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,

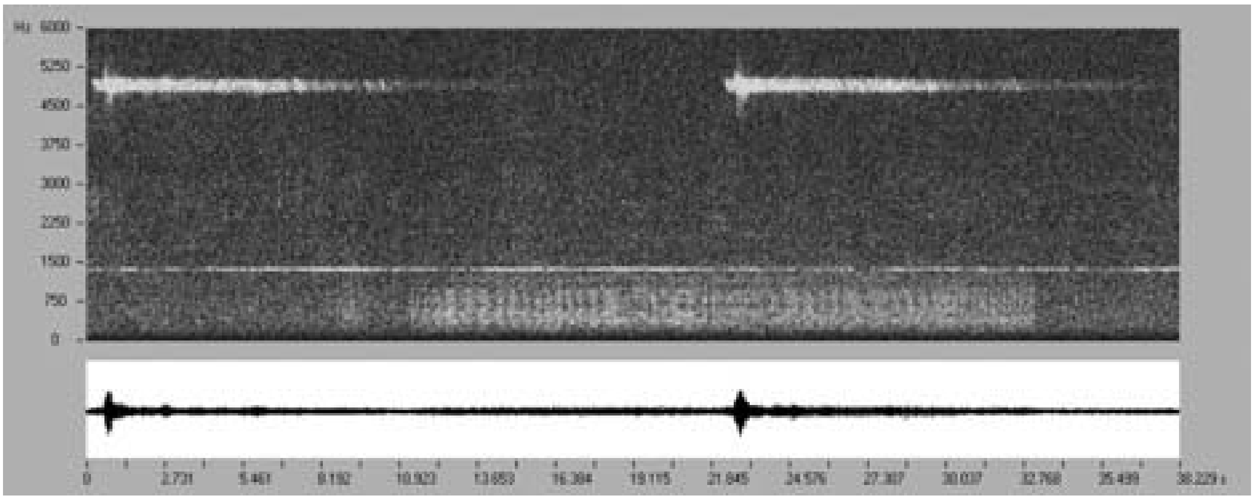


Figure 1.1—Spectrogram showing noises from three different man-made sources recorded in the Ligurian Sea: impulse noises (below 1kHz) from a jack hammer used for construction in Monaco Harbour (about 40km away), a fast ferry passing by (the line at 1.4kHz), and a sonar operating from an unknown location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

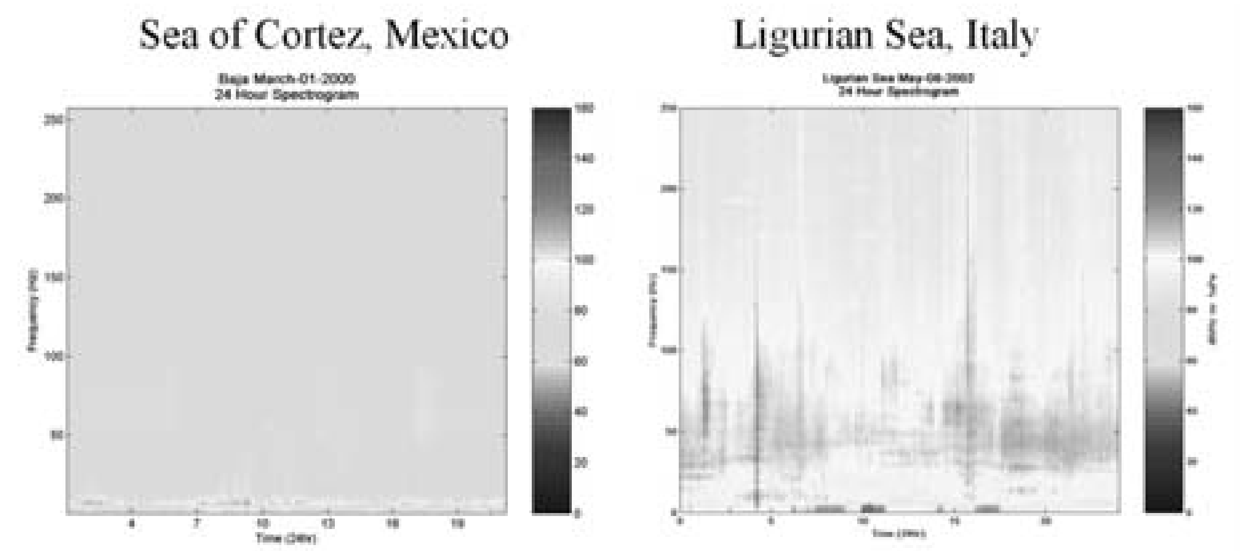


Figure 1.2—24-hour noise map of the Ligurian Sea compared with the Sea of Cortez (Mexico): the noise level in the 0-250Hz band is up to 40dB higher in the Ligurian Sea. Courtesy of C. Clark (Cornell University, USA; For the color version of this figure please refer to the Annex at the end of the document)

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 20ng·l^{-1}. The harbours of Mersin (Turkey) (936ng·l^{-1}), and Livorno (Italy) (810ng·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 736ng·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 252ng·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 138ng·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 (<0.6–5.2ng·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 88ng·l^{-1}, respectively), as well as in the marinas of

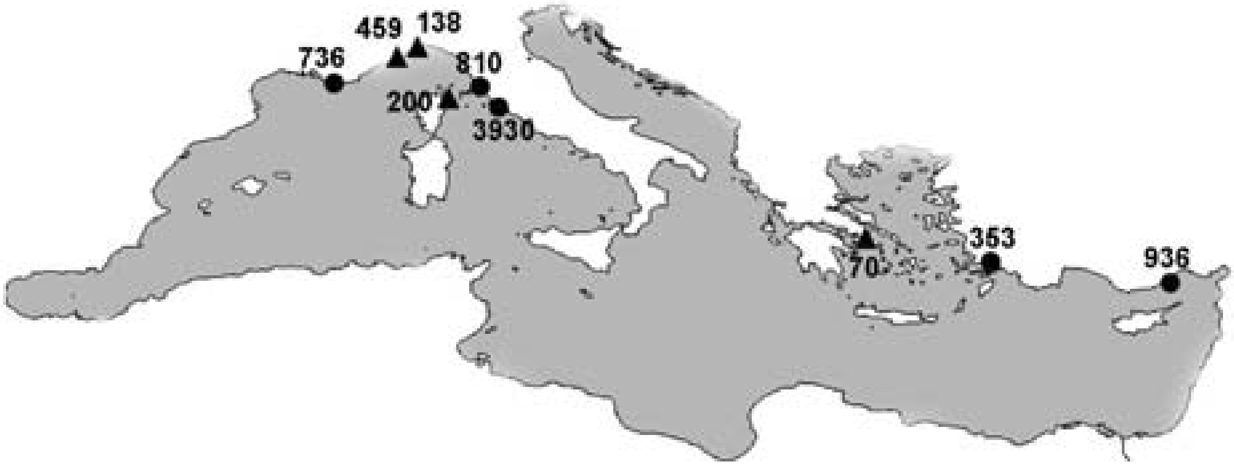


Figure 1.6—Concentrations of TBT in seawater (ng·l⁻¹) before and after the enactment of the ban on organotin-based antifouling paints in the Mediterranean in 1991. Sites sampled in 1988 (●): Mersin port, Turkey; Marmaris port, Turkey; Livorno port, Italy; Cecina marina, Italy; Marseille vieux port (marina), 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·l^{-1} (Smith, 1981). Other morphological changes include shell malformation and reproductive failure in bivalves at concentrations of 20ng·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·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

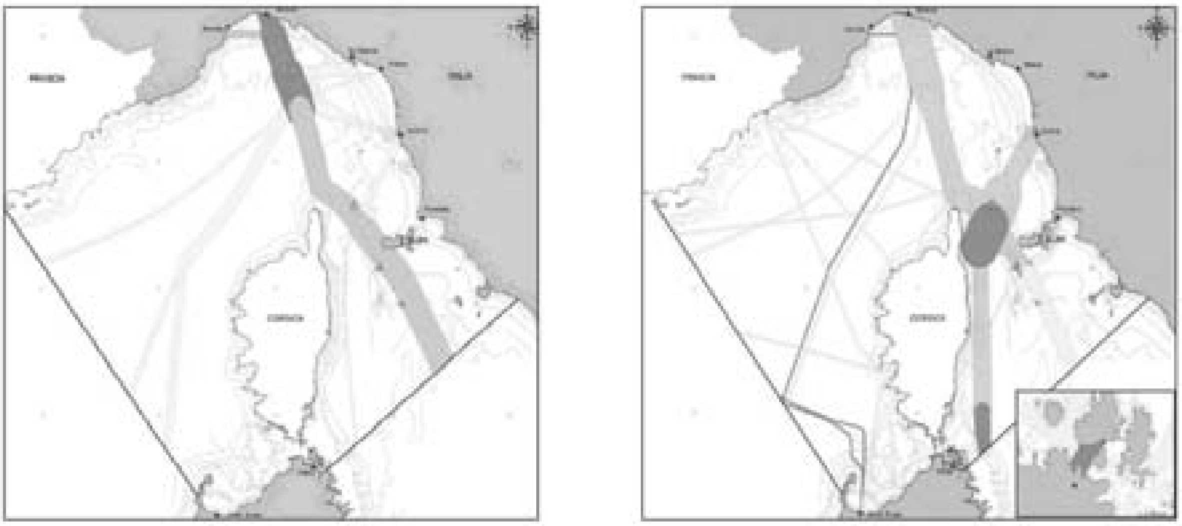
Whales and dolphins often congregate during the summer months to feed precisely in areas where vessel traffic is highest. One such aggregation area for fin whales, striped dolphins and other cetacean species is the Pelagos Sanctuary in the Ligurian Sea, where particular oceanographic features support 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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