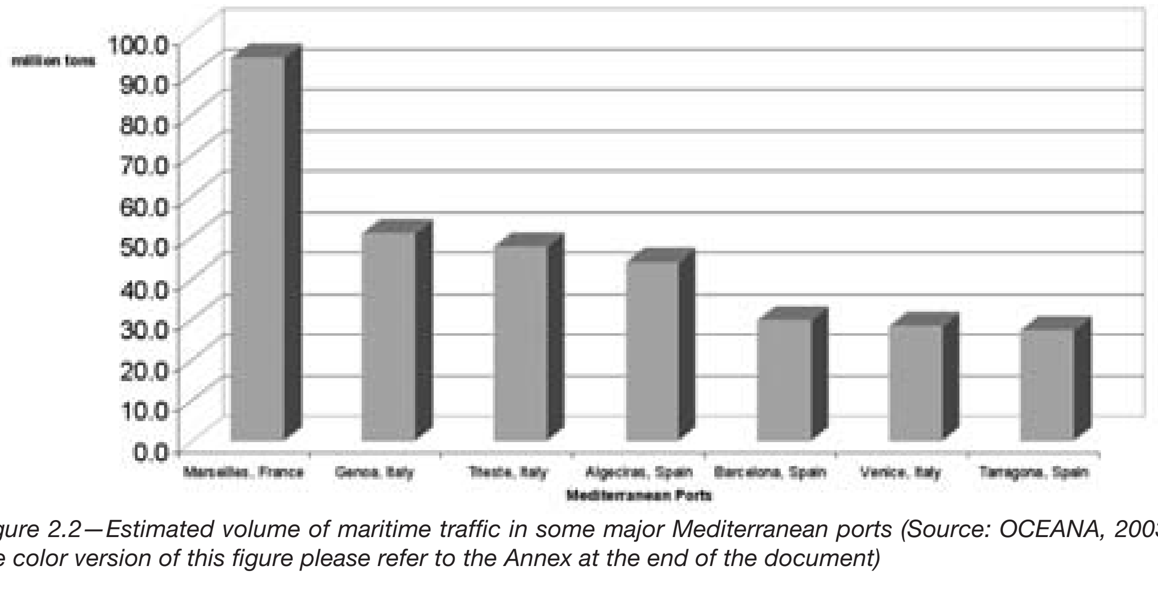


Figure 2.2—Estimated volume of maritime traffic in some major Mediterranean ports (Source: OCEANA, 2003; For the color version of this figure please refer to the Annex at the end of the document)

ships passed through them in 1998, an average of 137 vessels a day (Kesgin & Vardar, 2001). As a result, maritime traffic by volume in the main Mediterranean ports is significant (Figure 2.2).

It is estimated that around 360 million tons of oil and refined products cross the Mediterranean every year (UNEP, 2006). The development of Caspian oil pipelines in the near future is expected to increase this figure by bringing into the Mediterranean an additional 100 million tons of oil annually. General maritime traffic is also expected to increase as the future implementation of 'motorways of the sea' as part of the Trans-European Transport Network will considerably increase the volume of traffic in the region (REMPEC, 2002).

3. Volume and spatial distribution of discharged oil in the Mediterranean Sea

Out of the above volume of oil and derived products, close to 400,000 tons are deliberately dumped every year into this semi-enclosed basin (Solberg & Theophilopoulos, 1997; UNEP, 2006). This pollution is driven by routine ship operations, which are regarded as the source of the most serious type of oil pollution in the Mediterranean, posing an acute, long-term threat to marine and coastal ecosystems (UNEP, 2006).

Around a decade ago, it was estimated that about 250,000 tons of petroleum hydrocarbons were discharged annually due to port shipping operations such as deballasting, tank washing, dry-docking, bunkering (refuelling), discharging of bilge oil, etc. (Figure 2.3). This volume is less than the 500,000 tons estimated in 1977 by Le Lourd (1977), which was considered reasonable by IMO (UNEP/IOC, 1988).

Because it is so vulnerable to pollution, the Mediterranean Sea was among the first regional seas to be declared a Special Area by the MARPOL 73/78 Convention. The Mediterranean coastal states that are Contracting Parties to the Barcelona Convention of 1976 are bound to ensure the effective implementation of international agreements aimed at safeguarding the Mediterranean from marine pollution. So far, attention has been focused on the states' preparedness and their containment of accidental oil spills, which are seen as one of the worst types of visible threat due to the intensity of oil traffic in the Mediterranean. On the other hand, little has been done to safeguard the Mediterranean from the oil spillages caused by routine ship operations.

It is only recently that the maritime community has had regional statistics available that shed light on the significant hazard of the oil spillages that routine ship operations cause. Knowledge of the volume and extent of such pollution has been hampered by very limited aerial surveillance in the Mediterranean Sea compared to that occurring in other Special Areas, such as the Baltic Sea and the North-West European Waters (European Commission, 2001).

Routine ship operations pollute the sea by releasing their ballast water, tank washings and engine-room effluents into the sea. Studies conducted

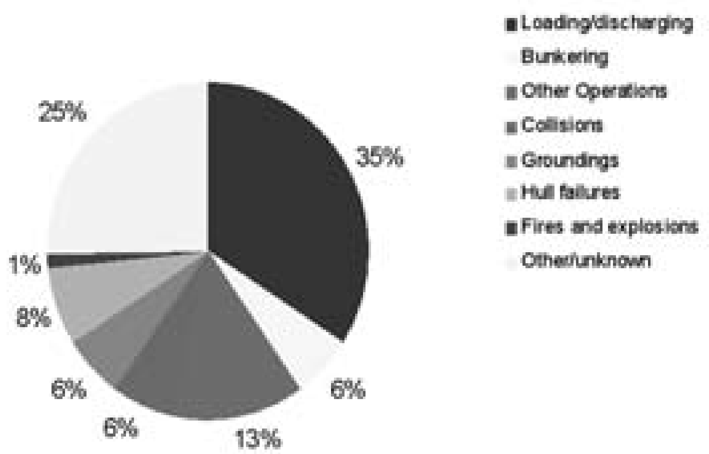


Figure 2.3—The shipping operations and accidents that contributed most to oil pollution in the Mediterranean, 1974–2006 (Source: ITOPF, 2007; For the color version of this figure please refer to the Annex at the end of the document)

by ITOPF show that globally most oil spillages originate from routine operations that very often occur in port or oil terminals (Figure 2.4). Luckily, most of these spills involve small quantities of less than 7 tons. However, accidents (such as collisions and groundings) do also happen, resulting in large oil spills. About 20% of large spills are of over 700 tons of oil (ITOPF, 2001).

Naturally, oil tankers are not the only vessels that contaminate the sea with petroleum hydrocarbons. There are cargo ships, fishing boats, leisure craft and naval vessels that also discharge oily waste, adding more tons of oil into the sea.

Recently, the European Commission Joint Research Centre (JRC) in Ispra, Italy, has made available a series of maps of all possible spills in the Mediterranean Sea during the period 1999–2004 as detected by space-borne instruments (European Commission, 2005). Figure 2.5 is the latest of such maps, showing concentrations of oil spills along the major shipping routes, particularly in the Ligurian Sea and the Gulf of Lion as well as very close to the eastern coast of Corsica and in the Straits of Sicily. All the maps show a strong scattering of possible oil spillages throughout the Mediterranean, which make it more difficult to implement effective operational patrolling.

During 1999, space-borne techniques detected a total spill area of 17,141 km2. Based on the assumptions made by Parker and Cormack (1984), the JRC maintains that this amounts to a total of 1,540 metric tonnes. However, this gives only a partial picture of the total amount, since according Hollinger and Mannella (1984) 90% of the oil usually remains in the thicker parts of the spill, or 10% of its total area. A revised estimate based on this claim adds up to an annual discharge of 13,858 metric tonnes (Ferraro *et al*., 2007).

It should be noted that the use of synthetic aperture radar (SAR) imagery for the detection of oil slicks encounters a limitation when the wind speed is either too low (typically lower than 2–3$m \cdot s^{-1}$; Donelan & Pierson, 1987), or too high (typically above 15$m \cdot s^{-1}$; Alpers & Hühnerfuss, 1988). This is due to the physical relationship between the incoming radar wave and the sea surface. Thus, while low wind speeds do not produce sufficient surface roughness in the surrounding water to contrast with the oil, strong

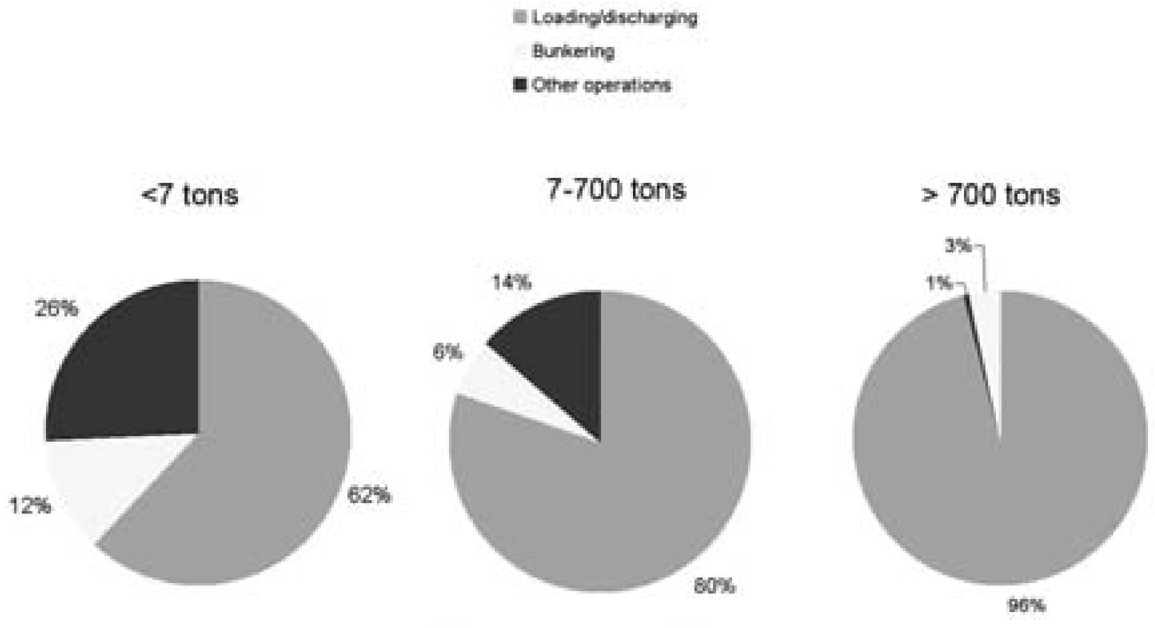


Figure 2.4—Distribution of operational causal factors of spills from oil tankers by size of spill, 1974–2006 (Source: ITOPF, 2007; For the color version of this figure please refer to the Annex at the end of the document)

winds lead to increased microwave backscattering from the spill area, reducing its contrast with the surrounding sea.

As for the detection of oil pollution, remote-sensing is hampered by two main restrictions, in that it is able neither to assess the quantity of oil, nor to identify the polluter. The accurate estimation of oil spillage can only be achieved by means of accurate knowledge of the spill thickness, which can only be measured *in situ* (Ferraro *et al*., 2007).

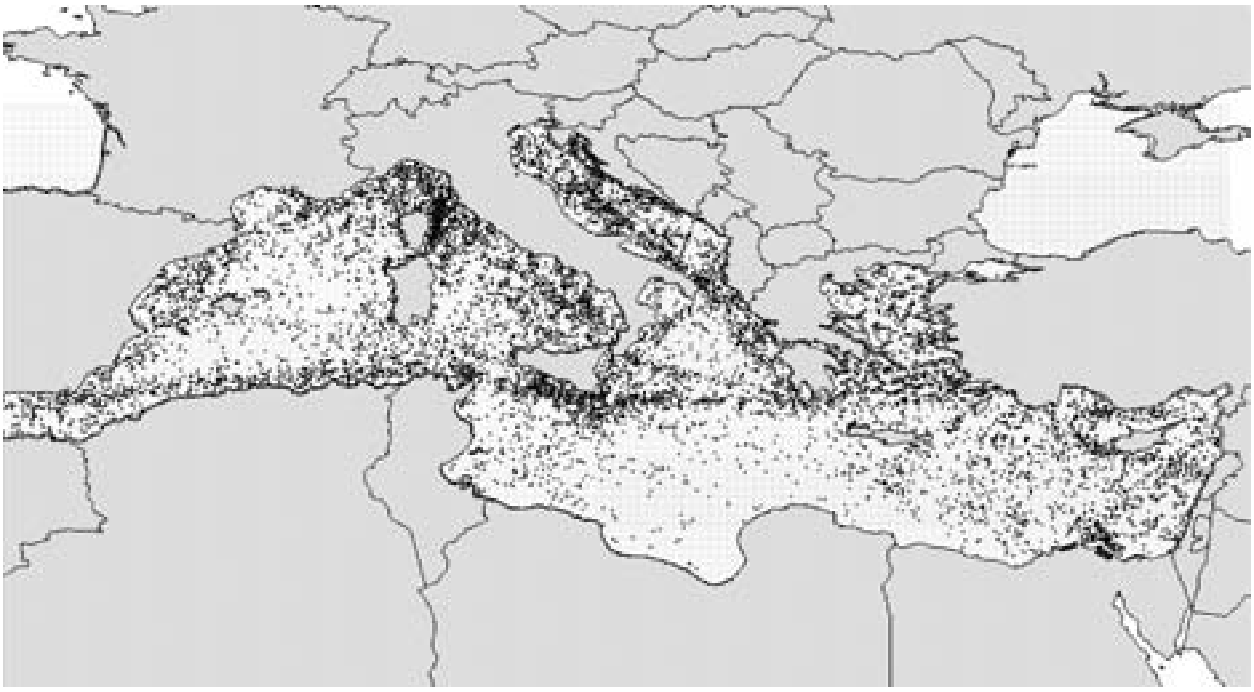


Figure 2.5—Probable oil spills (black spots) detected in the Mediterranean Sea during the period 1999–2004, using space-borne methods and image processing techniques. (Source: Ferraro et al., 2007; For the color version of this figure please refer to the Annex at the end of the document)

4. Generation and management of hydrocarbon wastes

4.1. Oil tankers

Estimates show that the amount of waste fuel oil on tankers docking in European ports could amount to more than 160,000 tons (OCEANA, 2003), which needs to be safely collected and treated appropriately so as not to cause any additional pollution. This collection and treatment of ship-generated waste and cargo residues is a prerequisite for countries that are signatories to the MARPOL Convention, in that they have to provide waste reception facilities. This provision also stipulates the ports' responsibility to monitor the waste being discharged by ships.

Although mandatory, the International Convention for the Prevention of Pollution from Ships (MARPOL 73/78) of the International Maritime Organization has long been considered inadequate, particularly for small vessels (Carpenter & Macgill, 2001). In 2000, the European Union issued Directive 2000/59/EC on port reception facilities for ship-generated waste and cargo residues, with the purpose of reducing the 'discharges of ship-generated waste and cargo residues into the sea, especially illegal discharges, from ships using ports in the Community, by improving the availability and use of port reception facilities for ship-generated waste and cargo residues, thereby enhancing the protection of the marine environment' (European Community, 2000). Since it is a directive, it does not dictate to individual Member States the means by which to achieve the result.

This directive places the burden of the cost of providing adequate reception facilities on the ships visiting the ports by applying a number of principles to all vessels (excluding fishing vessels and leisure craft carrying 12 passengers or less). It does so by opting for a 'no-special-fee' system whereby vessels pay irrespective of whether they use the facilities, in order to remove any economic incentive to discharge illegally, to recoup a sufficient level of cost to support progressive improvement in technology, and to achieve an equitable distribution of costs. According to Carpenter and Macgill (2001), this system should provide opportunities and incentives for vessels to reduce their illegal discharges. The stipulated system of fines for those caught polluting or with falsified record books should also provide a disincentive.

At the same time shipping experts argue that such a charging system does not encourage ship owners to introduce cleaner technologies. Moreover, it would have a negative impact on the EU policy to promote short maritime motorways between EU ports (Carpenter & Macgill, 2001).

Four years later, the EU commissioned a study on the functioning of port reception facilities for ship-generated waste in Community ports (EMSA, 2005). As in similar studies to measure the effectiveness of MARPOL 73/78 (Carpenter & Macgill, 2000), the resulting picture is of a very complex, varied and confusing situation for the ports surveyed. The report underlines the fact that there are only 50 port reception facilities for hydrocarbon waste and, of these, only 15 exceed the minimum capacity requirement (EMSA, 2005). It also shows that even though the Directive has been transposed in all Member States through national and/or regional legislation, its level of implementation differs between States.

On the positive side, the directive has led to improved ship-waste handling and greater awareness among stakeholders of the environmental impact of illegal discharges into the sea, resulting in increased delivery of waste to these ports. The directive was designed with flexibility in mind because of the wide variety of European ports, resulting from the fact that the various Member States currently operate different charging systems, while the type of ports range from government-owned or subsidized, through local government ownership to wholly privatized ports (Carpenter & Macgill, 2001).

On the negative side, the study shows that stakeholders interpret the directive differently, which leads to misunderstandings. As a result, it calls for more detailed, clear and uniform guidelines when these are not provided by central or regional governments.

In view of the above, the study recommends that detailed and clear guidelines be provided at the EU level to ensure the uniform implementation of the directive, specifying (a) the role of the port authorities, (b) cost-recovery details (principles and methods for calculation of fees), (c) waste notification procedures, and (d) the contractual framework with waste operators. The information should also include a common delivery certificate to prevent fraud.

4.2. Fishing vessels

According to the GAIN Report (USDA, 2004), in 2002 there were 91,000 fishing vessels registered in the EU. The number of vessels has been reduced by 11% over a five-year period, from 102,000 to 90,595. The average age of EU fishing vessels in 2002 was 22.3 years.

As with cargo vessels, waste oils produced by fishing vessels can be categorized into three types: waste lubricating oils, bilge water, and oil sludge. Unlike waste lubricating oil, which can be recycled and re-used after treatment (Lin *et al*., 2007), bilge water has low economic value for recycling. This waste is often discharged into the sea without being properly treated or recycled, since the small space on board fishing vessels makes it difficult to store a lot of waste oil (Lin *et al*., 2007). According to the MARPOL Convention, for non-tankers of 400 tons and above, the waste oil or bilge water generated can only be discharged if the oil content is below 15 parts per million. The oil content of the waste lubricating oil and bilge water produced varies considerably, and often depends on the type of engine and its level of maintenance, fuel type and number of sea-faring days.

5. Areas in the Mediterranean Sea with a high frequency of oil spills

Areas in the Mediterranean Sea with a high frequency of oil spills are found along the major shipping lanes (Figures 2.1 and 2.5) and next to oil terminals and transboundary shipping ports.

Major oil spills of an accidental nature could occur at any time in any part of the Mediterranean, particularly along the major sea routes and in or around the more important oil loading and unloading terminals (EEA, 1999). The risk of unpredictable accidental spillages is on the increase due to the aging tankers operating in the Mediterranean Sea. Accidental oil spills have caused damage to a number of local areas of the Mediterranean coastal and marine ecosystems (EEA, 1999). According to the EEA (1999) and REMPEC (2002) 82 accidents involving oil spills were recorded during the period 1990 to December 1999, with an estimated volume of more than 22,000 tons of oil spilt into the sea.

The other component of oil pollution in the Mediterranean—that of illegal operational discharges by ships—is likely to continue as long as regular surveillance is absent in the region. To ensure uniform compliance everywhere with agreed regulations, effective monitoring and intervention capabilities are necessary. In the case of deliberate ship-based pollution, monitoring is effective only when supported by continuous airborne surveillance. Currently, aerial monitoring is being carried out only over limited spatial areas, since it is neither technically nor financially feasible to spread such operations over the entire Mediterranean basin.

The lack of infrastructure and data makes a preliminary estimate of the extent, the hot spots and the temporal and spatial trends of operational oil discharges difficult. However, a recent five-year study carried out by the European Joint Research Centre and the University of Ljubljana highlights the critical extent of illegal ship-based oil discharges in the Mediterranean Sea (Ferraro *et al*., 2007). This study makes use of space-borne remote sensing, which has an important complementary and supporting role in detecting and deterring marine pollution from ships (Perrotta & Xefteris, 1996). Moreover, satellite imaging has the potential to support the coastguard service in the identification of polluting ships and prosecution of offenders (Ferraro *et al*., 2007).

For this study, a total of 18,947 images captured by an orbiting synthetic aperture radar were analysed, out of which a total 9,299 *possible*

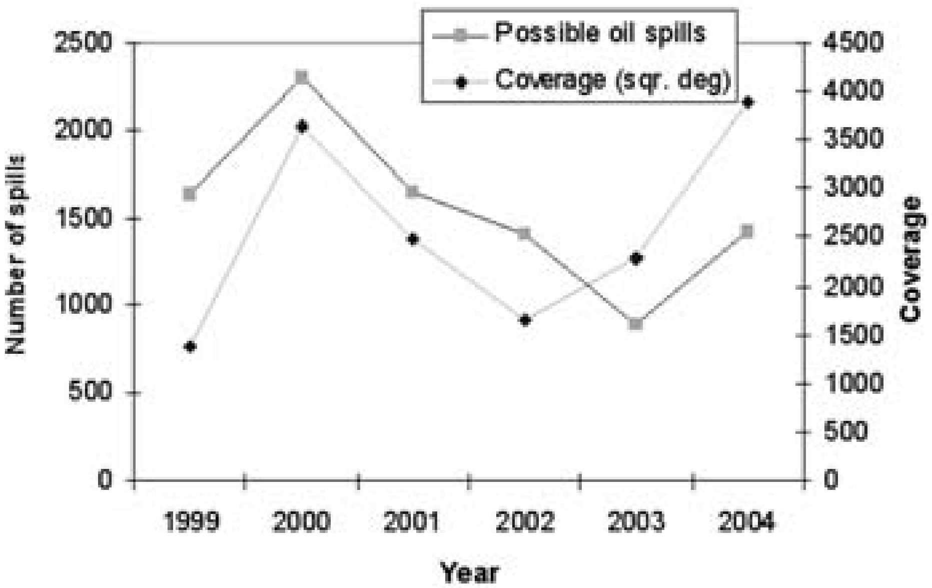


Figure 2.6—Yearly imaging coverage and number of possible oil slicks detected over the whole Mediterranean basin in the period 1999–2004 (Source: Tarchi et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

oil spills were detected from high-confidence imagery features. SAR sensors provide wide-area reconnaissance, and are independent of sunlight or cloud cover. This study was conducted using archived imagery for the period 1999–2004, and therefore the spills detected could not be validated by aerial or vessel surveillance.

The results obtained for the whole Mediterranean for the period 1999–2004 (Tarchi *et al*., 2006) indicate that the oil spill density remained fairly constant between 2000 and 2002, followed by a significant reduction during 2003 and 2004 (Figure 2.6).

The oil spill density in the Mediterranean basin is shown in Figure 2.7, which is based on the information obtained from all images analysed during the 1999–2004 period.[1] The most evident feature from the density map is that the high-spill areas are located away from the coasts, beyond territorial waters, which may indicate deliberate discharge so as to avoid legitimate punitive measures by the respective coastal states. With regard to the spatial distribution of the spillages, it should be noted that the images analysed were not uniformly distributed. The low number of images available for analysis along the Libyan coast, compared to the large number available over the Italian seas for example, led to somewhat biased results.

These data provide valuable information on the spatial distribution of possible oil spills as well as the identification of hot spots. There is evidence that the distribution of the spills is correlated with the major shipping routes, particularly in the Ionian Sea, the Adriatic Sea, the Messina Straits, the Sicily Channel, the Ligurian Sea, the Gulf of Lion and east of Corsica. Tankers release their ballast to make room for the oil which they are about to load, and therefore the lanes leading to oil terminals and at some distance from it are likely to be contaminated with oil or tar balls, especially if no reception facilities for ballast water are available at the oil terminal. Although space-borne radar techniques cannot provide information on the thickness of these oil slicks, indirect methods have been used to estimate the total volume from SAR imagery.

5.1. Seasonal trends

Ferraro *et al*. (2007) observe that the number of possible spills peaked during the summer months. Similar results have been obtained in the past

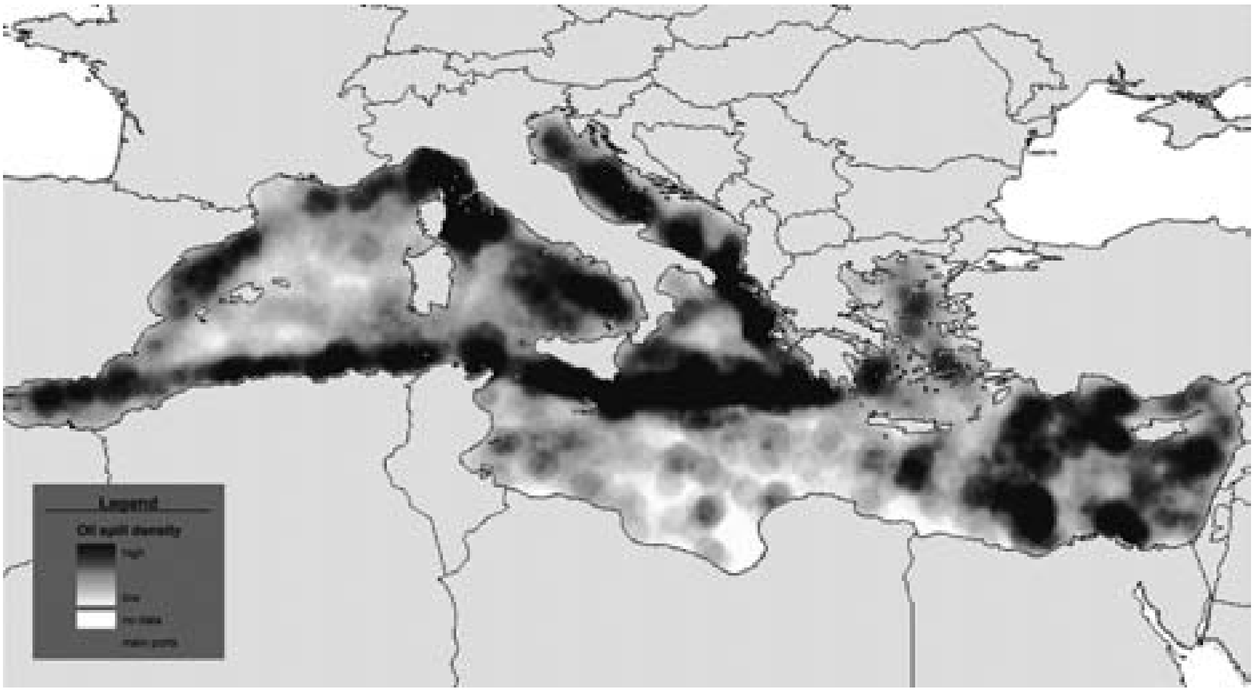


Figure 2.7—Normalized probable oil spill density for the Mediterranean basin, 1999–2004. (Source: Tarchi et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

1 The number of observed possible oil slicks in a given area was normalized by the total number of observations available for that area. Such a procedure basically removes any bias effect and accounts for uneven coverage of the area.

by other studies using space-borne imagery to detect oil spills, and this phenomenon has been attributed to higher visibility of oil films from space due to lower surface wind magnitudes (Gade *et al*., 2000). Nevertheless, the larger number of spills detected during summer in areas between Corsica, Sardinia and the Italian peninsula, which are strongly correlated with local shipping routes, strongly suggest that this seasonality may also be due to increased maritime traffic associated with tourism.

5.2. Localized high-spill areas

5.2.1. Tyrrhenian, Adriatic and Ionian Seas

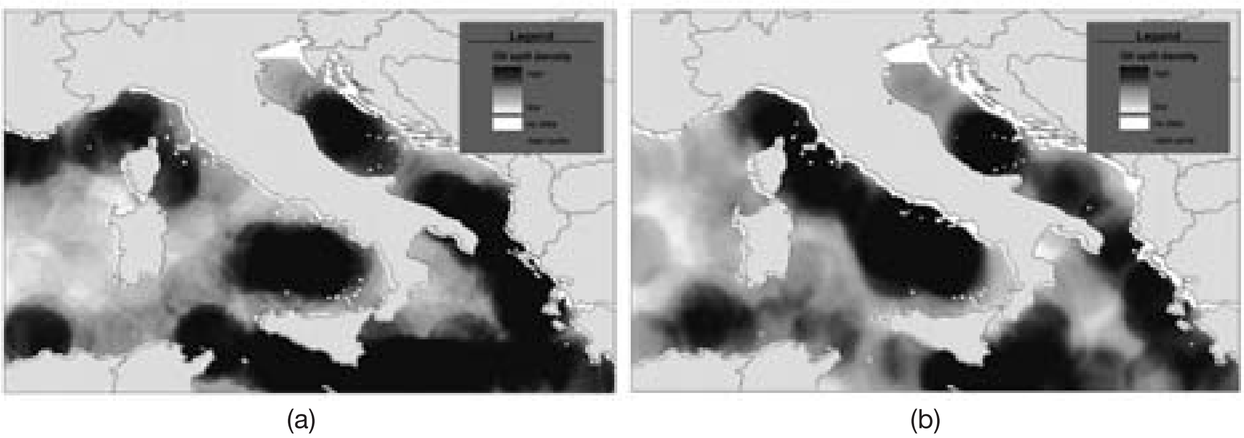
The Adriatic Sea is a well-known hot spot due to its intense traffic. This study showed the presence of 1,049 possible oil spills during the period 1999–2004 (Bernardini *et al*., 2005; Tarchi *et al*., 2006). It was observed that the number of possible oil spills detected in the Italian Search and Rescue Area[2] slightly decreased during the entire period of study,[3] in line with the general trend for the entire basin. In Italian territorial waters, a significant reduction in the number of oil spills—estimated at over 50%—was noted during the period 2003–2004 compared with the previous 3 years (Figure 2.8). Based on these results, Tarchi *et al*. (2006) deduced that the cause for the significant reduction of possible oil spills in Italian waters can be attributed to reasons other than lack of coverage by satellite images.

In 2003, Croatia declared an ecological protection and fisheries zone in the Adriatic Sea that will come into force at some time in the future. The extended jurisdiction will enable the country to exercise those obligations allowed by international law to protect vulnerable marine ecosystems so as to promote efficient and sustainable use of fisheries resources. Tarchi *et al*. (2006) suggest that the declared Croatian zone may be the reason behind the slight increase in detected oil spills in the rest of the Adriatic Sea, even though regular aerial surveillance has not yet been implemented. At the same time, the density of detected oil spills in the Croatian zone exceeds that in the rest of the Adriatic Sea.

5.2.2. Ligurian Sea and the French Environmental Protection Zone (ZPE)

The recently declared ZPE has been designated for the protection of marine biodiversity and fisheries conservation. It is an area where France can implement and enforce laws and regulations in conformity with UNCLOS (Chevalier, 2005).

It is interesting to note that while no variation in the number of possible spills detected in the zone occurred between 2000 and 2003, a decrease of approximately 70% was observed in 2004 (Figure 2.9). This reduction in detected spills was



(a) (b)

Figure 2.8— Normalized probable oil spill density for the Tyrrhenian, Adriatic and part of the Ionian Seas during (a) 1999 and (b) 2004. (Source: Tarchi et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

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2 The Italian Search and Rescue Area, including the territorial waters, covers an area of approximately 52 square degrees in total.

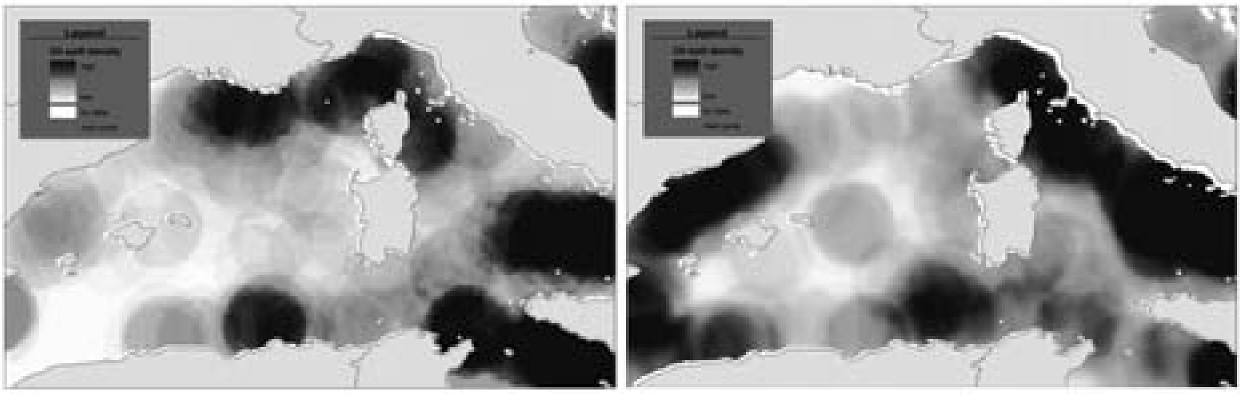
3 There was uniform SAR imagery cover throughout the entire study period over both the Search and Rescue Area and the territorial waters.

interpreted as a result of the declaration of the French zone at the beginning of 2004 (Ferraro *et al*., 2006). The overall results, however, may suggest that the pollution problem has migrated to other areas, although this requires further investigation. Thus, in the area outside the zone, a similar constant trend was observed between 2000 and 2002 with a slight reduction during 2003, but this was then followed by an increase in 2004 (Ferraro *et al*., 2006).

One of the preliminary conclusions made by the study is that the fall in detected oil spills in the French zone can be attributed to the regular aerial surveillance that has been in place in order to implement the new measures. This may also explain the rise in oil spill detection in zones around it.

Overall, these studies have for the first time revealed the realistic extent of the problem of routine illicit operational discharges at a Mediterranean-wide level. This calls for a number of decisive actions in order for ships to conform to international environmental law.

A number of interesting projects focusing on sea-based oil pollution monitoring using remote-sensing techniques have been carried out in the Mediterranean during the last decade.[4] Currently, a two-year project is being implemented with the goal of assessing the feasibility and capabilities of space-borne radar imagery to support aerial and naval monitoring of sea-based pollution in the Mediterranean. The Automated Electronic System for Ocean Pollution (AESOP) is utilizing all the experience gained so far from previous monitoring projects in order to make a significant step forward in combating marine pollution (Ferraro *et al*., 2007).



(a) (b)

Figure 2.9— Normalized probable oil spill density for the Tyrrhenian Sea and Gulf of Lion during (a) 1999 and (b) 2004. (Source: Ferraro et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

4 They include projects funded by the European Commission and the European Space Agency (ESA), such as RAMSES, GAIANET, VASCO and CLEOPATRA, as well as the studies performed by the EC-JRC in the field.

6. Correlation between high-spill areas and oil contamination in water, sediments and biota

Crude oil is composed of thousands of complex gaseous, liquid and solid organic compounds, of which hydrocarbons are the most abundant (Kennish, 1992). Important constituents are alkanes (paraffins), cycloalkanes (cycloparaffins, naphthalenes), alkenes, alkynes and aromatic hydrocarbons, including polynuclear or polycyclic hydrocarbons (Ernst *et al*., 2006).

These compounds form the most toxic part of the oil and toxicity generally increases from alkanes, cycloalkanes and alkenes to the aromatics. Toxicity is mostly related to the low molecular weight of aromatic compounds, such as benzene, xylene and toluene (Durako *et al*., 1993). While some of these substances easily evaporate, others readily dissolve in water and become incorporated into the water-soluble fraction (Zieman *et al*., 1984).
Aromatic hydrocarbons are also emitted through natural processes such as volcanoes and forest fires, but they are increasingly the result of man-made environmental pollution. These contaminants enter the air through the burning of fuels such as coal, oil and gas, and may then attach to solids entering the aquatic environment through fallout or stormwater runoff.

The absorption of this water-soluble fraction by marine organisms represents the route of exposure that generates primary sub-lethal toxic effects in marine organisms, in particular those at the lower levels of the food chain (Wells, 1999). Continued bioaccumulation of these chemical pollutants continues to manifest itself at different levels of biological complexity, from molecules to communities—a phenomenon termed biomagnification (IUPAC, 1993).

Although it is well documented that bioaccumulation phenomena in marine organisms

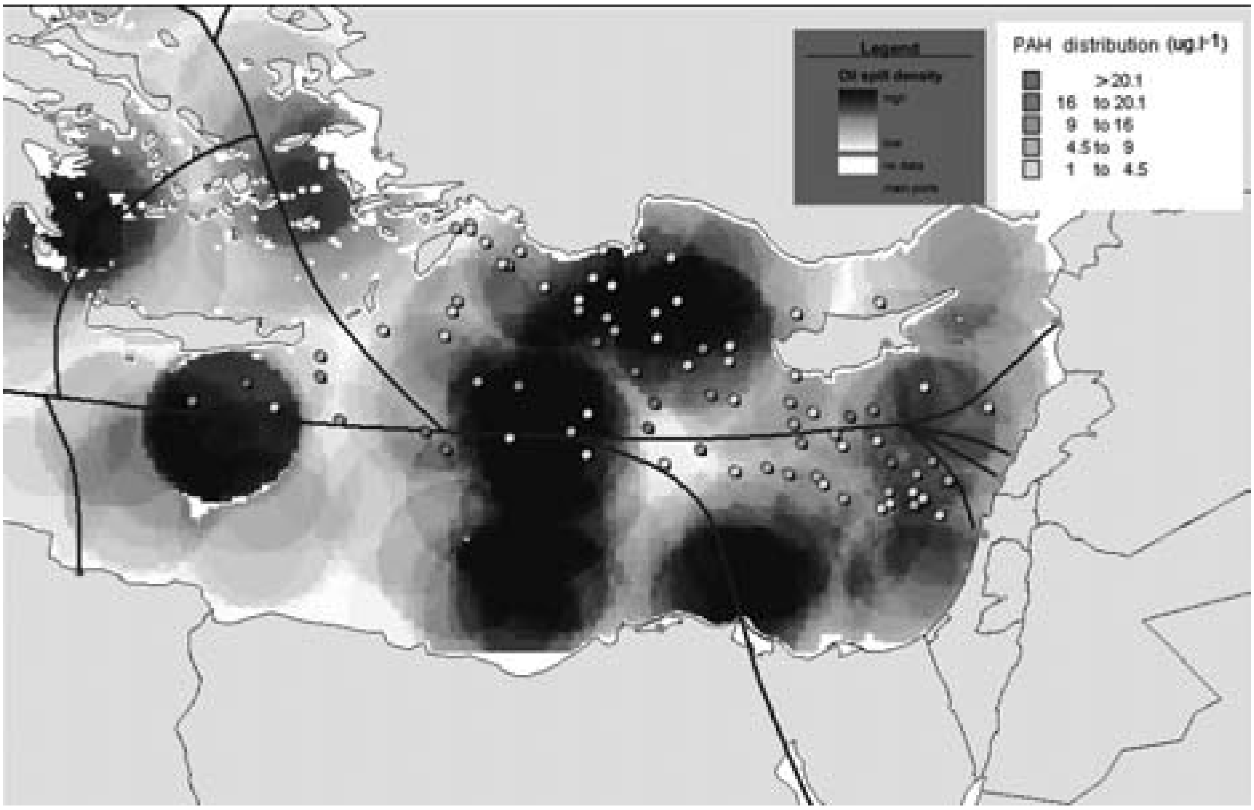


Figure 2.10—Dissolved and dispersed polyaromatic hydrocarbons (PAH) in samples taken at 1m depth superimposed over the main shipping lines and oil spill density detected during 1999 in the Eastern Mediterranean (Source: Tarchi et al., 2006; Ravid et al., 1985; EEA, 1999; For the color version of this figure please refer to the Annex at the end of the document)

may result from food-chain biomagnification processes or from concentration of pollutants by filter feeders, there is a relative paucity of analytical data concerning incidences of biomagnification in the Mediterranean (De Flora *et al*., 1991).

6.1. Water

A large number of determinations of oil contaminants in seawater were performed in the 1980s within the MEDPOL Programme. However, recent data usually refer to individual compounds, and are therefore not comparable with previous bulk data.

In 1985 studies confirmed that all areas investigated in the Eastern Mediterranean were polluted by aromatic hydrocarbons, with concentrations ranging from 0.3 to over 20μg·1^{-1}. The spatial correlation between the distribution of these point concentrations and the main shipping lanes can easily be seen (Figure 2.10).

More recently, levels of PAHs in water samples taken from the North Aegean Sea were found to be lower in range than those observed along the Turkish coast (GEF, 2002). Higher concentrations were observed in offshore samples, which were most probably due to direct discharges from ships. The GEF (2002) reports that the PAH content in the dissolved phase in the open Mediterranean Sea was 0.4–0.9ng·1^{-1}, with values around 2ng·1^{-1} in coastal areas (Table 2.1).

Table 2.1—Concentrations of polycyclic aromatic hydrocarbons (PAHs) measured in seawater collected from different hot spots in the Mediterranean (Source: GEF, 2002).

	Levels of PAHs (ng·l^{-1})
North Aegean Sea	10–30
Turkish coast	20–40,000
İzmit Bay (Marmara Sea)	0.2–7.4
Rhône & Ebro outflows	570–970

The above studies demonstrate the gaps in PAH flux data in the open Mediterranean water column and the almost total absence of flux measurement in deep water. This lack of information limits our understanding of the transport and fate of PAHs, and of the environmental quality of open and deep Mediterranean waters (Bouloubassi *et al*., 2006).

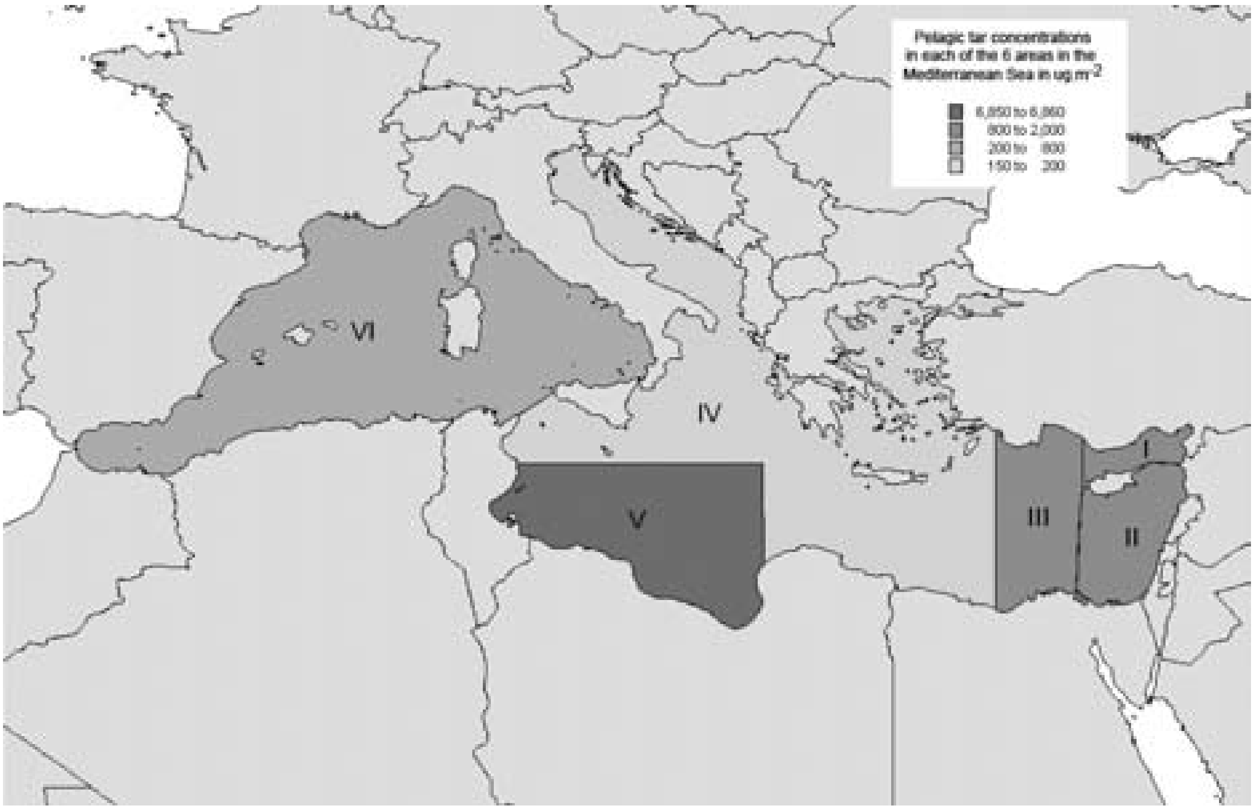


Figure 2.11—Pelagic tar concentrations in the six Mediterranean areas. (Source: Golik et al., 1988; For the color version of this figure please refer to the Annex at the end of the document)

This also hinders the development of a PAH mass budget in the region.

6.2. Pelagic tar balls

Tar lumps in the marine environment are formed from oil or oily compounds which are released into the sea. The light fraction of the oil evaporates, leaving its viscous, heavy fraction floating on the water in the form of tar balls. The release of these tar balls into the water may result from natural, deliberate or accidental causes, and once on the surface, they can drift with tides and currents and pile up in high seas to wash up on shores.

The Mediterranean basin is subdivided into 6 regions. Area I, which includes the Gulf of İskenderun and the tanker lanes leading to the Dortyol oil terminal, were found to be significantly more polluted than Area II, which includes the oil terminals of Syria, Lebanon and Israel (Figure 2.11). Hot spots of tar pollution were identified along the shipping lanes south and southwest of Cyprus, indicating activities in this area. The Gulf of Sirte (Area V) was found to be the most polluted area by Golik *et al*. (1988). It has been suggested that this high spill area originates from the intense activity of the oil terminals on the Libyan coast. It is relevant that at the four main Libyan terminals, namely those in Tripoli, Misurata, Khoms and Zawia, no waste reception facilities are available except for the one in Tripoli, which according to REMPEC lacks 'adequate and organized reception and treatment facilities for oily waste' (REMPEC, 2005).

6.3. Sediments

Polycyclic aromatic hydrocarbons are the main oil-derived contaminants in marine coastal sediments. Their hydrophobic character makes them easily adsorb on to suspended particles, and once in this form they become even more resistent to biological degradation in comparison to dissolved PAHs. This explains why their concentration in sediments may be 100 times or more higher than in the overlying water column (De Luca *et al*., 2005).

The distribution of oil-derived contaminants in Mediterranean Sea coastal sediments has been the subject of several studies. Unfortunately, most of them focus on the north-western part of the Mediterranean Sea (Figure 2.12). The

Figure 2.12—Degree of oil contamination in coastal marine sediments (Source: Zaghden et al., 2005; Kucuksezgin, et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

concentrations of PAHs in sediments vary greatly, depending on the proximity of the sampling site to the source and the degree of exposure to contaminants. However, the reported values do not provide the level of contamination by particular species of PAHs, and therefore the relative toxic equivalency factor (Sprovieri *et al*., 2007) for the region cannot be estimated. Different PAH species vary in their degree of bioavailability, and this in turn is affected by their degree of sequestration and metabolism by biota.

Woodhead *et al*. (1999) refer to studies showing that levels of total PAHs in marine sediments as low as 1,000 and 3,100ng·g^{-1} dry weight respectively are able to induce biological effects in winter flounder and spot that are considered to be early warning signals of detrimental changes to biological systems. They also mention that sediment total PAH levels as low as 181 and as high as 41,200ng·g^{-1} dry weight have been linked to carcinogenesis in fish.

6.4. Biota

Organisms are routinely collected from coastal areas for water-quality monitoring, which may also include the quantification of oil contaminants in their tissues. Marine organisms tend to concentrate such contaminants in their tissues, either directly from the water column, or through ingestion of food and sediments. In the Mediterranean, the mussel *Mytilus galloprovincialis* and the benthic red mullet *(Mullus barbatus)* have been largely used for such monitoring, since they tend to reflect the level of contamination of the water and sediments in coastal areas (EEA, 1999).

Concentrations of PAHs in marine organisms indicate a high degree of contamination that can be correlated with hot spots, including oil refineries, terminals and ports (Narbonne *et al*., 2001) (Figure 2.13). Systematic monitoring has been carried out mostly along the north-western part of the Mediterranean coast, reflecting a significant data gap for other parts of the region.

Sediment quality thresholds have been assigned by Johnson *et al*. (2002), linking the levels of PAHs found in sediments and biota to their observed effects in English sole. At concentrations greater than 1,000ng·g^{-1} dry weight, there appears to be a substantial increase in the risk of liver disease and reproductive impairment, and potential

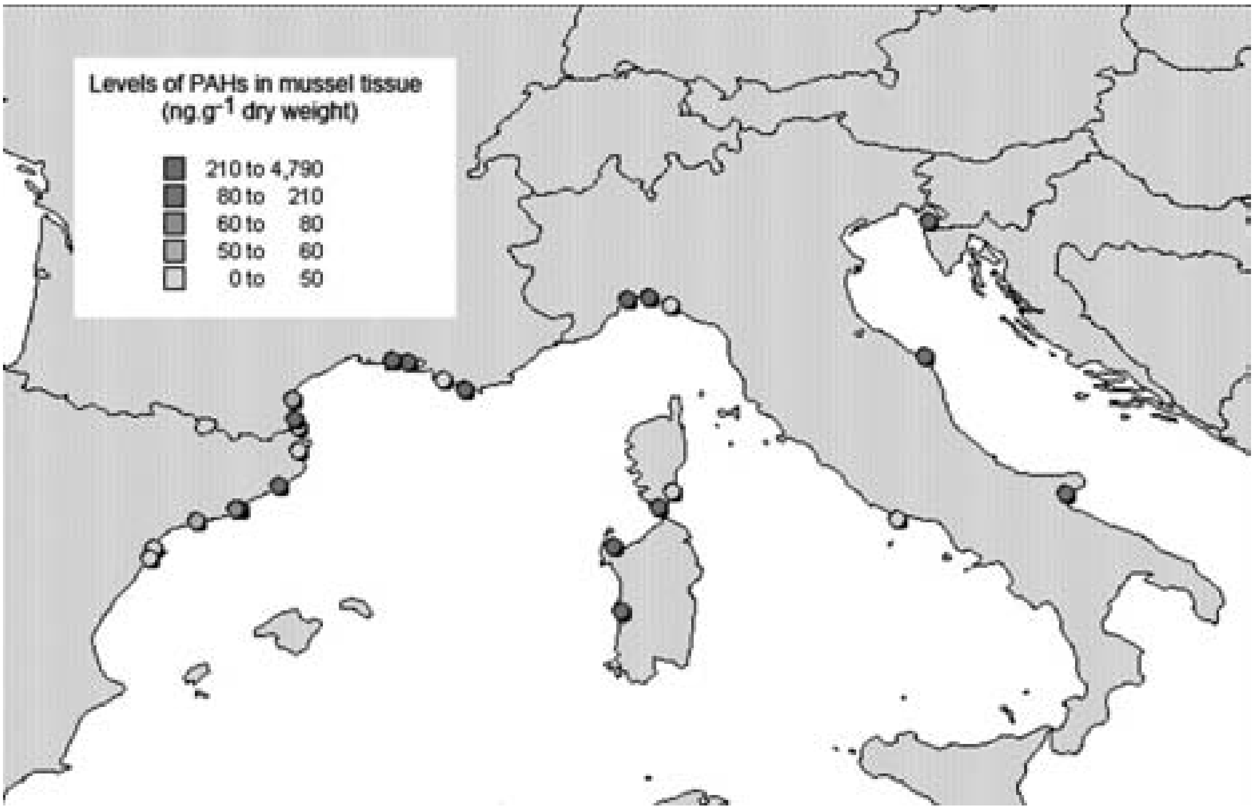


Figure 2.13—Distribution of polycyclic aromatic hydrocarbon concentrations in mussels collected from hot spot areas in the Mediterranean (Source: Narbonne et al., 2001; Zorita et al., 2007; For the color version of this figure please refer to the Annex at the end of the document)

effects on growth. At concentrations higher than 5,000ng·g^{-1} dry weight, the levels of hepatic DNA adducts would be approximately 10 times the normal levels. At 10,000 ng·g^{-1} dry weight, 50% of the fish would be expected to have liver disease, nearly 30% of the females would show inhibition of gonad growth, and over 40% would show inhibition of spawning (Figure 2.14).

In the case of benthic invertebrates (Long *et al*., 1995), total PAH concentrations (dry weight basis) below the 'effects range-low' value (4,022ng·g-$^{-1}$) are interpreted as being rarely associated with adverse effects; levels between the 'effects range-low' and 'effects range-median' values (44,792ng·g^{-1}) are occasionally associated with adverse affects; and total PAH concentrations above this threshold are interpreted as being frequently associated with adverse effects.

Data on PAH levels in the deep-sea fish *Mora moro* are reported by Solé *et al*. (2001). Fish samples collected from the Gulf of Lion (NW Mediterranean) at an approximate depth of 1000m showed a bioaccumulation trend in muscle, gills, digestive tube and liver tissues, in decreasing order. The levels were low compared to those found in shallower-water fish, but of the same order as those reported for other deep-sea fish, except for those collected in the Arctic. It is interesting to note that most determinations of organic pollutants in NW Mediterranean fish have been conducted in coastal areas, and this study is the first comprehensive study of a deep-sea species in this region, where little is known about their pollutant burden.

Based on the above toxicity thresholds and on the levels shown in Figure 2.12, the PAH levels in Mediterranean coastal sediments are suggestive of the induction of detrimental effects in marine organisms, particularly fish. However, the level in marine invertebrates suggests lower induction of adverse effects along the selected Mediterranean coasts.

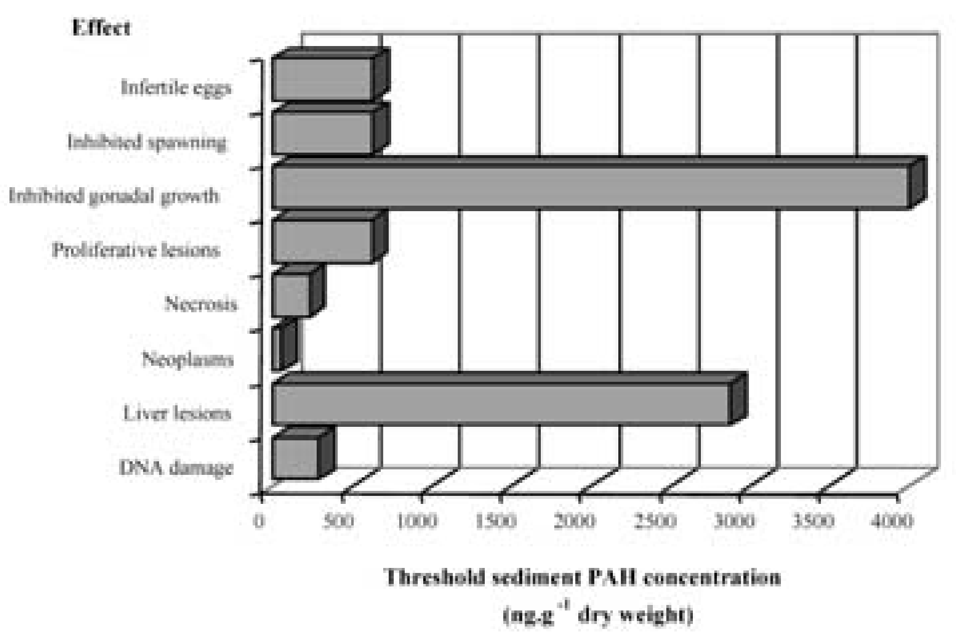


Figure 2.14—Relationship between sediment PAH concentration and effects on biota (Source: Johnson et al., 2002; For the color version of this figure please refer to the Annex at the end of the document)

7. Ecological and physiological effects of oil on marine organisms

7.1. Levels of toxicity

Whereas mortality is a good indicator of the level of toxicity because it is easy to detect, its significance for ecological management is questionable due to its irreversible nature. Detecting carefully studied enzymatic, metabolic or physiological stresses resulting from sublethal toxicity levels can give a clearer indication of the onset of damage, at which point the organisms may still be able to survive. Based on this, the detection of sublethal toxicity effects provides an important early-warning signal of the detrimental impact on biological communities and marine ecosystems as a whole.

7.1.1. Sublethal toxicity in biological systems

7.1.1.1. Genetic level

Several studies show that PAHs may induce genetic damage in living organisms, even at environmentally low concentrations (Cavallo *et al*., 2006, Bolognesi *et al*., 2006a; Cebulska-Wasilewska *et al*., 2005). Such damage includes DNA base modification, strand breaks, depurination and cross-linkages. Organisms have their own DNA repair mechanisms, but when these do not function efficiently DNA mutations are induced. This may ultimately result in chromosomal aberrations, birth defects and long-term effects, such as cancer in vertebrates (UNEP, 1995). For example, Singh (2000) observed a direct relationship between DNA damage in erythrocytes extracted from bullheads (*Ameiurus nebulosus*) and carp (*Cyprinus carpio*) and contaminant levels in aquatic sediments.

Based on its sensitivity, the degree of DNA integrity has been proposed as an indicator of genotoxicity and an effective biomarker for environmental monitoring (Shugart, 1990). Specific techniques, such as the comet assay (Frenzilli *et al*., 2004), are now available to ecotoxicologists to evaluate the impact of PAHs upon marine organisms.

The outcome of a long-term biomarker study to evaluate the long-term risk and impact of the 1991 *Haven* oil spill disaster along the Ligurian coast of Italy[5] revealed that many years after the accident significant genotoxic damage is still measurable in various indicator species living in the affected area (Bolognesi *et al*., 2006b). This study used mussels, oysters, and fish with different feeding habits. Ten years after the accident, a statistically significant increase in DNA damage and micronucleus frequency is still evident in samples collected from different areas of the wreck. These results indicate that a residual source of carcinogenic and mutagenic compounds still exists in the area, with adverse impacts on the marine ecosystem.

7.1.1.2. Cellular, biochemical and physiological level

It has been shown that the water-soluble fraction of PAHs preferentially accumulates in membrane lipids and other lipidic compartments (Di Toro *et al*., 2001), and in doing so it has the potential to disrupt the biochemical and physiological properties of cell membranes causing toxic effects (Lavarías *et al*., 2007). Changes in the composition and packing of cell membranes, have been shown to affect enzymatic and receptor activities (Spector & Yorek, 1985) and the immunological system (Reynaud & Deschaux, 2006). Other studies have demonstrated the relationship between exposure to petroleum hydrocarbons and haemolysis, and an increase in the haematocrit of fish exposed to the water-soluble fraction (Simonato *et al*., 2008).

5 On 14 April 1991, the Cypriot oil tanker *Haven*, carrying 144,000 tonnes of Iranian heavy oil, sank near the port of Genoa, Italy, after having been on fire for three days. The accident was probably due to an explosion occurring during tank-cleaning operations, and caused the death of five crew. Much of the oil was consumed during the fire, and an estimated quantity of 10,000 tonnes was spilt in the sea, whilst an additional 90,000 tonnes sank with the ship. The *Haven* is now lying at a depth of 75m one mile off Arenzano, a town close to Genoa. This is the worst oil tanker accident that has ever happened in the Mediterranean Sea and is one of the few cases where oil has affected subtidal sediments.

Chronic exposure to oil pollutants can also produce an increase in oxygen-derived free radicals, generating oxidative stress in organisms (Di Giulio *et al*., 1993). In addition, numerous field (e.g. Porte *et al*., 2002—Figure 2.15a) and *in vitro* studies (Bonacci *et al*., 2003) have demonstrated that these pollutants can induce the activity of 7-ethoxyresorufin-O-deethylase (EROD), which can be used as an indicator of the inductive response of the cytochrome P4501A1 (CYP1A1) in response to PAH pollution (Goksoyr & Forlin, 1992), including benzo[*a*]pyrenes and beta-naphthoflavone (Bonacci *et al*., 2003). P-450 dependent microsomal monooxygenase enzymes form part of the detoxification mechanisms to biotransform xenobiotic compounds. EROD was successfully used to monitor oil exposure related to the *Erika* and *Exxon-Valdez* oil spillages (Jewett *et al*., 2002).

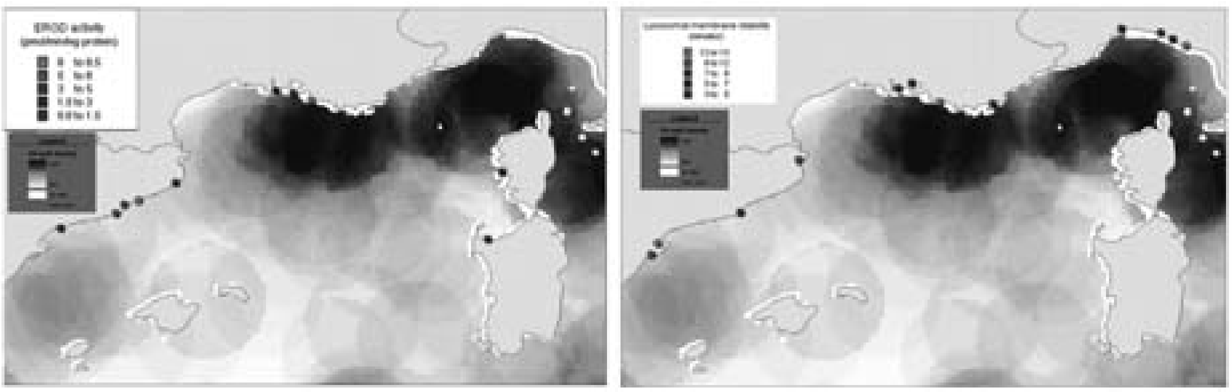
Contaminant-induced alterations in cell structure and function have been investigated by several studies, in particular the liability of lysosomal membrane to respond to chemical exposure (Martins *et al*., 2005). Based on this sensitive response, Lowe and Pipe (1994) developed a biomarker method based on lysosomal uptake, retention, and reflux of a red dye,[6] which appears to be more closely linked to organic chemical pollution (Domouhtsidou & Dimitriadis, 2001). The application of this technique to bivalve haemocytes is non-invasive (Lowe *et al*., 1995), and studies have demonstrated its utility in identifying pollution problems related to PAHs (e.g. Zorita *et al*., 2007) (Figure 2.15b). However, when using this biomarker, results should be interpreted with caution since it is likely that organic chemical pollutants can exert biphasic effects, stabilizing lysosomal membranes at low exposures and destabilizing them at high exposures (Moore, 1988; Lekube *et al*., 2000).

These reactions at the biochemical and physiological level may act as the starting point for a sequence of functional alterations that may alter vital functions, affecting the survival of the organisms and damaging populations and communities (Martínez-Gómez *et al*., 2006).

7.1.1.3. Organ level

Teleost hepatic lesions—including neoplastic, pre-neoplastic, focal and necrotic lesions resembling those experimentally induced in fish by chronic exposure to PAH-contaminated sediments and diets—are commonly detected in marine bottom fish from contaminated environments (Myers *et al*., 1998). Supported by other field studies (Moore & Myers, 1994), specific histopathological changes caused by xenobiotic exposure have been recognised as valuable tools for detection of adverse chronic effects of contaminant exposure and uptake on marine organisms (Myers *et al*., 1998).

Nine years after the *Haven* accident, which took place in front of the port of Genoa in 1991, fish collected from the accident area in Savona still exhibited evidence of liver damage, indicating the continued presence of xenobiotics. Significant tar masses are still seen on the sea floor in the area and it is reasonable to argue that fish living there



(a) (b)

Figure 2.15—(a) EROD activity and (b) lysosomal membrane stability, measured in marine organisms collected from selected hot spots and control sites in the north-western Mediterranean basin. The point measurements are overlaid on the map of oil spill density in 1999 (Source: Zorita et al., 2007; Porte et al., 2002; Ferraro et al., 2006; For the color version of this figure please refer to the Annex at the end of the document)

6 Known as the neutral red retention (NRR) assay.

are being constantly subjected to low, chronic doses of oil pollutants (Pietrapiana *et al*., 2002).

7.1.1.4. Behavioural response

Behavioural studies on the fiddler crab *Uca pugnax* found that crabs exposed to oil-polluted marsh environments, resulting from an oil spillage 38 years ago, avoided burrowing into oiled layers, suffered delayed escape responses and lower feeding rates, and achieved lower densities (Culbertson *et al*., 2007). Knowledge on the long-term consequences of spilt oil should be taken into account when assessing oil-affected areas in terms of restoration and rehabilitation.

7.1.1.5. Community level

It is evident that in order to better understand the effect that oil pollution has on marine assemblages, knowledge of toxicity in individual organisms needs to be supplemented with information on its impact on many other levels, such as physiology, behaviour and predator-prey interactions under various exposure scenarios (Yamamoto *et al*., 2003; Koshikawa *et al*., 2007). Changes in community structure after disturbance are the result of the competitive selection of the most tolerant species (Damasio *et al*., 2007).

Mesocosm studies, which are useful intermediates between bioassays and ecosystem experiments, have yielded important information on the direct toxic effects of oil pollution on phyto- and zooplankton assemblages as well as its indirect effects on food-web interactions (Siron *et al*., 1991), including microbial food webs (Koshikawa *et al*., 2007).

Applying mescocosm studies, Ernst *et al*. (2006) observed a dual response to oil-induced pollution by foraminifera. While the mortality of certain taxa increased, the growth patterns of deformed formaniferal tests due to exposure were anomalous, and led to changes in the taxonomic composition of the assemblage.

Following the *Agip Abruzzo* oil spill that occurred outside Livorno port in April 1991, Danovaro *et al*. (1995) observed that meiobenthic assemblages situated in the Ligurian Sea appeared to be highly sensitive to hydrocarbon stress in the area. A decline in population density was observed in particular with nematodes, turbellarians and foraminifera. The oil-induced disturbance had no influence on copepods, which increased slightly after the oil spill, probably because they were dominated by epibenthic forms which were able to escape from the toxic effects of hydrocarbon pollution. The decline in the population of the non-selective, deposit-feeding nematodes may indicate possible repercussions on the functional characteristics of the meiobenthic assemblages. It is interesting to note however, that afterwards the meiobenthos assemblages seemed to recover from the contamination shock, indicating their high resilience to the oil pollution accident.

Similarly, Johansson *et al*. (1980) observed that primary production and the phytoplankton biomass exposed to the spill of No 5 fuel oil from the tanker *Tsesis* near Stockholm in 1977 did not decrease, but on the contrary increased. This effect was attributed to the decline of the zooplankton biomass and hence a reduction in grazing.

Other studies show, however, that at least two to three years, and sometimes even up to fifteen years (Hawkins *et al*., 2002), are needed for intertidal animal communities to recover to their original level after being exposed to oil spills . Moreover, the impact of oil on benthic animals usually varies from species to species. Species such as *Cellana* and *Monodonta* spp. have been observed to increase rapidly after an oil spill, whereas others such as patellids and *Septifer* spp. do not show any apparent temporal trends. These differences in response to oil pollution among benthic animals are considered to be caused by the differences in habitat use, susceptibility to heavy oil, life history and ability to migrate (Yamamoto *et al*., 2003).

Such results derived from community-based indices should, however, be treated with caution. Investigators claim that these indices should not be considered reliable indicators of ecological impairment caused by specific contaminants (Baird & Burton, 2001) since they may be responding to a variety of disturbances acting in tandem at the community level (Damasio *et al*., 2007).

The long-term impact of benthic tar aggregates deriving from oil spills (such as from the 1991 *Haven* accident) on soft-bottom macrobenthic communities reveals significant resilience. Guidetti *et al*. (2000) observed no significant differences in abundance of either the whole benthic assemblage or the main benthic taxonomic groups between contaminated and control sites. Such real case studies suggest that tar aggregates do not induce appreciable detrimental effects on the soft-bottom macrobenthos.

Clear evidence of detrimental impacts of the *Haven* spill, however, comes from the study conducted by Peirano *et al*. (2005). They observed that the shoot density of the shallow-water *Posidonia oceanica* meadows sampled along 300km of the Ligurian coast eight years after the accident pointed to a generalized state of regression. In samples from the Arenzano area, where the spilt oil washed ashore during the accident, no rhizome more than 8 years old was found, thus confirming the shoot mortality induced by the oil spill event.

8. Significance of ship-generated emissions on the marine environment

The continuous growth of the global marine shipping sector has been accompanied by a corresponding increase in the sector's contribution to global atmospheric emissions, which so far has only been moderately controlled (Friedrich *et al*., 2007). A briefing report issued by the European Environment Bureau together with other agencies states that, at the European level, ocean-going vessels contribute significantly to global emissions of sulphur oxides, nitrogen oxides and particulate matter (EEB *et al*., 2004). Moreover, it states that even after accounting for enforcement of MARPOL Annex VI, which sets limits on the sulphur content of marine fuels for its 'special areas', emissions of SO_2 from international shipping are expected to increase by more than 42% by 2020, and those of NO_x by two-thirds. In both cases, the estimates are that by 2020 the European nitrogen and sulphur oxides inventories from shipping will exceed the total emissions from all land-based sources, whether mobile or stationary, generated by its 25 member states combined. On a regional scale, sulphur emissions contribute to acid rain.

The cause of such pollution stems from the fact that cargo ships are powered by diesel engines, which account for around 17% of NO_x emissions on summer days in certain regions of the world (Gallagher, 2005). Studies show that ship emissions are most evident in the Northern Hemisphere oceans, where over 60% of the SO_2 concentration in the atmosphere and 30% of all sulphates can be attributed to ships. Except for the area around Australia, the Southern Hemisphere oceans are almost unaffected. This is because of the heavier shipping traffic that occurs in the North (Capaldo *et al*., 1999).

The impact of ship-generated emissions on air quality is significantly felt in port cities and in states with extensive coastlines adjacent to shipping lanes, often resulting in higher public health risks in many regions (EEB *et al*., 2004). Studies conducted by the International Maritime Organization show that about 70–80% of all ship emissions occur within 400km of the coast (IMO, 2000).

IMO's inventories show that global shipping is a small contributor to the world's total CO_2 emissions, however, accounting for just 1.8% in 1996. A significant reduction of 10% in CO_2 emissions from shipping would represent less than a 0.2% reduction in total world emissions (IMO, 2000).

8.1. Estimates of emissions resulting from shipping activities in the Mediterranean

The amount of exhaust emissions from marine engines can be deduced using emission models. These models are based on emission factors adopted from onboard engine measurements and theoretical factors derived from the respective chemical reaction equations,[7] which are then combined with actual global fuel consumption. According to the Global Environment Facility (GEF, 2002), these model emission factors often generate a certain degree of uncertainty, which can only be fine-tuned if countries provide more complete information on emissions (such as estimation techniques used, sector-specific emissions, etc.)

Figure 2.16 summarizes the pollutants emitted by each vessel type in tons per day for 2004. These data are for the total emissions generated by the main and auxiliary engines working at a typical cruising speed. From these data, it is evident that container ships, followed by tankers, generate most emissions when compared to other vessel types.

The main factors affecting ship emissions include engine type, meteorological factors, fuel quantity

7 Estimated from current best-practice models based on route distance, speed, fuel type, ship characteristics, and ship operating profile (Fournier, 2006).

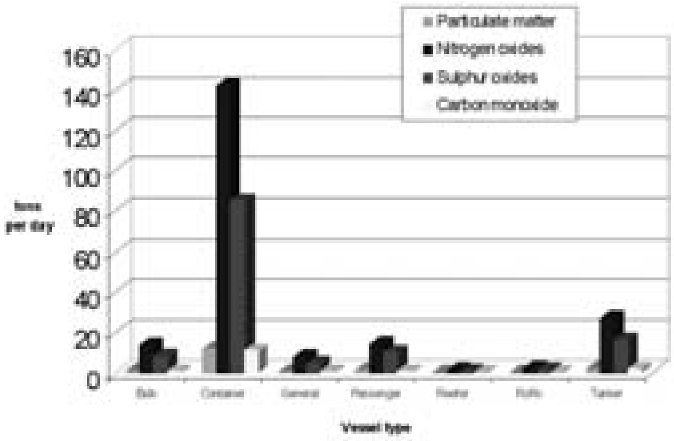


Figure 2.16—Air pollutants emitted by each vessel type (tons per day) for 2004. (Source: California Air Resources Board, 2005; For the color version of this figure please refer to the Annex at the end of the document)

and quality (Krozer *et al*., 2003), ship operational mode, age of the engine and use of emission reduction technologies (Cooper & Gustafsson, 2004).

As already indicated in this chapter, around 250–300 oil tankers operate in the Mediterranean every day (UNEP, 2006). Based on the above estimates (Figure 2.16), these ocean-going vessels could potentially emit around 7,400 tons per day of NO_x and 4,400 tons per day of SO_2 in the Mediterranean basin.

These figures are supported by the study conducted by Georgakaki *et al*. (2005) using EUROSTAT data to calculate emissions arising from maritime transport during the year 2000 in EU waters. The estimates (within the stated limitations of the study itself) for the emissions generated within EU15 (except for Italy) are in the region of 1 million tons of SO_x and around 1.5 million tons of NO_x. The traffic between EU15 and the accession countries was estimated to have generated levels of just under 300,000 tons and 400,000 tonnes of SO_x and NO_x respectively in the year 2000.

In a study conducted by Lloyd's Register for the European Commission, Lavender (1999) similarly estimated that on an annual basis ships in passage or arriving at or departing from a berth in the Mediterranean and Black Seas emitted approximately 1,725,000 tons of NO_x, 1,246,000 tons of SO_2, 147,000 tons of CO and 35,000 tons of hydrocarbons.[8] The overall level constituted around 87% of the emission totals for the North-East Atlantic region previously estimated by Lloyd's Register. This study shows a strong relationship between ship traffic density and pollutant emissions, with the highest emissions occurring along the Atlantic–Suez corridor and the coastal routes around the Gulfs of Valencia and Lion, as well as in the Ligurian, Tyrrhenian and Adriatic Seas. It is important to note that emissions from fishing and naval vessels were omitted from this study.

In a separate investigation to study vessel emissions in the Mediterranean Sea during a 21-month period, Marmer and Langmann (2005) found that, due to the high vessel traffic, the region exhibits mean summer sulphate levels (7.8mg·m^{-2}) that are much higher than those found throughout the rest of Europe (4.7mg·m^{-2}). By focusing on the emissions generated by vessels, the authors found that 54% of the total sulphate aerosol column over the region during summertime was due to ships, which in turn contributed over 50% to the direct radiative forcing in the area. When the model was run without the ship emission contributions there was a significant reduction of sulphates (29%), ozone (15%), and nitric acid (66%), amongst other contaminants.

Kesgin and Vardar (2001) calculated NO_x emissions in the Bosphorus to be 12,818 tons emitted from transit ships passing through the Straits in 1994 (Figure 2.17). The amount of nitrogen oxide emissions from domestic passenger ships used for public transport in the Istanbul Strait is equal to approximately 4% of the nitrogen oxide emissions from motor vehicles in Istanbul. Values estimated by Trozzi *et al*. (1995) show the importance of atmospheric emissions in locations such as the

8 A number of key parameters needed for the calculation of ship-generated emissions were either unavailable or unsuitable due to limitations in time, resources and data-handling capabilities, and a number of assumptions and exclusions were made (Lavender, 1999).

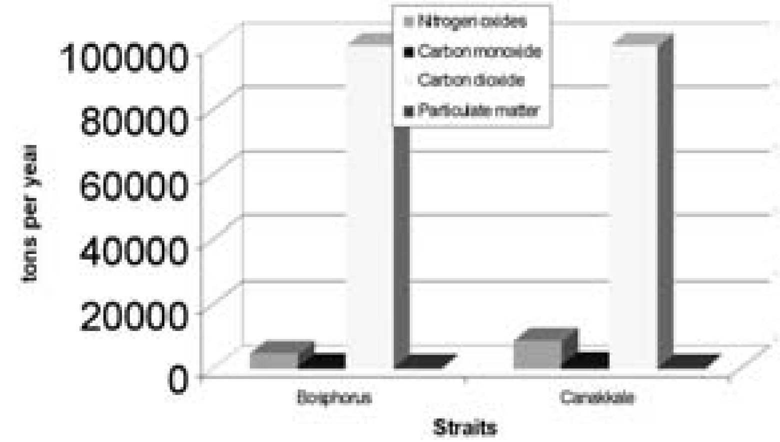
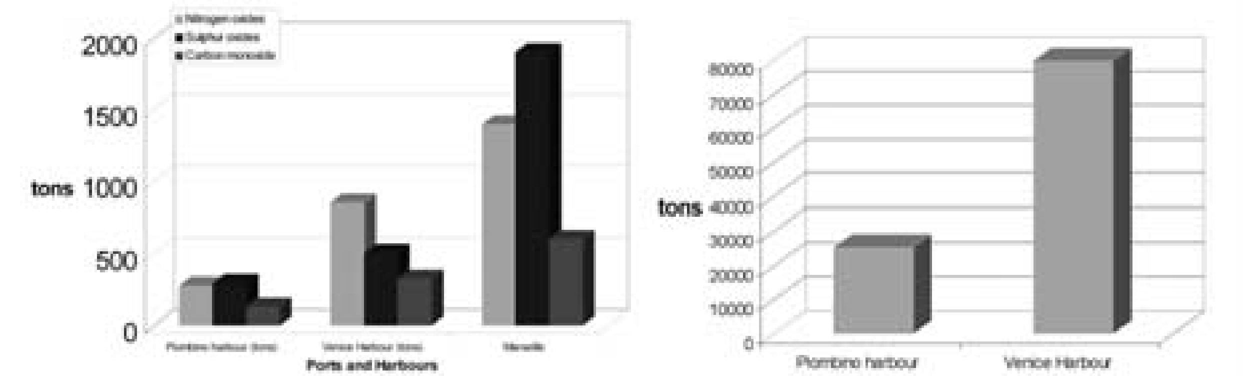


Figure 2.17—Estimated yearly pollutant emissions from transit ships in the Turkish Straits. (Source: Kesgin & Vardar, 2001; For the color version of this figure please refer to the Annex at the end of the document)



(a) (b)

Figure 2.18—Annual emissions in Mediterranean harbours: (a) NO_x, SO_x and CO in Piombino, Venice and Marseille, and (b) emission of CO_2 in Piombino and Venice harbours. (Source: Trozzi et al., 1995; For the color version of this figure please refer to the Annex at the end of the document)

harbour of Venice (Figure 2.18), which is situated in a particular natural environment (lagoon).

8.2. Impact of ship-generated emissions

8.2.1. Climate change and radiative forcing

Indirect estimates of radiative forcing as a result of ship-generated CO_2 emissions indicate that ships may account for 1.8% of current IPCC estimates (IMO, 2000). NO_x emissions are highly likely to produce non-zero, positive forcing effects that will contribute to global warming and that could be in the same range as (or longer than) direct forcing from CO_2. It has been deduced that, globally, the warming effects due to direct CO_2 emissions and tropospheric ozone may be offset by cooling effects from ship NO_x emissions and indirect cloud effects from ship sulphur aerosol emissions. However, it is important to note that these estimates are highly uncertain and much more research is required before these initial conclusions can be considered reliable (IMO, 2000).

Ship-generated emissions of sulphur aerosols are known to contribute to the formation of clouds over the ocean (Kaufman *et al*., 1991) as they have a significant role in the formation of aerosols on which water condenses to form clouds. The interactions between aerosols and clouds constitutes one of the most important uncertainties in the understanding of the rate of climate change, since clouds reflect energy and thus reduce the warming effect of greenhouse gases (Graf, 2004).

8.2.2. Ocean acidification and its impact on marine organisms

Chemical reactions in the atmosphere can convert emitted NO_x and SO_x into nitric and sulphuric acids respectively. The process of altering the gases into their acid counterparts can take several days, during which these pollutants can be transported hundreds of kilometres from their original sources.

The SO_2 emitted into the atmosphere is responsible for 60–70% of the acid deposition that occurs globally (Environment Canada, 1999). More than 90% of the sulphur and 95% of the NO_x in the atmosphere is of human origin, but their concentrations are much lower than that of atmospheric CO_2, which is mainly responsible for making natural rainwater slightly acidic. However, their effective impact on acidification comes from the fact that these gases are much more water-soluble than CO_2 and therefore have a much greater effect on the pH of rainwater.

The oceans are naturally alkaline, with an average pH of around 8.2, although this can vary by up to 0.3 units depending on location and season. When CO_2 dissolves in seawater it forms a weak acid, lowering the pH. The rate at which extra CO_2 is being injected into the oceans far exceeds the rate at which natural processes can neutralize its acidity. Studies show that the average pH of the oceans has already fallen by about 0.1 units, or a 30% increase of hydrogen ions, compared with pre-industrial levels. Even if all carbon emissions stopped today, it would still take thousands of years for the oceans to recover from this acidification. On the other hand, if global emissions of CO_2 continue to rise, the average pH of the oceans could fall by 0.5 units by 2100, equivalent to a threefold increase in the concentration of hydrogen ions (Caldeira & Wickett, 2003).

Little is known about how marine organisms will be affected by both long- and short-term ocean acidification, increased CO_2 influx and other pH-associated changes to ocean chemistry (Turley *et al*., 2006). However, the likely scenario is that marine organisms will be affected on a number of levels, including acid–base regulation, growth, reproduction, feeding and ultimately mortality.

Marine organisms which rely on the production of calcium carbonate shells and tests, including corals, molluscs, crustaceans and calcified algal species, are the most vulnerable creatures to acidification. Under normal conditions, their shells and tests do not dissolve because the upper layers of the oceans are supersaturated with calcium carbonate. However, acidification reduces carbonate ion concentrations, making it harder for these organisms to construct their shells. When the sea's carbonate ion concentration drops below saturation point, these structures start to dissolve (Orr *et al*., 2005). If atmospheric CO_2 concentrations reach 800ppm (in 2004 they stood at 377ppm) the corresponding drop in carbonate ion availability in surface waters worldwide will be around 60% (Feely *et al*., 2004). This fall in carbonate availability means organisms will be less able to deposit calcium carbonate structures and, if the reduction in carbonate availability is severe enough, calcite dissolution can occur (Raven *et al*., 2005).

Their poor capacity for ionic regulation and dependency on a magnesium calcite test, among other reasons, make echinoids particularly vulnerable to ocean acidification. It has been shown that chronic ocean acidification to below 7.5 would be severely detrimental to the acid–base balance of intertidal species such as *Psammechinus miliaris* despite its ability to tolerate fluctuations in pCO_2 and pH in its environment (Miles *et al*., 2007).

8.3. Reduction of ship-generated emissions

Sea-borne trade has been confirmed to be a significant contributor to the development of environmentally sustainable transport. Even if specific air pollutants are emitted more by shipping than any other transportation means, its relatively lower energy consumption is still a driving force that promotes sea-borne transportation in an inter-modal transportation chain (IMO, 2000).

A series of actions have been proposed by the IMO (2000) to reduce greenhouse gas emissions through technical and operational measures. Actions taken towards better hull and propeller design are identified as general measures for the saving of energy. Other measures related to machinery have been identified and could lead to reductions of certain types of emissions at the expense of increased emissions of other gases.

Studies conducted by the IMO show that the single main measure that could result in the greatest emission reduction is to reduce the speed of ships. The implementation of new and improved technologies has been deemed the second best alterative to reduce emissions. However, at the same time, it is very difficult to achieve the first measure through effective policy instruments (such as market-based approaches) because of the expected increase in demand for international shipping services in the years to come. This means that the total emissions from international shipping for the coming years will be on the increase (IMO, 2000).

However, these technical and operational measures have a limited potential for contributing to a reduction in ship-generated emissions. The IMO stresses that if the present demand for more shipping and increased speed remain, then technical measures alone will not be able to prevent overall growth in vessel emissions.

The International Council on Clean Transportation has made a number of recommendations to accelerate the use of cleaner marine fuels and the widespread deployment of existing pollution control technologies and emission reduction strategies to improve the environmental performance of the shipping sector (ICCT, 2007). Its policy recommendations are aimed at achieving steady, incremental progress in reducing emissions from marine vessels that will result in significant environmental and public health benefits in the short, medium and long terms. The ICCT claims that the proposed recommendations could be implemented in each of several distinct categories, including (a) marine fuels, (b) new engines, (c) new vessels, (d) existing engines and vessels, (e) greenhouse gas emissions and (f) in-port emissions.

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Chapter 3

Impacts of shipping on the biodiversity of large marine vertebrates: persistent organic pollutants, sewage and debris

Maria Cristina Fossi and Giancarlo Lauriano

1. Introduction

Although the Mediterranean occupies only 0.7% of the area of the world's oceans, it is a major reservoir of marine and coastal biodiversity, with 7.5% of the world's marine fauna and 18% of its marine flora; 28% of its species are endemic. This small semi-enclosed sea has many islands and underwater beds and is also a major wintering, reproduction and migration area (RAC/SPA, 2007a).

Intensive unbridled urbanization, over-exploitation of resources, maritime transport and pollution have led directly to loss of biodiversity. At least 306 animal and plant species are now threatened in the Mediterranean (RAC/SPA, 2007a).

Over the past half century, shipping has increased enormously in the Mediterranean Sea. In the last two decades there has been a 77% increase in the volume of ship cargo loaded and unloaded in Mediterranean ports. An estimated 200,000 commercial ships cross the Mediterranean annually and this volume is expected to grow three- or fourfold in the next 20 years (Dobler, 2002).

Because the Mediterranean Sea is a land-locked basin with little water exchange, it is highly subject to pollution by natural contaminants such as heavy metals and PAHs (Augier *et al*., 1993; Monaci *et al*., 1998; Marsili & Focardi, 1997; Marsili *et al*., 1997) and also by xenobiotics such as persistent organic pollutants (POPs). Organisms at the top of the marine food web and/or long-living animals, such as marine birds and reptiles, large predatory fish and odontocete cetaceans, are the most exposed to contamination through biomagnification (Corsolini *et al*., 2000). Recent studies on Mediterranean swordfish have shown that typically female proteins (vitellogenin and zona radiata proteins) are being induced in adult males (Fossi *et al*., 2001a, 2004a, 2006a). This is an early warning signal of the risk to which the reproductive function of Mediterranean organisms is exposed and a threat to the region's biodiversity.

The charisma of top vertebrate predators is often used by conservationists as a lever for financial support and to establish protected areas—a strategy that has been criticized (*see* Sergio *et al*., 2005). Sergio and colleagues (2005) consider that conservation focusing on top predators can be ecologically justified because it delivers broader biodiversity benefits. They report that sites occupied by top predators are consistently associated with high biodiversity.

This chapter reviews the main impacts of discharges from shipping vessels (POPs, sewage and plastic refuse) on Mediterranean fauna, particularly large, long-living marine vertebrates, including top predators such as odontocete cetaceans, pelagic fish and turtles. The final section of the chapter is devoted to recommendations and outlook.

2. Persistent organic pollutants and their ecological effects on top predators

Persistent organic pollutants (POPs) are chemical substances that persist in the environment, bioaccumulate through the food web, and threaten human and environmental health.

POPs are ubiquitous pollutants in the marine environment, particularly in industrialized coastal areas with shipping traffic. POPs such as organochlorine (OC) pesticides, polychlorinated biphenyls (PCBs) and polybrominated diphenyl ethers (PBDEs) have been implicated in disruption of endocrine systems in a wide range of marine organisms (Tanabe, 2002). Sources of organochlorines are mainly agriculture and sanitary uses. PCBs had a wide range of industrial applications as dielectric and hydraulic fluids prior to being banned in many countries. All these compounds enter the marine environment by leaching from waste deposits or other materials, through shipping traffic, wastewater discharge and atmospheric deposition (Wurl & Obbard, 2005). POPs, however, make up only a small percentage of the approximately 250,000 man-made chemicals—the so called chemical universe—present in the marine environment (Roose & Brinkman, 2005).

In response to evidence of long-range transport of these substances even to regions where they have never been used or produced, threatening the environment of the whole globe, the international community has called for urgent global action to reduce and eliminate the release of these chemicals (EPA, 2007).

Some geographical areas are potentially more threatened than others by POPs: one of these is the Mediterranean Sea. This basin has limited exchange of water with the Atlantic Ocean and is surrounded by some of the most heavily populated and industrialized countries in the world. Levels of certain xenobiotics are much higher here than in other seas and oceans (Aguilar *et al*., 2002). In this environment top predators such as large pelagic fish and marine mammals tend to accumulate large quantities of organochlorine contaminants (OCs) and toxic metals (Corsolini *et al*., 1995; Marsili, 2000). Levels of OCs in the striped dolphin *(Stenella coeruleoalba)* in the Mediterranean are 1–2 orders of magnitude higher than in Atlantic and Pacific dolphins of the same species.

Here we report the final results of two case studies on the toxicological hazard to swordfish *(Xiphias gladius)* and to the blue-fin tuna *(Thunnus thynnus thynnus)* due to the potential oestrogenic effects of POPs such as OCs. These effects were investigated using sensitive biomarkers such as vitellogenin (Vtg), zona radiata proteins (Zrp) and CYP1A activities. Induction of CYP1A is associated with exposure to polycyclic aromatic hydrocarbons (PAHs) and halogenated aromatic hydrocarbons. Both compounds are known to bind to aryl hydrocarbon receptors (AHR) and initiate the down-stream cascade of events that results in transcription of CYP1A, an enzyme whose activity can be assessed in liver microsomes through the activity of ethoxyresorufin-O-deethylase (EROD) and benzo(*a*)pyrene monooxygenase (BPMO).

Non-lethal techniques, such as performing skin biopsies to investigate biomarkers including the induction of CYP1A1 activities, assessed mainly through BPMO activity, were also used (Fossi *et al*., 1992, 2003) in order to carry out hazard assessment of threatened cetacean species exposed to OCs, such as striped dolphin, common bottlenose dolphin *(Tursiops truncatus)*, short-beaked common dolphin *(Delphinus delphis)* and fin whale *(Balaenoptera physalus)*. Potential effects of contaminants on the loggerhead turtle *(Caretta caretta)* are also reported as a case study.

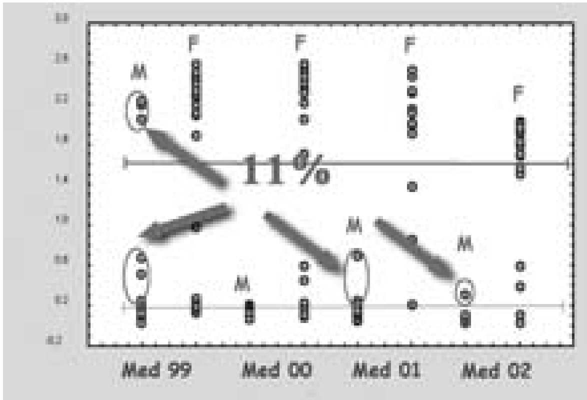
2.1. Case study 1: POPs in large pelagic fish

The rapid worldwide depletion of top predatory fish has raised serious concerns about the ecological effects of industrialized fishing. Myers and Worm (2003) estimate that large predatory fish biomass (including swordfish and tuna) is today only about 10% of its pre-industrial level. Serious concern about Mediterranean pelagic long lines has recently been expressed, since the Mediterranean swordfish population is particularly affected by this type of fishing gear. However, there is another unexplored factor that could drastically interfere with the stability of populations of Mediterranean top predators, including large pelagic fish: the toxic effects of a class of POPs known as endocrine disrupting chemicals (EDCs).

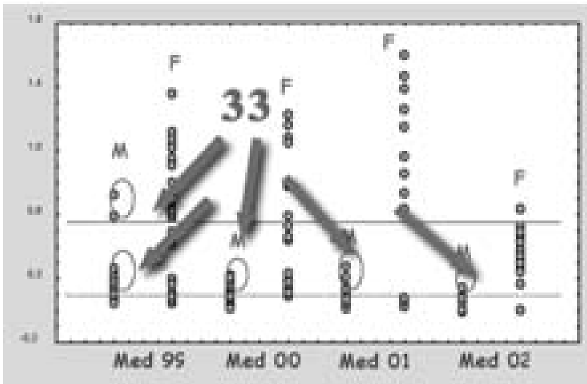
EDCs are a structurally diverse group of compounds that may adversely affect the health of humans and wildlife by interacting with the endocrine system (Colborn *et al*., 1993; Adami *et al*., 1995; Colborn, 1998; Gillesby & Zacharewski, 1998). They include chemicals used heavily in the past in industry and agriculture, such as polychlorinated biphenyls (PCBs) and organochlorine pesticides (DDTs, lindane, aldrin, dieldrin, etc.). EDCs also include currently used chemicals, such as plasticizers and surfactants. Many known EDCs are oestrogenic and affect reproductive function. Because of the lipophilic and persistent nature of most xenobiotic oestrogens and their metabolites, many bioaccumulate and are subject to biomagnification (Arukwe *et al*., 1996 ; Colborn, 1998).

In a four-year survey on the Mediterranean populations of swordfish (Fossi *et al*., 2004a) and tuna (Fossi *et al*., 2002), the potential toxicological effects of POPs such OCs were investigated in 192 specimens of swordfish and 20 specimens of tuna caught in the spawning seasons from 1999 to 2002 in the Straits of Messina, Sicily (Italy). A combination of biomarkers and contaminant analysis was used (Fossi *et al*., 2002, 2006a). Vitellogenin and Zrp (Goksoyr, 1991) levels were measured in plasma samples as biomarkers of EDCs; CYP1A activities (through ethoxyresorufin-O-deethylase (EROD) and benzo(*a*)pyrene monooxygenase (BPMO)) (Kurelec *et al*., 1977; Lubet *et al*., 1985), a biomarker of exposure to OCs, were detected in liver. OCs were also quantified in muscle, liver and gonads.

The data were compared with data for the same species from the Azores; the results sounded the first warning of the toxicological hazard of EDCs to swordfish and blue-fin tuna. The results confirmed the finding of a dramatic induction of Vtg and Zrp in adult male swordfish (Fossi *et al*., 2004a). It is noteworthy that several Mediterranean male specimens showed values of Vtg and Zrp (Figure 3.1 a,b) that were higher than average male values and/or in the same range as those of reproductive females, which suggests that this species is exposed to xenoestrogen in the Mediterranean Sea. This result was particularly evident in 1999 when the swordfish specimens examined were older. In general, levels of Vtg and Zrp were higher in Mediterranean than in Atlantic specimens. Atlantic swordfish showed a mean Vtg level of 0.17 (Abs at 490 nm) and a mean Zrp level of 0.14 (Abs at 490 nm) (Fossi *et al*., 2001b). CYP1A (BPMO and EROD) activities were found to be more than twice as high in Mediterranean specimens as in the control population.



(a)



(b)

Figure 3.1—(a) Zona radiata proteins (Zrp) and (b) vitellogenin (Vtg) of male and female swordfish captured in the Mediterranean Sea (Straits of Messina, Sicily, Italy) in summer 1999, 2000, 2001 and 2002, during the spawning period. Arrows indicate male specimens with higher values than the mean for males (line) and/or in the same range as reproductive females. (Modified from Fossi et al., 2004a.; For the color version of this figure please refer to the Annex at the end of the document)

The role of organochlorines (range of PCBs in liver 128–22,847ppb dry weight) in this induction phenomenon in Mediterranean swordfish is suggested by the statistically significant correlations between Zrp levels in plasma and PCB concentrations in muscle (Kendall's Tau-b = 0.312; $p < 0.032$) and between Vtg levels in plasma and PCB concentrations in liver (Kendall's Tau-b = 0.618; $p < 0.034$) in male specimens. Organochlorine levels (PCBs in liver) were also correlated with total length of male swordfish (Kendall's Tau-b = 0.377; $p < 0.021$).

These data, and those published by De Metrio *et al*. (2003) demonstrating a high percentage of intersex in Mediterranean swordfish, sound a warning about reproductive alterations in large pelagic fish (Hashimoto *et al*., 2000; Segner *et al*., 2003; Van den Belt *et al*., 2003) and suggest the need for continuous monitoring to avoid population reductions and loss of Mediterranean biodiversity (Porte et al., 2006; Burger *et al*., 2007).

2.2. Case study 2: POPs in cetaceans

In the last 20 years there has also been growing concern about the risk to Mediterranean cetaceans associated with high bioaccumulation of OCs (Fossi *et al*., 1992; Fossi et al., 2003; Fossi & Marsili, 2003) and emerging pollutants, such as polybrominated diphenyl ethers (PBDEs) (Alaee *et al*., 2003; Fossi *et al*., 2006b). OC pollutants have been claimed to impair reproduction (Reddy *et al*., 2001) and depress the immune system in dolphins. Both reproduction and the immune system are critical to long-term population maintenance. While levels of organochlorine compounds are decreasing in the Mediterranean Sea (Borrel & Aguilar, 2007), concentrations of PBDEs seem to be increasing. Polybrominated diphenyl ethers (PBDEs) are a major family of brominated flame retardants (BFRs) which are lipophilic, persistent, and toxic to both fauna and humans. Since the end of the 1990s, brominated flame retardants have attracted increasing attention. Today there is growing concern about the accumulation of brominated organic compounds in the food chain. The highest levels of PBDEs have been found in animals at the top of the marine food chain. However, information on the effects of BFRs in wildlife and man is still lacking (Alaee *et al*., 2003; Fossi *et al*., 2006b).

2.2.1. Species differences

In a study performed by Fossi and colleagues (Fossi *et al*., 2003), subcutaneous tissues (skin and blubber) (Figure 3.2) were obtained from three odontocete species (striped, bottlenose and common dolphins) and one mysticete species (fin whale) from the western Ligurian Sea (in the

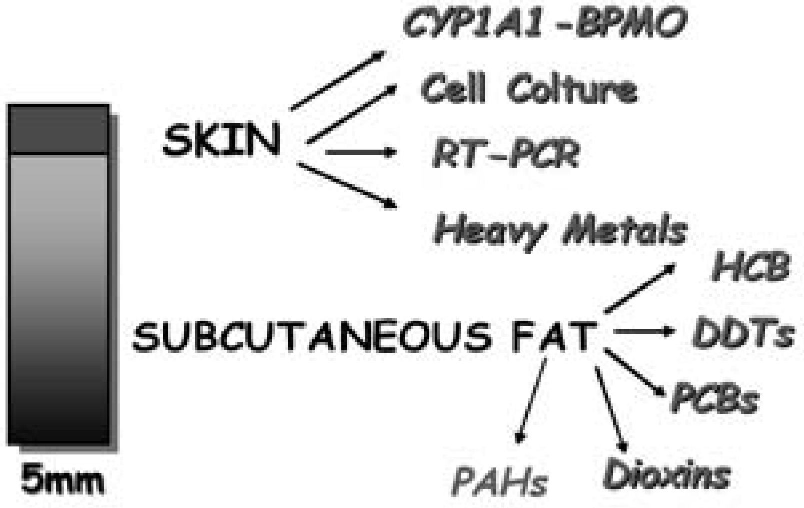


Figure 3.2—Analytical versatility of skin biopsies from free-ranging cetaceans for contaminant and biomarker analysis. Skin biopsy is a powerful tool for toxicological studies for the following reasons: (a) it allows collection of a large number of samples across a wide geographic range; (b) it allows collection of sequential samples from the same animal if identified by photo-identification or other techniques; (c) it is suitable for residue analysis of OCs (HCB, DDTs, PCBs), including dioxin group chemicals (suitable for calculation of TEQs), heavy metals and polycyclic aromatic hydrocarbons (PAHs); (d) it is suitable for biomarker analysis: induction of CYP1A1 by enzyme assay (BPMO) and immunohistochemical assay (CYP1A1); and also for DNA damage, real-time PCR (RT-PCR) and fibroblast cell culture. (For the color version of this figure please refer to the Annex at the end of the document)

Pelagos Sanctuary between Corsica and the French–Italian coast) and from the Ionian Sea. CYP1A1 (BPMO) activity was measured in skin biopsies, and OC concentrations were assayed. In this study, levels of OCs in striped dolphin were from 10 to 20 times higher than in swordfish (Fossi & Marsili, 2003). The relative oestrogenic power of these chemicals, identified by *in vitro* and *in vivo* screening methods (Safe, 1995, 2000) is rather weak (10^{-3} or less) compared with the reference power of 17-estradiol and DES (Miyamoto & Klein, 1998). Moreover, OC levels in the striped dolphin in the Mediterranean were 1–2 orders of magnitude higher than in specimens from the Atlantic and Pacific (Figure 3.3).

Some general considerations on the potential hazard to these Mediterranean species can be drawn from a comparison of the levels commonly detected in Mediterranean cetaceans and those of other cetacean species with known reproductive impairment (Fossi & Marsili, 2003) (Figure 3.3). Several examples suggest that exposure to OCs and PCBs has affected endocrine function and reproduction in marine mammals. Levels of PCBs found in Mediterranean free-ranging odontocetes (Fossi *et al*., 2003) are similar to those measured in the population of beluga whales in the St Lawrence estuary which included a hermaphrodite specimen (mean PCB value = 78,900ng·g^{-1} lipid weight) (Muir *et al*., 1996).

Organochlorine (HCB, DDT and PCB) concentrations and CYP1A1 activities (Fossi *et al*., 1992; Marsili, 2000) indicated that there were marked differences between fin whales and odontocete species in the levels of all contaminants (Figure 3.4 A, B, C). The same was found for CYP1A1 activities, which were four times higher in certain odontocete species, such as the striped dolphin, than in mysticetes, with levels of OCs one order of magnitude higher in odontocetes (Figure 3.4 D) (Fossi *et al*., 2003). The difference in organochlorine bioaccumulation and consequently in CYP1A1 induction between the two groups is obviously related to their different positions in the marine food chain, with odontocetes as terminal consumers and fin whales as macroplankton feeders.

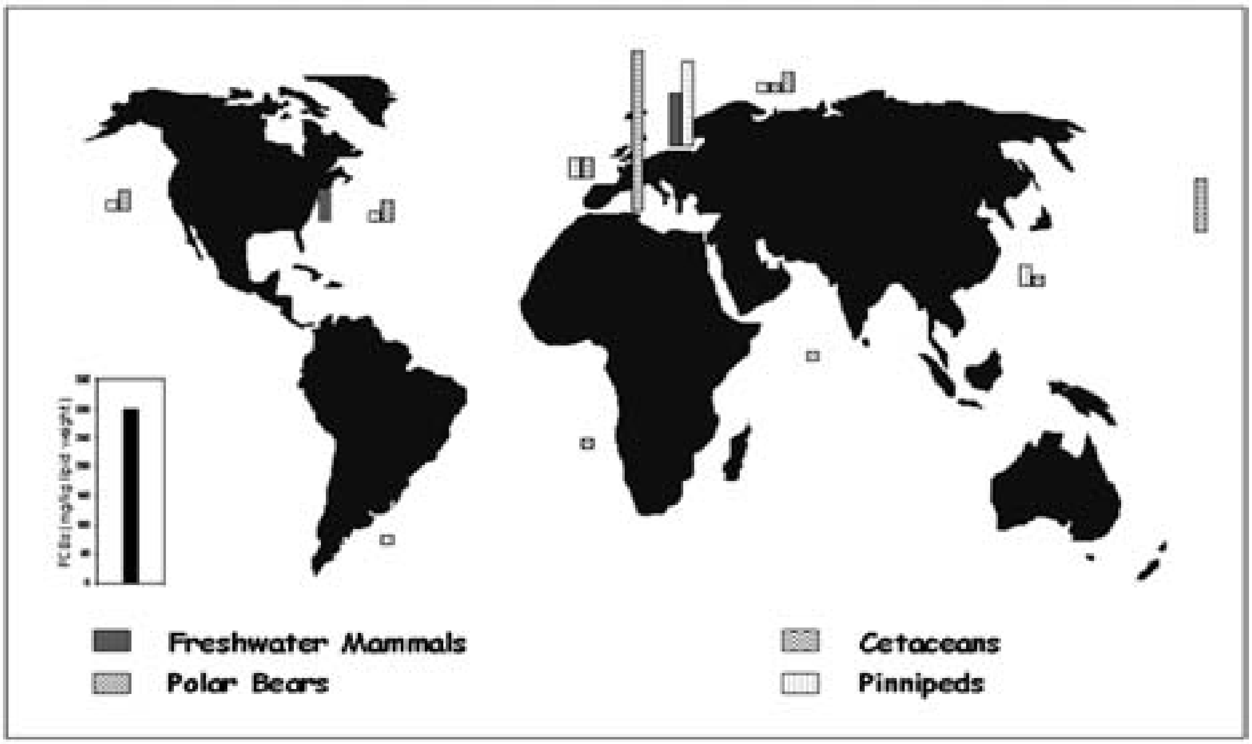


Figure 3.3—Range of total PCB concentrations (μg·g^{-1} lipid weight) in marine fish-eating mammals from various parts of the world. (Modified from Fossi & Marsili, 2003.)

In ecotoxicological studies of Mediterranean cetaceans, several questions remain unanswered. The need for new biomarkers for EDCs and a 'cell model' to explore different susceptibilities to several classes of EDCs led Fossi's research team to culture fibroblasts of different cetacean species ('dolphins in test tubes') (Marsili *et al*., 2000). Fossi and colleagues proposed and applied three new methodological tools to detect cultured fibroblast responses to OC-EDCs and PBDEs: immunofluorescence, Western blot and real-time PCR (Fossi *et al*., 2006b). With these 'prognostic' tools, they explored interspecies and gender susceptibility to OC-EDCs and PBDEs using qualitative and semi-quantitative evaluation (Western blot and immunofluorescence) of the target proteins CYP1A1 and CYP2B in cultured fibroblasts. Particular attention was paid to the role of detoxification enzymes (CYP2B) and the related biochemical susceptibility of the various species to different classes of chemicals. The role of CYP2B in the metabolism of two tetrachlorobiphenyl congeners *in vitro* had previously been studied in beluga and pilot whales (White *et al*., 2000).

The main results of these innovative experiments were: 1) detection of cytochromes (CYP) 1A1, 1A2 and 2B4 in bottlenose dolphin, striped dolphin and fin whale fibroblasts, revealed by fibroblast fluorescence detected by immunofluorescence and by cross-reaction of the antibody used in Western blot analysis; 2) increasing induction (CYP2B) with increasing doses of contaminants, revealed by both methods, with greater induction by PBDE than by OC treatment in bottlenose dolphin; 3) different increases in fluorescence (2B cytochromes) in the two species of dolphins in relation to contaminant doses, with higher induction responses in striped dolphin and bottlenose dolphin than fin whale; 4) different patterns of induction (cytochromes 2B) in male and female striped dolphins.

The information obtained in this project (Fossi *et al*., 2007) will be the basis for further applications and validation of biomarker methodologies to study different species and the gender susceptibility of marine mammals to different mixtures of endocrine-disrupting xenobiotics, including emerging contaminants (PBDEs).

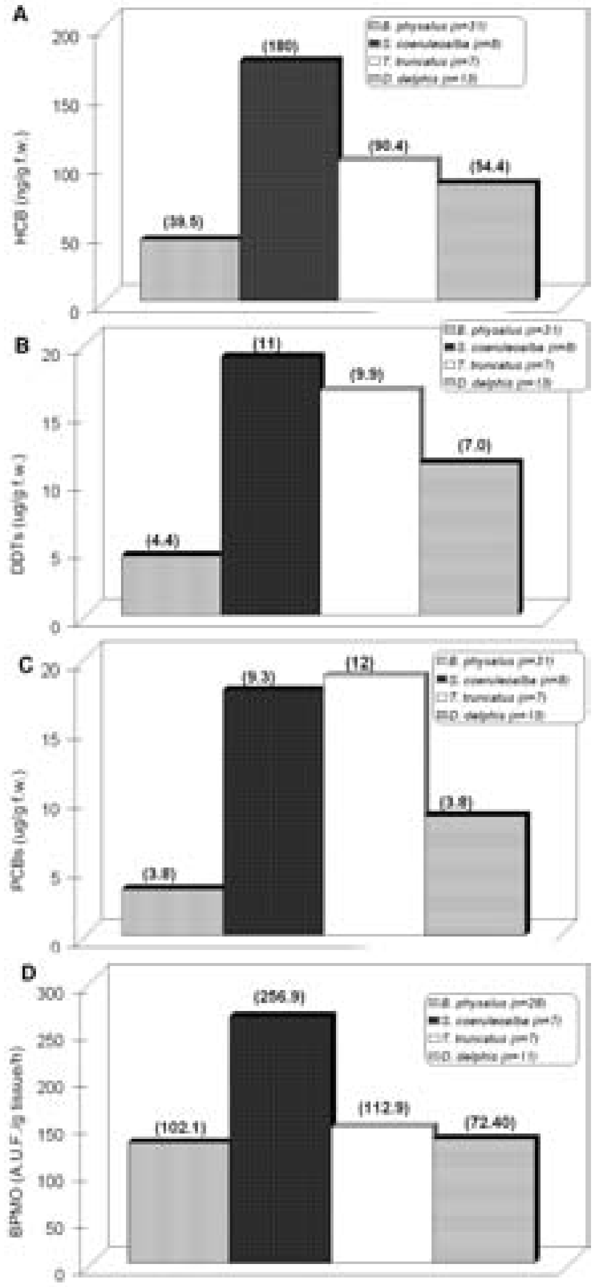


Figure 3.4—A: HCB, B: DDT and C: PCB concentrations ($ng \cdot g^{-1}$, $\mu g \cdot g^{-1}$ and $\mu g \cdot g^{-1}$ fresh weight, respectively); and D: CYP1A1 (BPMO) activity (arbitrary units of fluorescence (AUF)$\cdot$g tissue$^{-1} \cdot h^{-1}$) in skin biopsies from Mediterranean cetaceans. Arithmetic means, with standard deviation in brackets; n = number of samples. (Modified from Fossi et al., 2003).

2.2.2. Geographical differences

A geographical contamination trend was also found for striped dolphin in the Mediterranean basin: PCB and DDT levels decreased from the north-west (Ligurian Sea) to the south-east (Ionian

Sea) (Marsili, 2000). Limited information exists on the ecotoxicological status of south-western Mediterranean groups of this top predator.

Fossi and colleagues used a non-lethal approach (skin biopsy, Figure 3.2) to investigate bioaccumulation of OCs and trace elements (Hg, Cd, Pb) as well as CYP1A induction (BPMO) as a diagnostic biomarker in nine specimens of striped dolphin biopsied during a survey of the Aeolian area in summer 2002 (Fossi *et al*., 2004b). The same samples were also analysed to investigate polychlorodibenzo-p-dioxins (PCDDs), polychlorodibenzofurans (PCDFs) and coplanar PCBs. One aim was to compare the resulting data with those of striped dolphin from other Mediterranean areas (Pelagos Sanctuary and Ionian Sea), obtained by the same methodological approach (Fossi *et al*., 2000).

Its interesting that the CYP1A activities (BPMO) found in the Aeolian population (43.5 AUF·g^{-1}·h^{-1}) were roughly five and three times lower than levels previously found (Marsili, 2000) in Ligurian (199.7 AUF·g $tissue^{-1}$·h^{-1}) and Ionian (125.5 AUF·g $tissue^{-1}$·h^{-1}) striped dolphin groups, respectively (Figure 3.5). Moreover OC levels were significantly higher in the Ligurian samples (Marsili, 2000) than in the Aeolian ones. These data are the first warning of a threat to these marine mammals in the Pelagos Sanctuary.

In conclusion, the two case studies (the large pelagic fish and the cetacean projects) show that: (a) Vtg and Zrp can be used as diagnostic tools for fish stock hazard assessment in the Mediterranean Sea; (b) CYP1A1 (BPMO) induction in cetacean skin biopsies may be used as an early sign of exposure to EDCs such as OCs, and a potential alert to trans-generational effects; and (c) skin biopsies are a suitable material for assessing the ecotoxicological status of the various Mediterranean species and sub-populations of dolphins.

This research provides a warning of reproductive alterations in marine top predators and suggests the need for continuous monitoring to avoid population reductions and a loss of biodiversity in the Mediterranean Sea.

2.3. Case study 3: POPs and loggerhead turtles

Of the three species of marine turtles living regularly in the Mediterranean Sea, the loggerhead turtle is the most common and its distribution covers the whole basin. Demographic studies

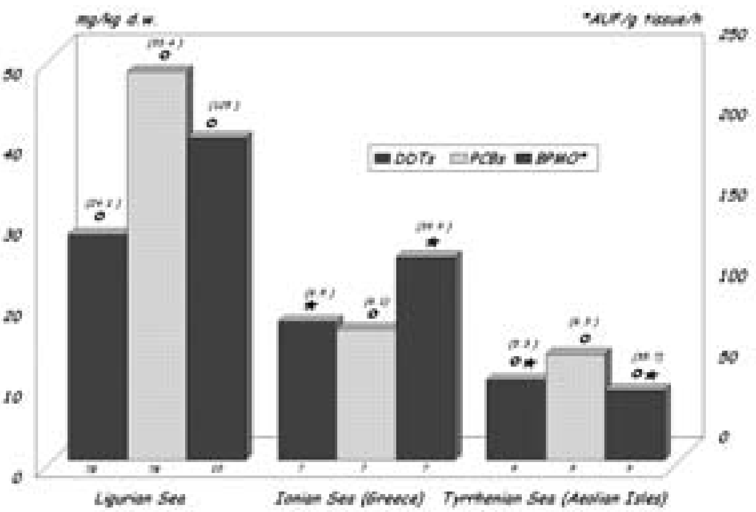


*Figure 3.5—Mean levels of Total DDT and Total PCB (mg/kg dry weight) and CYP1A (BPMO) activity (AUF·g $tissue^{-1}$·h^{-1}) in biopsies of striped dolphins sampled in three different areas of the Mediterranean Sea (Marsili, 2000; Fossi et al., 2004b). Significant differences (p<0.05) between the Ligurian and the other two groups are indicated with ° and significant differences (p<0.05) between the Ionian and the other two groups are marked with *. Standard deviations are in brackets and sample sizes are at the foot of the columns. (Source: Fossi et al., 2004b.)*

indicate that the loggerhead turtle population is in decline with only 2,000 nesting females per year in the Mediterranean region (Groombridge, 1990; Godley *et al*., 1999; Storelli *et al*., 2007). Mediterranean populations of marine turtles have been decreasing steadily in the last few decades. Overexploitation of eggs and meat as food in the past, incidental mortality in fishing nets, and degradation of marine and nesting habitats due to shipping and tourism (Groombridge, 1990) have had a dramatic impact on marine turtle populations. Pollution is another possible threat to them, though mortality due to marine pollution (PAHs, POPs and heavy metals) is one of the most difficult to estimate of all man-made sources of mortality (Storelli *et al*., 2007).

Low levels of PCBs (total PCB in fat 334 $ng \cdot g^{-1}$ wet weight) were found by Corsolini and colleagues (2000) in *Caretta caretta* from the Adriatic Sea and were similar to those found in Mediterranean fish. The longevity of marine turtles, the long-term toxic effects of mono-*ortho* congeners and the fact that total toxic potential in turtles depends largely on concentrations of non–*ortho* coplanar PCB 77 are all factors increasing the level of risk to turtle health (Storelli *et al*., 2007).

The effects of endocrine disruptors have never been investigated in this emblematic species of Mediterranean biodiversity. In a study to develop and validate a series of diagnostic and prognostic biomarkers to evaluate exposure to oestrogenic/androgenic compounds in Mediterranean specimens (Fossi *et al*., 2004c; Casini *et al*., 2007), turtles from Tuscany and Sicily (Italy) were sampled at local rescue centres. All the specimens were found stranded alive, classed as juveniles and not sexed.

Differences in vitellogenin and zona radiata protein levels in plasma were found between the two areas, indicating that these biomarkers are suitable for future monitoring studies. Esterase activity in plasma was in the range 0.01–0.09$\mu mol \cdot ml^{-1} \cdot min^{-1}$ for AChE and 0.01–0.04 for BChE. Porphyrin analysis showed the presence of uro-, proto- and copro-porphyrins. PCB levels in blood were lower than those detected in other tissues of loggerhead turtles from the Adriatic Sea, although the congener profile was similar (Corsolini *et al*., 2000). Hexa- and hepta-CBs were the most abundant classes of isomers, constituting over 70% of the total PCB load in turtles from Tuscany and Sicily; most were PCBs 138, 153 and 180. Profiles based on PCB isomer classes in both groups resembled the composition of Aroclor: hexa-CB > penta-CB > hepta-CB. Toxic equivalents were calculated in samples from Tuscany as the sum of coplanar PCBs, PCDDs and PCDFs, using bird TEFs. Although coplanar PCBs were more abundant, dioxins and furans contributed most to toxicity (Corsolini, personal communication).

In conclusion this study suggests that the application of the proposed non-lethal protocol of sampling and analysis for sea turtle ecotoxicological studies may provide insights into unexplored phenomena in a species potentially susceptible to the adverse effects of environmental pollutants that can alter reproductive processes.

3. Sewage discharge and plastics: effects of ingested plastic on large marine vertebrates, including marine mammals and turtles

3.1. Sources and types of pollutants and debris

Sewage and marine debris have a number of different sources, the most important of which are land based. All the wastewater produced by a city may overflow into a river system, which will carry the pollutants downstream and eventually deposit them in the sea. Such heavily polluted urban runoff that flows untreated to the ocean may be carrying gasoline, heavy metals, pesticides and even solid rubbish. Where urban effluent is treated in a conventional sewage treatment plant, bacteria will remove up to 90% of the biodegradable matter, which is the organic portion of such waste; nevertheless, the remainder will still contain solids and polluted water which may reach the sea. In addition, commercial and recreational vessels contribute substantially to sewage discharges into the Mediterranean Sea, leading to the deterioration of water quality and an overall degradation of marine habitats.

One of the most important and widespread types of water degradation is an increased concentration of faecal coliform bacteria, which are commonly found in the intestines of all warm-blooded animals. Discharge of untreated or partially treated human waste from vessels as well as from urban centres contributes to high bacteria counts and is responsible for an increased risk to human health. This problem can be particularly serious in lakes, slow-moving rivers, marinas and other bodies of water with low flushing rates (EPA, 2007). When concentrations of faecal coliform bacteria rise above safe levels and pose a risk to human health, local health boards act to close swimming areas. Other waste items include plastics—synthetic organic polymers which have existed for a century. Characteristics such as their light weight, hardiness, durability and cheapness (Laist, 1997) make plastics extremely versatile and widely used in most sectors of the economy. As a consequence, plastics form up to 80% of all marine debris (Derraik, 2002) (Table 3.1). They have become a problematic rubbish component because they are resistant to natural degradation processes and they can be a danger to marine wildlife through ingestion or entanglement.

Furthermore, marine debris can also be a serious aesthetic problem on shores and beaches where plastic debris accumulates and may discourage tourist and recreational activities (Sheavly, 2005).

Table 3.1—Proportion of plastic debris in the Mediterranean (Med.) area. (Modified from Derraik, 2002.)

Area	Debris type	% of debris items represented by plastics
French Med. coast	Deep sea floor	>70
Med. beaches (5)	Beach	60–80
Mediterranean Sea	Surface waters	60–70

It is generally accepted that most debris in the sea is released as a result of human behaviour, such as littering or dumping. The main marine sources of plastic debris include the following:

- Commercial fishing vessels: debris resulting from commercial fishing includes lines and ropes, bait boxes and fishing gear. All these materials become debris when fishermen fail to retrieve their fishing gear or when they discard equipment or plastic containers overboard. In 1975 the world's fishing fleet dumped 135,400 tons of plastic fishing gear and 23,600 tons of packing materials into the sea (Cawthorn *et al*., 1989).

- Recreational craft: these generate mainly plastic bags and various kinds of food packaging (Sheavly, 2005).

- Merchant and military vessels: these generate large amounts of plastic rubbish, especially in the case of large vessels

- carrying supplies for several months. Horsman (1982) estimates that merchant ships dump 639,000 plastic containers daily.

- Offshore oil and gas platforms: these could be treated as land-based sources, since they generate a wide variety of marine debris and wastewater.

Debris of land-based origin is generally transported by rivers down to the sea. There are 69 main rivers in the Mediterranean basin, which discharge 283km^3 of water annually. As a consequence, an estimated 5.5 million items of plastic debris are floating in the Gulf of Lion in the north-western Mediterranean, and up to 750 million in the whole basin.

3.2. Effects on marine life

Although plastic debris is widespread in the sea as a result of the increased use of plastics in the last three decades, there is still little information on its impact on the marine ecosystem as a whole (Quayle, 1992); the danger posed by plastics to the marine environment has only recently been recognised (Stefatos *et al*., 1999). Nevertheless, is it well known that plastic debris affects marine life through the ingestion of plastic items (Laist, 1997; Derraik, 2002) or entanglement in ropes or ghost fishing gear. The majority of studies on the ingestion of plastic debris have been performed on sea turtles and seabirds, while other marine groups have been less studied. It is generally recognised that animals mistake plastic debris for food and consequently ingest it; this is reported to have happened to at least 267 marine vertebrate species worldwide, which are summarized in Table 3.2. Such events can easily be under-estimated, since most dead animals neither reach shore nor are picked up at sea; instead they may sink or be eaten by predators.

Some marine animals' feeding strategies are not well known; some species may feed selectively, as is the case of some seabirds and fish (Derraik, 2002). Seabirds might mistake floating plastic debris for cuttlebones and ingest parts of it (Cadée, 2002). Among sea birds, surface feeders as well as plankton-feeding divers seem to be most liable to ingest plastics (Robards *et al*., 1995); moreover the latter are more likely to mistake plastic particles for prey (Derraik, 2002).

3.2.1. Loggerhead turtles

The loggerhead turtle inhabits Mediterranean coastal environments as well as open waters and is known to feed upon a wide variety of prey, such as molluscs, crustaceans, fish, sea urchins, salps, jellyfish and even dead organisms. This turtle seem to selectively ingest white plastic debris as a consequence of its selective feeding strategy in the central Mediterranean Sea (Gramentz, 1998). Such behaviour exposes loggerhead turtles to the toxic effects of plastics (Corsolini *et al*., 2000; EPA, 1992). This is the case of plastic pellets, which concentrate and adsorb DDEs and PCBs onto their surface. Individuals can also ingest other xenobiotic compounds, especially OCs (from shipping or land), tar balls, petroleum and fatty tissues of prey (Corsolini *et al*., 2000). Tar balls and plastic bags (which look very similar to

Table 3.2—Number of marine species worldwide reported to have ingested plastics. (Source: Greenpeace, 2006.)

Species group	Total species worldwide	Percentage of species with ingestion records
Sea turtles	7	6 (86%)
Seabirds	312	111 (36%)
Marine mammals	115	26 (23%)
• Baleen whales	10	2 (20%)
• Toothed whales	65	21 (32%)
• Otariidae	14	1 (7%)
• Phocidae	19	1 (5%)
• Sirenia	4	1 (25%)
• Mustelidae	1	0

jellyfish) are dangerous items for sea turtles since their digestive and respiratory ducts coincide, making them vulnerable to suffocation.

Even though loggerhead turtles demonstrate considerable tolerance to debris ingestion, as suggested by the low mortality reported in the literature (Tomás *et al*., 2002), the sublethal effects of debris ingestion, such as dietary dilution, should also be considered. The result may be malnutrition, and this may seriously affect the population in the long term (Bjorndal *et al*., 1994). Moreover, this food dilution effect may be more detrimental for juveniles than for adults: the former are pelagic and are frequently highly exposed to rubbish in the convergence zones, where plastic debris accumulates.

The ingestion of plastics can produce two main effects:

- The debris may accumulate in the animal's gut, producing a false sense of fullness which causes the animal to stop eating and leads to malnutrition.
- The debris may cause internal injuries and block the animal's digestive tract leading to death. This is the especially the case with the ingestion of barbed and/or sharp objects (such as broken PVC objects).

3.2.2. Cetaceans

Amongst cetaceans there are published reports for 21 species of odontocetes that have ingested plastic debris (Laist, 1997); nevertheless, in many cases the ingestion of plastic debris may be under-estimated, since dead or sick animals may go unnoticed and sink upon death and therefore not be counted (Baird & Hooker, 2000). The cetacean stranding reports published annually by the Milan Natural History Museum include Risso's dolphin *(Grampus griseus)*, as well as striped and common bottlenose dolphins among the cetaceans most at risk of debris ingestion (CSC, 1987–2004). Data show that the deaths of some animals were due to obstruction of the gut, throat or digestive tract.

4. Conclusions and recommendations

The general outcome of this chapter is that it sounds a warning about the biodiversity of the large fauna (pelagic fishes, cetaceans and sea turtles) in the Mediterranean Sea. The conservation status of important and charismatic members of the Mediterranean fauna is threatened because of the risks posed by POPs to their reproductive systems, and by polluted waters and the ingestion of plastic debris to their health and survival.

The first part of the chapter highlights the high risk of reproductive alterations in large marine vertebrates and suggests the need for ongoing monitoring of POPs and emerging pollutants to prevent population reductions and a loss of biodiversity in the Mediterranean Sea.

An Action Plan for the conservation of cetaceans in the Mediterranean Sea was drawn up by the Mediterranean Action Plan (MAP) secretariat together with national experts and international organizations and adopted by the contracting parties in 1991. The following year, the National Focal Points for Specially Protected Areas developed it further for implementation (RAC/SPA, 2007b). An Action Plan for the conservation of marine Mediterranean turtles was adopted in 1989 (RAC/SPA, 2007c).

Plastic debris and wastewater have multiple origins, which are both land and sea based. Because of the complexity of these sources, which include domestic, industrial, urban and recreational activities, there can be no single solution to the problems they cause.

The main attempt to mitigate the problems of waste in the world's oceans has been through international legislation; the principal legal framework for addressing marine pollution is the International Convention for the Prevention of Pollution from Ships (MARPOL). Annex V of MARPOL 'restricts at sea discharge of garbage and bans at sea disposal of plastics and other synthetic materials such as ropes, fishing nets, and plastic garbage bags with limited exceptions'. The Mediterranean and Black Seas, both ACCOBAMS areas, are among the special areas where Annex V lays down much stricter discharge regulations (Derraik, 2002).

The mitigation of marine pollution is a complex matter, so a combination of legal, educational and public awareness measures is required. Since most marine debris is released as a result of human behaviour, education must be a priority action in preventing marine pollution and improving public awareness, by encouraging people to change their ways and choose better options. These are fundamental measures that need to be put in place by the countries involved. This should be linked with the need to enforce legislation, which is a major problem in most Mediterranean countries. The difficulty of enforcement derives mainly from the large size of the Mediterranean Sea.

Recommended actions include:

- **Improved coordination**: *Work with them not against them* should be the slogan. The private sector and industrial companies should be involved in a common process.

- **Improved garbage management**

- **More effective enforcement of legislation**: Governments and local authorities need to enforce anti-waste laws more effectively and introduce deterrents to dumping.

- **Public awareness and education programmes**: *Getting the message out*. Simple messages should be used to change people's behaviour. Governmental and non-governmental organizations and groups should cooperate in putting out the right messages.

- **Measures to decrease waste products**

- **Improved research**: Research is needed into the unknown impact of marine debris on the ecosystem as well as on the marine organisms. The Adriatic Sea and the north-western Mediterranean Basin could be recognised as key areas for the Mediterranean Sea.

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Chapter 4

Biodiversity impacts of species introductions via marine vessels

Bella S. Galil, Anna Occhipinti-Ambrogi and Stephan Gollasch

1. Introduction

The Mediterranean Sea has sustained extensive maritime activity since the early Bronze Age (Knapp, 1993; Wachsmann, 1998). Until the 15th century, when mercantile horizons expanded with geographical discoveries, direct maritime shipping was mostly confined to the Mediterranean, the nearby provinces of the eastern Atlantic Ocean, and the Black Sea. Following the opening of the Suez Canal in 1869, the Mediterranean has increasingly served as a hub of international commercial shipping, with about 220,000 vessels of more than 100 tonnes crossing the Mediterranean annually, carrying 30% of global sea-borne trade volume (Dobler, 2002).

There is growing awareness worldwide that marine invasions constitute one of the most significant components of global change, with often harmful effects on biodiversity, economy and human health. This concern, coupled with the wide publicity given to some cases of invasive alien species[1] noted for their conspicuous economic or ecological impacts, has drawn the attention of scientists, managers and media alike. Adding to the wide perception that the littoral and infralittoral biota of the sea has been undergoing a rapid and profound change, a series of atlases published by the Mediterranean Science Commission (CIESM) presented to biologists and to a wider public the disturbing extent to which a variety of alien fish, molluscs, and decapod and stomatopod crustaceans have been recorded in the Mediterranean Sea (CIESM, 2002–2004).

Shipping has been implicated in the transfer of numerous neritic organisms to foreign shores (e.g. Carlton, 1985; CIESM, 2002; Galil, 2006). Such unintentional transport by vessels is exacerbated by the dense global traffic patterns, the large number of vessels (both commercial and recreational), and the diverse transport niches they provide, such as in ballast water and sediments or through hull fouling (Carlton, 1985; Minchin & Gollasch, 2002). It has been estimated that 3,000–4,000 species are being transported by vessels every day (Carlton & Geller, 1993). It is seldom possible to ascertain the precise means of transmission as some organisms may conceivably be transported by several vectors.[2]

The transport of boring, fouling, crevicolous or adherent species on the hulls of ships has long been recognised as the most ancient method of aquatic species introduction. In 1873 the tri-masted Karikal arrived at the port of Marseille from India carrying on its flanks a '... petite forêt d'êtres vivants … peuplée de Crustacés' (Catta, 1876:4), including the crab species *Planes minutus* (Linnaeus, 1758), *Pachygrapsus transversus* (Gibbes, 1850) (as *P. advena*) and *Plagusia squamosa* (Herbst, 1790), the last numbering in the hundreds of specimens. The first alien species recorded as such in the Mediterranean were a pair of fouling serpulid polychaetes, *Hydroides dianthus* (Verrill, 1873) and *H. diramphus* (Mörch, 1863) collected in the harbours of İzmir and Naples in 1865 and 1870 respectively (Carus, 1889; Zibrowius, 1973). Fouling generally involves small-sized sedentary, burrow-dwelling or clinging species, though large species whose life history includes an appropriate life stage may be disseminated as well (Zibrowius, 1979; Gollasch, 2002).

Ballast (formerly solid, but for the past 130 years aqueous) is usually taken into dedicated ballast tanks or into empty cargo holds when offloading cargo, and discharged when loading cargo or bunkering (fuelling), to increase vessel stability in heavy weather and to submerge the propeller when empty. Ballast water therefore

1 An alien species is defined as an organism, inclusive of parts, gametes or propagules that may survive and subsequently reproduce, occurring outside its known or consensual native range, as documented in scientific publications. An invasive alien is an alien whose population has undergone an exponential growth stage and is rapidly extending its range (Occhipinti-Ambrogi and Galil, 2004).

2 Vector: any living or non-living carrier that transports living organisms intentionally or unintentionally (ICES, 2005).

had originated mostly in or near port waters until the recent recommendation that the exchange of ballast water be carried out in the open ocean (IMO, 1997, 2004). Water and sediment carried in ballast tanks have been found to contain viable organisms, even after voyages of several weeks' duration (e.g. Gollasch *et al*., 2000a, 2000b; Wonham *et al*., 2001; Drake *et al*., 2002). Since the volume of ballast water may be as much as a third of the vessel's deadweight tonnage, it is a cause of great concern. Indeed, the global maritime trade connections of Mediterranean ports provide examples of transoceanic ballast dispersal: the Erythrean[3] alien portunid crab *Charybdis hellerii* (A. Milne Edwards, 1867) was recorded in 1987 in Cuba, and in rapid succession in Venezuela, Colombia, Florida and Brazil (Gómez & Martínez-Iglesias, 1990; Mantelatto & Dias, 1999). Transport in ballast tanks is the most probable mode of dispersal since the crab's arrival corresponds with increased coal shipping from Port Drummond, Colombia, to Israel. On return voyages the vessels, ballasted in Israeli coastal waters, discharged the waters and their entrained biota off the Colombian coast. Also the presence of two Erythrean alien fish, *Alepes djedaba* (Forsskål, 1775) and *Stephanolepis diaspros* Fraser-Brunner, 1940, identified along with four other fish species in floodable cargo holds and dedicated ballast tanks arriving in Baltimore, USA, from Israel, attests that this is a major pathway for transoceanic dispersal (Wonham *et al*., 2000). A recent study carried out within the framework of the EU-funded programme 'Delivering Alien Invasive Species Inventories for Europe' (http://www.europe-aliens.org) revealed that shipping served as the vector of primary or secondary[4] introduction for one third of the species (Galil *et al*., in prep.; *see* Appendix). However, this percentage could be even higher since the introduction vector for some alien species is still uncertain or unknown.

It is often difficult to identify a pathway for a specific introduction: pathways may differ between regions or the introduction may have occurred through multiple pathways. Therefore, most sources of introductions cannot be positively identified, except for intentional introductions (i.e. for aquaculture). Understanding the introduction of a species at one site does not necessarily help to predict other introductions of the same species, as the possible introduction pathways and factors determining establishment success and further spread are site or time specific. From initial inoculation sites—large ports or aquaculture facilities, for instance—alien species may spread by natural dispersal, by local translocation of aquaculture stock, or by coastal and recreational shipping. New findings of a species may also be due to further introductions from its original source region.

Putative pathways are often proposed in the literature, but the possibility of determining with any certainty a particular pathway in a particular location where an alien species has been spotted is small. Although possible pathways have been considered in depth, including detailed timelines that are important in understanding the mechanisms of transportation and introduction, most cases lack critical evaluation of the proposed pathways. Moreover, inference from other introduction sites of the same species may result in undue generalizations. Nevertheless, the practical importance of distinguishing between pathways is clear, and must prompt every effort towards reducing the risk of new introductions. With this in mind, our account makes use of the available literature and of the subsequent critical work performed in the framework of certain Mediterranean projects, with the aim of presenting the best hypothesis on the relative importance of pathways. The large number of marine alien species recorded in the Mediterranean should spur the urgent adoption of the international and national measures needed to control bioinvasions. These actions should fully address the dominant pathways for introductions in the Mediterranean: the Suez Canal, ballast water and fouling of marine vessels and installations, and translocations for aquaculture. Future research should focus on the invasion process, interactions among alien species and between the alien and native biota, and the impacts of introductions, and it should

3 Erythrean alien: an Indo-West-Pacific organism that enters the Mediterranean through the Suez Canal as a result of 'natural' dispersal, by autochthonous active or passive larval or adult movements.

4 Secondary introduction is the dispersal of an alien beyond its primary location of introduction (Occhipinti-Ambrogi & Galil, 2004).

support the prevention and management of introductions.

This chapter presents a review of the shipping-introduced and shipping-dispersed alien species in the Mediterranean Sea. Patterns of abundance and distribution of those species and an analysis of their vectors are discussed, together with their environmental, economic and social impacts. As eradication and mitigation options are restricted, potential biodiversity offsets for introducing alien species are presented.

2. Vessel-transported alien species in the Mediterranean

The surge in long-distance shipping and the opening of the Suez Canal predate modern studies of Mediterranean marine taxa, save molluscs and fishes, by at least half a century. Since the beginning of the 20th century extensive biological surveys have been conducted in the Mediterranean, allowing for a reasonable measure of confidence in separating the recently introduced alien from the native biota in the better-known taxa. Unicellular organisms have been excluded from our list because of uncertainties as to their identification, status (native, cryptogenic[5] or alien) and geographic spread.

As the likelihood of encountering a vagrant in the sea is vanishingly small, we consider most records to represent self-maintaining populations of some duration. It is recognised that some alien species may fail to maintain populations over time, and therefore a single record dating back several decades may be considered an ephemeral entry. An authoritative dataset has been recently completed tracing the origin, date and mode of introduction, current distribution, rate of spread, and actual and potential impacts of the aquatic alien species recorded in European seas (DAISIE, 2008). The list of vessel-transported species presented in this chapter is largely based on this source, and is geographically restricted to the Mediterranean Sea *sensu stricto*, between the Straits of Gibraltar and the Dardanelles.

The 191 metazoan species identified in the present work as vessel-transported aliens in the Mediterranean Sea are listed in the Appendix. The list is clearly an under-estimate, due to incomplete knowledge of some taxa, the inability to distinguish aliens from some native species, and the lack of concerted and standardized efforts to survey port and port-proximate environments for alien biota. A taxonomic classification of the primary and secondary vessel-transported alien species (Figure 4.1) shows that the most frequently recorded taxa are Mollusca (27%),

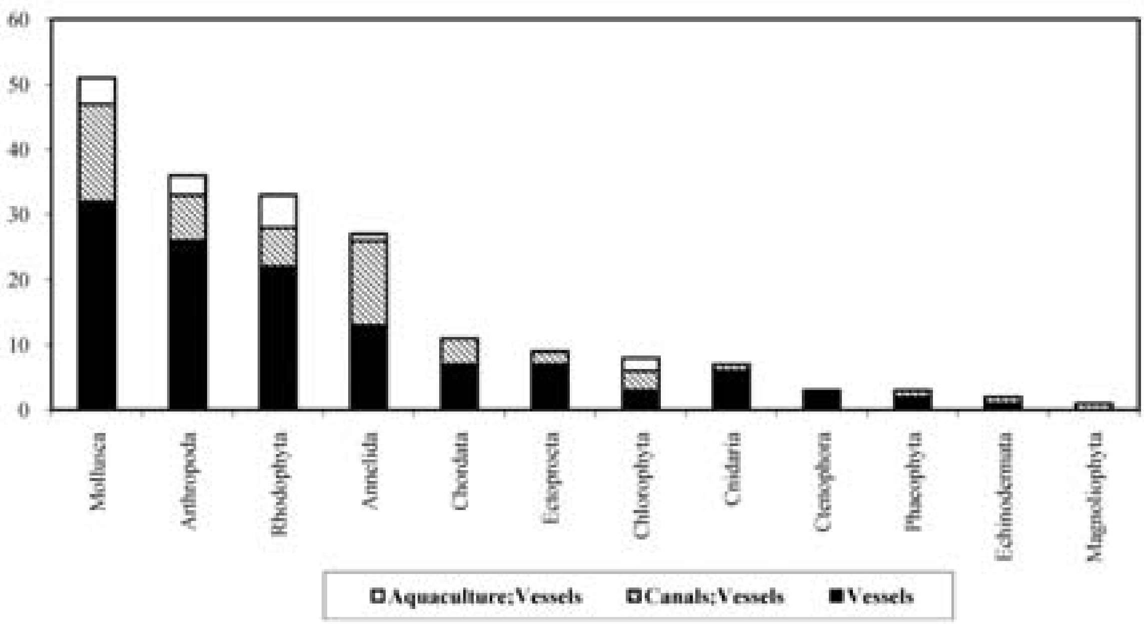

Figure 4.1—Taxonomic classification of the vessel-transported alien species.

5 Cryptogenic: of obscure or unknown origin, 'a species that is not demonstrably native or introduced' (CABI, 2007).

Arthropoda (19%), Rhodophyta (17%) and Annelida (14%).

The date of the establishment of the first population in a new locality is significant for the study of the patterns and processes of invasion. As research efforts vary greatly along the coasts of the Mediterranean, and even the better-studied localities suffer from temporal and taxonomical lacunae, we accept that the date of collection (or, when missing, the date of publication) may be years behind the actual date of arrival, and that taxonomic identification and publication may lag behind collection. As an example, the tube worm *Hydroides brachyacanthus* Rijoa, 1941 was collected in Jaffa, Israel, in 1933 but identified only nearly 60 years later (Ben-Eliahu, 1991).

The number of introductions has been rising steadily, with a distinct surge in records since the 1960s (Figures 4.2, 4.3). The increase in vessel-transported aquatic aliens may be attributed to global political, economical and societal changes. The development of the Middle Eastern oil fields

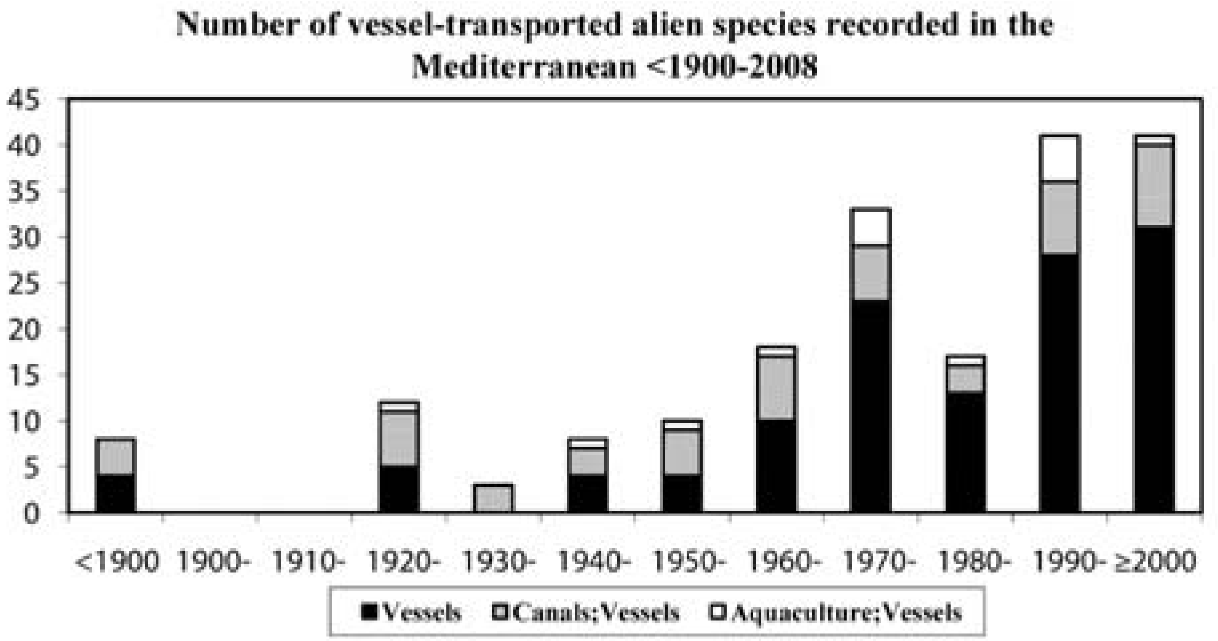

Figure 4.2—Number of vessel-transported alien species recorded per decade in the Mediterranean, <1900–2008.

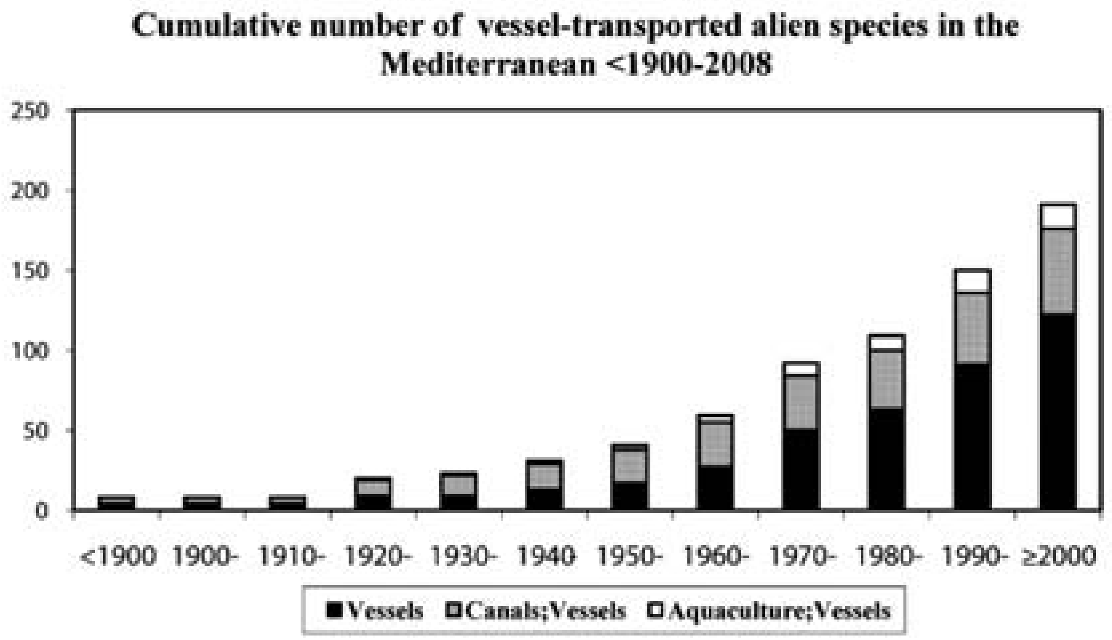

Figure 4.3—Cumulative number of vessel-transported alien species in the Mediterranean, <1900–2008.

and trade globalization, improved water quality in port environments, increased overlap with other introduction pathways (particularly the Suez Canal and aquaculture) and growing research interest and effort have influenced the rate and means of species introductions. An increase in new records in the 1920s and 1970s (Figure 4.2) reflects the publication of the results of the Cambridge Expedition to the Suez Canal, and the joint programme by the Smithsonian Institution, the Hebrew University of Jerusalem and the Haifa Sea Fisheries Research Station, respectively. The decline in the 1980s may reflect the closure of the Suez Canal to maritime traffic between 1967 and 1975 (a slight time lag can be expected due to population growth, research serendipity and publication time).

The native range of vessel-transported alien taxa recorded in the Mediterranean (Figure 4.4) is most commonly the Indo-Pacific Ocean (29%), pantropical (17%) and the Indian Ocean (14%). However, the native range is not necessarily the area of origin for Mediterranean populations, as propagules may have been secondarily introduced from established alien populations elsewhere. With a few notable exceptions, the source populations of alien species in the Mediterranean have not been ascertained by molecular means (Jousson *et al*., 1998; Meusnier *et al*., 2004; Terranova *et al*., 2006; Andreakis *et al*., 2004, 2007). However, it is quite clear that most of the alien species in the Mediterranean are thermophilic,[6] originating in tropical waters (Galil, 2007b).

The dynamics of the vessel-transported alien species in the Mediterranean vary. Of the 191 species listed, 102 are known from a single country and sometimes from a single record, whereas 30 species have been recorded from 5 or more peri-Mediterranean countries. In some cases the interval between the initial establishment and spread has been exceedingly short: *Caulerpa taxifolia* (Vahl) C. Agardh, 1817, spread from a tiny patch off Monaco in 1984 to Mallorca, Elba, Sicily and Tunisia by 2000; *C. racemosa* var. *cylindracea* (Sonder) Verlaque, Huisman and Boudouresque, 2003, spread throughout much of the Mediterranean in the 1990s; and the grapsid crab *Percnon gibbesi* (H. Milne Edwards, 1853), first collected off the Balearic Islands in 1999, was sighted off the southern coast of Turkey in 2005. In other cases a time lag occurs, sometimes extending over half a century: *Fulvia fragilis* (Forsskål in Niehbur, 1775) was collected in Haifa Bay in 1937, but nearly five decades later off Tunisia and Spain and over six decades later off Italy; *Brachidontes pharaonis* (Fischer P., 1870), sighted off Lebanon in 1929, was collected in the eastern Adriatic in 1998 and off Corsica in 1999. These hiatuses cannot be considered artefacts stemming from poor collection and identification,

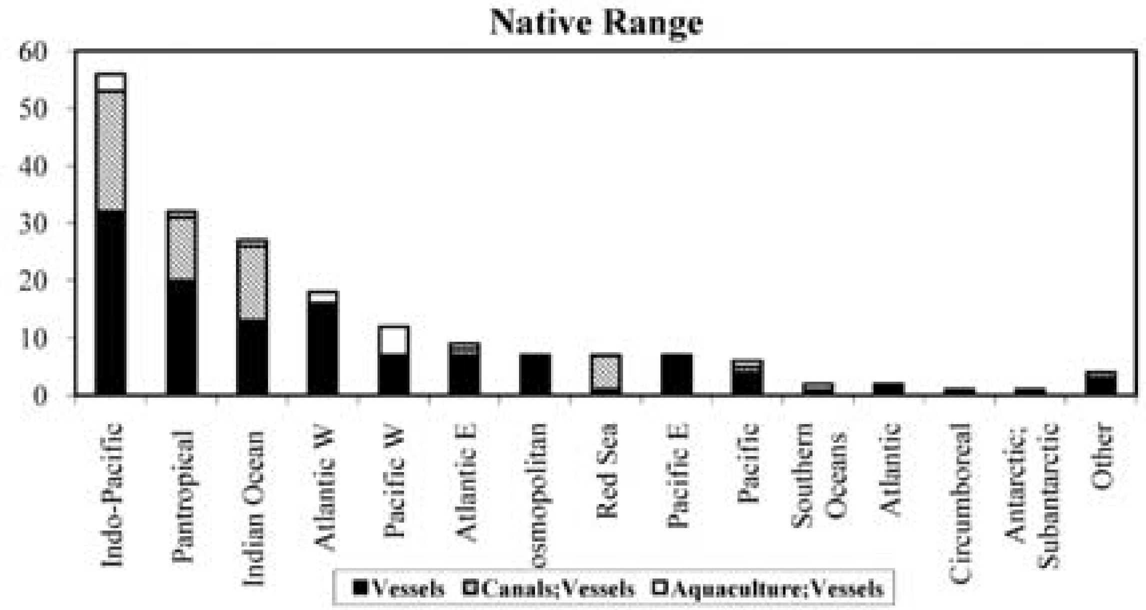

Figure 4.4—Native range of vessel-transported alien taxa recorded in the Mediterranean.

6 Thermophile: an organism which thrives at higher temperatures.

as Mediterranean molluscs had been assiduously studied throughout that period (CIESM, 2002-2004).

Perusal of the dates of first record and of geographical spread discloses a distinct pattern: of the 30 vessel-transported aliens widely distributed in the Mediterranean (recorded from 5 or more countries), seven (*Asparagopsis taxiformis* (Delile) Trevisan de Saint-Léon, 1845; *Cerithium scabridum* (Philippi, 1848); *Halophila stipulacea* (Forsskål) Ascherson; *Hydroides dianthus* (Verrill, 1871); *H. diramphus* Morch, 1863; *H. elegans* (Haswell, 1883); and *Portunus pelagicus* (Linnaeus, 1758)) were first recorded in this sea over a century ago, and 11 more before 1950. The most widely spread aliens entered the Mediterranean through the Suez Canal and dispersed further with maritime traffic (Figure 4.5).

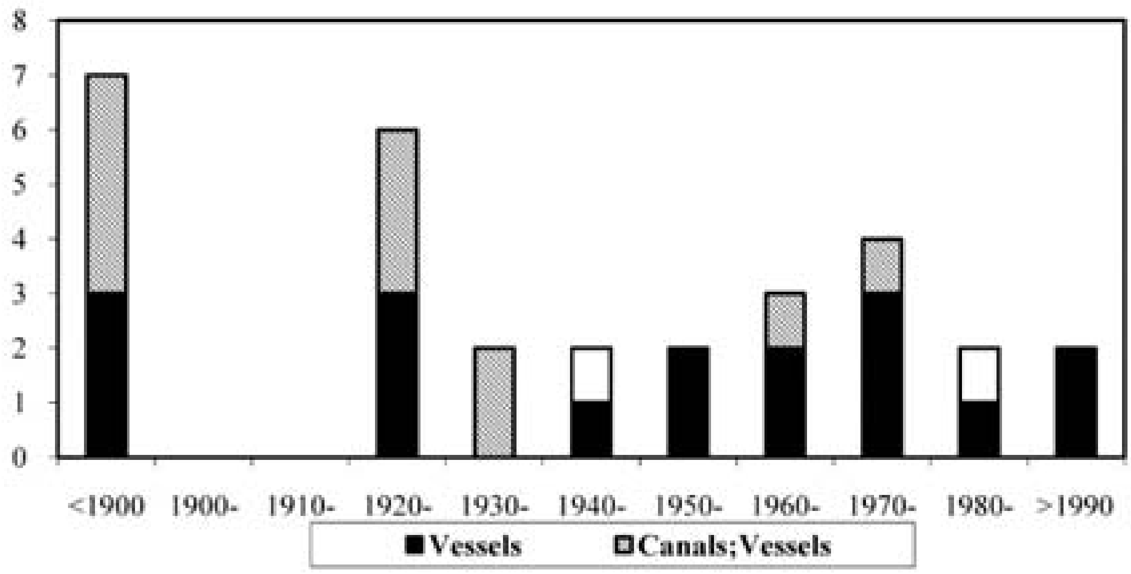

Figure 4.5—Number of most widely spread alien species recorded per decade in the Mediterranean.

3. What is the most important vector and why?

The role of commercial shipping in transporting alien species in ballast tanks and as hull fouling has been widely acknowledged as the most dominant vector for transporting alien species (Cohen & Carlton, 1998, Ruiz *et al*., 2000a; Hewitt *et al*., 2004). Yet few studies have been conducted in the Mediterranean on the relative contribution of either ballast tanks or hull fouling to the burgeoning alien biota (but see Mineur *et al*., 2007). Beginning with the *Karikal* (*see above*), there have been sporadic reports on alien species arriving on heavily fouled vessels. Slower-moving and frequently moored vessels, such as research vessels or drilling platforms employed in offshore oil and gas exploration in the Mediterranean, serve as large artificial reefs and therefore pose a high risk of alien species transmission. Two alien species, *Hydroides elegans* and *Balanus trigonus Darwin*, 1854, were found in the fouling biota of the oceanographic research vessel *Bannock*, which was studied after it spent a year in the Mediterranean (Relini *et al*., 1972). The oil platform *Discovery II* arrived in Genoa, Italy, in 1977 from the Indian Ocean carrying 'una ricca fauna tropicale vivente tra cui Teleosti Blennidi e Scorpenidi e Decapodi'[7] (Relini Orsi and Mori, 1979). The oil-drilling platform *Southern Cross* was brought from Australia to Haifa Bay, Israel, in 2003 for maintenance work including in-water scraping of its extensive fouling. The local divers employed described unfamiliar fish and crustaceans among the dense fauna, and from the shells that had been collected by the divers 12 species of alien molluscs were identified (Mienis, 2004). As the operational scenarios of these vessels differ greatly from those prevailing in commercial shipping, it is assumed that the data concerning the presence of fouling alien biota is not necessarily representative of commercial traffic.

Rich biotic assemblages composed of protists, phyto- and zooplankton, invertebrates and fish have been recorded from ballast tanks (e.g. Gollasch *et al*., 2000b). Some of the vessels involved in the earliest studies into the patterns of survivorship of tank-entrained biota during transit and of taxa that might comprise an inoculant pool originated in the Mediterranean. Over 70 planktonic taxa (mostly copepods, dinoflagellates and meroplankton) were collected from the cargo hold of a coal carrier transporting ballast water from Israel to Chesapeake Bay, USA, on 13 voyages between 1994 and 1997 (Smith *et al*., 1999). Ballast sampling and open-ocean exchange experiments were also conducted in 1995 aboard a coal carrier ballasting in Hadera, Israel, and deballasting in Baltimore, USA. Fifty live taxa were identified and taxon-specific trends in plankton mortality were quantified (Wonham *et al*., 2001). In 1999 the microbial community in exchanged and unexchanged ballast-water holds was examined during another voyage from Hadera to Baltimore (Drake *et al*., 2002). In 2003 ballast water was sampled in 15 vessels arriving in the port of Koper, Slovenia. All the voyages but one originated in the Mediterranean Sea. The presence of rich entrained biota, including an alien calanoid copepod, *Acartia tonsa* Dana, 1849, underscores the importance of discharged ballast in secondary introductions (David *et al*., 2007). Similarly, three alien dinoflagellates were identified in ballast water sampled from 19 vessels arriving in Taranto Harbour, Italy (Saracino & Rubino, 2006).

Assuming that the likelihood of finding shipping-transported alien species is highest near their point of inoculation, one would assume that sampling would be concentrated on habitats and sites in ports and adjacent areas. Though Mediterranean ports have been surveyed for half a century, most studies have focused on pollution monitoring (e.g. Bellan-Santini, 1968; Ergen and Önen, 1983; Rebzani-Zahaf and Bellan, 2004). However, the realization that ports may become focal points of biotic invasion have

7 '…a rich live tropical fauna, including blennid and scorpaenid Teleostei and Decapoda.'

induced researchers to pay attention to alien taxa. Recently, a study of the soft-bottom communities of Alsancak port, İzmir, Turkey, following recovery from pollution, found that two of the six alien species—*Streblospio gynobranchiata* Rice and Levin, 1998, and *Polydora cornuta* Bosc, 1802—accounted for more than 70% of the specimens, while the alien bivalve *Anadara demiri* (Piani, 1981) contributed most to the biomass, making up 93% of the biomass at one site (Çinar *et al*., 2006b). Studies of port and port-proximate areas yield on occasion single specimens of alien species, such as *Heteropanope laevis* (Dana, 1852) in the port of Anzio (Taramelli, 1957); *Thalamita gloriensis* Crosnier, 1962 from Genoa Harbour (Relini Orsi & Mori, 1979); *Menaethius monoceros* (Latreille, 1825) from Punta Ala, Italy (Falciai, 2003); and *Actumnus globulus* Heller, 1861, near Punta Ala, Italy (Galil *et al*., 2006). On other occasions, evidence of already well-established populations of alien species comes to light. In 2001 the semelid bivalve *Theora (Endopleura) lubrica* Gould, 1861 was collected inside Livorno Harbour, Italy, where a thriving population was found 'in an area utilized by ships for goods in transit' (Balena *et al*., 2002), and again in 2006, in Haifa Bay, next to Haifa port (Bogi & Galil, 2007). Thriving populations of the crabs *Grapsus granulosus* H. Milne Edwards, 1853, and *Plagusia squamosa* (Herbst, 1790), were recently collected from the ports of Zarzis, Tunisia and Tripoli, Libya, where they may have arrived in the fouling community of vessels that had traversed the Suez Canal (Zaouali *et al*., 2007).

The first basin-wide port-survey programme targeting alien species was initiated by the Mediterranean Science Commission in 2003 (CIESM, 2005). The survey targeted fouling organisms such as macrophytes, bryozoans, serpulids, hydroids, ascidians, molluscs and barnacles inhabiting port and port-proximate man-made hard substrates, as well as organisms that pose a significant risk to human health that might be disseminated by shipping (e.g. *Vibrio cholerae* and dinoflagellate cysts). The survey methods followed the CRIMP protocols for baseline port surveys for alien species developed by Hewitt and Martin (1996), updated (Hewitt & Martin, 2001) and later adopted by the Global Ballast Water Management Programme (GloBallast, 2008). Ten sites were sampled: Barcelona harbour, Spain; Rades, Skhira and Zarzis harbours, Tunisia; İzmir harbour, Turkey; Hadera port, Israel; Venice, Taranto and Livorno ports, Italy; and Valetta harbour, Malta. Among the material identified were alien serpulids of species that can be expected to occur in port environments. *Microcosmus squamifer* Michaelsen, 1927, was identified from Livorno; *Ascidia* cf. *savigni* and *Phallusia nigra* Savigny, 1816—both Erythrean species—were found in the Israeli samples, as was *Balanus reticulatus* Utinomi, 1967. The widely invasive amphipod *Caprella scaura* Templeton, 1836, was identified from Livorno port. Of special concern are possible human pathogens such as the bacteria *Vibrio cholerae* Pacini, 1854, 01 and 0139, agents of human cholera. *Vibrio cholerae* is endemic in the Mediterranean and, indeed, a survey of plankton arriving in ballast water in Chesapeake Bay, USA, from the Mediterranean revealed viable *Vibrio* bacteria (Ruiz *et al*., 2000b). Samples collected from these ports were analysed for the presence of the two toxigenic serogroups of *Vibrio cholerae* by fluorescent antibodies. The serogroups were determined in 4 of the 12 ports sampled. A subset of the live samples was tested for the presence of *V. cholerae* (no information on their toxicity or serogroup): half of the 10 samples tested were positive (Drake *et al*., 2007).

In the same way as for primary introductions, shipping is also an important vector for secondary introduction. The widely invasive algae *Sargassum muticum* (Yendo) Fensholt, 1955, and *Caulerpa taxifolia* (Vahl) C. Agardh, 1817, spread across the Mediterranean on fishing boats and recreational craft (Knoepffler-Péguy *et al*., 1985; Meinesz, 1992). The Erythrean mytilid *Brachidontes pharaonis* has spread as far west as Sicily in ship fouling.

Examination of the relative importance of all the different pathways in the western and eastern basins in the Mediterranean shows that in the western basin a slightly lower number of vessel-transported aliens has been recorded (117 vs 129; Figure 4.6) than in the eastern Mediterranean. In the western basin 72% of the aliens are primary introductions by vessels, 17% entered the Mediterranean through the Suez Canal and were subsequently transported by vessels (secondary introductions), and 11% were introduced with aquaculture and secondarily transferred by vessels. In the eastern basin 55% are primary introductions by vessels, 40% were transferred by vessels after entering the Mediterranean through the Suez Canal (secondary introductions), and only 5% were introduced with aquaculture and secondarily transferred by vessels. Examination of the table in the Appendix shows that of the 191 ship-transported aliens, 134 are presumed[8] to have been transported in the fouling community and only 21 in ballast tanks. The vectors of all other species are unclear: some may have entered as ship-foulants or in the fouling community of aquaculture shellfish; others moved on their own through the Suez Canal, or in the fouling of vessels transiting through the Canal; and for some 28 it is unclear whether ballast or fouling served as the vector.

In 2001, 33,000 vessels entered the Mediterranean according to Lloyd's List. With the exception of the South Pacific and the North-East Pacific '… the Mediterranean is connected by considerable sea trade with most global regions.' (CIESM, 2002:6). Inbound commercial traffic arrives through the Straits of Gibraltar (81%) and the Suez Canal (19%). Of the Gibraltar entries, most vessels arrive from the Atlantic north-east (74%); the remainder come mostly from the Caribbean, Gulf of Mexico and Central America (8%), Atlantic north-west (6%), West Africa (5%) and Atlantic South America (5%). However, because of the complex pattern of commercial traffic, the last port of call for vessels entering the Mediterranean differs frequently from the voyage point of origin, thus 56% of the vessels entering the Mediterranean had their last port of call on the Atlantic coast of Europe and North Africa, 38% in tropical or subtropical waters, and 6% in other temperate seas (Dobler, 2002).

The disparity between the shipping patterns and the origins of the alien species in the Mediterranean is striking: whereas two-thirds of the vessels originate in North Atlantic ports, species originating in the North Atlantic Ocean compose only 14% of the alien biota; nearly half of the species originate in the Red Sea, Indian Ocean and the Indo-West Pacific, but less than one-fifth of the inbound traffic originates in these waters.

A successful ship-introduced invasion is a complex process in which organisms must survive both the significantly selective transit (either in fouling or in ballast) and the conditions in the recipient port, and also need to arrive in sufficient numbers to establish a self-sustaining population outside their native range. The apparent disparity between the number of invasion opportunities (transport frequency) and actual species invasions (invasion success) in the Mediterranean may be due to the considerable difference in salinity and temperature between the North Atlantic and Mediterranean ports. These differences apparently reduce the risk of the establishment of most organisms habituated to North Atlantic conditions, but allow ready inoculation and establishment by thermophilic alien species.

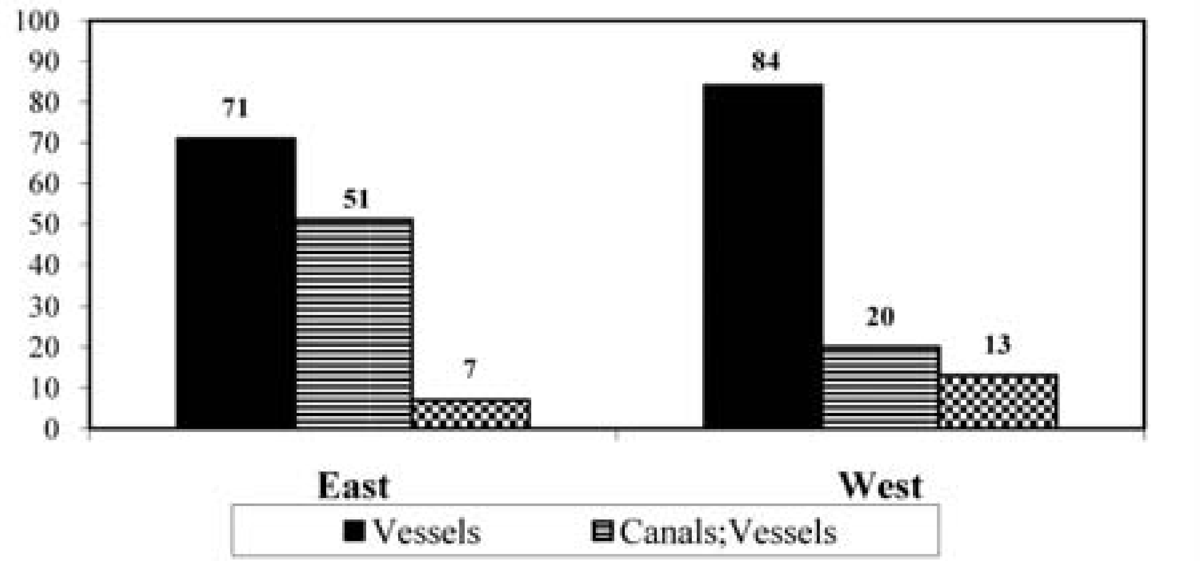

Figure 4.6—Number of species introduced by each pathway in the eastern and western basins of the Mediterranean.

8 Analysis is speculative because of the uncertainty concerning the vectors responsible for any given introduction.

4. Effects of vessel-introduced marine species on ecological communities

The impacts of introduced alien species have been extensively reported and many cases of their ecological, economic and health threats have been recorded (Parker *et al*., 1999; Ruiz *et al*., 1999).

The most conspicuous impacts are posed by species that have reached the 'massive development' stage (Occhipinti-Ambrogi & Galil, 2004), and thus have given rise to 'biological invasions' (Carlton, 1996), which often includes adverse effects on man-made structures (fouling and clogging) or economic activities (declining fisheries and aquaculture output). Potential impacts on public health, such as those arising from the documented transport of *Vibrio cholerae* and other species of *Vibrio* by ship ballast water (Drake & Galil, 2007) or from harmful algal blooms (HAB) of alien microalgae, are obviously of high concern to the public. The presence of the toxic dinoflagellate *Alexandrium catenella* (Whedon and Kofoid, 1936) Balech, 1985, has been assessed during various algal blooms (Vila *et al*., 2001). In Thau Lagoon, France, it has been attributed to introduction via ballast water (Lilly *et al*., 2002). A genetic analysis suggests that it originated in Japan, and subsequently spread to Spain, France and Italy (Penna *et al*., 2005).

As far as biodiversity is concerned, impacts may be more subtle and much more widespread. The establishment of alien species in native communities induces a complex range of consequences, depending on the interaction between species, the ability of the introduced species to modify the habitat or the ecosystem energy flow in the new environment (Crooks, 2002; Wallentinus & Nyberg, 2007).

Marine alien species have been considered a potential cause of the local decimation of native species (Carlton, 1993; Carlton *et al*., 1999; Clavero & García-Berthou, 2005; Galil, 2007a), even though in some cases the evidence is only circumstantial (Gurevitch & Padilla, 2004). Dramatic changes in the dominance patterns of species have been documented in numerous ecosystems after the introduction of alien species; in one of the most spectacular examples, the plankton of the Black Sea shifted from crustacean assemblages to the introduced ctenophore *Mnemiopsis leidyi* A. Agassiz, 1865, and this in turn caused profound changes in the fish assemblages feeding on microplankton and eventually, in conjunction with water pollution and overfishing, to the collapse of the fisheries (Kideys, 2002).

Irrespective of the difference in the total number of species present in any given community before and after the entry of aliens, there is a tendency towards homogenization of species composition in some marine coastal environments. A few common species distributed everywhere are likely to dominate particular biological assemblages. The unique traits of many coastal and even oceanic communities are disappearing. This tendency is in itself a subtle, yet evident threat to global biodiversity (Lewis *et al*., 2003).

The Mediterranean Sea, with its high endemism and unique communities, has one of the highest numbers of marine species introductions (Drake & Lodge, 2004; Flagella & Abdulla 2005; Streftaris *et al*., 2005; Galil, 2007a, 2008a, 2008b); hence all the causes for concern mentioned above can apply, acting, *inter alia*, as potent drivers of change to its biodiversity (Gritti, 2006; Bianchi, 2007). Some actual or potential impacts caused by alien species in the Mediterranean have been summarized elsewhere (Galil, 1999, 2000, 2007a, 2008a, 2008b; Boudouresque & Verlaque, 2002; Zibrowius, 2002; Zenetos *et al*., 2006).

The following paragraphs (summarized in Table 4.1) include Mediterranean examples of (a) high population growth rates of alien species that affect extensive areas and result in local ecosystem changes; (b) changes and shifts in native communities induced by multiple introductions of alien species; and (c) alien–native interactions that culminate in community and ecosystem changes.

Table 4.1--Summary of the impacts of the species discussed in the text.

Case study / species	Main type of impact	Severity of threat	Spread	Reference
Caulerpa taxifolia	Competition for space; habitat modifier	* * *	vr/large areas	Meinesz et al., 2001
Caulerpa racemosa var. cylindracea	Competition for space; habitat modifier	* * *	er/large areas	Piazzi & Cinelli, 2003
Acrothamnion preissii	Extensive cover over other species	*	r	Boudouresque & Verlaque, 2002
Womersleyella setacea	Extensive cover over other species	*	r	Boudouresque & Verlaque, 2002
Asparagopsis taxiformis	Extensive cover over other species	*	r	Andreakis et al., 2007
Asparagopsis armata	Extensive cover over other species; toxicity	* *	r	Sala & Boudouresque, 1997
Ficopomatus enigmaticus	Habitat modifier	i	s/localized	Fornós et al., 1997
Other serpulids	Competition for space	i	s	Zibrowius, 2002
Rapana venosa	Predation; food web modifier	* *	s/regional	Savini & Occhipinti Ambrogi, 2006
Dyspanopeus sayii	Competition with native species	* *	er/localized	Mizzan et al., 2005
Tricellaria inopinata	Competition with native species	* *	r/localized	Occhipinti-Ambrogi, 2000a
Musculista senhousia	Habitat and food web modifier	* * *	vr/localized	Mistri, 2003
Codium fragile ssp tomentosoides	Extensive cover over other species	* *	r/large areas	Bulleri et al., 2006a
Percnon gibbesi	Grazing	i	vr/large areas	Yokes & Galil, 2006

Severity of threat: i indifferent; * weak; * * moderate; * * *strong.
Spread: s slow; r rapid; vr very rapid; er extremely rapid.

4.1. Large scale invasions and ecosystem changes

4.1.1. *Caulerpa* species

One of the most notorious cases of a biological invasion is that of *Caulerpa taxifolia*. Originating in an apparently innocuous discharge from the Oceanographic Museum and Aquarium of Monaco (Jousson *et al*., 1998), this green alga has spread first along the coasts of France, Italy, Croatia, Tunisia and Spain (Balearic Islands) forming dense meadows in some coastal areas (Meinesz *et al*., 2001). Recently, a different strain of *C. taxifolia* has been identified from the Gulf of İskenderun, Turkey, and is considered to have been either vessel-introduced or aquarium-released (Cevik *et al*., 2007). The regression of formerly dominant *Posidonia* meadows, which were already under pressure due to anthropogenic impact, has presumably accelerated under the onslaught of the alien *Caulerpa*, and their native biota has been significantly reduced (Marbá *et al*., 2005). However, a compact airborne multispectral imaging survey of an area off the French Mediterranean coast colonized by *C. taxifolia*, which was validated by an extensive underwater survey, indicates that its cover had been overestimated by at least one order of magnitude and that *C. taxifolia* might not have had a substantial impact on the cover of *P. oceanica* (L.) Delile (Jaubert *et al*., 2003). A congener, *Caulerpa racemosa* var. *cylindracea* (Sonder) Verlaque, Huisman and Boudouresque, 2003, native to southwestern Australia, has accomplished an even more spectacular invasion. Its colonization of the western Mediterranean started in the 1990s and within one decade it had been documented along the coast of 11 states, from France to Croatia, growing on every kind of substrate at depths ranging from the surface to 70 m (Piazzi *et al*., 2005).

4.1.2. Other algal invasions

In Mediterranean harbours the large number of introduced macrophytes (more than 90 taxa) represents 6.5% of the sea's rich flora (about 1,400 species). Some of the alien algae are believed to have been introduced by ships: *Acrothamnion preissii* (Sonder) E.M.Wollaston, 1968, and *Womersleyella setacea* (Hollenberg) R.E. Norris, 1992. Both form dense carpets that prevent the development of other species and reduce species diversity and equitability (Ribera & Boudouresque, 1995; Boudouresque & Verlaque, 2002).

An alien red alga, *Asparagopsis taxiformis* (Delile) Trevisan de Saint-Léon, 1845, was recorded from the Egyptian coast in the nineteenth century, and beginning in the 1970s it reappeared in the eastern Adriatic, Greece and Turkey (Güven & Öztiğ, 1971; Koussouris *et al*., 1973; Span *et al*., 1998; Andreakis et al., 2004, 2007). In 1993 it was found off Lampedusa Island, in the straits of Sicily, and in fast succession expanded along the Italian peninsula, reaching the Gulf of Naples. On rocky shores it forms a continuous belt, especially during the cold season, coexisting in many sites with *Caulerpa racemosa* (Flagella *et al*., 2003). It has significantly changed the rocky littoral in the Gulf of Naples within a decade. Another red alga, *Asparagopsis armata* Harvey, 1855, native to Australia and New Zealand, has probably entered through the Gibraltar Strait and has colonized the western Mediterranean, starting from the Algerian coast in 1923. The toxic secondary metabolites produced by this alga are effective against native grazers such as amphipods, abalone and sea hares (Paul *et al*., 2006) and probably induce large grazers such as the sea urchin *Paracentrotus lividus* (Lamarck, 1816) and the fish *Sarpa salpa* (Linnaeus, 1758) to shun it, freeing it to form dense stands (Sala & Boudouresque, 1997).

4.1.3. Invasive serpulid worms

The Indo-Pacific reef-building worm *Ficopomatus enigmaticus* (Fauvel, 1923) was first noted in the Mediterranean in the 1920s off the coast of Spain (as *Mercierella enigmatica*). In both its natural and invaded habitats the reefs it builds are well-known features in many lagoons or enclosed environments of variable salinity. The biogenic construction may start on a patch of hard substrate but may overgrow onto soft substrates. *F. enigmaticus* is considered an ecosystem engineer as its extensive reefs alter local physical features and hydrodynamics. The presence of these reefs may increase the abundance and diversity of benthic species, but in some places they may interfere with coastal structures, and may clog pipelines and cooling water canals (Bianchi & Morri, 1996, 2001; Fornós *et al*., 1997). *Hydroides elegans*, *H. dianthus* and *Spirorbis marioni* Caullery and Mesnil, 1897, are invasive in harbour environments all over the Mediterranean. *Pomatoleios kraussii* (Baird, 1865) has been highly successful in the Levantine basin. In addition to *P. kraussii*, other Erythrean serpulids have spread over the Levantine area (e.g. *Hydroides minax* (Grube, 1878)) (Zibrowius, 1991). According to a recent study of the Levantine coasts of Turkey, the alien serpulid species dominated rocks, molluscs and artificial substrates (such as dock pilings, ropes and tyres), making up more than 95% of the individuals found in these habitats. They also constituted more than 85% of the epiphytic polychaetes on algae such as *Ulva* and *Cystoseira* (Çinar, 2006a, 2006b).

4.1.4. The veined rapa whelk

Rapana venosa (Valenciennes, 1846), which originated in the Indian Ocean and the Sea of Japan, is reported as a voracious predator on indigenous bivalves from the Black Sea, and is considered a major pest. This snail was also introduced to Chesapeake Bay, Río de la Plata and the French Atlantic coast (ICES, 2004). It was found in 1986 near natural oyster and mussel banks, next to the port of Thessaloniki, Greece (Koutsoubas & Voultsiadou-Koukoura, 1991). Though dense populations of *R. venosa* have been recorded from the northern Adriatic, no clear impact has been noted in the 30 or more years since its arrival. Experimental studies on its reproduction and diet have been conducted on a high-density population on artificial breakwaters in the north-western Adriatic (Savini & Occhipinti-Ambrogi, 2006). Recently, live specimens have been reported from Santa Gilla Lagoon (Sardinia), and Sabaudia Lagoon (Pontine region, in central

Italy), raising concern about new outbreaks (Crocetta & Soppelsa, 2006).

4.2. Changes in native communities: Benthic macrofauna in the lagoon of Venice and northern Adriatic Sea

Thirty-seven alien species have been recorded from the Lagoon of Venice, a large brackish-water basin in the northern Adriatic (Occhipinti-Ambrogi, 2000a). Several of these species may have been introduced by shipping. Decapod crustaceans are more likely to be introduced by commercial maritime traffic than by any other vector; their larvae can survive long periods in ballast water and adults of smaller decapods are frequently found during hull fouling studies. The panopeid crab *Dyspanopeus sayi* (Smith, 1869) from the north-western Atlantic coast was first reported in the Venice Lagoon in 1992 (Froglia & Speranza, 1993), but according to local fishermen it had occurred in the lagoon since the 1980s. It may have been vessel-transported or unintentionally imported with shellfish seed. Its abundance peaked in the 1990s, exceeding the abundance of the native crabs *Carcinus aestuarii* Nardo, 1847, and *Pilumnus hirtellus* (Linnaeus, 1761) in many areas of the lagoon, but it has now regressed (Mizzan *et al*., 2005). The Lagoon of Venice was probably the starting point for the spread of the Indo-Pacific bryozoan *Tricellaria inopinata* (d'Hondt and Occhipinti-Ambrogi, 1985) in European waters before it was transported by fouling towards the English Channel and the North Sea (Dyrynda *et al*., 2000; Occhipinti-Ambrogi, 2000a, 2000b).

The Asian date mussel, *Musculista senhousia* (Benson in Cantor, 1842) is found in many Mediterranean localities, including the harbours of Taranto in the Ionian Sea, and Leghorn and Olbia in the Tyrrhenian Sea (Mastrototaro *et al*., 2003; Campani *et al*., 2004). It has formed dense mats on subtidal soft sediments in the Po Delta: its effects on the benthic communities have been documented after a 10-year period of successful colonization (Mistri, 2003, 2004; Mistri *et al*., 2004). Since 1994, when the species was first found, *M. senhousia* has spread throughout the Goro lagoon and within a few years reached densities exceeding 104 individuals per m^2, and biomasses of up to 87g ash-free dry weight (AFDW) per m^2. Annual production estimates were around 50g AFDW per m^2 per year. It lives in subtidal soft sediments, forming byssal mats, which are typically raised a few millimetres above the sediment. The mats alter hydrodynamic, sedimentary, and microtopographic features on the bottom, as well as active and passive responses of larvae, juveniles, and adults of other species in the community. As a result, suspension-feeding infaunal bivalves may be adversely affected by *M. senhousia*. In the Goro lagoon, the abundance of macrobenthic species was significantly higher in the presence of mussel mats. Some native faunal groups have taken advantage of the mats: small gastropods (such as *Hydrobia ulvae* (Pennant, 1777)), amphipods (*Microdeutopus gryllotalpa* Costa, 1853; *Gammarus aequicauda* (Martynov, 1931)) and tube-building spionids (*Streblospio shrubsolii* (Buchanan, 1890)). All these species are surface-feeders and probably benefited from the enhanced deposition of particulate matter due to reduced water flow through the mussel matrix, as well as from mussel faeces and pseudofaeces. The growth and mortality of two species of deep-dwelling clams with long siphons, the indigenous *Tapes decussatus* (Linnaeus, 1758) and the alien *Ruditapes philippinarum* (Adams and Reeve, 1850), were not affected by the presence of *M. senhousia* mats.

4.3. Alien-native interactions

4.3.1. Introduced Caulerpales and native macrophytes in the Tyrrhenian Sea

Where four invasive alien algae—*C. racemosa*, *Acrothamnion preissii, A. armata* and *Womersleyella setacea*—co-occur along the Tyrrhenian coast of Italy, with *C. racemosa* displaying the highest biomass, they have significantly affected native communities (Piazzi & Cinelli, 2003). The interaction of the two invasive alien species of *Caulerpa* has been studied (Piazzi & Ceccherelli, 2002; Piazzi *et al*., 2003a; Balata *et al*., 2004). *C. racemosa* was the superior competitor despite having arrived later than *C. taxifolia*, and had worse impacts on native algae.

A possible competitive advantage of *C. racemosa* over indigenous species may be its resistance to high sedimentation rates: the synergetic occurrence of increased sedimentation and colonization by *Caulerpa* has caused a considerable decrease in the percentage cover of the main native algae (Piazzi *et al*., 2005). On the other hand, native turf-forming algae, which locally increase the sedimentation rate, offer a good cohesive surface, enhance the attachment of rhizoids of *C. racemosa* and favour its initial settlement (Piazzi *et al*., 2003b).

4.3.2. Competition for space on hard substrates and shifting equilibria in unstable environments

Codium fragile ssp. *tomentosoides* (Suringar) Hariot, 1889, native to the north-west Pacific, is abundant on breakwaters placed in shallow sandy bottoms parallel to the north-western Adriatic coastline (Bulleri *et al*., 2006a). This alga grows larger and more branched in sheltered habitats, leading to greater biomass on the landward side of the breakwaters. The breakwaters also bear large populations of the native mussel, *Mytilus galloprovincialis* Lamarck, 1819. Experiments have demonstrated that by settling first the mussels are able to reduce recruitment of the alga. It seems that young specimens of *Codium* form complex three-dimensional clumps which help mussels to settle on the smooth, bare surface of the quarried blocks used to build the breakwaters. However, the developed canopies of mature *Codium* affect the recruitment, survival and growth of under-storey mussels (Bulleri *et al*., 2006b).

4.3.3. Lack of competitors leading to rapid expansion

The grapsid crab *Percnon gibbesi* was first collected in the Mediterranean in the summer of 1999 from Linosa Island and the Balearic Islands. Later records attest to its spread in the Sicilian Channel, on the Catalan coast and northwards along the Tyrrhenian coast of Italy to Giglio Island and along the Ionian coast of Calabria, Greece and Turkey (Yokes & Galil, 2006). In these areas it formed thriving populations in an amazingly short period of time. Its subtidal habitat and crevicolous habits, its recent records in or near ports and marinas, and its rapid dispersal—within six years—across the Mediterranean Sea, suggest primary and secondary ship-borne transport, by commercial, recreational or fishing vessels (Cannicci *et al*., 2006). The morphological characteristics of its pincers and stomach and the analysis of its stomach contents indicate that *P. gibbesi* is a strictly herbivorous species. Few native Mediterranean crab species inhabit the rocky infralittoral, and none is as large as the invasive grapsid or as strictly herbivorous (Puccio *et al*., 2006).

5. Conclusions

International conferences on the environment in the past decade, including the meetings of the Conference of Parties to the Convention on Biological Diversity, the World Summit on Sustainable Development in 2002, and the Fifth World Congress on Protected Areas in 2003, have invariably highlighted the issue of invasive alien species and called upon governments to act 'to prevent the introduction of, control or eradicate those alien species which threaten ecosystems, habitats or species' (CBD, 1992: Article 8h).

The changes in marine biodiversity in the Mediterranean Sea are of major concern. Though no extinction of a native species is known, sudden declines in abundance and even local extirpations of native species concurrent with the proliferation of aliens have been recorded (Galil, 2007a). The establishment of alien biota and the adverse changes in the native communities are part of an anthropogenic ecosystem shift in the Mediterranean Sea, the consequence of a synergetic complex of stressors—pollution, eutrophication, destruction and fragmentation of habitats, overexploitation of fisheries, fast-expanding mariculture, widening of the Suez Canal, climate change, and a steep increase in maritime traffic—superimposed on a unique physiographical and climatic history (Galil, 2007a, 2008a, 2008b; Occhipinti-Ambrogi, 2007).

Though modern anti-fouling technology has decreased the chances of dispersal of fouling species, and rapid port turn-around times and fast ships are less conducive to the settlement and survival of transported biota, recent inventories of alien biota in the Mediterranean Sea have shown a growing trend in the number of ship-transported species, with a distinct surge in records since the 1960s. This has raised concerns among researchers and policy and management personnel dealing with anthropogenic changes in the marine environment. With 191 ship-transported aliens in the Mediterranean Sea, some habitats have already been significantly and adversely affected. The most frequently recorded ship-transported alien taxa are Mollusca, Arthropoda, Rhodophyta, and Annelida. The alien species are not uniformly distributed: heavily invaded areas occur in ports

Figure 4.7—Map of the Mediterranean Sea with the number of vessel-transported alien taxa recorded from each region/country. The asterisk denotes the region covering Libya, Malta and Tunisia.

and in anthropogenically impacted coastal habitats (which serve as contagion reservoirs *sensu* Floerl and Inglis, 2005), although even 'pristine' coastal areas are not wholly free from alien species. The highest numbers of vessel-transported alien species have been recorded from Italy (87), Turkey (65) and Israel (54) (Figure 4.7). These numbers reflect research intensity and, for the latter countries, the overlap of pathways, with primary introduction through the Suez Canal and secondary dispersal by shipping. Most of the vessel-transported alien species in the Mediterranean are thermophilic, originating in tropical waters. An increase in the average temperature of Mediterranean waters, which is already under way (Bianchi & Morri, 2003), is likely to have a significant influence on the establishment and distribution of thermophilic species. Rising seawater temperature may change the pool of species which could establish themselves in the Mediterranean, enable warm-water species (native and alien) to expand beyond their present distributions, and affect a suite of population characteristics (including reproduction and survival) that determine interspecific interactions and, therefore, the dominance and prevalence patterns of both native and alien species, and provide thermophilic aliens with a distinct advantage over the native Mediterranean biota.

6. Recommendations

There is a clear need to increase awareness of ship-transported alien marine biota as a major issue affecting native biodiversity in the Mediterranean.

The peri-Mediterranean countries possess a valuable instrument, were they ready to tackle marine bioinvasions. The Barcelona Convention (UNEP, 1976) and its relevant protocols, initially aimed at reducing pollution, have been updated with the adoption of new protocols. The Protocol concerning Mediterranean Specially Protected Areas (SPA), which was originally adopted in 1982 and came into force in 1986, prohibits 'the introduction of exotic species' (UNEP, 1982: Article 7e). In 2002 the Mediterranean Action Plan (MAP) of the United Nations Environment Programme (UNEP) drafted an 'Action Plan concerning species introductions and invasive species in the Mediterranean Sea' (UNEP, 2002). Both versions of the disputed Article 7 recognise that shipping is a major vector for introductions into the Mediterranean Sea. Article 22 of the Action Plan strongly recommends that, 'Given the importance of shipping-mediated introductions of non-indigenous species into the Mediterranean, … a regional project be developed to overcome gaps for the Mediterranean countries, and strengthen the capacities of the countries to reduce the transfer of aquatic organisms via ships' ballast water and sediments and hull fouling.' In 2005 the Regional Activity Centre for Specially Protected Areas (RAC/SPA) convened a meeting of scientists to identify the most important taxonomic and geographic data gaps concerning Mediterranean alien biota, and to propose guidelines for the prevention of biodiversity loss caused by vessel and mariculture-introduced alien species. These are meant to assist the Contracting Parties to the Barcelona Convention in implementing the new SPA Protocol, which calls on them to take '… all appropriate measures to regulate the intentional or non-intentional introduction of non-indigenous species … into the wild and prohibit those that may have harmful impacts on the ecosystems, habitats or species' (UNEP, 1995: Article 13.1). The desired goal would be the creation of a new protocol specifically aimed at preventing further changes to biological diversity due to the deleterious effects of the intentional and unintentional introductions of alien biota in the Mediterranean.

In addition it is recommended that the peri-Mediterranean countries sign the International Convention for the Control and Management of Ships' Ballast Water and Sediments (IMO, 2004) developed by the International Maritime Organization (IMO), the United Nations body which addresses shipping. This convention recommends ballast water exchange at sea as an intermediate management option until a more stringent standard is required. This standard may only be reached with ballast water treatment systems, which are still under development. To enter into force the convention must be ratified by 30 states, representing 35% of world merchant shipping tonnage. Only three Mediterranean countries, Egypt, Spain and Syria, have signed the Convention so far (IMO, 2008).

7. Biodiversity offsets

The ethics of environmental stewardship and the wish to reduce the footprint of shipping demand that the impacts of ship-transported aliens be addressed. The IMO examined the industry's role in transporting alien aquatic species and recognised that it has a vital role—alongside governments—in working towards sustaining marine biodiversity, hence the Convention (*see above*). The impacts of vessel-transported aliens in the Mediterranean cannot readily be mitigated without significant resources and require cooperation among governments and a range of stakeholders.

The concept of biodiversity offsets[9] is under consideration as a route to finance biodiversity-enhancing actions while balancing demands for development (ten Kate *et al*., 2004). There is as yet no consensus on whether biodiversity offsets are viable and valid conservation mechanisms, and no agreement as to activities likely to give rise to the greatest conservation benefit. There are no data on how shipping impacts on biodiversity can be measured and no agreement whether they should include both primary (resulting from vessels alone) and secondary impacts (impacts of ports and shipping canals), or how this measured value can be compared to the value of the offsets. Secondary impacts are particularly controversial as they are outside the control of the shipping industry, and are greatly influenced by local government policy.

Unlike terrestrial environments, the interconnectivity of the marine environment means that the trading of degraded areas of relatively minor biodiversity for areas of greater biodiversity value or the trading of smaller, unmanaged habitats for larger, consolidated conservation areas is not applicable. Many vessel-transported alien species in the Mediterranean have spread beyond their original point of arrival, either on their own or by overlap with other vectors. There is no way that alien species can be kept from establishing populations even in the legally protected marine SPAs or nature reserves.

However, one single act, if undertaken, could drastically limit the influx of alien species into the Mediterranean. Since '[t]he magnitude of Erythrean inv[asion] is directly related to the decline in salinity in the Bitter Lakes', Hewitt *et al*. (2006) suggest that '[t]he most efficient method [to reduce the passage of species from the Red Sea into the Mediterranean] may be the insertion of an environmental barrier such as a saline lock to mimic the most successful obstacle to aquatic translocation in canal systems known to us …'. In an era of heightened environmental concern, the shipping industry has already recognised that its activities may have an impact which cannot be mitigated and which lies outside the control and responsibility of shipping companies. Biodiversity offsets may offer a potential mechanism for financing a salinity lock in the Suez Canal that may offer benefits to biodiversity while allowing the continued operation of trans-canal shipping. The ongoing widening of the Suez Canal lends urgency to the need for such invasion barriers.

9 Biodiversity offsets: sustainable conservation actions intended to compensate for currently unavoidable harm to biodiversity (ten Kate *et al*., 2004).

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Appendix to Chapter 4—List of the vessel-transported alien species in the Mediterranean. F=fouling, B=ballast, Eg=Egypt, IL=Israel, Lb,Sy=Lebanon, Syria, Cy=Cyprus, Tr=Turkey, Gr=Greece, EAdr=Eastern Adriatic, It=Italy, Fr=France, Sp=Spain, Al,Mr=Algeria, Morocco, LTM=Libya, Tunisia, Malta.

Species	**Phylum**	**Native Range**	**Pathway**	**Vector**	**Eg**	**IL**	**Lb,Sy**	**Cy**	**Tr**	**Gr**	**EAdr**	**It**	**Fr**	**Sp**	**Al,Mr**	**LTM***
Acartia tonsa	Arthropoda	Pantropical	Vessels	F?B?					+			+	+			
Acetabularia calyculus	Chlorophyta	Pantropical	Canals;Vessels	F?S?	+	+								+	+	
Acrochaetium codicola	Rhodophyta	Subcosmopolitan	Vessels	F?					+			+	+	+	+	+
Acrothamnion preissii	Rhodophyta	Indo Pacific	Vessels	F								+	+	+		
Actumnus globulus	Arthropoda	Indian Ocean	Vessels	F								+				
Aeolidiella indica	Mollusca	Pantropical	Vessels	F								+				+
Aglaothamnion feldmanniae	Rhodophyta	Atlantic NE	Vessels	F								+	+			
Ammothea hilgendorfi	Arthropoda	Indo Pacific	Vessels	F								+				
Anadara demiri	Mollusca	Indian Ocean	Aquaculture;Vessels	F/B					+	+		+				
Anadara inaequivalvis	Mollusca	Indo Pacific	Aquaculture;Vessels	F/B	+							+				
Anadara inflata	Mollusca	Pacific NW	Aquaculture;Vessels	F/B					+							
Anoplodactylus californicus	Arthropoda	Subcosmopolitan	Vessels	F	+	+			+			+				
Anoplodactylus digitatus	Arthropoda	Indo Pacific	Canals;Vessels	F		+										
Antithamnion amphigeneum	Rhodophyta	Pacific SW	Vessels	F								+	+	+	+	
Antithamnionella spirographidis	Rhodophyta	Indo Pacific	Aquaculture;Vessels	F?								+	+			
Antithamnionella ternifolia	Rhodophyta	Subcosmopolitan	Vessels	F									+			
Asparagopsis armata	Rhodophyta	Pacific SW	Vessels	F								+	+	+	+	+
Asparagopsis taxiformis	Rhodophyta	Pantropical	Canals;Vessels	F?	+				+	+	+	+				+
Audouinella robusta	Rhodophyta	Subcosmopolitan	Vessels	F	+											
Audouinella spathoglossi	Rhodophyta	Indo Pacific	Vessels	F?	+											
Balanus improvisus	Arthropoda	Atlantic NW	Vessels	F								+				
Balanus reticulatus	Arthropoda	Circumtropical	Canals;Vessels	F		+										
Balanus trigonus	Arthropoda	Pantropical	Vessels	F							+					
Barbatia trapezina	Mollusca	Indo Pacific	Vessels	F		+										

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Bemlos leptocheirus	Arthropoda	Indian Ocean	Vessels	F	+											
Beroe cucumis	Ctenophora	Atlantic NW	Vessels							+		+				
Beroe ovata	Ctenophora	Atlantin N	Vessels							+		+				
Bonnemaisonia intricata	Rhodophyta	Indo Pacific	Aquaculture;Vessels	F?								+				
Botryocladia madagascariensis	Rhodophyta	Indian Ocean	Vessels	F?S?					+			+				+
Brachidontes pharaonis	Mollusca	Indian Ocean	Canals;Vessels	F	+	+	+	+	+	+	+	+	+			
Branchiomma boholense	Annelida	Indo Pacific	Canals;Vessels	F	+	+	+									+
Branchiomma luctuosum	Annelida	Indo Pacific	Vessels	F				+	+	+		+		+		
Bregmaceros atlanticus	Chordata	Circumtropical	Canals;Vessels	B?		+			+			+				
Bugula serrata	Entoprocta	Indo Pacific	Canals;Vessels	F								+	+			
Calappa pelii	Arthropoda	Atlantic E	Vessels	F								+				
Callinectes danae	Arthropoda	Atlantic W	Vessels	B								+				
Callinectes sapidus	Arthropoda	Atlantic W	Vessels	B	+	+	+	+	+	+	+	+	+	+		+
Caprella scaura	Arthropoda	Indo Pacific	Vessels	F								+		+		
Cardites akabana	Mollusca	Indo Pacific	Vessels	F/B					+							
Caulerpa racemosa var cylindracea	Chlorophyta	Indian Ocean	Vessels	F?				+	+	+	+	+	+	+		+
Caulerpa racemosa var turbinata uvifera	Chlorophyta	Red Sea	Canals;Vessels	F?	+		+	+								+
Caulerpa taxifolia	Chlorophyta	Pantropical	Aquaculture;Vessels	F?					+		+	+	+	+		+
Celleporaria aperta	Entoprocta	Pantropical	Vessels	F	+	+										+
Celleporaria brunnea	Entoprocta	Pacific NE	Vessels	F					+							
Celleporaria pilaefera	Entoprocta	Indo Pacific	Vessels	F												+
Celleporella carolinensis	Entoprocta	Atlantic W	Vessels	F								+				
Ceramium strobiliforme	Rhodophyta	Atlantic E Tropical	Vessels	F								+				
Cerithium litteratum	Mollusca	Atlantic W	Vessels	F/B						+						
Cerithium scabridum	Mollusca	Indian Ocean	Canals;Vessels	F/B	+	+	+	+	+			+				+

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Chama asperella	Mollusca	Indo Pacific	Vessels	F		+										
Chama brassica	Mollusca	Indo Pacific	Vessels	F		+										
Charybdis feriata	Arthropoda	Indo Pacific	Vessels	Other										+		
Chlamys lischkei	Mollusca	Atlantic SW	Vessels	F/B?								+		+		
Chondria collinsiana	Rhodophyta	Pantropical	Canals;Vessels	F?					+	+						
Chondria curvilineata	Rhodophyta	Pantropical	Vessels	F	+								+			
Chondria polyrhiza	Rhodophyta	Indian Ocean	Vessels	F						+		+				
Chondria pygmaea	Rhodophyta	Indo Pacific	Canals;Vessels	F?							+	+				+
Chromodoris annulata	Mollusca	Indian Ocean	Vessels							+						
Clytia hummelincki	Cnidaria	Pantropical	Vessels	F								+				
Clytia linearis	Cnidaria	Circumtropical	Vessels	F								+	+			
Codium fragile ssp. tomentosoides	Chlorophyta	Pacific NW	Aquaculture;Vessels	F					+	+		+	+	+	+	+
Codium taylorii	Chlorophyta	Pantropical	Vessels	F?	+	+	+									
Crepidula aculeata	Mollusca	Subcosmopolitan	Vessels	F										+		
Ctenocalanus citer	Arthropoda	Subcosmopolitan	Vessels	B?					+							
Cuthona perca	Mollusca	Indo Pacific	Vessels	F								+				
Dendrostrea frons	Mollusca	Indo Pacific	Canals;Vessels	F					+							
Diadema setosum	Echinodermata	Indo Pacific	Canals;Vessels	F					+							
Diadumene cincta	Cnidaria	Atlantic NE	Canals;Vessels	F?								+				
Dictyota ciliolata	Phaeophyta	Cosmopolitan	Vessels	F?										+		
Diodora funiculata	Mollusca	Indo Pacific	Vessels	F/B		+										
Distaplia bermudensis	Chordata	Atlantic W	Vessels	F								+				
Dromia spinirostris	Arthropoda	Atlantic E	Vessels	F								+				
Dyspanopeus sayi	Arthropoda	Atlantic NW	Aquaculture;Vessels	F/B								+				
Elasmopus pectenicrus	Arthropoda	Circumtropical	Canals;Vessels	F	+	+			+			+		+		
Electra tenella	Ectoprocta	Atlantic W	Vessels	F								+				
Electroma vexillum	Mollusca	Indo Pacific	Vessels	F/B					+							
Elysia grandifolia	Mollusca	Indian Ocean	Canals;Vessels	F			+		+							
Elysia tomentosa	Mollusca	Indo Pacific	Canals;Vessels	F					+							
Eucheilota paradoxica	Cnidaria	Pantropical	Vessels	F							+		+	+		

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Eucidaris tribuloides	Echinodermata	Atlantic E	Vessels	F												+
Eusyllis kupfferi	Annelida	Atlantic	Vessels	F				+								
Favorinus ghanensis	Mollusca	Atlantic E Tropical	Vessels	F												+
Ficopomatus enigmaticus	Annelida	Southern Ocean	Vessels	F	+	+			+	+	+	+	+	+		+
Fulvia fragilis	Mollusca	Indian Ocean	Canals;Vessels	F/B	+	+		+	+	+		+		+		+
Galaxaura rugosa	Rhodophyta	Pantropical	Vessels	F?		+	+									
Garveia franciscana	Cnidaria	Indo Pacific	Vessels	F								+				
Goniotrichiopsis sublittoralis	Rhodophyta	Pacific N	Vessels	F?									+	+		
Grapsus granulosus	Arthropoda	Red Sea	Canals;Vessels	F												+
Halgerda willeyi	Mollusca	Indo Pacific	Canals;Vessels	F?B?							+					
Haliplanella lineata	Cnidaria	Pacific NW	Vessels	F									+			
Halophila stipulacea	Magnoliophyta	Indian Ocean	Canals;Vessels	F?	+		+	+	+	+	+	+				+
Halothrix lumbricalis	Phaeophyta	Circumboreal	Vessels	F?B?					+			+	+			
Hemigrapsus sanguineus	Arthropoda	Pacific W	Vessels	B							+					+
Herbstia nitida	Arthropoda	Atlantic E	Vessels	F?								+				
Herdmania momus	Chordata	Indo Pacific	Vessels	F	+	+	+	+	+							
Hydroides brachyacantha	Annelida	Red Sea	Canals;Vessels	F		+			+							
Hydroides dianthus	Annelida	Atlantic NW	Vessels	F	+		+		+	+	+	+	+	+		+
Hydroides dirampus	Annelida	Circumtropical	Vessels	F	+		+		+			+		+		+
Hydroides elegans	Annelida	Circumtropical	Vessels	F	+	+	+		+	+		+	+	+		+
Hydroides heterocerus	Annelida	Indian Ocean	Canals;Vessels	F		+	+	+	+							
Hydroides homoceros	Annelida	Indian Ocean	Canals;Vessels	F		+			+							
Hydroides minax	Annelida	Indo Pacific	Canals;Vessels	F		+	+		+							
Hydroides operculatus	Annelida	Indian Ocean	Canals;Vessels	F		+	+		+							
Hyotissa hyotis	Mollusca	Indo Pacific	Vessels	F/B		+										

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Hypnea cornuta	Rhodophyta	Pantropical	Canals;Vessels	F?	+	+						+				
Hypnea spicifera	Rhodophyta	Indian Ocean	Canals;Vessels	F?	+											
Hypnea spinella	Rhodophyta	Pantropical	Vessels	F					+	+			+	+		
Iolaea neofelixoides	Mollusca	Pacific	Vessels	F/B					+							
Isognomon ephippium	Mollusca	Indian Ocean	Vessels	F		+										
Laurencia caduciramulosa	Rhodophyta	Pacific W	Vessels	F?								+	+			
Leiosolenus hanleyanus	Mollusca	Indo Pacific	Vessels	F/B		+										
Leucotina cf. eva	Mollusca	Indo Pacific	Canals;Vessels	F?B?					+							
Libinia dubia	Arthropoda	Atlantic W	Vessels	B												+
Lomentaria flaccida	Rhodophyta	Pacific NW	Aquaculture;Vessels	F?									+			
Lomentaria hakodatensis	Rhodophyta	Pacific	Aquaculture;Vessels	F?								+	+			
Lumbrineris inflata	Annelida	Indo Pacific	Vessels	B?								+				
Melibe viridis	Mollusca	Indo Pacific	Vessels	B?					+	+	+	+				+
Menaethius monoceros	Arthropoda	Indo Pacific	Vessels	F?								+				
Metacalanus acutioperculum	Arthropoda	Pacific NW	Vessels	B?								+				
Metasychis gotoi	Annelida	Indo Pacific	Canals;Vessels	F?B?	+	+			+	+	+					
Microcosmus exasperatus	Chordata	Red Sea	Canals;Vessels	F			+									+
Microcosmus squamiger	Chordata	Indian Ocean SE	Vessels	F								+	+	+	+	+
Mnemiopsis leidyi	Ctenophora	Atlantic W	Vessels	B?					+	+						
Mytilopsis sallei	Mollusca	Atlantic	Vessels		+											
Nanostrea exigua	Mollusca	Indo Pacific	Canals;Vessels	F		+										
Neanthes willeyi	Annelida	Southern Ocean	Aquaculture;Vessels	F?B?	+											
Neomeris annulata	Chlorophyta	Pantropical	Canals;Vessels	F?			+									
Neosiphonia sphaerocarpa	Rhodophyta	Pantropical	Vessels	F						+			+	+		+
Oculina patagonica	Cnidaria	Atlantic SW	Vessels	F	+	+	+		+	+		+		+	+	+

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Omobranchus punctatus	Chordata	Indo Pacific	Canals;Vessels	F/B		+										
Ophryotrocha japonica	Annelida	Pacific	Vessels	F?B?								+				
Oxynoe viridis	Mollusca	Indo Pacific	Canals;Vessels	F?B?					+							
Pampus argenteus	Chordata	Indo Pacific	Vessels	B							+					
Paracerceis sculpta	Arthropoda	Pantropical	Vessels	F?	+							+		+		+
Paradella dianae	Arthropoda	Pacific NE	Vessels	F?	+				+			+				+
Parahyotissa imbricata	Mollusca	Indian Ocean	Vessels	F		+										
Percnon gibbesi	Arthropoda	Pacific E;Atlantic	Vessels	F					+	+		+		+		+
Perinereis nuntia	Annelida	Indian Ocean	Canals;Vessels	F?B?	+											
Phallusia nigra	Chordata	Atlantic W Tropical	Vessels	F		+	+		+							
Pherusella brevituba	Ectoprocta	Pacific	Vessels	F								+				
Pigrogromitus timsanus	Arthropoda	Circumtropical	Canals;Vessels	F		+										
Pileolaria berkeleyana	Annelida	Pacific E	Vessels	F									+			
Pinguipes brasilianus	Chordata	Atlantic SW	Vessels	B?								+				
Plagusia squamosa	Arthropoda	Indo Pacific	Canals;Vessels	F			+									+
Planostrea pestigris	Mollusca	Indian Ocean	Vessels	F		+										
Pleonosporium caribaeum	Rhodophyta	Pantropical	Vessels	F									+	+		
Plicatula chinensis	Mollusca	Indian Ocean	Vessels	F		+										
Plocamium secundatum	Rhodophyta	Antarctic;Subantarctic	Vessels	F?								+				
Polyandrocarpa zorritensis	Chordata	Pacific E	Vessels	F								+				
Polycera hedgpethi	Mollusca	Pacific NE	Vessels	F								+				
Polycerella emertoni	Mollusca	Pantropical	Vessels	F						+		+				+
Polydora cornuta	Annelida	Pantropical	Vessels	B?					+					+		
Polysiphonia harveyi	Rhodophyta	Pacific NW	Aquaculture;Vessels	F?									+			
Polysiphonia paniculata	Rhodophyta	Pacific NE	Vessels	F?									+			

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Pomatoleios kraussii	Annelida	Indo Pacific	Canals;Vessels	F		+	+		+							
Portunus pelagicus	Arthropoda	Indo Pacific	Canals;Vessels	B	+	+	+	+	+	+		+				
Pseudocyclops xiphophorus	Arthropoda	Indian Ocean	Vessels	F?								+				
Pseudopolydora paucibranchiata	Annelida	Pacific	Vessels													
Rapana venosa	Mollusca	Pacific NW	Aquaculture;Vessels	B					+	+		+				
Redicrice sulcata	Mollusca	Indo Pacific	Canals;Vessels	F?B?		+										
Reteroporella jermanensis	Ectoprocta	Red Sea	Canals;Vessels	F		+										
Rhithropanopeus harrisii	Arthropoda	Atlantic NW	Aquaculture;Vessels	B								+	+			+
Rhodymenia erythraea	Rhodophyta	Red Sea	Canals;Vessels	F?S?	+											
Saccostrea cucullata	Mollusca	Indo Pacific	Vessels	F	+				+							
Scyllarus caparti	Arthropoda	Atlantic E	Aquaculture;Vessels	F?B?								+				
Semipallium coruscans	Mollusca	Indo Pacific	Vessels	F?							+					
Septifer bilocularis	Mollusca	Red Sea	Vessels	F?B?		+										
Smittina malleolus	Ectoprocta	Pantropical	Vessels	F		+										
Sphaeroma walkeri	Arthropoda	Indian Ocean	Vessels	F?	+				+				+	+		+
Sphaerosyllis longipapillata	Annelida	Pacific	Canals;Vessels	F				+								
Sphenia ruppellii	Mollusca	Indian Ocean	Canals;Vessels	F		+										
Spirobranchus tetraceros	Annelida	Circumtropical	Canals;Vessels	F	+		+	+	+	+						
Spirorbis marioni	Annelida	Pacific E	Vessels	F					+			+	+	+	+	
Spondylus cf. multisetosus	Mollusca	Indo Pacific	Vessels	F					+							
Spondylus groschi	Mollusca	Indian Ocean	Canals;Vessels	F		+										
Spondylus nicobaricus	Mollusca	Indo Pacific	Vessels	F		+										
Spondylus spinosus	Mollusca	Indo Pacific	Canals;Vessels	F		+	+		+							
Stenothoe gallensis	Arthropoda	Pantropical	Vessels	F					+			+	+			
Stomatella impertusa	Mollusca	Indo Pacific	Canals;Vessels	F?B?					+							

Species	Phylum	Native Range	Pathway	Vector	Eg	IL	Lb,Sy	Cy	Tr	Gr	EAdr	It	Fr	Sp	Al,Mr	LTM*
Streblospio gynobranchiata	Annelida	Atlantic W Tropical	Vessels	F					+							
Stypopodium schimperi	Phaeophyta	Pantropical	Canals;Vessels	F?S?	+	+	+	+	+	+						+
Symphyocladia marchantioides	Rhodophyta	Indo Pacific	Vessels									+				
Synagrops japonicus	Chordata	Indo Pacific	Vessels	B								+				
Terebella ehrenbergi	Annelida	Indo Pacific	Canals;Vessels	F		+		+								
Thais lacera	Mollusca	Indian Ocean	Canals;Vessels	F?B?	+	+		+	+			+				+
Thalamita gloriensis	Arthropoda	Indo Pacific	Vessels	B								+				
Tharyx dorsobranchialis	Annelida	Unknown	Canals;Vessels	F						+						
Theora lubrica	Mollusca	Indo Pacific	Vessels	F?B?		+						+				
Tremoctopus gracilis	Mollusca	Indo Pacific	Vessels	F?								+				
Ulva ohnoi	Chlorophyta	Pacific NW	Vessels	F?	+											
Womersleyella setacea	Rhodophyta	Indo Pacific	Vessels	F						+		+	+	+		+

Chapter 5

Maritime traffic effects on biodiversity and potential mitigation measures: a synthesis

Ameer Abdulla and Olof Linden

1. Introduction

This review has shown that the Mediterranean Sea is particularly sensitive to a multitude of deleterious effects that are associated with maritime transport. The high volume and frequency of maritime traffic, the sea's semi-enclosed nature and its long history of human use suggest that the Mediterranean is one of the most vulnerable marine ecosystems. It is clear that maritime effects in the Mediterranean need to be better documented and their impacts better managed. Here we synthesize the review's main findings in order to allow readers and decision-makers faster access to the information and the conclusions.

2. Maritime impacts in the Mediterranean

2.1. The effects of noise, collisions, physical impacts and antifouling TBT paint

Although we know that anthropogenic sound in the ocean is a serious threat, we do not currently have sufficient information to understand the full extent of the problem on Mediterranean food webs. Most of the current research has focused on marine mammals, mainly cetaceans and pinnipeds, and a few other vertebrates such as sea turtles (*see* Chapter 1) but has largely ignored other species. The Mediterranean Sea in general and the Ligurian Sea in particular may be disproportionately and severely affected by many different sources of man-made noise. The most important of these are maritime traffic (commercial and recreational), seismic surveys, military sonar, drilling operations, coastal construction works and underwater explosions originating from military exercises. The high volume of shipping in the Mediterranean Sea results in high background noise levels that do not allow for silent areas or refugia and are likely to make it difficult for cetaceans to communicate with each other or to receive acoustic cues, for example to detect approaching vessels or other hazards (Chapter 1). There appears to be a link between noise and collisions, where collisions may be related to high-density maritime traffic, increased masking ambient noise, possible hearing impairment in cetaceans, and cetaceans' inability to avoid the collision area because of the high density of shipping noise all around.

A review by Panigada and colleagues of fin whale strandings and deaths since 1972 shows that 16.0% of these cetaceans were killed by ship strikes. Chapter 1 provides evidence that the majority of the accidents (76.7%) occur within the feeding season as whales and dolphins often congregate during the summer months to feed precisely in areas where vessel traffic is highest. One such aggregation area for fin whales, striped dolphins and other cetacean species is the Pelagos Sanctuary in the Ligurian Sea, where particular oceanographic features support high levels of prey and consequently a large number of cetaceans. The majority of reported strikes (82.2%) were recorded in the Pelagos Sanctuary and the Gulf of Lion or adjacent waters, suggesting that these are high-risk areas for whale collisions; the remaining strikes were reported in Spanish and southern Italian waters. In those cases where it was possible to ascertain the vessel type involved in a strike, standard ferries were most frequently implicated (62.5%), followed by merchant ships (16.7%), fast ferries (12.5%) and yachts. High-speed ferries were introduced into the area in 1996 and in the six years following accounted for almost 50% of the total known collisions. At least 70% of dead sperm whales found along the Greek coast have clear propeller marks on their body and their deaths are likely to have been caused by collisions with large ships (Chapter 1). Most of them are young and immature individuals, which live in social units usually comprising 8–14 members. Other marine vertebrates, such as sea turtles, which need to come to the sea surface at regular intervals to breathe, are also exposed to the risk of ship strikes. On most occasions, a turtle does not survive an impact or is left seriously injured, with limited movement and diving ability. In such cases, injuries occur most frequently in the anterior area, with severe cuts or amputation of the fore flippers, head and front scutes of the carapace. Vallini proposes that such collisions are usually due to recreational and not commercial vessels (Chapter 1).

Aside from clear impact associated with direct collisions with species, physical impacts of vessels on Mediterranean marine habitats and species also include anchoring, abrasion by ship hulls in shallow waters, propeller scarring, groundings, disturbance of soft sediments during navigation, and shading (Chapter 1). Borg suggests that these impacts have substantial effects on marine organisms, which include dislodgement leading to tissue damage and death, and increased turbidity from sediment resuspension, which decreases photosynthetic ability. However, published studies deal almost exclusively with the effects of recreational boat anchoring on seagrass (*Posidonia oceanica*) meadows and there is a lack of information on the effects of anchoring by commercial vessels in the vicinity of deep-water habitats, and on other physical effects resulting from recreational boating and commercial shipping.

Although currently a banned substance that is being phased out of use, TBT still occurs on the hulls of many recreational and commercial vessels in the Mediterranean. TBT is considered the most toxic substance intentionally introduced into marine environments (Galil, Chapter 1) and affects non-target biota especially in areas with high vessel density and restricted water circulation such as harbours and marinas. Marine invertebrates are notably sensitive to the substance, and suffer well-documented sublethal impacts such as morphological change, growth inhibition, suppressed immunity, reduced reproductive potential and changes in population structure. TBT and its degradation products also bioaccumulate within tissues of marine organisms and move up the food chain (Galil, Chapter 1). Very high concentrations have been found in Mediterranean top predators such as the bottlenose dolphin, bluefin tuna and blue shark.

2.2. The effects of ship-generated oil discharges and exhaust emissions in the Mediterranean basin

Issues relating to oil spills and air emissions generated by ships are reviewed extensively by Galdies in Chapter 2. Maritime traffic in the Mediterranean is dramatically increasing in large part due to the growing number of oil tankers and changes in the distribution of oil producers. Oil from fields around the Caspian Sea is seen as a major reason for the increase in tanker traffic. Although tankers (as well as most other ships) are safer now than they were some 20 or 30 years ago, the increasing frequency of their movements results in greater accident risks . In addition, routine operational spills, most of which are illegal, remain a serious problem. Every year approximately 400,000 tons of oil are deliberately released into the Mediterranean. This figure is not decreasing as there is little enforcement of regulations against oil pollution (Galdies, Chapter 2).

Chapter 2 also reviews the chemical fate and biological impacts of petroleum hydrocarbons with a focus on the Mediterranean. It discusses the levels of such pollutants, including polycyclic aromatic hydrocarbons (PAHs), in sediments and water. Somewhat surprisingly, in some areas PAH levels are higher in offshore samples than in those from near-shore waters, which is due to intensive ship traffic. As expected, the levels of PAHs in bottom sediments are generally higher near ports and industrial areas. Most studies, however, have been carried out in the north-western part of the Mediterranean. In addition, there is an almost total lack of information on levels in deep-sea areas. In order to be able to assess the state of contamination by petroleum hydrocarbons in the Mediterranean, more studies are needed, particularly along the southern coasts and also along major shipping routes.

Potential ecological and physiological effects of petroleum hydrocarbons on marine organisms are discussed but relatively few data are currently available from the Mediterranean region. PAHs are known to affect different species at the genetic, cellular, biochemical and physiological levels. Genetic damage may result in chromosomal aberrations, impacts on embryonic stages and long-term effects such as carcinogenic and mutagenic growth in vertebrates. Some of these effects have been found after an oil spill off the coast of Liguria in Italy. Species collected in this area were affected as long as 10 years after the spill. Some water-soluble PAHs have been shown to disrupt biochemical membranes, causing changes in enzymatic and receptor activities in affected organisms. Experimental studies have also shown that exposure to petroleum hydrocarbons may lead to various impacts on blood chemistry, such as haemolysis and increased haematocrit counts. Both laboratory and field studies have shown that petroleum hydrocarbons may result in oxidative stress, particularly in marine fish, leading to activation of the cytochrome P450 system, which is a detoxification mechanism for biotransforming xenobiotic compounds. Fish exposed to petroleum hydrocarbons under chronic conditions often exhibit various types of lesions, including neoplastic, focal and necrotic lesions. Several field studies have shown such histopathological changes in waters contaminated by oil. For example, after the *Haven* accident outside the port of Genoa, fish sampled in the contaminated area showed such lesions as long as nine years after the accident. Behavioural changes are also observed among aquatic organisms exposed to oil spills (Chapter 2). Behavioural abnormalities result in lowered chances of survival in the natural environment.

On a different scale, community level changes were detected off Livorno after the 1991 *Agip Abruzzo* oil spill, where several different species in the meiobenthic community reacted differently by either decreasing or increasing their population numbers. Foraminifera, turbellarians and nematodes were particularly sensitive while populations of copepods increased. Further research indicated an acute initial response among these organisms with rapid recovery indicating high resilience to single oil spill incidents. Benthic macro-fauna may also be resilient to the effects of petroleum hydrocarbons, particularly weathered oil. Samples from the *Haven* oil spill showed no significant differences between tar contaminated sites and control sites. However, other studies indicate clear impacts on the growth of shallow-water seagrass over extensive areas and for as long as eight years after a spill.

Ship traffic also generates gas emissions that are released into the atmosphere. These emissions have been assessed by the European Environment Bureau, which has shown that ships contribute significantly to atmospheric contamination by NO_x and SO_x as well as CO_2. By 2020, the figures for SO_x input from shipping into the air will have increased by more than 42% and for NO_x by more than 60%. Annually it is estimated that 1,725,000 tons of NO_x and 1,246,000 tons of SO_x are released by ships in passage or arriving at/departing from ports in the Mediterranean and the Black Sea. Little is known about the impacts of this pollution, particularly on the marine environment of the Mediterranean. However, research in other regions shows that SO_x, NO_x, and CO_2 contribute to ocean acidification, which affects a variety of biochemical and physiological processes. Research to assess the extent of emission-induced acidification and its potential impacts on Mediterranean species should be a priority.

2.3. The effects of persistent organic pollutants, sewage and debris

As the Mediterranean is surrounded by heavily industrialized countries and is an enclosed water body with limited exchange with the Atlantic Ocean, it is an ecosystem that is particularly sensitive to the build-up of persistent natural substances (metals, PAHs) and man-made organics (such as pesticides and PCBs). In Chapter 3 Fossi and Lauriano review the impacts of these persistent organic substances on top predators and long-lived organisms in the Mediterranean and particularly focus on the Mediterranean swordfish, bluefin tuna and loggerhead turtle. Clearly, organisms at the top of the food chain, such as marine birds, large predatory fish and cetaceans, bioaccumulate such substances at high levels. The authors suggest that levels are currently serious and that, given the importance of top predators in maintaining high biodiversity in marine food chains, careful monitoring of the situation should be a priority.

Persistent organic pollutants occur at elevated concentrations in the Mediterranean. In addition, organochlorine pesticides, polychlorinated biphenyls (PCBs), and polybrominated diphenyl ethers have been documented at high levels. These substances have been shown to disrupt the endocrine system in a number of organisms. Organochlorine pesticides are produced by the agricultural and household sectors while PCBs used to have a wide range of industrial applications, although most countries have now banned their use. These substances enter the Mediterranean ecosystem though runoff from agricultural land, leaking dumpsites, industrial operations and via the atmosphere. These chemicals have been reported to modify reproductive systems of Mediterranean swordfish, which may constitute a threat to the survival of the species, and there is evidence for potential trans-generational effects in small cetaceans. Furthermore, poorly treated sewage and debris from ships increases eutrophication and contamination from pathogens. Debris in the form of plastics—synthetic organic polymers—released from fishing vessels, recreational craft, ships and land makes up almost 80% of all garbage found on shorelines and on the sea floor. Many marine organisms such as seabirds, turtles and whales suffer from ingesting floating plastics.

2.4. The effects of marine invasive species

The introduction and establishment of non-native species is the result of a wide array of synergistic stressors and vectors, of which maritime traffic is an important one. Although to date there has been no recorded extinction of a Mediterranean native species due to an introduced species, Chapter 4 shows that there have been sudden declines in abundance and local extinctions of native species correlated with the introduction of non-indigenous species. The most significant biodiversity impacts of invasive species in the Mediterranean include: (a) high population growth rates of alien species, affecting extensive areas and resulting in local ecosystem changes; (b) changes and shifts in native communities induced by multiple introductions of alien species; and (c) alien–native interactions that culminate in community and ecosystem changes (Galil *et al*., Chapter 4). The overall impact is a noticeable change in biodiversity and an eventual local and regional homogenization of species diversity. There are currently 191 ship-transported aliens in the Mediterranean of the taxa Mollusca, Arthropoda, Rhodophyta, and Annelida. Galil *et al*. show that these species are not uniformly distributed and heavily invaded areas occur in ports and in anthropogenically-impacted coastal habitats that act as contagion reservoirs. Italy (87), Turkey (65) and Israel (54) have recorded the highest numbers of invasive species; while these numbers may be explained by vector intensity (Suez Canal and shipping), they may also be a reflection of research investment. Most of these vessel-transported alien species originate from tropical waters. Therefore an increase in the average temperature of Mediterranean waters due to climate change is likely to enable warm-water alien species to expand beyond their present distributions, and may have an impact on the reproduction and survival of native Mediterranean biota.

3. Potential measures to manage and mitigate maritime impacts

3.1. Managing the effects of noise, collisions, physical impacts and antifouling TBT paint

According to Pavan (Chapter 1), few scientific papers dealing with noise and little basic information on the main sources of noise are available for the purposes of setting up noise management strategies. The author recommends that given the current state of our knowledge it is essential to take a precautionary approach to noise regulation. Efforts to protect and preserve marine mammals should use geographic noise exclusion zones or MPAs that take noise pollution into account; these would act as more silent refugia to ensure the protection of critical and productive habitats for vulnerable and endangered species in particular. The authors suggest that underwater noise should be explicitly classified by states as a pollutant and managed accordingly. In the absence of specific laws, and given the fact that underwater noise is a transboundary pollutant, the European Union Habitats Directive is a possible framework for developing a regulatory system for Mediterranean waters. The system would comply with ACCOBAMS, the International Whaling Commission Scientific Committee, the European Parliament and the United Nations Convention on the Law of the Sea (UNCLOS), which consider underwater noise marine pollution. Following better practice examples from the USA, the International Maritime Organization (IMO) should adjust routes, merge existing routes and/or create new routeing measures or speed restrictions to minimize exposure of marine mammals sensitive to noise and preserve critical habitats from commercial shipping and other large ocean-going vessel traffic. Other recommendations include raising awareness within the shipping industry, creating alliances across various stakeholder groups, and engaging maritime industries in the development of innovative and practical solutions to minimize vessel noise.

Also in Chapter 1, Borg proposes that there are three main priority areas for the management of physical impacts. These include investing in applied research to understand the magnitude and extent of damage resulting from direct physical effects; implementing habitat restoration programmes in areas that have been adversely affected; and developing permanent moorings in sensitive habitats (and/or MPAs) to prevent anchoring. To better manage species collisions, Panigada and Vallini (Chapter 1) suggest that reducing ship speed when crossing areas of high cetacean and turtle density would both allow the animals more time to avoid the oncoming vessel and give the operator more time to react to the animals' presence. This could be aided by the presence of trained observers onboard to alert the crew of approaching cetaceans. Such measures could be limited to particularly risky vessel types or activities, or areas characterized by particularly high concentrations of cetaceans or turtles. A ban on motorboat racing within the 12-mile limit of Italian territorial waters in the Pelagos Sanctuary is an example of such regulations. These measures should be coupled with a thorough understanding of potential triggers that may instigate avoidance behaviour in cetacean and turtles. In addition, particular areas characterized by high densities of both cetaceans and ships should be identified as critical cetacean habitats, where speed reductions or the shifting of ferry routes to areas of lower cetacean density may be proposed. Finally, the use of recent technology such as the Automatic Identification System (AIS) can help model specific probabilities of collision, derive estimates of true mortality rates, as well as identify causative factors and regional collision hotspots.

In 1989 the Contracting Parties to the Barcelona Convention adopted measures including a ban on organotin-based antifouling paints, which entered into effect in 1991. However, Chapter 1 presents post-1991 data that show continuing high levels of TBT, raising suspicions that the legislation banning the paints has been largely ignored. In 2001 the IMO adopted the International Convention on the Control of Harmful Anti-Fouling Systems on Ships (AFS Convention), which called for a global prohibition on the application of organotin compounds and is due to enter into force on 17 September 2008. With the forthcoming ban on TBT-based paints, booster biocides are increasingly being used as alternative antifouling products. Unfortunately, like TBT, detailed testing of the toxicity, persistence and sorptive behaviour of the alternative biocides has not been completed, nor their impact on the marine environment clarified. Regulations for use based on comprehensive testing should be developed prior to licensing them. Galil (Chapter 1) asserts that it is critical then to monitor the expected decline in the levels of TBT and its derivatives in Mediterranean ports and marinas, as well as to monitor the levels of labile copper and organic booster biocides.

3.2. Managing ship-generated oil discharges and exhaust emissions in the Mediterranean basin

In order to reduce the emissions of greenhouse gases from ships, a number of technical and operational measures have been proposed by organizations such as the IMO, the International Council on Clean Transportation (ICCT), UNEP/MAP and EEA. These measures emphasise the use of cleaner marine fuels, installation of pollution control facilities onboard, design of new engines and vessels, and in-port operational procedures. Considering the current trends of accelerated environmental degradation, countries of the region must step up their efforts nationally as well as in regional and international fora to control emissions from ships in the Mediterranean.

Chapter 2 reviews a number of attempts to control oil spilling in the Mediterranean, beginning with the designation of the Mediterranean as a ‘Special Area’ under MARPOL 73/78. In addition, several countries have attempted, with varying success, to increase surveillance in their waters to control the problem of oil spills. For example, in French waters improved surveillance and enforcement of regulations have decreased the number of spills significantly. There are, however, worrying indications that oil still finds its way into the sea, since it is pumped out in other neighbouring areas. In order to bring down the number of oil spills substantially countries of the region must strengthen their surveillance nationally while simultaneously coordinating their efforts on a regional scale.

3.3. Managing persistent organic pollutants, sewage and debris

Chapter 3 suggests that the diversity of large fauna such as pelagic fishes, cetaceans and sea turtles may be in danger in the Mediterranean Sea. These important and charismatic species are threatened because of the risks posed by polluted waters and POPs to their reproductive systems and by the ingestion of plastic debris to their health and survival. These substances enter the Mediterranean ecosystem though runoff from agricultural land, leaking dumpsites, industrial operations and via the atmosphere. Urgent measures to further control and limit the release of these substances have been called for repeatedly in various fora.

The authors of Chapter 3 suggest that although the Action Plan for the Conservation of Cetaceans in the Mediterranean Sea was adopted by the Contracting Parties to the Barcelona Convention in 1991 it has not been regionally implemented by state members. In 1992 the National Focal Points for Specially Protected Areas further developed the Action Plan to comprehensively address all potential impacts to cetaceans. Furthermore, an Action Plan for the Conservation of Mediterranean Marine Turtles was adopted in 1989. However, as plastic debris and wastewater continue to be released into the Mediterranean these problems are increasingly acute. The pollution comes from multiple sources and therefore multilateral and interdisciplinary actions are needed to manage the problems. These actions will need to include different stakeholders such as the private sector, households, urban authorities, tourism and the recreational sector, the shipping industry, and fisheries. Actions must be taken both nationally and on a regional basis.

At the national level a combination of public awareness and educational, procedural and legal measures are urgently required. Since humans release most marine debris, education regarding the impacts of this debris is a key priority. Increased public awareness of the problems combined with actions for the safe disposal and recycling of garbage should involve national and local authorities, NGOs and the private sector. Countries of southern Europe, North Africa and the Eastern Mediterranean need to improve their efforts to collect garbage and waste, to recycle, and to enhance sewage treatment. Activities to influence the public's behaviour should be coupled with better enforcement of rules and regulations. Although in some cases better legislation is needed, in the majority of countries existing laws and regulations are sufficient but need proper enforcement.

At an international level, the principal legal framework for addressing marine pollution originating from ships is the International Convention for the Prevention of Pollution from Ships (MARPOL). Annex V of MARPOL 'restricts at sea discharge of garbage and bans at sea disposal of plastics and other synthetic materials such as ropes, fishing nets, and plastic garbage bags with limited exceptions'. The Mediterranean and Black Seas, both ACCOBAMS areas, are among the special areas where Annex V lays down much stricter discharge regulations. Here once again the difficulty is enforcement, as few countries in the Mediterranean have effective surveillance of the sea. National coast guard units in signatory parties must be equipped and trained to deal with the illegal discharge of oil and sewage and the dumping of garbage and other waste from ships.

3.4. Managing invasive species in the Mediterranean

Although new anti-fouling technology has reduced the probability of dispersal of invasive species and short port visits reduce the opportunity for settlement and survival of transported biota, there has been a distinct and consistent increase in invasive species since the 1960s (Chapter 4). The Barcelona Convention (1976) and its Protocol concerning Specially Protected Areas (SPA), which was adopted in 1982 and came into force in 1986, prohibits 'the introduction of exotic species' (Article 7e). In 2002 the Mediterranean Action Plan (MAP) of the United Nations Environment Programme (UNEP) drafted an 'Action Plan concerning species introductions and invasive species in the Mediterranean Sea' (UNEP(DEC)MED WG.213/4). Furthermore, Galil and co-authors (Chapter 4) recommend that the Mediterranean countries sign the International Convention for the Control and Management of Ships' Ballast Water and Sediments developed by the International Maritime Organization (IMO, 2004) and implement its recommendations. This convention proposes ballast water exchange at sea as an intermediate management option until a more stringent standard is adopted. This standard may only be reached with ballast water treatment systems that are still under development. To enter into force the convention must be ratified by 30 states, representing 35% of world merchant shipping tonnage. Only three Mediterranean countries, Spain, Syria and Egypt, have signed the Convention thus far. It is important that these regional policy tools are utilized and enforced by the different signatory parties with the aim of preventing further change of biological diversity due to intentional and unintentional introductions into the Mediterranean. One mitigation measure that may dramatically reduce the introduction of alien species into the Mediterranean involves the development of an ecological obstacle such as a saline lock in the Suez Canal (Chapter 4). Such a lock would provide the best barrier against introductions from the Red Sea and would be timely in the face of an observed reduction in the natural salinity of the bitter lakes and the current plans to widen the Canal. Shipping companies could finance the engineering and establishment of such a structure as a form of biodiversity offset.

4. Conclusions and priorities for management

Although there is clearly a need for more information and research, the impacts of maritime traffic in the Mediterranean Sea have been shown to be high in intensity, frequency, duration and geographic distribution. While some impacts require more research to understand the ecological effects on habitats and species before specific mitigation measures can be identified, a number of concrete priority management steps can be outlined and acted upon in the short, medium and long terms.

4.1. Short term

a) Develop permanent mooring stations in and around sensitive marine habitats and MPAs;

b) Restore physically damaged habitats;

c) Integrate maritime noise, physical impacts, TBT effects, and emissions into the design of Marine Protected Areas or Particularly Sensitive Sea Areas (PSSAs);

d) Urge IMO to adjust routes, merge existing routes and/or create new routeing measures and speed restrictions to minimize exposure of marine mammals sensitive to noise; and

e) Reduce ship speed when crossing areas of high cetacean and turtle density.

4.2. Medium term

a) Classify underwater noise as a pollutant in national legislation and manage accordingly, and use the European Union Habitats Directive as a framework for developing regulations in the Mediterranean;

b) Monitor the levels of TBT and alternative copper and organic biocides;

c) Strengthen coastguard surveillance nationally to reduce oil spills, and simultaneously coordinate multilateral efforts in order to enforce MARPOL;

d) Enforce regulations to curb runoff from agricultural land, leakage from dumpsites and industrial operations, and garbage dumping;

e) Urge all Mediterranean countries to sign the International Convention for the Control and Management of Ships' Ballast Water and Sediments (IMO 2004) developed by the International Maritime Organization (IMO), and implement its recommendations.

4.3. Long term

a) Develop an environmental barrier in the Suez Canal, potentially funded by the shipping sector, to reduce or impede Lessepsian migration from the Red Sea;

b) Increase public awareness regarding the issues and the effects of maritime traffic on biodiversity;

c) Create multilateral alliances among stakeholders associated with maritime industries and their impacts;

d) Encourage the use of cleaner marine fuels, installation of pollution control facilities onboard, design of new engines and vessels, and in-port operational procedures to reduce ship emissions.

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