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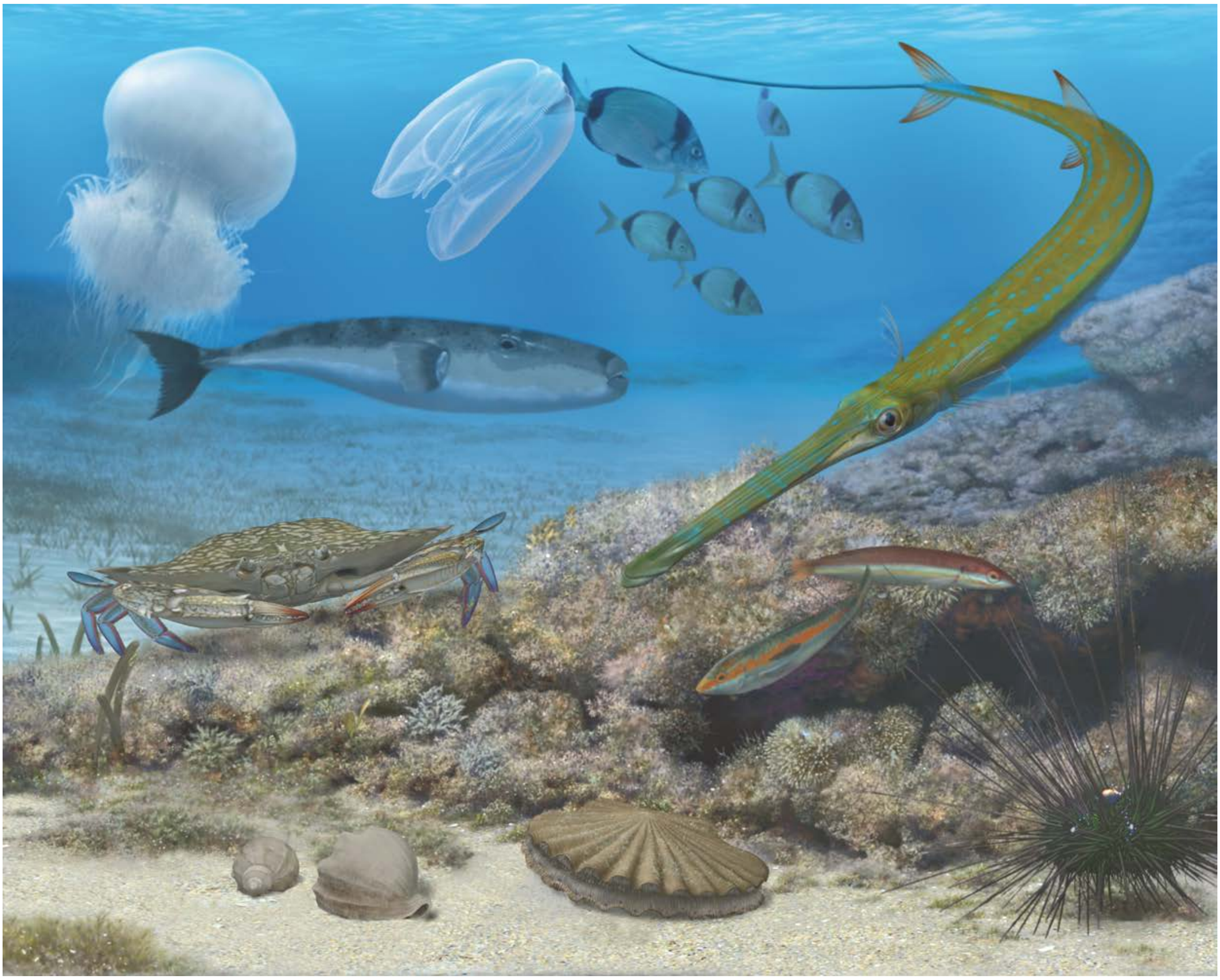
# NON-INDIGENOUS SPECIES IN THE MEDITERRANEAN AND THE BLACK SEA



Cover illustration: Alberto Gennari

GENERAL FISHERIES COMMISSION FOR THE MEDITERRANEAN

# STUDIES AND REVIEWS 87

# NON-INDIGENOUS SPECIES IN THE MEDITERRANEAN AND THE BLACK SEA

Bayram Öztürk

FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS
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# Preparation of this document

Reliable scientific data relating to changes in the distribution and abundance of non-indigenous species in the Mediterranean and the Black Sea over time are essential for understanding their ecological and economic impacts, including on human health. While the production of this monograph has taken some time in gathering the best available information and reviewing it extensively, it now provides the foundation for future comparative studies. Besides the usefulness of the images and descriptions of the various non-indigenous species, some of which are increasing in economic importance, others of which are detrimental to local ecosystems, tourism or human health, this document presents a unique historical and regional perspective of the impacts of these species.

This document was prepared by Bayram Öztürk, Professor at the Faculty of Aquatic Sciences, Istanbul University, and Head of the Turkish Marine Research Foundation, Istanbul, Turkey.

# Abstract

The biota of the Mediterranean and the Black Sea has started to change significantly over the last few decades, due to the introduction of non-indigenous species as a result of Lessepsian migration, ship ballast water, range expansion of Atlantic species, intentional or unintentional introduction and climate change.

The Black Sea has suffered from the non-indigenous invasive comb jelly *Mnemiopsis leidyi* since the late 1980s, which has led to biodiversity loss and a decrease in fish catch, mainly driven by a sudden collapse of small pelagic species, in particular of the European anchovy. The trend of Mediterranization has also contributed to changes in Black Sea biota, as many species that used to be restricted to the Mediterranean have infiltrated the Black Sea, presumably due to climate change. Rapa whelk (*Rapana venosa),* a sea snail, is the most harvested non-indigenous invasive species by Black Sea countries and has also been commercially exploited in the Marmara Sea. The Marmara Sea, situated between the Black Sea and the Aegean Sea, plays a crucial role in the range shifts of invasive species due to its exchange of water mass and marine biota. Like the Black Sea, the Marmara Sea has also been negatively impacted by the comb jelly *M. leidyi*, showing a recovery of small pelagic fish species only in recent years. Lessepsian fishes and invertebrates are increasingly found in the Marmara Sea, but overall impacts are not yet known.

Numerous Lessepsian species have invaded the eastern Mediterranean Sea entering through the Suez Canal, while others have expanded through the central and western sectors of the basin. A number of fish and invertebrate species originating in the Atlantic have also reached the eastern part of the Mediterranean Sea. In total, over 900 non-indigenous species have been reported in the Mediterranean Sea, and almost 300 non-indigenous species in the Black Sea. Over half have established permanent populations and are geographically expanding.

Many Lessepsian fishes have become abundant and commercially relevant, gaining influence in local markets, mostly in the eastern Mediterranean region. Indeed, the new market for non-indigenous species is growing. Due to the lack of overall statistics, the total catch in the entire basin is neither known nor predictable. While non-indigenous species are targeted through various fishing techniques, many others are simply discarded due to a lack of value and there are even some, such as lionfishes, pufferfishes and several jellyfish species, that present immediate dangers to human health. Additionally, important impacts on biodiversity have already been reported, due to habitat displacement and competition with native species.

The creation of marine protected areas and the protection of native species may represent effective measures for mitigating the negative impacts of non-indigenous species. Disseminating information and raising public awareness, in particular on harmful species, are also important tools at the regional level. For the purposes of monitoring and protecting marine biodiversity, regional cooperation on the enforcement of legal measures is essential in order to minimize and reduce the impacts of non-indigenous species both in the Mediterranean and the Black Sea.

# Contents

# Tables, figures, plates

# Abbreviations and acronyms

| | |
|---|---|
| **CBD** | Convention on Biological Diversity |
| **CPUE** | catch per unit effort |
| **CIESM** | International Commission for the Scientific Exploration of the Mediterranean Sea |
| **EEA** | European Environment Agency |
| **FAO** | Food and Agriculture Organization of the United Nations |
| **GEF** | Global Environment Facility |
| **GESAMP** | Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection |
| **GFCM** | General Fisheries Commission for the Mediterranean |
| **GISD** | Global Invasive Species Database |
| **IMO** | International Maritime Organization |
| **IOC** | Intergovernmental Oceanographic Commission |
| **IUCN** | International Union for Conservation of Nature |
| **MEPC** | Marine Environment Protection Committee |
| **MSFD** | Marine Strategy Framework Directive |
| **RAC/SPA** | Regional Activity Centre/Special Protected Areas |
| **STECF** | Scientific, Technical and Economic Committee for Fisheries |
| **TUIK** | Turkish Statistical Institute |
| **UNDP** | United Nations Development Programme |
| **UNEP/MAP** | United Nations Environment Programme/Mediterranean Action Plan |

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# Executive summary

Biota of the Mediterranean and the Black Sea have begun to change significantly with the arrival of non-indigenous species over the last few decades, in particular from Lessepsian migration, due to both intentional and unintentional introduction. The dispersion of non-indigenous species is a dynamic process occurring consistently more frequently and likely to continue into the future. This phenomenon causes severe ecological, socio-economic and human health problems across the entire basin. Recent studies indicate that almost 300 non-indigenous species have been reported from the Black Sea, while over 900 non-indigenous species have been detected in the Mediterranean Sea.

Shipping and intentional introduction are the main vectors of non-indigenous species introduction into the Black Sea. Due to the enclosed nature and low biodiversity of the Black Sea, several non-indigenous species represent threats to the native biota. A notable example, the comb jelly *Mnemiopsis leidyi,* which was transported to the Black Sea in ship ballast water, has caused significant ecological and economic damage to coastal fisheries due to its feeding primarily on the larvae and eggs of small pelagic fishes, particularly anchovy, horse mackerel and sprat. A sea snail, *Rapana venosa,* is the first non-indigenous invasive species in the Black Sea to gain prominent commercial status, and since the 1980s, it has been exported to Asian countries. Meanwhile, its impact on the native fauna, especially mussel and oyster beds, has been detrimental.

Mediterranization also represents a growing trend, and many species of Mediterranean origin have penetrated into the Black Sea. Recently, Lessepsian fish migrants such as the obtuse barracuda *Sphyraena obtusata*, a coral-dwelling fish, *Heniochus acuminatus*, and *Lagocephalus sceleratus* have been reported, as well as two non-indigenous invasive crustaceans. Conversely, an intentionally introduced fish species, *Liza haematocheilus*, has penetrated into the Aegean Sea (and consequently the Mediterranean) through the Turkish Straits, testifying to the close interaction between these two basins.

The Marmara Sea provides a link between the Mediterranean and the Black Sea and serves as a biological corridor, an acclimatization area and a barrier for non-indigenous species. *M. leidyi* has also had a detrimental effect on anchovy fisheries in the Marmara Sea and only recently have their stocks begun to recover. Additionally, the poisonous Lessepsian fish migrant, *Lagocephalus sceleratus,* and the non-indigenous stomatopod shrimp, *Erugosquilla massavensis,* have been reported recently in the Marmara Sea.

As for the Mediterranean Sea, non-indigenous species can enter from the Atlantic Ocean through the Strait of Gibraltar, from the Red Sea through the Suez Canal or from the Black Sea through the Strait of Çanakkale (Dardanelles), by intentional or unintentional human introduction. Some Lessepsian sprinter fish species have passed through the Strait of Sicily, which is regarded as a biogeographical boundary between the eastern and western Mediterranean Sea. On the other hand, some species of Atlantic origin have penetrated into the Mediterranean Sea farther east, reaching the coast of Sicily from their established areas near the Strait of Gibraltar.

The main vectors of arrival for Mediterranean non-indigenous species are the Suez Canal, shipping and aquaculture. A general trend shows that the number of non-indigenous species has increased in recent years and climate change seems to be encouraging biological invasion.

Non-indigenous species have had a variety of consequences on fisheries, biodiversity, human health and the economy in the Mediterranean Sea. Some of the non-indigenous fish species pertaining to different families have become economically important after the establishment of sustainable populations, such as lizardfish (Synodontidae), goatfish (Mullidae), mackerels (Scombridae) and round herrings (Clupeidae), mostly in the eastern Mediterranean region. Similarly, a number of non-indigenous crustacean species are also commercially valuable, including the kuruma prawn (*Penaeus japonicus*), green tiger prawn (*Penaeus semisulcatus*), mantis shrimps (Squillidae) and swimming crab (*Portunus pelagicus*). Meanwhile, some of the introduced mollusc species, such as the Pacific cupped oyster (*Magallana gigas*) and the Japanese carpet shell (*Ruditapes philippinarum*), already boast a market value. On the other hand, some non-indigenous species in the eastern Mediterranean present threats to human health, particularly pufferfishes (*Lagocephalus* spp.), devil firefish[1] (*Pterois miles*), *Plotosus lineatus* and the jellyfish *Rhopilema nomadica*. The introduction of the non-indigenous green algae, *Caulerpa taxifolia* and *Caulerpa cylindracea*, has also negatively impacted fisheries and ecosystems in the Mediterranean Sea. In summary, while some non-indigenous species can harm fishing gear through mesh clogging, fouling and other kinds of damage, other non-indigenous species have impacted native marine biodiversity, mainly through habitat competition and species displacement.

Regional cooperation is essential to obtain the most accurate risk assessments and to minimize the impacts of non-indigenous species both in the Mediterranean and the Black Sea. In this context, regional and international organizations should establish a common non-indigenous species database (Bonanno and Orlando-Bonaca, 2019), easily accessible to all stakeholders, or harmonize already existing databases on non-indigenous species[2]. In the meantime, data on non-indigenous species require standardization to avoid species confusion and to facilitate updates. Additionally, an early warning system should be developed for unpredictable non-indigenous species impacts or blooms with the help of citizen scientists organized by local authorities. Such an approach will allow local communities, especially fishers and divers, to be increasingly involved in monitoring programmes for non-indigenous species.

Countries should also collect catch statistics of the most commercially relevant non-indigenous species to allow for better fisheries management. Similarly, specific studies are required on the impacts of non-indigenous species on small-scale and industrial fisheries in order to improve fisheries reporting and monitoring systems. For some countries, it is highly recommended to implement capacity-building programmes in terms of species identification and data collection. Finally, key species and key habitats, for example Posidonia meadows, which are endemic to the Mediterranean, should be protected from non-indigenous species invasion.

[1] Also known as lionfish in the region.

[2] A joint initiative between the United Nations Environment Programme/Mediterranean Action Plan and the GFCM to monitor non-indigenous species in the eastern Mediterranean was launched in 2017, with a common priority for non-indigenous species and a combination of observation platforms towards a comprehensive coverage of non-indigenous species in the area.

# Introduction

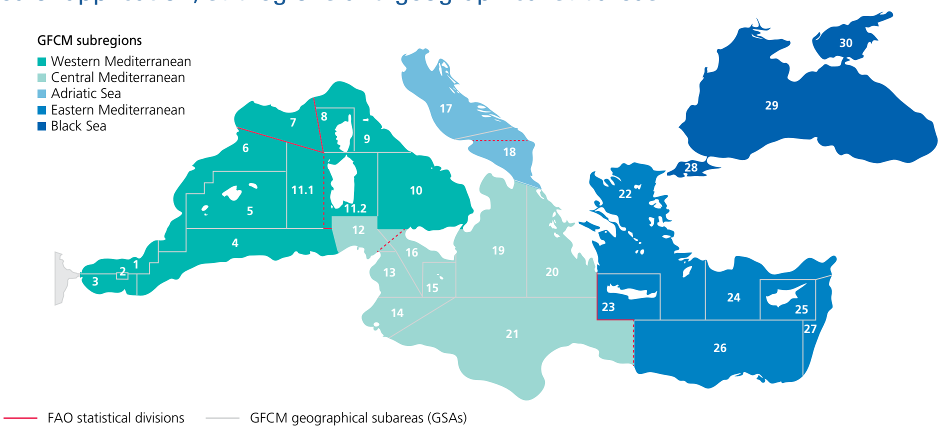
The invasion of non-indigenous or alien species into the Mediterranean and the Black Sea has been recorded for years. This process has speeded up in recent years, however, with many examples of negative impacts on marine ecosystems and the local marine fauna and flora, as well as on socio-economic activities, such as fisheries.

The Mediterranean and the Black Sea are interconnected through the Turkish Straits System (hereafter referred to as the Turkish Straits). These narrow straits act as a biological corridor, a barrier or an acclimatization zone for marine species (Öztürk and Öztürk, 1996). Ship-transported species, such as the rapa whelk *Rapana venosa*, or species introduced to the Black Sea, including the fish *Liza haematocheilus*, have dispersed to the Mediterranean Sea. Since the Suez Canal opened in 1869, the Mediterranean Sea has also been connected to the Red Sea, and consequently the Indian Ocean, which has allowed a massive invasion of tropical marine organisms, mostly to the eastern Mediterranean. Additionally, a number of Indo-Pacific species (Lessepsian migrants) have entered and started to colonize the western Mediterranean, as well as the Aegean, Marmara and Black seas. With strong temperature gradients from Gibraltar to the Strait of Kerch, creating tropical conditions in summer and temperate conditions in winter, the Mediterranean and the Black Sea comprise a highly favourable area for the introduction of species from other regions of the world.

In recent years, due to non-indigenous species, fish composition and catch quantities have shifted. Furthermore, some fish and invertebrate species, have established, mostly in the eastern Mediterranean Sea, causing problems for human health and biodiversity. It is likely that non-indigenous species will produce more serious impacts in the Mediterranean and Black Sea fisheries in the coming years.

In this review, the Mediterranean and the Black Sea are examined separately, although they are interconnected physically and ecologically. The information is presented, when relevant, at the scale of GFCM subregions (Figure 1).

**FIGURE 1**
**GFCM area of application, subregions and geographical subareas**




| | | | | |
|---|---|---|---|---|
| 01. Northern Alboran Sea | 07. Gulf of Lion | 13. Gulf of Hammamet | 19. Western Ionian Sea | 25. Cyprus |
| 02. Alboran island | 08. Corsica | 14. Gulf of Gabès | 20. Eastern Ionian Sea | 26. South Levant |
| 03. Southern Alboran Sea | 09. Ligurian Sea and northern Tyrrhenian Sea | 15. Malta | 21. Southern Ionian Sea | 27. Eastern Levant Sea |
| 04. Algeria | 10. South and central Tyrrhenian Sea | 16. South of Sicily | 22. Aegean Sea | 28. Marmara Sea |
| 05. Balearic islands | 11.1. Sardinia (west)<br>11.2. Sardinia (east) | 17. Northern Adriatic Sea | 23. Crete | 29. Black Sea |
| 06. Northern Spain | 12. Northern Tunisia | 18. Southern Adriatic Sea | 24. North Levant Sea | 30. Azov Sea |

**Source:** FAO, 2020.

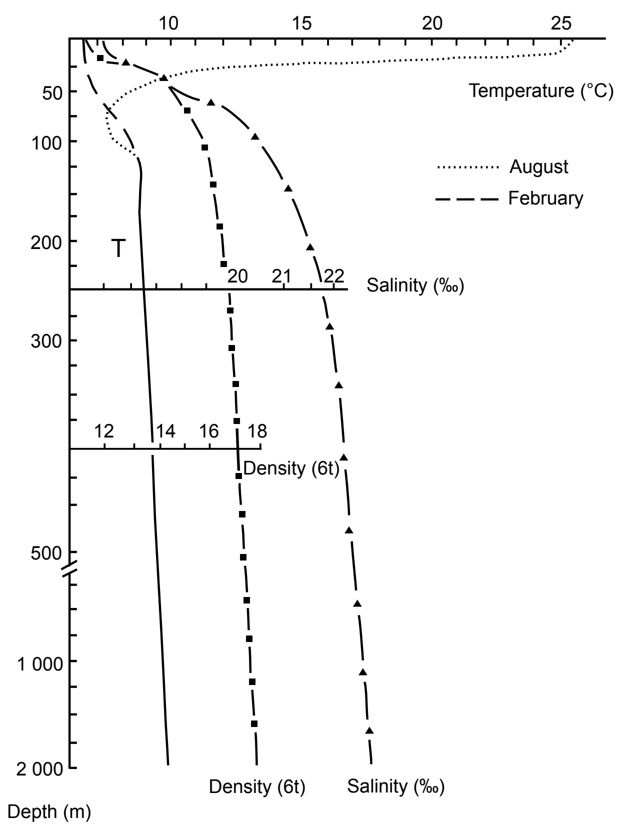
# 1. Non-indigenous species in the Black Sea

## 1.1 Main characteristics of the Black Sea

Of the world's seas, the Black Sea is one of the most isolated from a major ocean, and it is the largest anoxic body of water on the planet (87 percent of its volume is anoxic, i.e. the oxycline is quite shallow at about 100 m depth). The Black Sea boasts a total surface area of 423 000 square kilometres and its catchment area comprises over 2 million square kilometres. It is surrounded by Turkey, Bulgaria, Romania, Ukraine, the Russian Federation and Georgia. In the northeastern corner, the Black Sea is connected to the Azov Sea through the Strait of Kerch, while in the southwestern corner, it is linked to the Marmara Sea through the narrow Strait of Istanbul (Bosphorus). The sea's maximum depth is 2 212 m. However, the most striking characteristic of the Black Sea is probably its high level of hydrogen sulphide ($H_2S$): at depths greater than 150–200 m lies a permanent hydrogen sulphide zone with very little life; these conditions remain relatively stable, although seasonal and annual fluctuations are observed.

The presence of a permanent halocline between 150 and 200 m is another major distinguishing feature (Figure 2) of the Black Sea. Stratification is caused by the combination of freshwater input and the Mediterranean inflow of highly saline water. The average surface salinity represents about 18–18.5 per mille during the winter and increases by 1.0–1.5 per mille in summer. Temperatures show more variation than water salinity, both seasonally and regionally. The mean annual surface temperature varies from 16°C in the south to 13°C in the northeast and 11°C in the northwest. While the upper 50–70 m of water manifest seasonal fluctuations in temperature, particularly with vertical variation, the temperature of deeper waters remains constant throughout the year. Typically, the temperature at a depth of 1 000 m is about 9°C and only shows a slight increase of 0.1°C per 1 000 m towards the deepest sections (Balkas *et al.*, 1990).

**FIGURE 2**
Vertical profile of salinity, temperature and density in the Black Sea




Source: Balkas *et al.*, 1990.

The changes in Black Sea ecosystems that have taken place since the 1960s due to the concurrent impacts of eutrophication, overfishing, climatic fluctuation and non-indigenous species invasions have been studied extensively. A synthesis provided in the report on the *State of the environment of the Black Sea* (BSC, 2008) examines the changes which have occurred in both the pelagic and benthic ecosystems.

The biodiversity of the Black Sea clearly reflects its geological history. After the appearance of the Strait of Istanbul provided a connection to the Mediterranean Sea, about 7 000–10 000 years ago, the salinity of the Black Sea rose gradually, and Mediterranean species soon became established in the Black Sea. Today, 80 percent of the total fauna of the Black Sea is of Mediterranean origin. According to Slastenenko (1959), the Black Sea has received more than one third and about one fifth of its fauna from the Aegean Sea and the greater Mediterranean Sea, respectively. The last group of species to arrive, called non-indigenous species, includes those introduced either intentionally or unintentionally by human activities from the various seas and oceans of the world. Leppäkoski and Mihnea (1996) noted that low salinity, low species diversity and coastal ecosystems markedly impacted by eutrophication, combined with high shipping levels, have encouraged the establishment of non-indigenous species with high ecological plasticity in the Black Sea.

## 1.2 Vectors for non-indigenous species in the Black Sea

In recent years, the Black Sea has become home to a large number of non-indigenous plants and animals. There are three main vectors for non-indigenous species reaching the Black Sea: a) shipping activities, (the most common vector); b) intentional or unintentional introduction by humans; c) Mediterranization, or the process of species from the Mediterranean passing through all ecological barriers in the Turkish Straits and penetrating into the Black Sea.

### 1.2.1 Shipping activities

The introduction of non-indigenous species most commonly occurs via ocean-going ships. Marine organisms usually travel as part of the fouling attached to ships' hulls, as well as in ballast tank sediment and ballast water (Öztürk 2002a, 2002b; Zaitsev and Mamaev, 1997; Zaitsev and Öztürk, 2001; Streftaris, Zenetos and Papathanassiou, 2005; Gomoiu *et al.*, 2002). Shipping activities are intense in the Black Sea, mainly due to the transport of Caspian oil from Novorossiysk, in the Russian Federation, to Mediterranean countries via the Turkish Straits (Figure 3). It is also well known that the largest unintentional pathway for the transport of marine organisms is in the ballast water of commercial vessels; a typical commercial bulk vessel may carry over 30 000 metric tonnes of ballast water to provide the stability and trim adjustment necessary during a voyage (cited in Wonham *et al.*, 2000).

Hundreds of algal and animal species, either microorganisms or even smaller organisms, are known to travel by attaching themselves to ships' hulls. Most of them, like algae, clams and barnacles, usually survive attached to the living substrate. Active non-sessile forms, such as amphipods, shrimps, crabs and fish can also be found. When the ship is in motion, they hide in barnacles and other similar shelters, so as not to be swept away by the current.

Ballast water is pumped into tanks to stabilize a ship when it is not carrying any cargo. When ships fill their ballast tanks in ports, or sometimes in other areas, suspended matter and various planktonic organisms are also pumped into the tanks with the water. Many organisms, including their spores and eggs, sometimes survive the trip in the ballast water or sediment. Upon arrival at the ship's destination, the ballast water is discharged into the sea and the organisms find themselves in a new environment. If the conditions are favourable to their particular needs, these organisms may survive and even become naturalized. The huge number of ocean-going ships means that many new species are constantly being introduced into new environments. For example, according to Skolka and Preda (2010), about 60 percent of the non-indigenous species introduced

**FIGURE 3**
Main shipping routes and straits in the Mediterranean and the Black Sea


Source: UN, 2021 modified from UNEP-MAP-REMPEC, 1996.

into the Romanian part of the Black Sea arrived in ballast water. Indeed, only about 6 percent were intentionally introduced for economic purposes, including into freshwater ecosystems. An analysis of the geographic origin of the marine non-indigenous species in the Romanian waters of the Black Sea shows that most of them (43 percent) are cosmopolite planktonic species, while 12 percent have an Atlantic-Mediterranean origin, 27 percent are North Atlantic species and 18 percent are of Indo-Pacific origin. In fact, Anastasiu *et al.* (2016) reported 102 non-indigenous aquatic species, including 44 freshwater species and 58 marine species.

### 1.2.2 Intentional or unintentional introduction by humans

A number of non-indigenous species have been introduced into the Black Sea for aquaculture or other reasons. The mosquitofish, *Gambusia affinis*, a good example of this type of introduction, is well known for its ability to feed on neuston larvae and mosquito eggs, including those of species transmitting malaria. It was therefore introduced into the wetlands around the entire Black Sea basin in order to combat malaria. However, the mosquitofish adapted rapidly, turned into a euryhaline species, and today is widespread in the Black Sea basin, occupying a broad salinity range from 0 to 17 per mille (Kosarev, ed., 2007).

### 1.2.3 Mediterranization effects

The range expansion of Mediterranean species into the Black Sea appears to be a relatively new phenomenon and primarily related to increasing water temperatures as a consequence of climate change (Oğuz, 2005). Even though the Turkish Straits (Straits of Istanbul and Çanakkale) serve as an ecological barrier with very different oceanographic conditions from the Black Sea and the Mediterranean Sea, some phytoplankton and zooplankton species are able to penetrate into the Black Sea (Georgieva, 1993; Kovalev, 2006; Selifinova, Smeleva and Kideys, 2008).

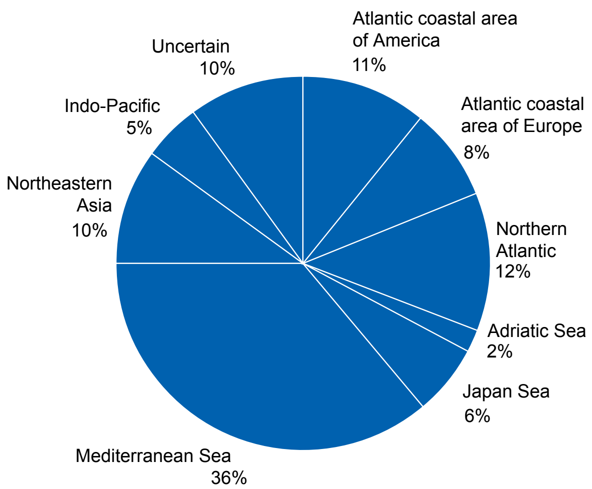
Because of increases in water temperature, some Mediterranean fish species, such as sardine (*Sardina pilchardus*), bogue (*Boops boops*) and wrasse, have also penetrated into the Black Sea in recent years. Reflecting these trends, Boltachev and Karpova (2013, 2014) reported the first occurrence in 2013 of the dogtooth grouper (*Epinephelus caninus)*, along the southwestern Crimean Peninsula, which clearly reflects Mediterranization impacts on the Black Sea, as this subtropical species is distributed widely in the eastern Atlantic Ocean, from Portugal to Angola, and in the Mediterranean Sea, but was never recorded before from the Black Sea.

## 1.3 Occurrence and impacts of non-indigenous species in the Black Sea

The number of non-indigenous species in the Black Sea is continually increasing (for example, Boltachev and Karpova, 2014). Zaitsev and Mamaev (1997) described 26 non-indigenous species from the Black Sea. Subsequently, Zaitsev and Öztürk (2001), recorded that 59 species of non-indigenous marine organisms were living in the Black Sea, and yet only a few species, such as the sea snail *Rapana venosa* and the comb jelly *Mnemiopsis leidyi,* had been well studied in terms of their impact on fisheries. Similarly, Çınar *et al.* (2005, 2011) reported 20 non-indigenous species from the Turkish part of the Black Sea. Recently, Topaloğlu and Öztürk (2017) documented the first reported sighting of the crustacean *Penaeus japonicus*, a Lessepsian migrant of commercial value in the Mediterranean, from the Turkish part of the Black Sea.

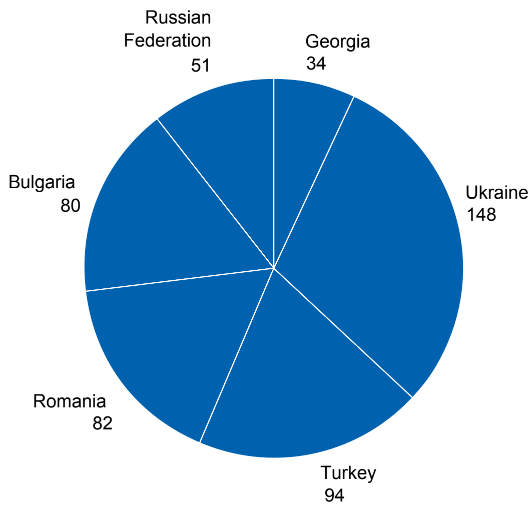
Other surveys indicate the presence of 156 non-indigenous species in the Black Sea, with most of these species coming from the Mediterranean (Shiganova and Öztürk, 2009; Figure 4). However, even more recently, according to Alexandrov, Minicheva and Zaitzev (2017), 261 non-indigenous marine species were registered in the database of the Permanent Secretariat of the Black Sea Commission in 2013, of which 148 were recorded in Ukraine, 94 in Turkey, 82 in Romania, 80 in Bulgaria, 51 in the Russian Federation and 34 in Georgia (Figure 5).

**FIGURE 4**
Areas of origin of non-indigenous species and their representation in the Black Sea




Source: Shiganova and Öztürk, 2009.

**FIGURE 5**
Number of non-indigenous marine species by Black Sea coastal state




Source: Alexandrov, Minicheva and Zaitzev, 2017.

### 1.3.1 Non-indigenous invertebrates in the Black Sea and their impacts on biodiversity

Over the last decades, many non-indigenous invasive species, including the sea snail, *Rapana venosa*, the bivalve species, *Mya arenaria* and *Anadara inaequivalvis,* as well as the carnivorous comb jelly species, *Mnemiopsis leidyi* and *Beroe ovata,* have developed mass populations and engendered major ecosystem transformations with considerable impacts on the pelagic and/or benthic food webs of the Black Sea.

***Mnemiopsis leidyi***

*Mnemiopsis leidyi* (Plate 1) is a carnivorous ctenophore (also known as a comb jelly) characterized by two big lobes referred to as lateral or oral lobes. The oral lobes are derived from the ctenophore body (spherosome). Four smaller lobes are situated under the two principal oral lobes. In the Black Sea, its size varies from 40 to 180 mm in length. The adult is about 100 mm long; specimens larger than this are rare. They boast four relatively short, simple auricles emerging from the sides of the body immediately above the mouth and close to the sides of the oral lobes. The introduction of *M. leidyi* took place through ballast water brought from northern American Atlantic areas at the beginning of the 1980s (Shiganova, 1998; Zaitsev and Öztürk, 2001).

**PLATE 1**
*Mnemiopsis leidyi*

©GFCM/David Salvatori

Mature specimens of *M. leidyi* spawn at night in summer temperatures of 20 to 23°C in the upper layer of the sea. Embryonic development takes about 20–24 hours. Larvae are 0.3–0.5 mm long (Zaika and Sergeeva, 1990). The average egg production of *M. leidyi* in the coastal zones of the Black Sea is very high and exceeds 1 000 eggs per individual per day, with between 2 000 and 4 000 eggs being laid in total. Equations to determine the wet weight (W, mg) on the basis of the total length (L, mm) of the body are:

$$W = 3.1 \cdot L^{2.22} \text{ for } L<45 \text{ mm or } W = 3.8 \cdot L^{2.22} \text{ for } L>45 \text{ mm}$$ (Vinogradov *et al.*, 2000).

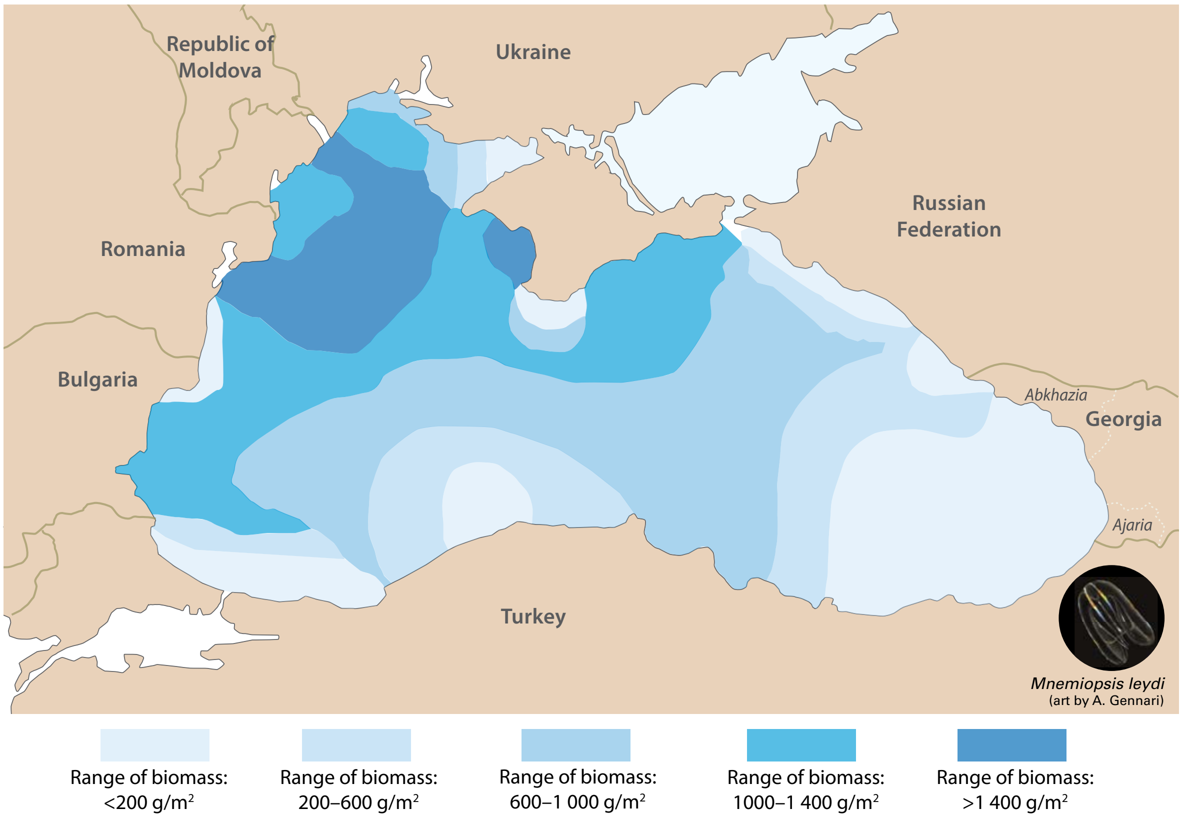
*M. leidyi* is abundant in the ports and harbours of the North American Atlantic coast and can be pumped (presumably as larvae or small juveniles) or gravitated (in the case of adults as well) with ballast water into cargo ships. While sufficient zooplankton may be available to healthily sustain this comb jelly on a voyage lasting 20 days or more to the Black Sea, food resources are not even necessary, since *M. leidyi* can live for three weeks or more without food, by reducing its body size at the same time (Reeve, Syms and Kremer, 1989). Like other ctenophores, *M. leidyi* is a simultaneous hermaphrodite, i.e. capable of self-fertilization, which means that in theory, a single animal could easily invade a new area.

The first record of *M. leidyi* in coastal Black Sea waters is from 1982 (Pereladov, 1988); not long after, in the winter of 1986–1987, it was sighted in open water (Zaitsev, Vorobyova and Alexandrov, 1988). The massive growth of the Black Sea population began in 1988 and at first covered only bays, gulfs and coastal waters. Its abundance reached 10–12 kilograms per square metre in several coastal areas (e.g. Anapa on the southwestern Bulgarian coast), although it did not exceed 1.5–3 kilograms per square metre in the open sea (Shushkina and Vinogradov, 1991). The highest occurrence of *M. leidyi* was registered in 1989 and 1990 (about 1 200 gram per cubic metre), but then its abundance started to decrease (Vinogradov *et al.*, 2000). For example, between 1991 and 1994, the average biomass of *M. leidyi* reached 2.2–3.5 gram per cubic metre along the Romanian littoral zone and decreased to 0.2 gram per cubic metre in 1995 (Radu, Nicolaev and Radu, 1996–1997). The same quantitative distribution was observed in the area of the Black Sea into which the Dnieper River drains, with the average biomass of *M. leidyi* at 3.2–5.1 gram per cubic metre between 1993 and 1997. On a larger geographical scale, its population density stabilized during these years between 300 and 800 gram per square metre in the Black Sea and between 500 and 600 gram per square metre in the Azov Sea (GESAMP, 1997).

Meanwhile, another comb jelly species, *Beroe ovata*, which is a predator of *M. leidyi*, was introduced into the Black Sea, presumably through ballast water, and observed for the first time in 1997 (Shiganova *et al.*, 2000). As explained later under *B. ovata*, this introduction helped mitigate the *M. leidyi* population explosion.

Distribution maps of *M. leidyi* in the Black Sea have been established according to different years and seasons. To produce a general map, the distribution of this comb jelly was modelled, based on characteristics of its biology (reproduction, growth, mortality) and water mass transportation in the Black Sea (Lebedeva, 1998). The map showing the general distribution of *M. leidyi* was generated for the water layer of 0–30 m (the layer this species typically inhabits) in September 1998 (Figure 6).

**FIGURE 6**
Distribution of *Mnemiopsis leidyi* in the Black Sea during its maximum development of September 1998




Source: UN, 2021 modified from Zaitsev and Öztürk, 2001.

*Mnemiopsis leidyi* is usually found close to shore, in bays and estuaries, although specimens have occasionally been collected several hundred kilometres offshore. This species is able to tolerate a wide range of salinities and temperatures, i.e. it can live and reproduce in salinities between 3.4 and 75 percent and in temperatures between $1.3^{\circ}$C and $32^{\circ}$C, while surviving well in oxygen-poor environments. *Mnemiopsis leidyi* is most abundant in brackish water with high levels of suspended material and appears to be hardly affected by contaminants. The only factors which seem to restrict its rapid population growth are temperature, the availability of food and the presence of predators (GESAMP, 1997).

*Mnemiopsis leidyi* is the most striking example of a non-indigenous species negatively impacting the Black Sea ecosystem. After its invasion, the structure of the planktonic communities in both coastal waters and the open part of the sea significantly changed. The general abundance of subsurface mesozooplankton declined on average by a factor of 2–2.5 or more, compared to previously. The biomass of some species (small copepods such as *Oithona sp.*, *Paracalanus sp.*, *Acartia sp.* and *Pseudocalanus sp.*) decreased by a factor of 3–10 or more. A sharp decrease (by a factor of approximately 2–10) of meroplankton in summer has also been observed, demonstrating the grazing impact of *M. leidyi* upon the larvae of benthic animals. Indeed, the subsequent decrease of the zoobenthos biomass has been estimated at about 30 percent (Shiganova, 1998; Volovik, Dubinina and Semenov, 1993).

In summary, the three major impacts of *M. leidyi* on fisheries have been identified as follows:

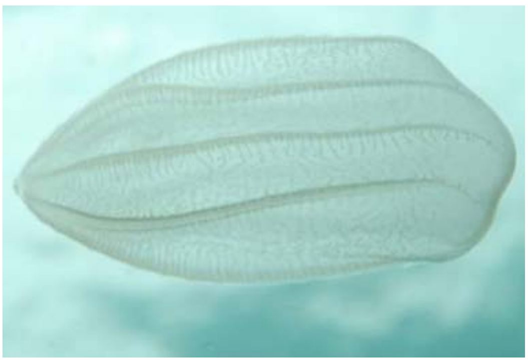
1) Predation on fish eggs and larvae. For example, in shelf waters, *M. leidyi* has been estimated to graze up to 70 percent of the total ichthyoplankton stock (Tsikhon-Lukanina, Reznichenko and Lukasheva, 1993);
2) Feeding on larvae and adult fish food, such as zooplankton, causing starvation in other species; and
3) Further accelerating ongoing ecological changes presently occurring due to eutrophication. For example, direct environmental impacts on pelagic and benthic systems (anoxia) are caused by mucus and dead comb jellies raining down in massive quantities to the floor of the shallow shelf.

All these impacts linked to *M. leidyi* have resulted in a drastic decrease of fish production, e.g. by a factor of 4–5 for Black Sea shad (*Alosa* spp.) and of over 10 for European anchovy (*Engraulis encrasicolus*). Indeed, the annual loss in fish catch attributed to *M. leidyi* has been calculated at approximately USD 200 million in the Black Sea and USD 30–40 million in the Azov Sea (GESAMP, 1997).

### *Beroe ovata*

*Beroe ovata* (Plate 2) is another non-indigenous comb jelly. This species is mitre-shaped (i.e. roughly oval), with a marked lateral compression is and a broad lateral diameter twice the width. Its eggs are 300–350 µm in size and contained in 0.9–1.0 mm diameter gelatinous capsules. The abundance of eggs produced depends on the size of the comb jelly, with approximately 2 000–3 000 eggs for those that are 5–6 cm long and 5 000–7 000 eggs for individuals 8–10 cm long (Volovik, 2004; Shiganova *et al.*, 2000).

**PLATE 2**
*Beroe ovata*



*Beroe ovata* was probably transported in ballast water into the Black Sea, as occurred with *M. leidyi*, and likely transferred from the estuaries along the North Atlantic Ocean, where this species is tolerant to lower salinity and is a native predator of *M. leidyi.* Another hypothesis is that *B. ovata,* which lives in the Mediterranean and the Marmara Sea, penetrated into the Black Sea during the abnormally warm winters of 1997–1998 and 1998–1999, particularly expanding during the summer of 1999 into the northeastern part, which allowed for its easy acclimatization. Interestingly, *B. ovata* was recommended to the Group of Experts on the Scientific Aspects of Marine Environmental Protection (GESAMP) to combat the rapid population growth of *M. leidyi*.

In 1998, the arrival to the Black Sea of *Beroe ovata*, through its predation of *M. leidyi*, led to a partial recovery of the planktonic food web structure. *Beroe ovata* predominantly inhabits the 30-mile width of the coastal zone of the Black Sea, important for fisheries, although its reproduction likely takes place in open waters.

Most experts believe that *B. ovata* feeds exclusively on other comb jellies during all of its developmental stages (Nelson, 1925; Kamshilov, 1955). It has been shown that over the course of a month, a 35 mm-long individual can consume 44 individuals of the 10–35 mm-long *Bolinopsis* and grow up to 44 mm (Zaitsev and Öztürk, 2001). However, *B. ovata* represents a food web dead-end due to its lack of natural enemies in the Black Sea. Therefore, either direct or indirect impacts may be anticipated across the food web, potentially repeating the problematic history of *M. leidyi* and adding to the issue of gelatinous non-indigenous species in the Black Sea. It is of concern, as reported for the first time by Öztürk, Mihneva and Shiganova (2011), that a considerable aggregation of the non-indigenous comb jelly *Bolinopsis vitrea* was found in Turkish waters at Şile on the Asian side and at Kilyos on the European side in 2007. In 2010, individuals of this species, otherwise known in the Mediterranean, were recorded at two locations in Bulgarian waters, into which they may have been carried by the currents or else released with ballast water.

***Rapana venosa***

*Rapana venosa* (Plate 3) is a sea snail or a whelk, also known by the name *Rapana thomasiana* and commonly referred to as rapa whelk. It is assumed that this sea snail was brought to the Black Sea in ballast water from its native habitat in the western Pacific (Gomoiu, 1972; Sorokin, 1982). Near the northeastern Black Sea coast, *R. venosa* has been found to become mature at 2–3 years, live 8–9 years and reproduce during warm periods (July–September). Pelagic larvae of *R. venosa*

**PLATE 3**
*Rapana venosa* (left) and *R. venosa* with egg capsules (right)

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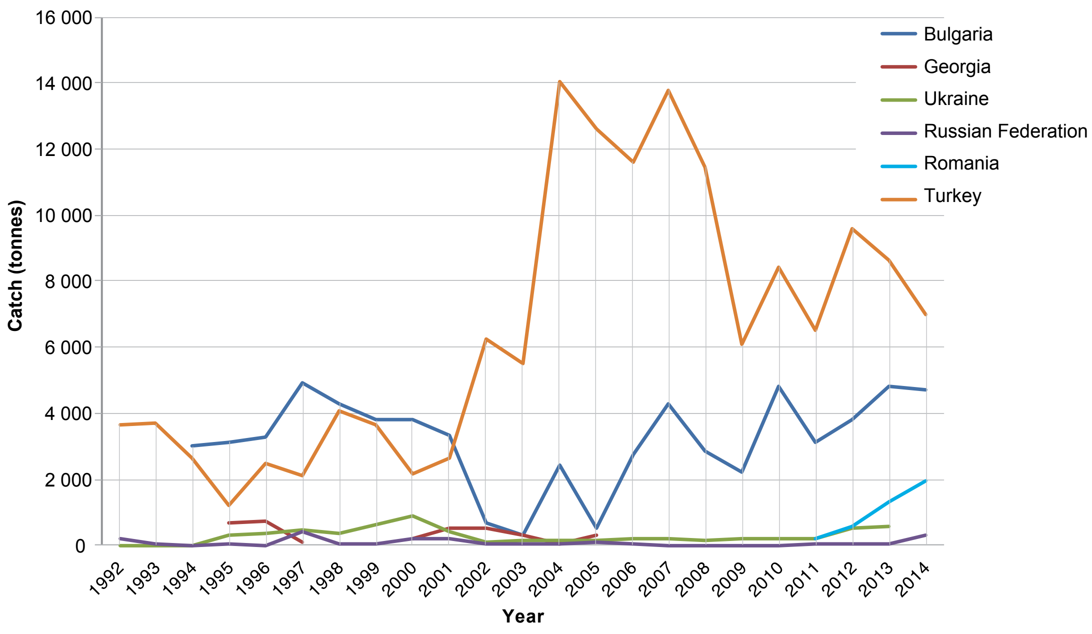
feed on nannoplankton algae while adults feed mainly on bivalves of the families Cardiidae, Mytilidae, Veneridae and Archidae.

*Rapana venosa* can travel over large distances in search of food, while in some periods of the year, it buries itself into the soft sediment. The introduction of this predatory mollusc into the Black Sea ecosystem has turned out to be detrimental to oyster and mussel communities. The distribution of *R. venosa* is associated with reductions in area and density of mussel settlement, particularly near the coasts of Anatolia and the Caucasus. This sea snail has also destroyed oyster banks in the area of the Strait of Kerch and in Karkinitsky Bay; other mollusc communities down to a depth of 30 m have suffered as well. At the same time, Turkey has been conducting large-scale harvesting of *R. venosa* since the mid-1980s (Bilecik, 1990), and other Black Sea countries have also started to exploit it commercially (Figure 7). Nevertheless, Turkish catch remains much higher than other countries, of which the greatest harvester is Bulgaria, where it became a valuable resource after 1994. Prior to regular harvesting, the biomass of *R. venosa* in the coastal areas between Kaliakra and Pomorie in Bulgaria was estimated at about 2 000 tonnes (Prodanov and Konsulova, 1995).

In Turkey, the catch of *R. venosa* has greatly increased in recent years. An analysis of fisheries along the eastern coast of Turkey (Samsun Province) has shown that the number of vessels using dredges to harvest sea snails grew considerably between 2000 and 2005, especially in the 33–149 hp vessel group. These are typical boats that combine sea snail dredging, bottom trawling and net fishing (Knudsen and Zengin, 2006). In recent years, some studies of new catch methods have been undertaken in the Black Sea, such as pots and surface supplied diving systems (Kideys, 2002; Kideys *et al.*, 2007; Sağlam *et al.*, 2007), though some diving accidents have been reported along the Turkish coast. However, these new, environmentally friendly systems have higher operational costs for fishers. Providing a local perspective with wider regional implications, Çulha *et al.* (2009)

**FIGURE 7**
*Rapana venosa* catch in the Black Sea (1992–2014)



reported that the highest density of *R. venosa* was found at 15 m depth in the sandy-muddy biotopes around the Sinop Peninsula of the Turkish coast, which provides an important income for fishers in the area (Knudsen, Zengin and Koçak, 2010).

Elsewhere in the Black Sea, along the northeastern Black Sea coast, *R. venosa* was collected by locals and fishers as souvenirs until the early 1990s. Subsequently, in 1990, 1994 and 1999, assessments were undertaken of the distribution and stock of this sea snail species from Takil Cape to Chauda Cape, with stocks of 2 800 tonnes, 1 500 tonnes and 1 300 tonnes, respectively recorded. The former two assessments corresponded to the initial commercial exploitation of this area, and the latter to a period of intensive fishing. The reduction in *R. venosa* stocks from 2 800–1 500 tonnes (virgin population) to 1 300 tonnes (exploited population) demonstrates the scale of the impacts of dredge fishing. Moreover, the use of knife-edge dredges has adversely affected bottom communities (Shlyakhov and Daskalov, 2008).

As reported by the Scientific, Technical and Economic Committee for Fisheries (STECF, 2011), in 1994, rapa whelk stocks were estimated to reach 14 000 tonnes along the northwestern coast of the Black Sea. As a consequence, the limit for harvesting *R. venosa* was established at 3 000 tonnes per year (Shlyakhov and Daskalov, 2008). Prodanova and Konsulova (1995), and Prodanov, Konsulova and Todorova (1995) have also presented stock assessments and growth rates of *R. venosa* in the Bulgarian waters of the Black Sea. A 1993 survey of ten fishing regions found that the biomass of the commercial stock (i.e. individuals with flesh weight above 60 g) and the total allowable catch were 7 482.6 and 3 217.6 tonnes, respectively.

The importance of the rapa whelk *Rapana venosa* as a Black Sea fishery resource has prompted the GFCM to make it one of its priority species in the Black Sea. The consequences of this decision include assessments of *Rapana venosa*'s status for the whole Black Sea – which have revealed its population to be approaching the maximum sustainable yield – and the adoption of Recommendation GFCM/42/2018/9, which has established a regional research programme for *R. venosa* fisheries in the Black Sea (GFCM, 2019).

### 1.3.2 Other non-indigenous species and their impacts on biodiversity

Concerning non-indigenous plankton species from the Black Sea, the phytoplankton *Phaeocystis pouchetii* is found to be extremely abundant and capable of clogging the gills of fish along the Bulgarian coasts of the Black Sea (Zaitsev and Öztürk, 2001). Furthermore, Moncheva, Petrova-Karadjova and Palasov (1995) reported the phytoplankton *Alexandrium monilatum* from the Bulgarian coast. There is also a list of non-indigenous species proposed by Moncheva and Kamburska (2002), although it is not fully in agreement with Gomez and Boicenco (2004).

The bivalve *Mya arenaria,* a native of the northern Atlantic, was first detected in 1966 and became abundant in a short time in the northwestern and western parts of the Black Sea, reaching its peak abundance in 1972. Although it was later imperilled by the regular hypoxia-anoxia crises that destroyed most of the benthos in the 1980s, it still remains in considerable abundance in the western coastal waters. This species competes for habitat with the small local bivalve, *Lentidium mediterraneum*, which avoids sandy bottoms siltated by *M. arenaria*.

Another non-indigenous bivalve of Indo-Pacific origin, *Anadara inaequivalvis,* was noted for the first time in the Black Sea in 1968 and has subsequently spread to the whole basin. This species has potential for commercial harvesting in Turkey (Sahin, Düzgünes and Okumus, 2006). Recent genetic evidence has shown that this species should be referred to as *A. kagoshimensis* (Krapal *et al.*, 2015)

In 2001, two new non-indigenous bivalve species were found in Odessa Bay: the edible *Mytilus edulis* and *Mytilus trossulus* (Alexandrov, 2004). *Mytilus edulis* was probably brought over in ballast water from the Mediterranean, where it is cultured along the Spanish and Italian shores. The Pacific species *M. trossulus* probably arrived with ships from the Russian Far East, where it is one of the primary species cultivated for aquaculture (Supronovich and Makarov, 1990). The shipworm, *Teredo navalis,* one of the oldest non-indigenous bivalve species found in the Black Sea, is present in small numbers and its impacts are insignificant at present since wood has generally been replaced by concrete or metallic underwater constructions, leading to a decrease in the abundance of shipworms.

Other non-indigenous species, such as the ivory barnacle, *Amphibalanus eburneus*[3], and the acorn barnacle, *Amphibalanus improvisus*[4], are organisms typically found in fouling communities, which may have adverse effects on, for example, the net cages of seabass aquaculture in Trabzon, along the Turkish Black Sea coast.

In November 2009, a few individuals of a jellyfish from the Red Sea, *Cassiopea andromeda* (Plate 4), were found on the shores of Kilyos, just off the Strait of Istanbul. Additionally, a non-indigenous crab species, *Callinectes sapidus* (blue crab; Plate 5), of potential economic importance, has been reported from the Black Sea (Zaitsev and Öztürk, 2001; Yağlıoğlu, Turan and Öğreden, 2014), but its ecological impacts on native species are not well known yet.

**PLATE 4**
*Cassiopea andromeda*

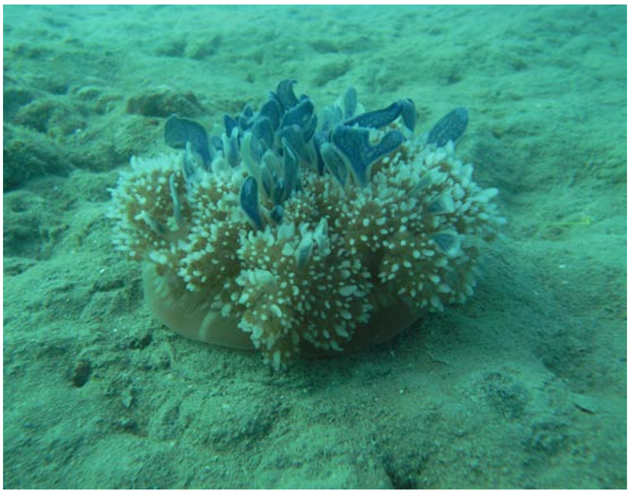


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**PLATE 5**
*Callinectes sapidus*

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### 1.3.3 Impacts of non-indigenous species on pelagic fisheries in the Black Sea

Several studies have focussed on non-indigenous species in the Black Sea (e.g. Zaitsev and Öztürk, 2001). Accurate data are lacking, however, to assess the impacts of non-indigenous species on the fisheries of certain commercial pelagic fish,

[3] Also known as *Balanus eburneus*.
[4] Also known as *Balanus improvisus*.

such as bluefish (*Pomatomus saltatrix*), horse mackerels (both the Atlantic horse mackerel *Trachurus trachurus* and the Mediterranean horse mackerel *Trachurus mediterraneus*) and bonito (*Sarda sarda*), as well as commercial demersal fish, including whiting (*Merlangius merlangus*), turbot (*Scophthalmus maximus*), red mullet (*Mullus barbatus*) and striped red mullet (*Mullus surmuletus*). On the other hand, the impacts of *Mnemiopsis leidyi* and *Beroe ovata* on small pelagic fish, such as anchovy and sprat, have been studied intensively, as described below.

***Sprattus sprattus***

European sprat (*Sprattus sprattus*) is one of the most abundant and commercially important pelagic fish species in the Black Sea and also serves as a major food source for larger fish (Ivanov and Beverton, 1985). European sprat reaches its maturity at one year and reproduces over the course of the whole year, though its peak spawning takes place between November and March.

Its spawning during winter and spring in deeper layers has been relatively unaffected by *Mnemiopsis leidyi* because of this comb jelly's low biomass at those depths. There is therefore little competition for prey and predation on sprat eggs and larvae (Shlyakov and Daskalov, 2008).

In summer, the juvenile and adult sprats leave the upper warmed layer, avoiding severe competition for food from other plankton-consumers, including from *M. leidyi*. During this period, the preferred food of *S. sprattus* mainly consists of the cold-water *Calanus* and *Pseudocalanus* copepod species living below the cold intermediate layer of the water column. It should be noted that these prey species are also available to *M. leidyi* as they migrate to the thermocline at night for their daily feeding, when they can be consumed by this comb jelly. This predation can partly explain the reduction in sprat stocks during the population explosion of *M. leidyi* in the early 1990s. Like other commercial stocks, heavy overfishing also took place before and during the *M. leidyi* outbreak, aggravating stock depletion (Prodanov *et al.*, 1997).

***Engraulis encrasicolus***

The European anchovy (*Engraulis encrasicolus*) is the most important commercial pelagic fish species throughout the Black Sea. Between October and November, it migrates to its wintering grounds along the Anatolian and Caucasian coasts, forming dense concentrations until March, at which point it becomes subject to intensive commercial harvesting.

*Engraulis encrasicolus* competes for food with *Mnemiopsis leidyi* (Grishin, Kovalenko and Sorokolit, 1994), which has probably impacted the anchovy's population growth (Oğuz, Fach and Salihoğlu, 2008). Combined with overfishing, the initial outbreak of *M. leidyi* appears to have caused a significant reduction in Black Sea anchovy stocks from the late 1980s into the early 1990s (Niermann, 2004; Grishin *et al.*, 2007). Moreover, it has been noted by Kideys (2002) that *M. leidyi*'s feeding on the eggs and larvae of *E. encrasicolus* represents another major factor contributing to the collapse of the anchovy fisheries.

Anchovy catch increased after the population explosion of *Beroe ovata* at the end of the 1990s, due to *B. ovata* putting pressure on *M. leidyi* populations. By feeding almost exclusively on *M. leidyi*, *B. ovata* helped the ecosystem to partly recover. As a result, anchovy catch by Turkish fishers, for example, stabilized from the 1990s into the 2000s in the Black Sea. However, the total economic losses caused by the reduction of anchovy catch between 1989 and 1992 can only be roughly estimated. According to Campbell (1993), the total annual loss to fish processing factories in the Black Sea was calculated at USD 11 million, and to the fishing industry itself at approximately

USD 330 million in 1992. In the Turkish fishing sector alone, for example, the economic damage was conservatively estimated at several hundred million dollars.

Damage caused by *M. leidyi* to the anchovy population is most likely due to food competition, as unusually low levels of zooplankton biomass were observed in the top 50 m of the water column during the summers of the early 1990s (Oğuz, Fach and Salihoğlu, 2008). Anchovy larvae could also be impacted by *M. leidyi* predation, given that anchovy larvae numbers peak in July and August, when *M. leidyi* biomass also experiences a seasonal peak (Grishin *et al*., 2007). Indeed, *M. leidyi* can consume a daily ration of several times its own weight (Lipskaya and Luchinskaya, 1990), and while its food spectrum is quite wide, its diet includes an abundance of anchovy eggs and larvae. Though anchovy larvae are predominantly found in narrow coastal zones and *M. leidyi* is distributed further offshore, there is some overlap between the two. Oğuz, Fach and Salihoğlu (2008) reported that the shift from a large marine ecosystem to a totally gelatinous invader-dominated state required an extremely strong environmental perturbation. As such, it can be clearly demonstrated that an environmental disturbance of this kind could create a suitable niche for a non-indigenous gelatinous invader to become a member of the food web structure and to share food resources with the native small pelagic fish communities.

***Trachurus trachurus***

Dietary studies on juvenile and adult Atlantic horse mackerel (*Trachurus trachurus*) have shown that both the habitats and the diet of juvenile Atlantic horse mackerel and *M. leidyi* overlap. Therefore, the strong feeding pressure exerted by *M. leidyi* on zooplankton directly impacts larval and juvenile Atlantic horse mackerel.

With the first outburst of *M. leidyi* in autumn 1988, the maximum production of zooplankton in summer did not suffer much. The copepods *Oithona nana* and *Oithona similis*, which constitute the main food sources for horse mackerel larvae (Revina, 1964), were especially abundant. However, favourable trophic conditions for horse mackerel larvae in summer 1988 failed to ensure the formation of a strong year-class, as juveniles were faced with strong feeding competition from *M. leidyi* later in the year. A sharp decline in *Oithona* under predation pressure from *M. leidyi* over the subsequent years further threatened the survival of the Atlantic horse mackerel (Vinogradov, Shushkina and Nikolaeva, 1993).

### 1.3.4 Non-indigenous fish species and their impacts

According to Slastenenko (1955–1956), a total of 189 fish species can be found in the Black Sea, of which 34 live in estuarine and lagoonal areas. In recent years, however, a number of fish species that are ecologically tolerant to wide-ranging temperature and salinity have settled in the Black Sea. These species have migrated from the Mediterranean Sea and are extending their northern distribution up to the Crimean Peninsula. For example, Indo-Pacific species, including the red barracuda (*Sphyraena pinguis*) and a coral-dwelling fish, *Heniochus acuminatus,* have recently expanded their distribution ranges to the Black Sea (Boltachev, 2009; Boltachev and Astakhov, 2004; Yankova *et al*., 2013). These fish species are Lessepsian migrants and, after passing through the Aegean Sea, have ultimately reached the Black Sea. Although temperature represents a primary factor in the dispersion of Lessepsian fish, with lower temperatures potentially impeding the progress of tropical fish, these species have actually penetrated into the Black Sea. Nevertheless, though these species need to be monitored in the Black Sea in terms of their distribution and abundance, there is little evidence to suggest that they have had strong impacts on ecosystems so far. Yankova *et al.* (2013) have reported twenty-one non-indigenous fish species in the Black Sea, including Lessepsian migrants belonging to eight genera.

**PLATE 6**
*Liza haematocheilus*

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An intentionally introduced fish, the haarder, *Liza haematocheilus* (Plate 6), has become an important commercial species and is now distributed around all the coastal waters of the Black Sea, Azov Sea, Marmara Sea and the Mediterranean. Its annual catch in the Black Sea exceeds 10 000 tonnes (Zaitsev and Öztürk, 2001). According to Okumuş and Basçınar (1997), in comparison with native mullet species, the growth rates of *L. haematocheilus* seem to be much higher, allowing it to reach sexual maturity earlier.

Sexual maturity in *L. haematocheilus* has been estimated at 3–4 years for males and 4–5 years for females, while the spawning period extends from the end of May to the beginning of July. Its pelagic eggs have a diameter of 0.8–0.9 mm and contain large oil droplets, which constitute up to 23 percent of the egg's volume. This characteristic provides high buoyancy, which allows the haarder´s eggs to develop in low saline water in coastal wetlands. Haarder fry feed on zooplankton and can therefore compete with local plankton-eating fish, while the adults consume small bottom-living organisms, mostly meiobenthos, thereby competing with plaice and turbot juveniles in coastal areas (Kazanskji, 1989). Some specific parasites (Trematoda, Monogenea) associated with haarder have been introduced into the Black Sea and turn up in the bodies of local grey mullets as well, the consequences of which on human health and on the biota of the Black Sea require further investigation. It is expected that in the future, this species will become even more commercialized in the Mediterranean and the Black Sea.

Among intentionally introduced species, rainbow trout (*Oncorhynchus mykiss*) and Atlantic salmon (*Salmo salar*), are also commercially produced, mostly in the Turkish waters of the Black Sea. A possible issue with such intentionally introduced species is the risk of hybridization with native species, in the event of their escape from sea cages, though no such case has yet been reported in the Black Sea.

### 1.3.5 Lessons learnt for the Black Sea

The introduction of non-indigenous species in the Black Sea is still ongoing and needs to be monitored at the national, regional and international levels. A special monitoring programme is crucial for key areas such as the Straits of Istanbul, Çanakkale and Kerch, to better understand the dispersal patterns of non-indigenous species.

The impact of non-indigenous species is complex and most of the time unpredictable due to the lack of monitoring and sufficient scientific knowledge about those species. Experts on non-indigenous species, including taxonomists, should be trained and encouraged to continue their important work. Capacity building for Black Sea coastal countries is essential for the monitoring of non-indigenous species. Moreover, initiatives to manage databases on *Mnemiopsis leidyi* and other non-indigenous species should be pursued and promoted by international organizations, such as the GFCM and the Black Sea Commission.

As a consequence of recent climate change and increases in water temperature, some Mediterranean species, such as the sardine, bogue and wrasse, have also penetrated into the Black Sea, likely impacting fishing catches in the future. Species that have extended their range from the Mediterranean therefore need to be monitored in the Black Sea. General trends appear to indicate that a miniature Mediterranean Sea will be established within the Black Sea. Additionally, the establishment of non-indigenous species in the Black Sea is putting pressure on endemic species, which appear to be retreating to brackish water areas and taking refuge in estuaries and deltas (Yankova *et al*., 2014).

Due to overfishing at the end of 1980s, the reduction of total fish biomass in the Black Sea to less than one third of its maximum value in the 1970s has caused a partial vacuum of ecological niches (Llope *et al*., 2011). Subsequently, following the motto *"Natura abhoret vacuum"*, some of these ecological niches have been occupied by the non-indigenous invader *M. leidyi*. Overfishing, therefore, should be avoided for all fish species in order to minimize the risk of non-indigenous species invasion.

Public awareness and sensibilization programmes for local people, fishers, boat crews, harbour masters and coast guards are required to recognize non-indigenous species and their impacts on nature, human health and fisheries. Fisheries cooperatives should also benefit from special education programmes to inform local fishers on the impacts of non-indigenous species.

Legal measures regarding the intentional introduction of non-indigenous species into the Black Sea should be taken by national authorities and within international conventions, such as the Convention on the Protection of the Black Sea Against Pollution (the Bucharest Convention) and the Convention on the Conservation of European Wildlife and Natural Habitats (the Bern Convention). Importantly, the International Convention for the Control and Management of Ships' Ballast Water and Sediments (the BWM Convention) came into force on 8 September 2017. This global convention can play an essential role in halting invasive non-indigenous aquatic species and facilitating regional and international cooperation. Indeed, the essential objective is to combat non-indigenous invasive species taking over entirely new ecosystems. The significant impacts of climate change and Mediterranization should also be taken into account for the Black Sea. Special studies are needed to genetically identify non-indigenous species, assess the economic losses caused by non-indigenous species to individual countries and obtain more reliable catch statistics, such as for rapa whelk (*Rapana venosa*).

A regional database and the continued development of networks are also recommended for the Black Sea to reinforce regional cooperation and improve early detection of all kinds of non-indigenous species, such as the invasive alga *Alexandrium monilatum.*

Additionally, citizen science activities should be encouraged in the Black Sea, as well as increased capacity building.

Finally, the Black Sea Commission should continue to regionally manage the non-indigenous species database and to monitor and predict future risks to biodiversity in the Black Sea.

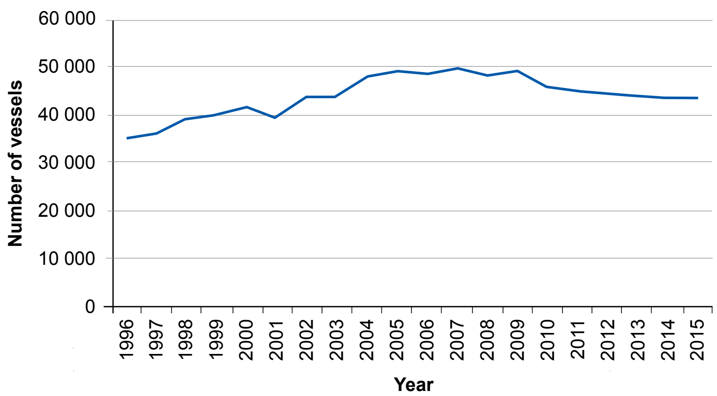
# 2. Non-indigenous species in the Turkish Straits

## 2.1 Introduction of non-indigenous species in the Marmara Sea

The Mediterranean and the Black Sea are interconnected by the Turkish Straits, which acts as a biological corridor, a barrier or an acclimatization zone for some marine species (Öztürk and Öztürk, 1996) (Figure 3).

With the opening of the Suez Canal in 1869, the Mediterranean Sea became connected to the Red Sea, and thereby the Indian Ocean. Some Lessepsian species entered and began to colonize the Mediterranean, Marmara Sea and Black Sea. Ship-transported species, such as *Rapana venosa*, or introduced species, such as *Liza haematocheilus*, colonized the Black Sea first and then dispersed to the Marmara Sea. Shipping traffic is at a significant level in the Strait of Istanbul and thus poses new risks for the Marmara and Black Seas since shipping is the main vector for alien species arrival in this region (Figure 8).

**FIGURE 8**
Shipping traffic in the Strait of Istanbul (1996–2015)




Source: Directorate General of Coastal Safety, 2021.

## 2.2 Occurrence and impacts of non-indigenous species in the Marmara Sea

Zaitsev and Öztürk (2001) reported 14 non-indigenous species, Çınar *et al.* (2005) 48 non-indigenous species and Çınar *et al.* (2011) 69 non-indigenous species in the Marmara Sea. More recently, Öztürk and Albayrak (2016) have recorded 95 non-indigenous species from 11 different systematic groups. At present, the Marmara Sea hosts 99 non-indigenous species belonging to 13 different systematic groups, of which the phytobenthos boast the highest number of species (30 species), followed by Polychaeta (25 species), Crustacea (14 species), Mollusca (6 species), Cnidaria (6 species), Foraminifera (5 species), Pisces (4 species), Ctenophora (2 species), Ciliophora (2 species), Echinodermata (2 species), Phytoplankton (1 species), Porifera (1 species) and Tunicata (1 species) (Figure 11).

Presently, only a few Lessepsian species are found in the Marmara Sea as it serves as a barrier for many thermophilic fish species. At the same time, the Marmara Sea likely acts as a major transitional acclimatization and colonization zone prior to species' expansions into the Black Sea. Katağan *et al.* (2004), Tunçer, Cihangir and Bilecenoğlu (2008), and Artüz and Kubanç (2015) reported the first occurrence of Lessepsian migrants in the Marmara Sea: a stomatopod shrimp, *Erugosquilla massavensis* (Plate 7), and two closely related pufferfish species, *Lagocephalus spadiceus* and *L. sceleratus*, none of which have been reported yet in the Black Sea. Currently, *Solea senegalensis*, a species of flatfish, provides the only example of a non-indigenous species from the Atlantic Ocean in the Marmara Sea. Apart from these non-indigenous arrivals, only one intentionally introduced

**PLATE 7**
*Erugosquilla massavensis*

©Bayram Öztürk

**PLATE 8**
*Penaeus japonicus*

©Bayram Öztürk

species specifically of Indo-Pacific origin – the kuruma prawn (*Penaeus japonicus*) – is found in the Marmara Sea; it is considered of high commercial value (Plate 8).

### 2.2.1 *Mnemiopsis leidyi* and its impacts on fisheries and fish stocks in the Marmara Sea

*Mnemiopsis leidyi* has had severe impacts on the Marmara Sea, which should be evaluated in terms of effects on fisheries and fish stocks. This species was first introduced to the Black Sea and spread via surface currents to the Marmara Sea and the Mediterranean. In October 1992, an extremely vigorous outbreak was recorded in the Marmara Sea (GESAMP, 1997), with the abundance of *M. leidyi* recorded as high as 4.3 kilograms per cubic metre near the Strait of Istanbul and 9.7 kilograms per cubic metre near the Strait of Çanakkale, mostly at depths of 10–30 m (Shiganova *et al.*, 1995). The pelagic fish stocks in the Marmara Sea declined markedly as these species feed mainly on copepods and cladocerans, which are also foraged by *M. leidyi*. Furthermore, *M. leidyi* feeds on fish eggs and larvae, seriously impacting economically important fishes, such as *Scomber scombrus, Sardina pilchardus, Sprattus sprattus*, *Engraulis encrasicolus*, *Trachurus trachurus* and *Pomatomus saltatrix,* which use the Marmara Sea as spawning grounds. Given that the Marmara Sea represents about 15 percent of Turkey's total fish catch, any harmful non-indigenous species can have substantial impacts on fish stocks and fishing communities.

İşinibilir (2007) reported, as shown also in the Black Sea, that *M. leidyi* abundance becomes limited in summer, when *Beroe ovata* is present.

In 1989, catch of the main pelagic commercial fish species in the Marmara Sea declined due to an outbreak of *Mnemiopsis* in the Black Sea (which can be assumed to have occurred in the Marmara Sea as well, since the northern end of the Marmara Sea region is affected by the Black Sea region in terms of fishery statistics), the fish catch increased steadily until 1999 (up to almost 55 000 tonnes). For Turkey only, between 1991 and 2000, the decline of fish stocks and economic losses to its fisheries were estimated at USD 400 000 (Öztürk and Öztürk, 2000). Between 2000 and 2013, dominant species of the catch of commercial small pelagic fish species in the Marmara Sea were European anchovy (*Engraulis encrasicolus*) and Atlantic horse mackerel (*Trachurus trachurus*), though the impact of *Mnemiopsis* on these species has not been calculated at present.

Another economic impact of *M. leidyi* was the invasion of this species into the freshwater reservoir of Istanbul City, which caused serious damage to the pipeline (Öztürk, Öztürk and Algan, 2001).

### 2.2.2 *Rapana venosa* and its impacts on fisheries in the Marmara Sea

The rapa whelk *Rapana venosa* is native to the western Pacific. After reaching the Black Sea, likely in ballast water or as eggs attached to ship hulls, *R. venosa* penetrated the Marmara Sea in the 1960s, and subsequently the Aegean Sea and the Adriatic Sea (Acar and Ates, 2017; Lipej *et al.*, 2012; Crocetta *et al.*, 2017) (Figure 9).

**FIGURE 9**
First records of *Rapana venosa* at various locations in the Black Sea, Marmara Sea and northern Aegean Sea

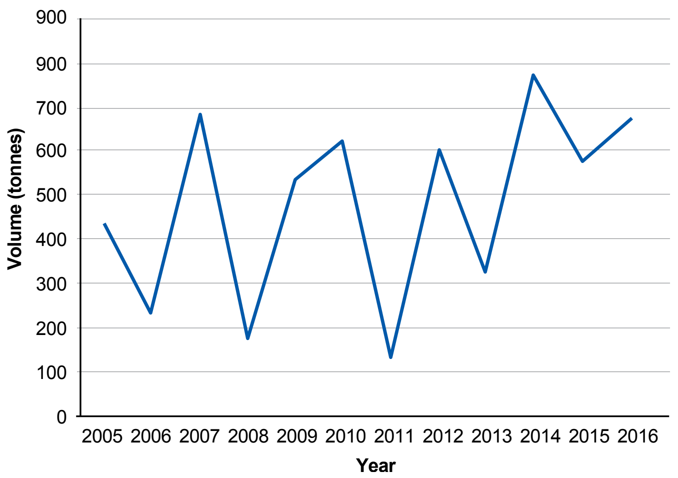

Source: UN, 2021 modifed from Zaitsev and Öztürk, 2001.

*Rapana venosa* feeds mainly on mussels and oysters on rocky bottoms. In the Marmara Sea, it is quite abundant at depths of 5–25 m (maximum density is 15–20 individuals per square metre). Due to high population densities of *R. venosa* along the Marmara Sea coast, oysters and mussels areas are largely absent from the area, whereas harvesting used to be commercially important. *Rapana venosa* is harvested by diving and dredging; however, impacts from dredging are much larger on benthic ecosystems, as it is a non-selective method (Aydin, Düzgüneş and Karadurmuş, 2016). In 1982, *R. venosa* became economically important for the first time, its meat being exported to Japan. The maximum catch for *R. venosa* was around 800 tonnes in 2014 and it remains significant (Figure 10).

### 2.2.3 Other non-indigenous species and their impacts on fisheries in the Marmara Sea

The Indo-Pacific kuruma prawn *Penaeus japonicus* was intentionally introduced to the Marmara Sea in the late 1960s from Iskenderun Bay on the Turkish coast of the Mediterranean Sea. However, its population did not increase as much as expected.

**FIGURE 10**
*Rapana venosa* catch in the Marmara Sea (2005–2016)




Source: Öztürk and Albayrak, 2016;TUIK, 2014.

Another non-indigenous species, *Erugosquilla massavensis*, a mantis shrimp, was found for the first time in the central Marmara Sea in 2004 (Katağan *et al.*, 2004), which is the second Indo-Pacific crustacean reported from the Marmara Sea. Mantis shrimps do have commercial value in the Turkish part of the Mediterranean Sea, but due to their small stock size in the Marmara Sea, they have not yet become economically important.

Albayrak (2005) reported the Japanese carpet shell (*Ruditapes philippinarum*), a non-indigenous bivalve, in the Marmara Sea had potential commercial value but was not being caught yet by fishers; subsequently, it was recorded in the Strait of Çanakkale (İşmen *et al.*, 2010). More recently, Çolakoğlu and Palaz (2014) noted that *R. philippinarum*, a dominant species in the sandy and muddy areas of coastal waters, was the target of intensive commercial harvesting. The non-indigenous starfish species *Asterias rubens* was also observed in the Marmara Sea and the Strait of Istanbul in 1996 (Albayrak, 1996), and its interaction with mussel communities should be monitored due to its ability to consume large quantities of mussels.

An intentionally introduced fish, the haarder, *Liza haematocheilus*, native to the Amu Darya River basin, reached the Turkish Black Sea coast from the Azov Sea, and subsequently the Marmara Sea and the coasts of the Aegean Sea. While this species is commercially important in its own right, data are not available evaluating it separately from native species of mullets, which together between 1995 and 2007 achieved a maximum of 5 000 tonnes. A possible concern is that *L. haematochelia* may displace other native mullet species, and this dynamic should be monitored.

*Lagocephalus spadiceus*, of Indo-Pacific origin, is one of the most abundant non-indigenous invasive pufferfishes of the eastern Mediterranean Sea, occurring along the entire Levantine coast from Port Said to the southern Aegean Sea (Golani *et al.*, 2002). This species requires monitoring to evaluate potential impacts on fisheries, human health and native fish fauna in the Marmara Sea. Recently, *Lagocephalus scelaratus* has also been reported by Artüz and Kubanç (2015) in the Marmara Sea, showing contents of a powerful and complex neurotoxin called tetradotoxin (TTX), but impacts of this fish are not well known yet.

The bivalves, *Anadara kagoshimensis* (formerly *A. inaequivalvis*; Krapal *et al.*, 2015) and *Mya arenaria,* are also remarkably invasive non-indigenous mollusc species in the Marmara Sea, and occur dominantly between depths of 3–15 m. *Mya arenaria* is preyed on by *Rapana venosa* and demersal fishes, such as turbot, goby and mullet.

In 2015, the non-indigenous upside-down jellyfish, *Cassiopea andromeda,* another Lessepsian migrant, was observed on diving expeditions at the entrance to the Strait of Çanakkale (Öztürk and Albayrak, 2016), but this species has not caused any impacts so far.

An updated list of non-indigenous species in the Marmara Sea is presented in Table 1, as well as their broad taxonomic classifications (Figure 11).

**TABLE 1 – List of non-indigenous species in the Marmara Sea**

| Group/species | Year of first record | Establishment success | Reference | Pathway | Origin | Potential impact |
|---|---|---|---|---|---|---|
| **PHYTOPLANKTON** | | | | | | |
| *Rhizosolenia calcar-avis* Schultze, 1858 | 1993 | Established | Kocataş *et al.*, 1993 | Shipping | Atlantic Ocean | |
| **PHYTOBENTHOS** | | | | | | |
| **Rhodophyta** | | | | | | |
| *Acanthophora nayadiformis* (Delile) Papenfuss, 1968 | 1973 | Established | Zeybek and Güner, 1973 | Suez Canal | Red Sea, Indian Ocean | |
| *Asparagopsis armata* Harvey, 1855 | 1986 | Invasive | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Bonnemaisonia hamifera* Hariot, 1891 | 1986 | Invasive | Zeybek *et al.*, 1986 | Shipping | Indo-Pacific | |
| *Chondria collinsiana* M.A. Howe, 1920 | 1986 | Invasive | Zeybek *et al.*, 1986 | Suez Canal | Atlantic Ocean, Pacific Ocean | |
| *Chondrophycus papillosus** (C. Agardh) D.J. Garbary & J.T. Harper, 1998 | 1957 | Established | Öztığ, 1957 | Suez Canal | Red Sea | |
| *Chondria curvilineata* F.S. Collins & Hervey, 1917 | 1984 | Established | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean | |
| *Colaconema codicola* (Børgesen) Stegenga, J.J. Bolton & R.J. Anderson, 1997 | 1984 | Established | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Ganonema farinosum* (J.V. Lamouroux) K.C. Fan & Yung C. Wang, 1974 | 1899 | Cryptogenic | Fritsch, 1899 | Suez Canal | Red Sea, Indo-Pacific | |
| *Gracilaria arcuata* Zanardini, 1858 | 1986 | Questionable | Zeybek *et al.*, 1986 | Suez Canal | Red Sea, Indo-Pacific | |
| *Griffithsia corallinoides* (Linnaeus) Trevisan, 1845 | 1993 | Established | Aysel *et al.*, 1993 | Gibraltar | Atlantic Ocean, Indo-Pacific | |
| *Hypnea variabilis* Okamura, 1909 | 1986 | Established | Zeybek *et al.*, 1986 | Shipping | Pacific Ocean | |
| *Radicilingua thysanorhizans* (Holmes) Papenfuss, 1956 | 1986 | Unknown | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean | |
| *Rhodophysema georgei* Batters, 1900 | 1986 | Casual | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean, Pacific Ocean | |
| **Heterokontophyta (Ochrophyta)** | | | | | | |
| *Botrytella parva* (Takamatsu) H.S. Kim, 1996 | 2012 | Established | Taşkin and Pedersen, 2012 | Shipping | | |
| *Chorda filum* (Linnaeus) Stackhouse, 1797 | 1986 | Casual | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Cladosiphon zosterae* (J. Agardh) Kylin, 1940 | 1984 | Cryptogenic | Aysel *et al.*, 1993 | Shipping | Atlantic Ocean | |
| *Colpomenia peregrina* Sauvageau, 1927 | 1998 | Invasive | Aysel *et al.*, 2000 | Shipping | Indo-Pacific | |
| *Ectocarpus siliculosus* var. *Hiemalis* (P.L. Crouan & H.M. Crouan) Gallardo, 1992 | 1899 | Cryptogenic | Fritsch, 1899 | Shipping | Atlantic Ocean | |
| *Halothrix lumbricalis* (Kützing) Reinke, 1888 | 1993 | Established | Aysel *et al.*, 1993 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Microspongium globosum* Reinke, 1888 | 2003 | Questionable | Taşkin *et al.*, 2006 | Shipping | Atlantic Ocean | |
| *Pylaiella littoralis* (Linnaeus) Kjellman, 1872 | 1993 | Cryptogenic | Aysel *et al.*, 1998 | Shipping | Atlantic Ocean, Pacific Ocean | |

* This name is currently regarded as a synonym of *Palisada perforata*.

**TABLE 1** (Continued)

| Group/species | Year of first record | Establishment success | Reference | Pathway | Origin | Potential impact |
|---|---|---|---|---|---|---|
| *Protectocarpus speciosus* (Børgesen) Kornmann, 1955 | 1993 | Unknown | Aysel *et al.*, 1998 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Sargassum latifolium* (Turner) C. Agardh, 1820 | 1986 | Questionable | Zeybek *et al.*, 1986 | Suez Canal | Red Sea | |
| *Scytosiphon dotyi* M.J. Wynne, 1969 | 2011 | Established | Taşkin, 2012 | Shipping | | |
| *Sphaerotrichia divaricata* (C. Agardh) Kylin, 1940 | 1986 | Established | Zeybek *et al.*, 1986 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Ulonema rhizophorum* Foslie, 1894 | 2012 | Established | Taşkin, 2013 | Shipping | | |
| **Chlorophyta** | | | | | | |
| *Bryopsis pennata* J.V. Lamouroux, 1809 | 1986 | Unknown | Aysel and Sukatar, 1987 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Codium fragile* subsp. *Fragile* (Suringar) Hariot, 1889 | 1998 | Invasive | Okudan *et al.*, 2003 | Shipping | Atlantic Ocean, Pacific Ocean | |
| *Ulva lactuca* Linnaeus, 1753 | 1986 | Cryptogenic | Aysel *et al.*, 1991 | Shipping/ Suez Canal | Cosmopolitan | |
| **FORAMINIFERA** | | | | | | |
| *Siphonaperta arenata* (Said, 1949) | 2002 | Casual | Kaminski *et al.*, 2002 | Shipping | Pacific Ocean | |
| *Amphistegina lobifera* Larsen, 1976 | 2004 | Invasive | Meriç *et al.*, 2005 | Suez Canal | Circumtropical | |
| *Cushmanina striatopunctata* Parker & Jones, 1865 | 2007 | Established | Meriç *et al.*, 2009 | Shipping | Tropical Atlantic | |
| *Spiroloculina antillarum* d'Orbigny, 1839 | 2007 | Established | Meriç *et al.*, 2008 | Suez Canal | Circumtropical | |
| *Spiroloculina angulata* Cushman, 1917 | 2001 | Unknown | Avşar *et al.*, 2001 | Shipping | | |
| **PORIFERA** | | | | | | |
| *Paraleucilla magna* Klautau, Monteiro & Borojevic, 2004 | 2016 | Invasive | Topaloğlu *et al.*, 2016 | Shipping | Western Atlantic | |
| **CNIDARIA** | | | | | | |
| *Aequorea vitrina* Gosse, 1853 | 2017 | Invasive | Yilmaz *et al.*, 2017 | Shipping | Northeastern Atlantic | |
| *Coryne eximia* Allman, 1859 | 1952 | Established | Demir, 1952–1954 | Shipping | | |
| *Diadumene cincta* Stephenson, 1925 | 2011 | Established | Gökalp, 2011 | Shipping | | |
| *Eudendrium merulum* Watson, 1985 | 1953 | Established | Marques *et al.*, 2000 | Shipping | | |
| *Filellum serratum* (Clarke, 1879) | 1981 | Established | Ünsal, 1981 | Shipping | | |
| *Cylista lacerata* (Dalyell, 1848) | 2014 | Established | Çinar *et al.*, 2014b | Shipping | | |
| **CTENOPHORA** | | | | | | |
| *Beroe ovata* Bruguière, 1789 | 2004 | Established | İşinibilir *et al.*, 2004 | Shipping | Western Atlantic | Invasive |
| *Mnemiopsis leidyi* A. Agassiz, 1865 | 1994 | Invasive | Kideys and Niermann, 1994 | Shipping | Western Atlantic | Invasive |
| **CILIOPHORA** | | | | | | |
| *Eutintinnus lusus-undae* (Entz, 1884) | 2004 | Established | Balkis, 2004 | Shipping | | |
| *Eutintinnus apertus* Kofoid & Campbell, 1929 | 2004 | Established | Balkis, 2004 | Shipping | | |
| **POLYCHAETA** | | | | | | |
| *Synelmis rigida* (Fauvel, 1919) | 1959 | | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |
| *Capitellethus dispar* (Ehlers, 1907) | 1959 | Questionable | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |

**TABLE 1** (Continued)

| Group/species | Year of first record | Establishment success | Reference | Pathway | Origin | Potential impact |
|---|---|---|---|---|---|---|
| *Dasybranchus carneus* Grube, 1869 | 1959 | Questionable | Rullier, 1963 | Shipping | Red Sea | |
| *Desdemona ornata* Banse, 1957 | 2005 | Established | Çinar *et al.*, 2009 | Shipping | Indo-Pacific | |
| *Ficopomatus enigmaticus* (Fauvel, 1923) | 1952 | Invasive | Demir, 1952–1954 | Shipping | Subtropical | |
| *Glycera alba adspersa* Fauvel, 1939 | 1963 | Unknown | Rullier, 1963 | Shipping | Indo-Pacific | |
| *Hydroides dirampha* Mörch, 1863 | 1894 | Invasive | Ostrou-moff, 1894 | Shipping | Circumtropical | |
| *Paralepidonotus indicus* (Kinberg, 1856) | 1963 | Unknown | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |
| *Harmothoe minuta* (Potts, 1910)** | 1963 | Unknown | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |
| *Hydroides elegans* (Haswell, 1883) [nomen protectum] | 2014 | Invasive | Çinar *et al.*, 2014a | Shipping | Cosmopolitan | |
| *Lumbrineris debilis* Grube, 1878 | 1959 | | Rullier, 1963 | Shipping | Indo-Pacific | |
| *Lepidonotus carinulatus* (Grube, 1869) | 1959 | Questionable | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |
| *Loimia medusa* (Savigny, 1822) | 1959 | Questionable | Rullier, 1963 | Suez Canal | Red Sea | |
| *Metasychis gotoi* (Izuka, 1902) | 2008 | Established | Çinar *et al.*, 2011 | Suez Canal | Red Sea, Indo-Pacific | |
| *Nereis persica* Fauvel, 1911 | 1959 | Established | Rullier, 1963 | Suez Canal | Red Sea, Indo-Pacific | |
| *Notomastus aberans* Day, 1957 | 2014 | Established | Çinar *et al.*, 2014a | Shipping | Cosmopolitan | |
| *Paraprionospio coora* Wilson, 1990 | 2008 | Cryptogenic | Yokohama *et al.*, 2010 | Shipping | Indo-Pacific | |
| *Polydora cornuta* Bosc, 1802 | 2002 | Invasive | Dağli and Ergen, 2008 | Shipping | Western Atlantic | |
| *Prionospio pulchra* Imajima, 1990 | 2008 | Established | Çinar *et al.*, 2011 | Shipping | Indo-Pacific | |
| *Pseudopolydora paucibranchiata* (Okuda, 1937) | 2008 | Established | Çinar *et al.*, 2011 | Shipping | Indo-Pacific | |
| *Sigambra constricta* (Southern, 1921) | 1959 | Unknown | Rullier, 1963 | Shipping | Red Sea, Indo-Pacific | |
| *Streblospio gynobranchiata* Rice & Levin, 1998 | 2005 | Invasive | Çinar *et al.*, 2009 | Shipping | Western Atlantic | |
| *Timarete anchylochaeta* (Schmarda, 1861) | 1959 | Questionable | Rullier, 1963 | Shipping | Indo-Pacific | |
| *Timarete dasylophius* (Marenzeller, 1879) | 1959 | Questionable | Rullier, 1963 | Shipping | Indo-Pacific | |
| **CRUSTACEA** | | | | | | |
| **Copepoda** | | | | | | |
| *Oithona davisae* Ferrari and Orsi, 1984 | 2016 | Invasive | Doğan and İşinibilir Okyar, 2016 | Shipping | Black Sea | |
| *Acartia (Acanthacartia) tonsa* Dana, 1849 | 1990 | Established | Ünal *et al.*, 2000 | Shipping | Western Atlantic, Indo-Pacific | |
| *Acrocalanus longicornis* Giesbrecht, 1888 | 1988 | Established | Ünal *et al.*, 2000 | Suez Canal | | |
| *Acrocalanus monachus* Giesbrecht, 1888 | 1998 | Established | Ünal *et al.*, 2000 | Suez Canal | | |
| *Calanopia elliptica* (Dana, 1849) | 1998 | Established | Ünal *et al.*, 2000 | Suez Canal | | |
| *Centropages furcatus* (Dana, 1849) | 2000 | Established | Ünal *et al.*, 2000 | Suez Canal | Red Sea, Indo-Pacific | |
| *Parvocalanus elegans* Andronov, 1972 | 2000 | Established | Ünal *et al.*, 2000 | Suez Canal | Red Sea, Indo-Pacific | |

** Currently regarded as nomen dubium.

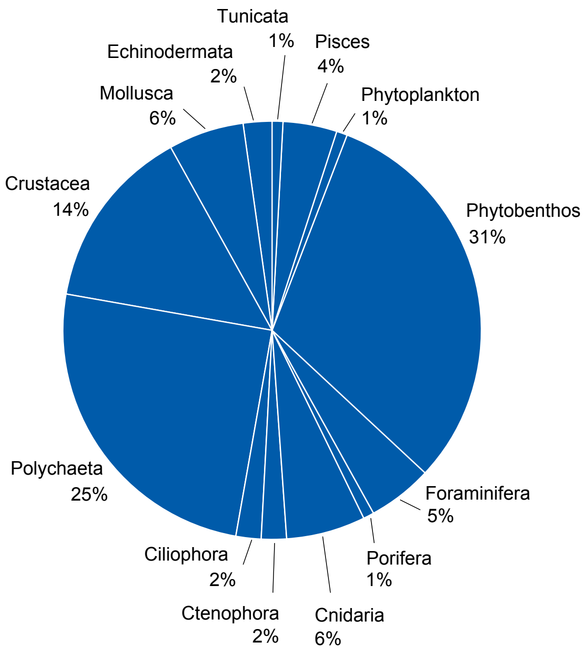
**TABLE 1** (Continued)

| Group/species | Year of first record | Establishment success | Reference | Pathway | Origin | Potential impact |
|---|---|---|---|---|---|---|
| *Parvocalanus latus* Andronov, 1972 | 2000 | Established | Ünal *et al.*, 2000 | Suez Canal | Indian Ocean | |
| **Amphipoda** | | | | | | |
| *Monocorophium sextonae* (Crawford, 1937) | 2006 | Casual | Aslan-Cihangir *et al.*, 2009 | Shipping | Pacific Ocean | |
| **Decapoda** | | | | | | |
| *Callinectes sapidus* Rathbun, 1896 | 2008 | Invasive | Tunçer and Bilgin, 2008 | Shipping | | Commercial |
| *Coleusia signata* (Paulson, 1875) | 2006 | Established | Artüz, 2007 | Suez Canal | Red Sea | |
| *Penaeus japonicus* (Bate, 1888) | 2001 | Invasive | Zaitsev and Öztürk, 2001 | Suez Canal | Indo-Pacific | Commercial |
| *Portunus segnis* (Forskål, 1775) | 2011 | Established | Altuğ *et al.*, 2011 | Unknown | | Commercial |
| **Stomatopoda** | | | | | | |
| *Erugosquilla massavensis* (Kossmann, 1880) | 2004 | Invasive | Katağan *et al.*, 2004 | Suez Canal | Indo-Pacific | Commercial |
| **MOLLUSCA** | | | | | | |
| **Gastropoda** | | | | | | |
| *Rapana venosa* (Valenciennes, 1846) | 1993 | Invasive | Albayrak and Balkis, 1996b | Shipping | Pacific Ocean | Commercial |
| **Bivalvia** | | | | | | |
| *Anadara kagoshimensis* (Tokunaga, 1906) | 1993 | Invasive Established | Albayrak and Balkis, 1996a | Shipping | Indo-Pacific | Commercial |
| *Chama asperella* Lamarck, 1819 | 1990 | Established | Çinar *et al.*, 2011 | Shipping | Red Sea, Indo-Pacific | |
| *Mya arenaria* Linnaeus, 1758 | 1993 | Invasive | Albayrak and Balkis, 1996a | Shipping | Western Atlantic | Commercial |
| *Ruditapes philippinarum* (A. Adams & Reeve, 1850) | 2004 | Invasive | Tunçer *et al.*, 2004 | Aquaculture | Pacific Ocean | Commercial |
| *Teredo navalis* Linnaeus, 1758 | 1966 | Invasive | Oberling, 1971 | Shipping | Circumtropical | |
| **ECHINODERMATA** | | | | | | |
| *Asterias rubens* Linnaeus, 1758 | 1993 | Established | Albayrak, 1996 | Shipping | Atlantic Ocean | |
| *Gambusia affinis* (Baird & Girard, 1853) | Not known | Established | Zaitsev and Öztürk, 2001 | Intentionally introduced | America | |
| **TUNICATA** | | | | | | |
| *Styela clava* Herdman, 1881 | 2016 | Invasive | Çinar, 2016 | Shipping | Northwestern Pacific | |
| **PISCES** | | | | | | |
| **Actinopterygii** | | | | | | |
| *Solea senegalensis* Kaup, 1858 | 1942 | Established | Erazi, 1942 | Gibraltar | Tropical Atlantic | |
| *Lagocephalus sceleratus* (Gmelin, 1789) | 2015 | Unknown | Artüz and Kubanç, 2015 | Unknown | Red Sea, Indo-Pacific | Poisonous |
| *Lagocephalus spadiceus* (Richardson, 1845) | 2007 | Established | Tunçer *et al.*, 2008 | Suez Canal | Red Sea, Indo-Pacific | |
| *Liza haematocheilus* (Temminck & Schlegel, 1845) | 1998 | Established | Kaya *et al.*, 1998 | Aquaculture | Pacific Ocean | Commercial |

Source: Modified from Zaitsev and Öztürk (2001); Çınar *et al.* (2005), Öztürk and Albayrak, 2016) and various sources.

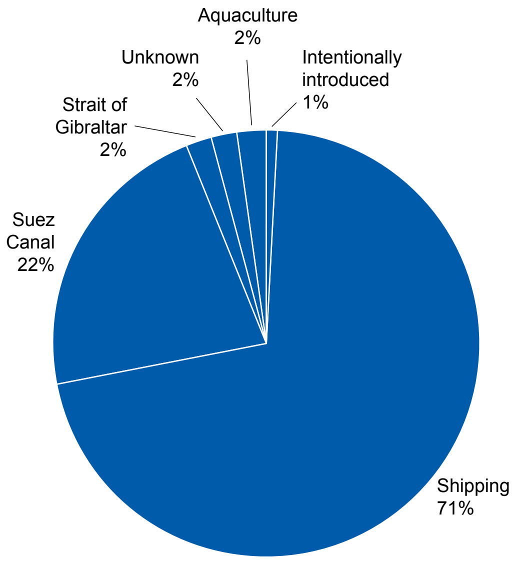
Figure 13 shows the increasing number of non-indigenous species recorded in the Marmara Sea since 1951, with a peak of 24 species between 2001 and 2010. Additionally, Figure 12 shows the percentage of different vectors and routes for the introduction of non-indigenous species into the Marmara Sea: via shipping (71 percent), the Suez Canal (22 percent), the Strait of Gibraltar

**FIGURE 11**
Distribution of non-indigenous species in the Marmara Sea according to taxa



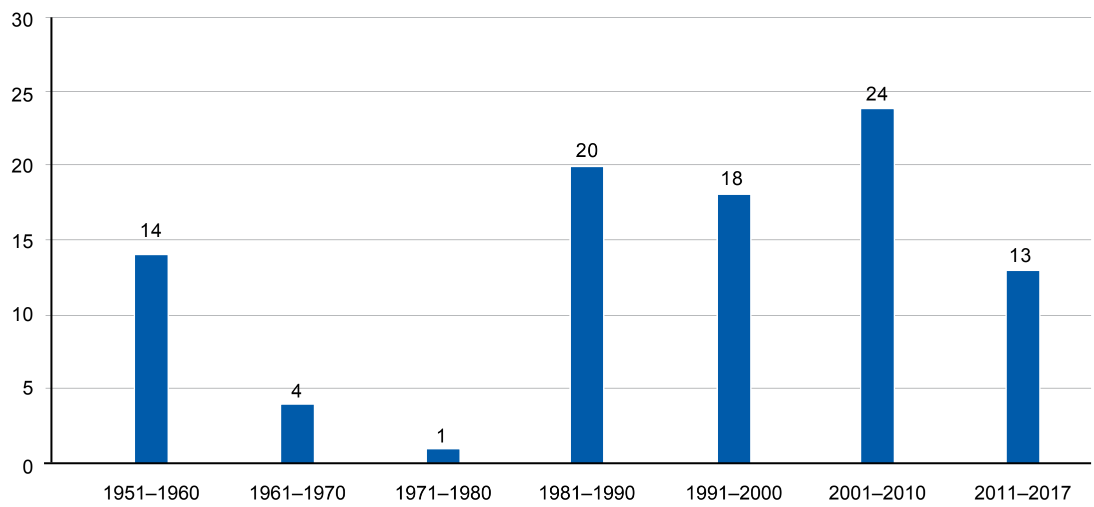

Source: revised from Öztürk and Albayrak, 2016.

**FIGURE 12**
Ways of introducing non-indigenous species into the Marmara Sea




Source: revised from Öztürk and Albayrak, 2016.

**FIGURE 13**
Number of new non-indigenous species reported from the Marmara Sea (1951–2017)



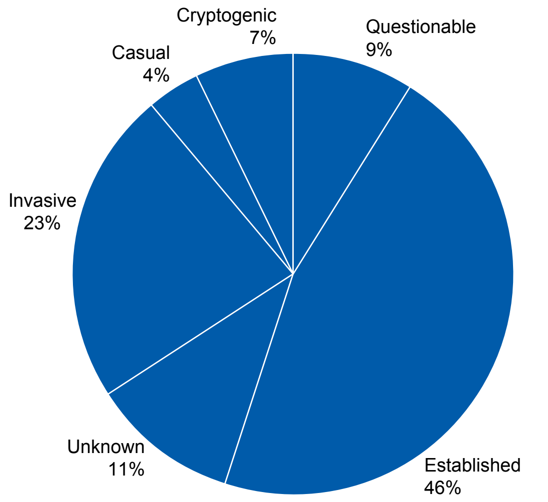

Source: revised from Öztürk and Albayrak, 2016.

(2 percent), aquaculture (2 percent), unknown (2 percent) and intentional introduction (1 percent). This graphic illustrates clearly how shipping remains by far the largest vector for transporting non-indigenous species, globally and not just for the Marmara Sea.

In Figure 14, the different non-indigenous species are grouped according to their status in the Marmara Sea: established (45 percent), cryptogenic (7 percent), questionable (9 percent), casual (4 percent), unknown (11 percent) and invasive (23 percent).

Altuğ *et al.* (2012) and Olgun, Başar and Aydönener (2012) have reported that ballast water increasingly threatens the ecology of the Marmara Sea, and therefore more stringent measures are required, i.e. prohibitions on deballasting operations in vulnerable and small seas such as the Mediterranean and the Black Sea can slow down non-indigenous species invasions into the Marmara Sea.

**FIGURE 14**
Status of non-indigenous species in the Marmara Sea




Source: Öztürk and Albayrak, 2016.

## 2.3 Conclusions

The Marmara Sea provides a "test laboratory" for non-indigenous species and monitoring them may help predict their impacts and evolution in the Black Sea and the Mediterranean. To collect new and accurate information on the occurrence of non-indigenous species, a robust reporting and monitoring system is required. Within this system, fishers can play a role as citizen scientists, reporting to fisheries cooperatives or relevant authorities on unusual organisms in their catch or damage to nets and other equipment. Additionally, other stakeholders, including divers, sailors and harbour authorities, should be closely linked to the appropriate scientific institutions in order to counter the ongoing threats of non-indigenous species in the Marmara Sea. It is likely that more non-indigenous species will be observed in the near future due to the heavy shipping activities occurring between the Mediterranean and the Black Sea.

# 3. Non-indigenous species in the Mediterranean Sea

## 3.1 Main characteristics of the Mediterranean Sea

The Mediterranean Sea is a mid-latitude semi-enclosed sea – almost an isolated oceanic system – and a remnant of the ancient Tethys Ocean. It is characterized by a narrow shelf and littoral zone, and a small drainage area, especially in the northern part. The Strait of Sicily (150 km width, 400 m depth) separates two distinct basins, the western and eastern, and functions as a geographical and hydrographical border. This and other channels play a significant role in determining the oceanographic characteristics of each regional sea, such as the Adriatic, Aegean and Levant Seas. From west to east, i.e. from Gibraltar to the Syrian Arab Republic, the Mediterranean Sea is about 4 000 km long. Its greatest breadth, from the coast of France to that of Algeria, covers a distance of 900 km. The Mediterranean, including all its adjacent seas except the Black Sea, has a surface area of 2 523 000 square kilometres and a volume of 3 708 000 cubic kilometres, with a mean depth of 1 470 m (Miller, 1983).

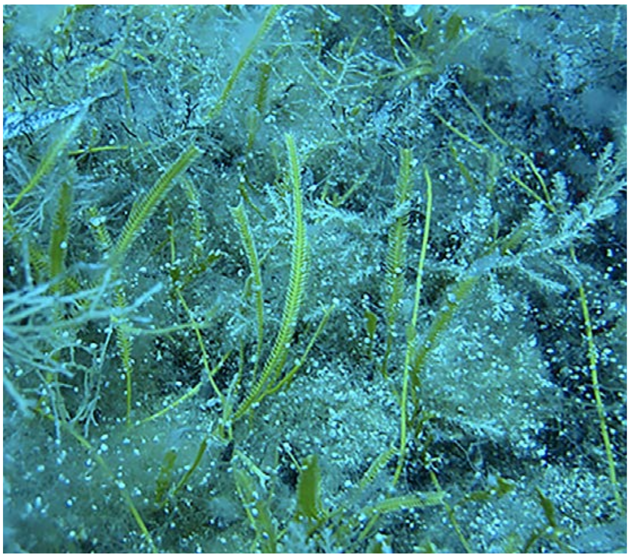
Oxygen levels are almost saturated in the surface layer (6 millitres per litre in the winter and 4.8 millilitres per litre in the summer). In deep water, the oxygen concentration is about 4.5 millitres per litre in the western basin and 4.2 millitres per litre in the eastern basin. The Mediterranean Sea also displays seasonal variation in surface temperature. During summer, warm surface water (warmer than 20°C) leads to strong stratification of the water column. During winter, cold water (12–15°C), distributed homogeneously between the surface and greater depths, causes important vertical convection (upwelling), thus recycling abundant nutrients found in deep waters (Fonteneau, 1996).

The circulation of water masses in the Strait of Gibraltar is vital for the biota of the Mediterranean. A permanent surface current towards the east brings superficial Atlantic water into the Mediterranean Sea, while a deep current flows westwards, transporting Mediterranean water. However, these circulation patterns may be altered by climate change in the future.

The geological history of the Mediterranean should also be taken into account in understanding the sea. During the Messinian salinity crisis, the Mediterranean became partly or nearly completely desiccated in the latter part of the Messinian age of the Miocene, 5.96–5.33 million years ago, and huge amounts of evaporitic sediments were deposited, causing the precursor of the Strait of Gibraltar to seal the Mediterranean off from the Atlantic (Hsü *et al*., 1977). The Messinian salinity crisis ended with the Strait of Gibraltar finally reopening 5.33 million years ago, at which point the Atlantic rapidly spilled in to fill the Mediterranean basin in what is known as the Zanclean flood. Even today, the Mediterranean is considerably saltier than the North Atlantic, owing to its near isolation by the Strait of Gibraltar and its high rate of evaporation.

The Mediterranean is an oligotrophic sea with both low phytoplankton biomass and primary production due to weak fluvial supplies and poor surface water input from the Atlantic. The Mediterranean fauna and flora, which have evolved over millions of years, include a large proportion of endemic species, with both temperate and subtropical elements (Fredj and Meinardi, 1992; Coll *et al.,* 2010). Over the last 50 years, however, many non-indigenous species have been observed in the Mediterranean Sea. Katsanevakis *et al.* (2013) have reported that the highest numbers of marine non-indigenous species recorded in the Mediterranean Sea are found in the coastal waters of Egypt, France, Greece, Israel, Italy and Turkey.

**PLATE 9**
*Caulerpa taxifolia* var. *distichophylla*



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As mentioned, ship transportation, with both ballast tanks and hull fouling, represents one of the most important introduction vectors for non-indigenous species. Meanwhile, after the opening of the Suez Canal, many other species were able to pass through to the eastern Mediterranean from the Red Sea. Some commercial species have also been intentionally introduced, like the Pacific cupped oyster, *Magallana gigas,* or the clam *Ruditapes philippinarum*, becoming established in the Mediterranean. Similarly, the accidentally introduced alga *Caulerpa taxifolia* (Plate 9) has also spread around the Mediterranean. All these non-indigenous species have affected the marine biodiversity and fisheries of this unique area.

The present Mediterranean fauna and flora represent a mixture of Atlantic, Mediterranean and Red Sea/Indo-Pacific (due to the Suez Canal) biotic components, with the eastern basin functioning as a Lessepsian province (Moraitou-Apostolopoulou, 2013). Coll *et al.* (2010) consider the invasion of non-indigenous species to be a crucial factor in ongoing changes in the region's biodiversity, mainly in the eastern basin, and believe it can rapidly spread northwards and westwards due to the warming of the Mediterranean Sea. The same authors note that the majority of non-indigenous species are of Indo-Pacific origin (41 percent), followed by those of Indian Ocean (16 percent) and of Red Sea (12 percent) origin; some species display a pantropical or circumtropical distribution (19 percent).

On the scale of the Mediterranean, Zenetos *et al.* (2012) have reported that the number of non-indigenous species continues to increase rapidly, by 2–3 species per year for macrophytes, molluscs and polychaetes, by 3–4 species per year for crustaceans, and by 6 species per year for fish. The same study revealed that the dominant group among non-indigenous species is molluscs (with 215 species), followed by crustaceans (159 species) and polychaetes (132 species).

Concerning non-indigenous crustacean species in the Mediterranean Sea, an increasing number have been observed by various experts. Galil, Froglia and Noel (2002) reported 59 decapod species and one stomatopod, i.e. a total of 60 non-indigenous species from the Mediterranean Sea, of which 37 species have established populations.

Furthermore, Zenetos *et al.* (2007) updated the number of non-indigenous species in Greek waters from 102 to 110. Of the eight new recorded species, five are zoobenthic, two are zooplanktonic and one is a teleost fish. The same authors also listed 745 non-indigenous species as present in 2005, which number had increased to 903 only a few years later (Zenetos *et al.,* 2008).

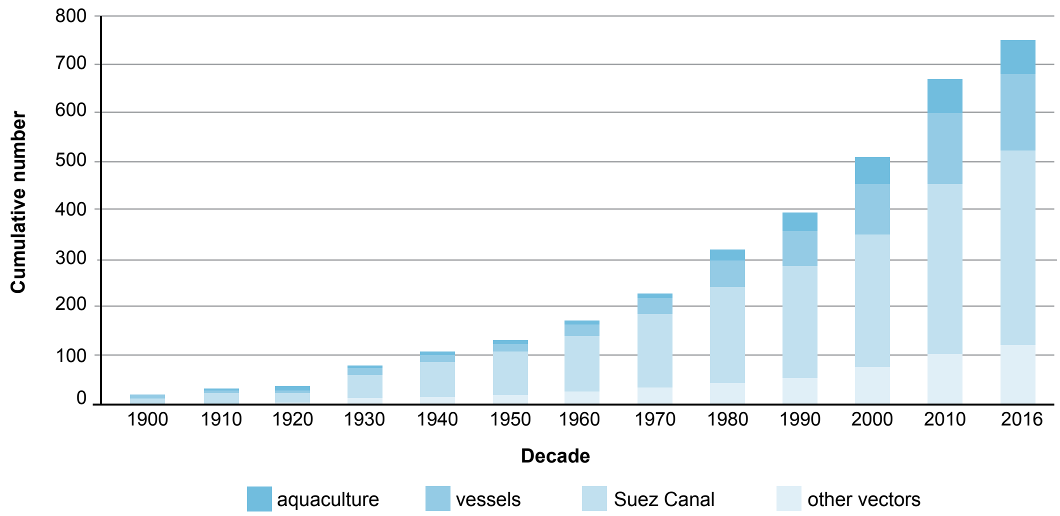
## 3.2 Main vectors for non-indigenous species into the Mediterranean Sea

The main vectors for non-indigenous species introduction into the Mediterranean Sea can be described as follows:

- The Suez Canal: one of the major vectors for species of Indo-Pacific origin or Lessepsian migrants;
- Shipping: ballast water, tank sediments and hull fouling;
- Connecting straits: the Kerch and the Turkish Straits for Black Sea species and the Strait of Gibraltar for Atlantic species. These straits also play an important role in the introduction, respectively, of non-indigenous species of Black Sea origin, such as *Liza haematocheilus*, *Mnemiopsis leidyi* and *Rapana venosa*, and of non-indigenous species of Atlantic origin,;
- Intentional or unintentional introduction by humans: this kind of introduction generally occurs for aquaculture purposes; and
- Minor vectors: aquariums, fish baits, recreational boats, among others.

According to the latest revision of Galil, Marchini and Occhipinti-Ambrogi (2018), the main vector for non-indigenous species introduction into the Mediterranean is the Suez Canal (63 percent), followed by shipping (commercial and recreational) and mariculture. The relative roles of the different introductory pathways shift through space and time (Galil, Marchini and Occhipinti-Ambrogi, 2018), and different estimates have been published in recent years. According to Streftaris, Zenetos and Papathanassiou (2005), although shipping has contributed more than aquaculture as a vector, it appears to have had a less significant role (20 percent) in the Mediterranean than the Suez Canal has had (52 percent). Galil (2009) noted that the majority of non-indigenous species in the eastern Mediterranean entered by crossing through the Suez Canal (68 percent of the total, 14 percent vessel-transported, 2 percent mariculture), whereas mariculture (42 percent), vessels (38 percent) or both (5 percent) represent the main means of introduction into the western Mediterranean. The number of species was estimated at the scale of thousands by Por (2009). Subsequently, Zenetos *et al.* (2012) reported that, of the total 986 non-indigenous species in the Mediterranean, 775 occurred in the eastern Mediterranean, 249 in the central Mediterranean, 190 in the Adriatic Sea and 308 in the western Mediterranean. Later, Galil *et al.* (2015) reported over 700 species from the Mediterranean Sea (Figure 15).

**FIGURE 15**
Cumulative number of non-indigenous species introduced per decade according to major vectors




Source: modified from Galil *et al.*, 2018.

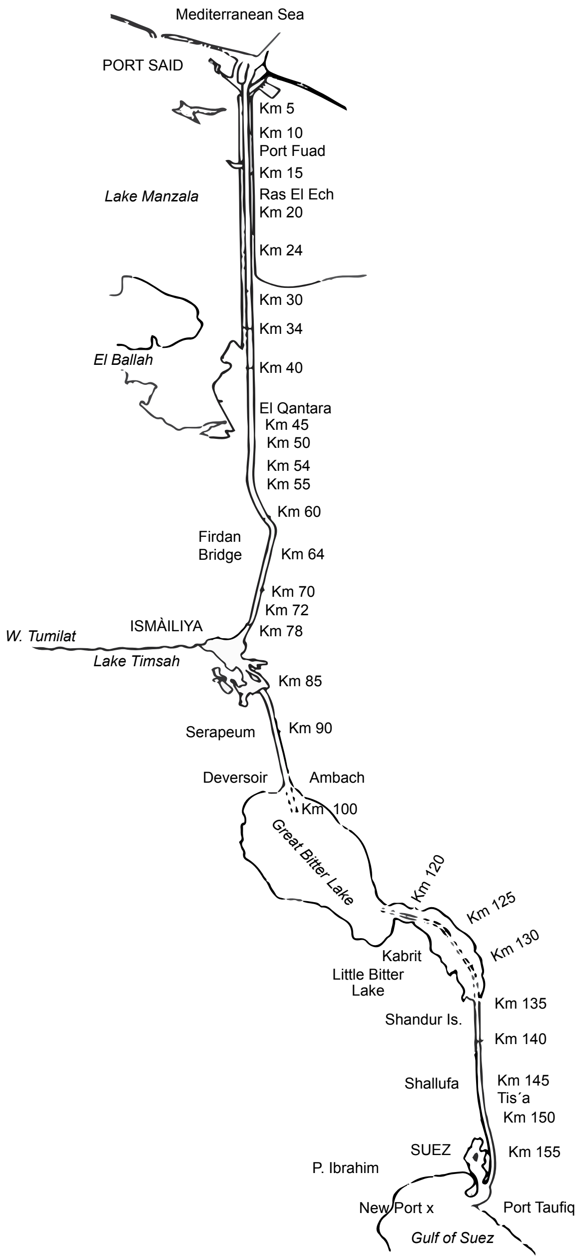
### 3.2.1 The Suez Canal

The Suez Canal was opened in 1869 in order to shorten the trade route between the Mediterranean and the Indian Ocean, creating a salt-water passage between the Mediterranean and the Red Sea (Figure 16). The French engineer in charge of the construction, Ferdinand de Lesseps (1805–1894), gives his name to those species expanding their ranges by crossing the Suez Canal, known as Lessepsian species. As the sea level of the Red Sea lies higher than that of the eastern Mediterranean, the canal serves as a tidal strait pouring Red Sea water into the Mediterranean. The Bitter Lakes, hypersaline natural lakes constituting a section of the canal, blocked the migration of Red Sea species into the Mediterranean for many decades. However, as the salinity of the lakes gradually equalized with that of the Red Sea, this migration barrier was removed, and plants and animals from the Red Sea began to colonize the eastern Mediterranean. The Red Sea is generally saltier and more nutrient-poor than the Atlantic, so Red Sea species boast advantages over Atlantic species in the less salty and more nutrient-rich Levant Sea.

Since its construction, the Suez Canal has been deepened and widened several times, with the last enlargement in 2015. Currently, the navigational depth is 24 m (Table 2).

The expansion of the Suez Canal has increased its prominence as a vector for the introduction of Red Sea species into the Mediterranean. Particularly responsible were the gradual reduction in salinity of the Bitter Lakes and the

**FIGURE 16**
Historical map of the Suez Canal




Source: Por, 1978.

**TABLE 2 – Dimensions of the Suez Canal through time**

| | 1869 | 1956 | 1962 | 1980 | 1994 | 1996 | 2001 | 2010 | 2015 |
|---|---|---|---|---|---|---|---|---|---|
| Overall length (km) | 164 | 175 | 175 | 189.80 | 189.80 | 189.80 | 191.80 | 193.30 | 193.30 |
| Width at 11 m depth (m) | – | 60 | 89 | 160–175 | 170–190 | 180–200 | 195–215 | 205–225 | 205–225 |
| Water depth max. (m) | 8 | 14 | 15.5 | 19.5 | 20.5 | 21 | 22.5 | 24 | 24 |
| Cross-sectional area (m$^2$) | 304 | 1 200 | 1 800 | 3 250–3 600 | 3 600–4 000 | 3 850–4 300 | 4 350–4 800 | 4 800–5 200 | 4 800–5 200 |

Source: Halim, 1990; SCA, 2015.

construction of the Aswan High Dam, which reduced the inflow of freshwater and nutrient-rich silt from the Nile into the eastern Mediterranean. Due to the removal of salinity barriers, many Lessepsian species have crossed via the Suez Canal and established permanent populations in the eastern Mediterranean with various ecological and economic consequences. On the other hand, only few have moved in the opposite direction (Ben-Tuvia, 1966; Avşar, 1999).

High water temperature and turbidity in the Canal have also been considered as possible barriers in the past. However, according to Galil *et al.* (2015, 2017), the recent expansion of the Canal could have direct consequences in accelerating the transfer of non-indigenous species into the Mediterranean.

Over the last few decades, these migration patterns have been monitored by various regional and international organizations, including the GFCM, the International Commission for the Scientific Exploration of the Mediterranean Sea (CIESM), the Regional Activity Centre/Special Protected Areas (RAC/SPA), the World Conservation Union, the European Union, the European Environment Agency (EEA) and others. Most of the research has been carried out on fish species, as fish offer higher economic value than other species.

### 3.2.2 Shipping

Many important shipping routes (Figure 3) traverse the Mediterranean, allowing non-indigenous species to be easily transported by ship ballast water or sediment, sessile (fouling) and vagile (clinging) on ship hulls, or even drilling platforms. As Bianchi and Morri (2003) have suggested, since most of the vessel-transported, non-indigenous species in the Mediterranean are thermophilic, having originated from tropical waters, an increase in average temperatures in Mediterranean waters facilitates this process. According to Galil, Occhipinti-Ambrogi and Gollasch (2008), the highest numbers of vessel-transported, non-indigenous species have been recorded from Italian (87 vessels), Turkish (65 vessels) and Israeli (54 vessels) waters.

Meanwhile, intra-Mediterranean, west–east (Atlantic Ocean–Mediterranean) and north–south (Black Sea–Mediterranean via the Turkish Straits, and Indian Ocean–Mediterranean via the Suez Canal) marine traffic is continually increasing due to the expansion of global trade.

Ballast water appears to play a less important role than fouling as a vector for the introduction of macrophytes into the Mediterranean (Boudouresque and Verlaque, 2002). For example, the transfer of oysters by fouling is probably responsible for the introduction of 44 macrophyte species into the Thau Lagoon in southern France (Ribera, 2002; Verlaque, 2001). In the Mediterranean, it has been noted that 13 percent of non-indigenous marine plant species were carried on ships' hulls and 3 percent arrived due to the deballasting process (Siguan, 2002). Wyatt and Carlton (2002) reported that seven phytoplankton species were possibly introduced to European and Mediterranean coastal waters with ballast water or oysters or via multiple vectors. Similarly, Por (1978) and Kovalev (2006) noted that several non-indigenous zooplankton species entered the Mediterranean Sea via the Suez Canal and Strait of Gibraltar. More comprehensively, Verlaque, Boudouresque and Mineur (2007) recorded a total of 110 non-indigenous marine macrophyte species in the CIESM Atlas for exotic species, citing the vectors already noted. Gollasch *et al.* (2003) indicated that in international shipping, ballast water has been identified as a major vector for the unintentional introduction of non-indigenous fauna and flora; for sampling ballast water, a wide range of techniques may be needed.

Concerning other taxonomic groups, Izquierdo-Muñoz, Ramos-Esplá and Díaz-Valdéz (2009) reported that ten non-indigenous tunicates have become established in some sections of the Mediterranean since 1958, the majority arriving through the Strait of Gibraltar and the Suez Canal or introduced by shellfish culture. Likewise, the majority of non-indigenous ascidians have probably been transported in the adult stage while attached to hulls.

Apart from normal shipping patterns, these problems are further aggravated, as Ulman *et al.* (2017) reported, by the Mediterranean Sea being home to over two thirds of the world's charter boat traffic and hosting an estimated 1.5 million recreational boats, with marinas providing major hubs for the stepping-stone transfer of non-indigenous species.

The International Maritime Organization (IMO) is running some important initiatives against non-indigenous species, such as Resolution MEPC.207(62), adopted on 15 July 2011, which provides guidelines for the control and management of ships' biofouling to minimize the transfer of invasive aquatic species. Furthermore, it is estimated that more than 10 000 marine species are transported daily across the oceans in the ballast water of cargo ships and introduced into non-native environments (EMSA, 2021). The IMO Ballast Water Management Convention works against the transfer of non-indigenous species (IMO, 2017), and the IMO has also executed the GloBallast Partnerships Programme (2008–2017), implemented by the United Nations Development Programme (UNDP) and funded by the Global Environmental Facility (GEF), to maintain the global momentum in tackling the ballast water problem and to catalyse innovative global partnerships to develop solutions (GloBallast, 2017).

While regular cleaning of hulls may reduce transfers of non-indigenous species, clean hulls immediately provide an area for new organisms to attach to and economic costs must be considered on both sides. Nevertheless, Seebens, Gastner and Blasius (2013) estimated that treating ballast water before discharge can reduce the risk of invasion by up to 82 percent.

### 3.2.3 Straits

The straits used for navigation in the Mediterranean and the Black Sea are also biological corridors (Figure 3). For example, *Liza haematocheilus*, a fish species intentionally introduced to the Azov Sea, passed subsequently through the Black Sea and the Turkish Straits (Istanbul and Çanakkale), and established itself successfully in the Mediterranean Sea (Kaya, Mater and Korkut, 1998). Similarly, as already noted, the sea-snail *Rapana venosa* and the comb jelly *Mnemiopsis leidyi*, first recorded in the Black Sea, have successfully migrated through the Turkish Straits to the Mediterranean.

At the western end of the Mediterranean, the Strait of Gibraltar presents another biological corridor for non-indigenous species arriving from the Atlantic Ocean. Quignard and Tomasini (2000) reported several fish species of Atlantic origin as neocolonizers. The Atlantic–Mediterranean faunistic connection has had a longer-lasting impact on Mediterranean biogeography than the Mediterranean–Red Sea connection, which dates back only to 1869 and the construction of the Suez Canal. Following the Messinian salinity crisis of 5 million years ago, the Zanclean flood reconnected the Atlantic to the Mediterranean via the Strait of Gibraltar, allowing Atlantic fauna to be established in the Mediterranean Sea. This process continues to this day, and Quignard and Tomasini (2000) have identified 26 fish species of Atlantic origin as recent colonizers. These species, migrating from west to east, include *Sphoeroides pachygaster* and *Sphyraena viridensis,* and are currently found off the coasts of Israel, Lebanon and Turkey, having extended their distribution range to the eastern Mediterranean as well. In fact, Lasram-Rais and Mouillot (2009) have reported that 62 Atlantic species have entered through the Strait of Gibraltar. Indeed, the first report of such Atlantic fish entering the Mediterranean was made in 1810 by Risso, when *Entelurus aequoreus* and *Pristis pectinata* were observed.

## 3.3 Lessepsian migration and species

The colonization of the Mediterranean by Red Sea marine species has been routinely reviewed and its records updated over the last nearly 70 years, i.e. since Koswigg (1953) and Ben-Tuvia (1953). Ben-Tuvia (1978) listed 36 species of Red Sea fish immigrants, which, interestingly, constituted at the time 12 percent of the fish population in the Levant Sea and 7 percent for the Mediterranean as a whole, but only 1 percent of the population in the central part. None of the Red Sea immigrants had reached the western basin. According to Por (1978, 1990), as far as the fauna were concerned, Lessepsian species represented about 4 percent of Mediterranean specific diversity and 10 percent of the Levant Sea's. However, as already noted, another major issue is presented by Lessepsian species' ability to migrate not only to the Levant Sea, but also beyond the Strait of Sicily to the western Mediterranean Sea or even up to the Black Sea.

Golani (1996) has reported that 59 Lessepsian fish species make up 14 percent of the ichthyofauna of the Levant Sea (i.e. east of the line connecting Antalya to Port Said). These species come from 42 families, 15 of which were not present in the Mediterranean prior to Lessepsian migration. Golani (2002) has also pointed out that 38 Lessepsian fish species have established sustainable populations in the eastern Mediterranean, with significant impacts on the local ecosystems.

Further north, in the Adriatic Sea, of the 407 fish species recorded by Dulčić, Lipej and Grbec (2002), six species were identified as Lessepsian migrants. Even more up to date, Dulčić, Dragičević and Tutman (2017) reported a total of 14 Lessepsian migrants from the Adriatic Sea, including the fish species *Fistularia commersonii*, *Siganus luridus* and *Lagocephalus sceleratus*, which appear to be successful colonizers, especially in the southern part of the sea, though the impact on native communities is not yet known. In 2009, CIESM listed 116 non-indigenous fish species across the entire Mediterranean basin, although this number does not distinguish between Lessepsian migrants and those originating from the Atlantic or from other areas.

Meanwhile, along the 7 375 km-long Italian coast, Occhipinti-Ambrogi *et al.* (2011) have recorded a total of 165 marine and brackish non-indigenous species, of which 25 species are likely to have been introduced via the Suez Canal. These species have been mainly observed in the southern Adriatic and Ionian seas, or in the Straits of Sicily and Messina, the areas most affected by Lessepsian introductions. Along the rocky coastlines of Albania and Montenegro, Katsanevakis *et al.* (2011) have modelled the distribution of five non-indigenous marine species.

In the CIESM Atlas of Exotic Species in the Mediterranean, Zenetos *et al*. (2003) reported 137 Lessepsian mollusc species. Research interest in non-indigenous mollusc species has grown markedly (Çevik, Öztürk and Buzzuro, 2001) in places like Iskenderun Bay, off Turkey, due to the importance of commercial species, such as *Magallana gigas* and *Saccostrea glomerata*. Çevik *et al*. (2007) also noted that in Iskenderun Bay, 67 of 181 mollusc species were non-indigenous, representing 37 percent of the total mollusc fauna.

In 2003–2004 in Greek waters, Daskos and Zenetos (2007) observed two non-indigenous opisthobranch species: *Goniobranchus annulatus*, which may have been introduced by shipping; and *Bursatella leachii*, which has long been considered to have immigrated via the Suez Canal.

In Croatian waters, Doğan and Nerlovic (2008) have recorded sightings of *Pinctada radiata*, a bivalve species of Red Sea origin, clearly demonstrating that Lessepsian molluscs have dispersed as far as the Adriatic Sea. Moreover, Sraieb, Sghaier and Cheikhrouha (2009) recorded the two already mentioned non-indigenous molluscs *B. leachii* and *P. radiata* from the Ghar El Melh lagoon in Tunisia.

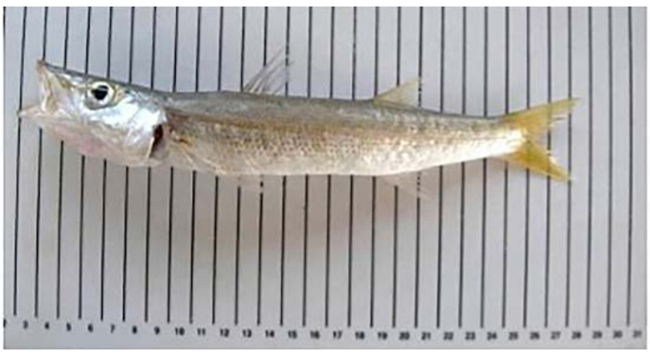
Çınar *et al.* (2005) additionally recorded 45 non-indigenous crustacean species along the Mediterranean Turkish coast, while Yokes and Galil (2006a), and Pancucci-Papadopoulou and Naletaki (2007) have since added to the list of observed non-indigenous crustacean species from the eastern Mediterranean Sea.

### 3.3.1 Lessepsian fish species

A number of important studies have been carried out on Lessepsian fish migrant species (e.g. Koswigg, 1953; Ben-Tuvia, 1953, 1966, 1978; Demetropoulos and Neocleous, 1969). The first Lessepsian fish species to be identified, the Red Sea hardyhead silverside (*Atherinomorus forskalii*), was recorded in 1902, 33 years after the opening of the Suez Canal (Ben-Tuvia, 1985). Golani *et al.* (2002) reported 90 exotic fish species (including fishes of Atlantic origin) from 56 different families, among a total of 650 fish species in the Mediterranean Sea. Unfortunately, not enough data are available from before the opening of the Suez Canal to draw any conclusive comparisons. One of the Lessepsian fish species, the dusky spinefoot (*Siganus luridus*), has extended its distribution up to the French coast from the Strait of Sicily (Daniel *et al.*, 2009). Other studies have also been conducted in the Tyrrhenian Sea (Psomadakis *et al.*, 2009), along the Tunisian coast (Ktari and Ktari, 1974; Ben Soussi *et al.*, 2004), in the Ligurian Sea (Garibaldi and Orsi-Relini, 2008), in the Adriatic Sea (Dulčić, Scordella and Guidetti, 2008; Dulčić *et al.*, 2004), along Montenegro's Adriatic coast (Joksimovic, Dragicević and Dulčić, 2009) and along the Algerian coast (Kara and Oudjane, 2009). Azzurro *et al.* (2017a, 2017b) have also reported a few Lessepsian species from French waters and the northern Adriatic Sea.

In Tunisia, Bradai *et al.* (2004a, 2004b) identified 15 non-indigenous species, among which eight species were Lessepsian and seven of Atlantic origin. In particular, Ben Soussi *et al.* (2004) reported the presence of the bluespotted cornetfish (*Fistularia commersonii*), while Zouari-Ktari *et al.* (2007) noted that the spawning of the yellowstripe barracuda (*Sphyraena chrysotaenia*) (Plate 10) took place between May and November in the Gulf of Gabès, where *Sphaeroides pachygaster* was also observed (Bradai, Gorbel and Bouain, 1993). Furthermore, Zouari-Ktari (2008) discovered that among the total of 22 non-indigenous species from Tunisia, 14 were Lessepsian migrants and 8 were Atlantic species.

**PLATE 10**
*Sphyraena chrysotaenia*



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It is clear that Lessepsian fish migrants can be found beyond the Strait of Sicily. Golani (1998) reported only four Lessepisan fish species between Tunisia and Italy at the time, though this number has increased to 14 in Tunisia (Lasram-Rais *et al.*, 2008). More recently, Ounifi-Ben Amor *et al.* (2016) have identified 136 non-indigenous species in Tunisian waters, with Crustacea the dominant taxa (24 percent), followed by Mollusca (23 percent), fishes (19 percent) and Annelida (13 percent). This same study highlighted the two main areas of origin for the non-indigenous species in Tunisian waters – Red Sea/Indo-Pacific and Atlantic – with slightly more species coming from the former (61.8 percent).

The Strait of Sicily represents a major biogeographic border between the eastern and western Mediterranean Sea, though the number of Atlantic-origin and Lessepsian species, especially fishes, increasingly observed to be crossing it suggests that the effectiveness of this boundary has decreased (Azzurro *et al.*, 2014). In the eastern Mediterranean Sea, Bariche, Harmelin-Vivien and Quingnard (2003) described the spawning and reproduction cycles of two Lessepsian siganid fish species, while Golani *et al.* (2007) recorded 108 non-indigenous fish species from 61 families, 37 of which were new to the Mediterranean ichthyofauna, representing a species increase of 20 percent over only four years.

In Cypriot waters, Tzomos *et al.* (2010) reported 25 Lessepsian fish species. More recently, Iglésias and Frotté (2015) noted that the number of Lessepsian fishes had increased to 35. The authors of this study consider the rapid increase of non-indigenous fish species in the waters surrounding Cyprus to coincide with the accelerating tropicalisation process observed elsewhere in the Mediterranean over the last decades.

The dispersal of Lessepsian fish in the Mediterranean Sea depends on several factors such as cyclonic Mediterranean shore currents directed toward the Levant Sea and similar temperature conditions. Most successful species are eurythermal and euryhaline species able to adapt to other ecological conditions (Mavruk and Avşar, 2008). Gücü and Gücü (2002) observed that low native species diversity affected the rate and extent of immigrant colonization. Furthermore, the absence of *Posidonia oceanica* meadows was identified as another important factor in the success of Lessepsian invasion. Indeed, *P. oceanica,* an endemic seagrass and the key species of the Mediterranean coastal ecosystem, was credited with defending the Levant basin's ecological integrity against invasion.

Marttin *et al.* (2006) have identified seven fish and three crustacean species that have managed to establish themselves and, according to the CIESM *Atlas of Exotic Species in the Mediterranean* (Verlaque *et al.*, 2015), become commercially important in the Levantine fisheries.

Bariche (2012) reported that some species are appreciated locally and command high market prices – e.g. fishes like the goldband goatfish (*Upeneus moluccensis*) (Plate 11), yellowstripe barracuda (*Sphyraena chrysotaenia*) and narrow-barred Spanish mackerel (*Scomberomorus commerson*) (Plate 12); and crustaceans

**PLATE 11**
*Upeneus moluccensis*

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like the kuruma prawn (*Penaeus japonicus*) (Plate 8) and speckled shrimp (*Metapenaeus monoceros*). Others are either sold separately – e.g. fishes like the marbled spinefoot (*Siganus rivulatus*) (Plate 13), dusky spinefoot (*Siganus luridus*) (Plate 14) and silver sillago (*Sillago sihama*); bivalves like the rayed pearl-oyster (*Pinctada radiata*) and spiny oyster (*Spondylus spinosus*); and the blue swimming crab (*Portunus segnis*) – or sold mixed with native Mediterranean species – e.g. fishes like the spotback herring (*Herklotsichthys punctatus*), red-eye round herring (*Etrumeus golanii*) (Plate 15) and Por's goatfish (*Upeneus pori*) (Plate 16) – at fairly reasonable prices. In the meantime, forty-two Lessepsian fish species have been documented in Egyptian waters (Halim and Rizkalla, 2011).

**PLATE 12**
*Scomberomorus commerson*

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**PLATE 13**
*Siganus rivulatus*

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**PLATE 14**
*Siganus luridus*

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The vertical distribution of Lessepsian fish species in Turkish waters has also been evaluated by several authors. Bilecenoğlu and Taşkavak (2002) have reported that a few specimens of *Upeneus moluccensis* (Plate 11) were found at depths of 180–190 m and that this species seemed to be very adaptive to the low temperatures prevailing at such depths throughout the eastern Levant Sea. Similarly, Golani (1996) found some Lessepsian species at a depth of 200 m off the coast of Ashdod in Israel. Edelist *et al.* (2012) compared historical (1990–1994) and (2008–2011) trawl survey data in the Levant Sea from the continental shelf with data from the upper slope in order to evaluate the relative abundance and biomass of Indo-Pacific fishes and their impacts on biodiversity, discovering that non-indigenous species constituted over 50 percent of the fish biomass.

In Greece, Corsini-Foka (2010) has pointed out that among the 31 Indo-Pacific fishes reported from the southeastern Aegean Sea, 48 percent have spread to the central and south Aegean Sea, 45 percent having reached the central Aegean Sea and 35 percent the south Aegean Sea (Crete). Corsini-Foka (2010) also observed, however, that major impediments obstruct easy colonization of the north Aegean Sea, particularly along its western coasts, where only two recent invaders, the bluespotted cornetfish (*Fistularia commersonii*) and silver-cheeked toadfish (*Lagocephalus sceleratus*), have been observed during summer. The keeled mullet (*Liza carinata*) and *L. sceleratus* have been reported from along the northeastern Aegean coast, which area likely presents more suitable environmental conditions for the establishment of these two tropical species.

Azzurro *et al.* (2017a) reported that the dusky spinefoot (*Siganus luridus*) has spread through much of the eastern Mediterranean since its introduction in 1920. The abundance of this invader around the island of Linosa (Strait of Sicily), where this species was first recorded in 2003, has been monitored; an average abundance of 0.36 individuals per 250 square metres was recorded across the 0–30 m depth range.

### 3.3.2 Evaluation of Lessepsian fish catch

Many Lessepsian fish species are commercially fished in the eastern Mediterranean Sea. However, insufficient accurate data are available for most of the species from the Mediterranean countries. Bariche *et al.* (2015) have reported that more than 90 marine fish species in the Mediterranean are Lessepsian migrants.

In Egyptian coastal waters, El-Sayad (1992) has recorded the presence of 32 Lessepsian migrants, though few data are available from Egyptian fisheries. Of economic importance are rabbitfishes (*Siganus rivulatus* and *Siganus luridus*) (Hamza, Mohammed and El Serafy, 2000), red-eye round herring (*Etrumeus golanii*) (Plate 15) and narrow-barred Spanish mackerel (*Scomberomorus commerson*) (Plate 12) (Di Natale *et al.* (2009).

**PLATE 15**
*Etrumeus golanii*

©Bayram Öztürk

**PLATE 16**
*Upeneus pori*

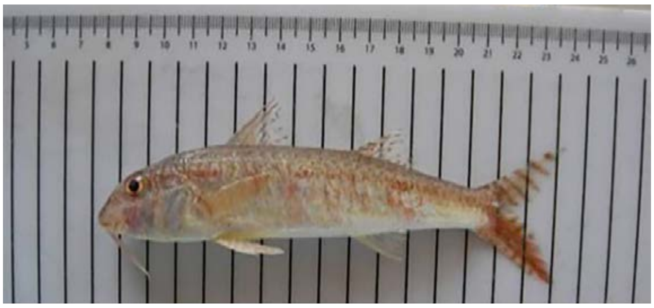

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**PLATE 17**
*Sargocentron rubrum*

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**PLATE 18**
*Saurida lessepsianus*

©Elif Özgür-Özbek

In Greek waters, Papaconstantinou (1990) reported that at least 11 species have reached the Aegean islands and are being fished commercially: redcoat (*Sargocentron rubrum*) (Plate 17), *Saurida lessepsianus* (Plate 18), *Pempheris rhomboidea* (Plate 19), black-barred halfbeak (*Hemiramphus far*) (Plate 20), *Siganus rivulatus, Siganus luridus, Lagocephalus spadiceus, Stephanolepis diaspros, Upeneus moluccensis, Equulites klunzingeri* and *Parexocoetus mento*. Corsini-Foka and Kalogirou (2008) and Corsini-Foka (2010) have also reported that *Scomberomorus commerson*, a fish with commercial potential, was found for the first time in waters around Rhodes, and that recent colonizers, such as the fish *Upeneus pori* (Plate 16) and the cephalopod *Sepioteuthis lessoniana* (Plate 21), were showing a rapid increase.

**PLATE 19**
*Pempheris rhomboidea*

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**PLATE 20**
*Hemiramphus far*

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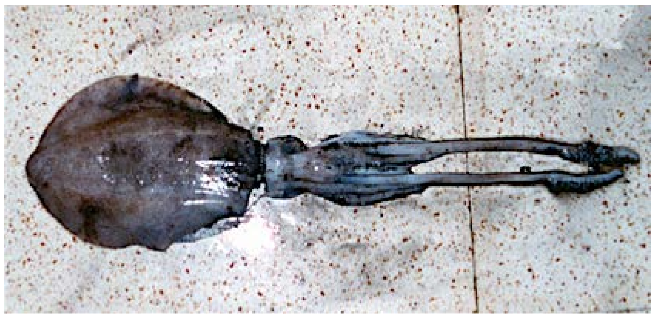
**PLATE 21**
*Sepioteuthis lessoniana*



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Comprehensive catch records have been kept by Israeli fisheries, with Lessepsian species estimated to comprise approximately a third of their total catch since 1954 (Galil, 1993). Moreover, nearly half of the trawl catch along the Israeli coast consists of Lessepsian fish species (Golani and Ben-Tuvia, 1995). The lizard fish, *Saurida lessepsianus* (Plate 18), was caught in Israeli waters for the first time in 1952; only three years later, 266 tonnes were landed by local trawlers, constituting almost 20 percent of the total trawler catch (Ben-Yami and Glaser, 1974). However, the most abundant fish in inshore fisheries (trammel-netting and hook-and-line) are the rabbitfish, *Siganus rivulatus* (Plate 13) and *S. luridus* (Plate 14), the yellowstripe barracuda (*Sphyraena chrysotaenia*), and the shrimp scad (*Alepes djedaba*). The above species, together with *Sillago sihama* and *Scomberomorus commerson*, are common in purse-seine landings.

The annual catch of lizardfish, which reached 400 tonnes in 1960 soon after its arrival, declined to 100 tonnes in the mid-1960s, but has been increasing since, with catch fluctuations likely correlated to the catch per unit of effort (CPUE). Catch statistics for mullids (mullets) do not distinguish between the native species *Mullus barbatus* and *M. surmuletus* and the non-indigenous *Upeneus moluccensis* and *U. pori*. However, a study of the frequency of the non-indigenous mullets in trawl catches conducted in the mid-1980s showed that they comprised 87 percent of the mullid catch off the coast of Israel at depths of 20 m and 50 percent at depths of 55 m, whereas the native mullids were more abundant in deeper waters (Golani and Ben-Tuvia, 1995). Indeed, the percentage of Lessepsian mullids within the total mullid catch has increased steadily, from 30 percent in 1980, to 42 percent in 1984, to 47 percent in 1989 (Golani and Ben-Tuvia, 1995).

Similarly, catch statistics for sphyraenids do not separate the invasive yellowstripe barracuda (*Sphyraena chrysotaenia*) from the native Mediterranean species, *S. sphyraena* and *S. viridensis*. However, an examination of the landed catch showed that the Lessepsian barracuda outnumbered the native sphyraenids in inshore trawl and purse-seine catches (Grofit, 1987).

In southern Lebanon, Lessepsian fish species constituted 37 percent of the weight of the total landings in small-scale fisheries (Carpentieri *et al*., 2008).

In Libyan waters, Shakman and Kinzelbach (2007) observed six Lessepsian fish species and described more than 37 percent of all present Lessepsian species as becoming commercially valuable, especially rabbitfish (*Siganus* spp.). These species are now found regularly in Libyan catches. Ten species (62.5 percent), however, are characterized as having no commercial value.

In Maltese waters, Sciberras and Schembri (2007) reported 13 Lessepsian fish species, though until recently, no non-indigenous species had established itself in the wild or been commercially exploited.

In Syrian waters, Saad (2005) observed 37 Lessepsian fish species, which represents 16.5 percent of all the bony fish species recorded. However, no commercial catch data have been reported for non-indigenous fish species. In Latakia harbour, some non-indigenous fish species, such as *U. moluccensis* and *U. pori,* were sold in 2007 and 2008 (unpublished data, B. Öztürk).

In Tunisian waters, Zouari-Ktari, Bradaï and Bouain (2008) recorded ten Lessepsian fish species with commercial importance. Indeed, a number of species have been observed occasionally in the markets, such as *Stephanolepis diaspros*, *Siganus luridus* and *Scomberomorus commerson* (in large quantities during summer 2009, personal communication). Additionally, the Senegalese sole (*Solea senegalensis*) can be purchased in the northern markets and represents 25 percent of Soleidae caught in Bizerte (Lake Ichkeul) (Chaouachi and Ben Hassine, 1998). However, there are no statistical data available.

According to Gücü and Bingel (1994), specific catch statistics in Turkish waters are lacking to evaluate the contribution of Red Sea species to total catches. However, these species are rather important in the complete demersal fish biomass, comprising 62 percent in the Gulf of Iskenderun, 34 percent in Mersin Bay and 27 percent in the coastal strip between Incekum and Anamur. Çiçek and Avşar (2003) reported that among 90 fish species collected by trawl sampling during 2002–2003 in the northeastern Mediterranean Sea, 17 species were Lessepsian migrants. They found that the CPUE for Lessepsian fish ranged from 3.39 kilograms per hour in November 2001 to 11.73 kilograms per hour in September 2002, and calculated the mean value as 5.28 ± 3.32 kilograms per hour; most of the Lessepsian biomass was caught close to shore, i.e. at a depth range of 0–20 m, and the number of Lessepsian fish species constituted 18.9 percent of the total number of fish species, while 26.7 percent of the total biomass was attributed to Lessepsian fish. Among these Lessepsian fishes, the most abundant was *S. lessepsianus* at 47.2 percent, followed by *Upeneus pori* (29.9 percent), and *Leiognathus kluzengeri* (13.3 percent). İşmen (2002) also noted that, in the eastern Mediterranean, 98 percent of the total biomass of *U. pori* was trawled in depths shallower than 50 m and that its market value had increased in recent years.

**PLATE 22**
*Pomadasys stridens*

©Bayram Öztürk

Lessepsian fishes include both demersal and pelagic species in the eastern Mediterranean Sea, with the latter recently experiencing increased importance. For example, Yilmaz and Hossucu (2003) reported that 360 tonnes of the round herring (*Etrumeus golanii*), a pelagic species, was caught in Antalya Bay in 2002, according to unofficial fish market records in Turkey.

**PLATE 23**
*Nemipterus randalli*

©Bayram Öztürk

Özgür-Özbek *et al.* (2010) reported on the abundance and biomass of 18 non-indigenous fish species caught during a bottom trawl survey carried out in the Gulf of Antalya in the summer of 2009. Among the 76 teleost species identified, non-indigenous species constituted 11.6 percent of the total number of individuals and 12.1 percent of the biomass of the catch. More recently, Akyol and Ünal (2016) reported the occurrence of a Lessepsian fish species, the striped piggy (*Pomadasys stridens*) (Plate 22), which has been caught commercially in large quantities in the Iskenderun Bay area since 2016.

In addition, *Siganus luridus, S. rivulatus* and Randall's threadfin bream (*Nemipterus randalli*) (Plate 23) boast commercial value in the eastern Mediterranean Sea, mainly in the İskenderun Bay region.

Apart from in the above-mentioned countries, some Lessepsian fish species are found in the waters of Croatia, Italy and Montenegro, but currently they are caught only as bycatch and have no market value.

### 3.3.3 Evaluation of Lessepsian crustacean catch

According to Doğan *et al.* (2007), the following crustacean species have commercial importance in the Turkish Aegean Sea and the entire Mediterranean Sea: *Penaeus merguiensis, Penaeus japonicus, Penaeus hathor* (Plate 24), *Metapenaeus monoceros, Metapenaeus stebbingi, Penaeus semisulcatus* (Plate 25), *Callinectes sapidus* and *Portunus segnis* (Plate 26). Geldiay and Kocataş (1972) reported that in Iskenderun Bay, *Penaeus kerathurus* was previously caught in substantial numbers by fishers but it had since been replaced by *P. japonicus.* Similarly, the blue crab,

**PLATE 24**
*Penaeus hathor*

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*C. sapidus*, has become common in the waters of Egypt, Greece, Israel, Lebanon, the Syrian Arab Republic and Turkey. Off the southeastern coast of Turkey, the Lessepsian shrimps *P. japonicus* and *P. semisulcatus* are the most valuable commercial species in landings (Duruer *et al.*, 2008). Furthermore, Lessepsian penaeid shrimps make up most of the shrimp catch along the southeastern Levantine coast, with *P. japonicus*, *M. monoceros* and *P. semisulcatus* especially highly prized (Galil, 2007b). Among crustaceans, *P. japonicus* is commercially exploited in Lebanon as well (Carpentieri *et al.*, 2008), while the mantis shrimp, *Erugosquilla massavensis*, is economically important in Turkey. In addition, the blue crab, *C. sapidus*, boasts a significant market value in Turkey: 22 tonnes were caught in 2007, 17 tonnes in 2008, and 87 tonnes in 2012 between Antalya and Iskenderun Gulf (B. Öztürk, unpublished data). Hasan, Zeini and Noel (2008) also reported 15 Lessepsian crustacean species of commercial importance in the Syrian Arab Republic.

**PLATE 25**
*Penaeus semisulcatus*

©Bayram Öztürk

**PLATE 26**
*Portunus segnis*

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Chaouachi and Ben Hassine (1998) have observed that *M. monoceros* has experienced a rapid expansion and may pose a threat to *Penaeus kerathurus* fisheries in the Gulf of Gabès in Tunisia. Indeed, Ben Abdallah *et al.* (2003) have reported that *M. monoceros* is well adapted to the environmental conditions of the Gulf of Gabès, where it has been reproducing since 1993 and has been caught subsequently along with the autochthonous shrimp *Penaeus kerathurus*, though its presence in catches shows significant fluctuations from year to year (Ben Abdallah *et al.*, 2003). In 1994–1998, after the appearance in the Gulf of Gabès of *M. monoceros*, the annual production of *P. kerathurus* dipped, failing to reach 300 tonnes. In 2000, *P. kerathurus* reached a maximum of 2 484 tonnes, before dropping again in 2001–2002. In 2003, *P. kerathurus* catch again increased to 1 829 tonnes (Ben Abdallah *et al.*, 2003).

The crab *Portunus segnis* and the shrimps *Penaeus japonicus* and *M. monoceros* have been caught commercially for many years in Egypt. Similarly, Corsini-Foka *et al.* (2017) have recorded *P. segnis* in Italian waters. Faccia, Alyakrinsky and Bianchi (2009) also reported the occurrence of a single individual of the red king crab (*Paralithodes camtschaticus*), a boreal species with economic potential, in the Ionian Sea, where it was most likely introduced by way of ballast water. More recently, Özcan and Özcan (2017) reported that the kuruma shrimp (*Penaeus japonicus*) has become a dominant commercial species in Iskenderun Bay.

Most of the commercialized non-indigenous crustacean species in the Mediterranean are listed in Table 3. A total of nine species are sold in markets.

**TABLE 3 – Commercialized non-indigenous crustacean species in the Mediterranean Sea**

| Species name | Country |
|---|---|
| *Callinectes sapidus* | Egypt, Greece, Lebanon, Syrian Arab Republic, Turkey |
| *Portunus segnis* | Egypt, Greece, Lebanon, Syrian Arab Republic, Tunisia, Turkey |
| *Penaeus japonicus* | Egypt, Israel, Lebanon, Syrian Arab Republic, Turkey |
| *Trachysalambria palaestinensis* | Lebanon, Syrian Arab Republic, Tunisia, Turkey |
| *Metapenaeus monoceros* | Egypt, Israel, Syrian Arab Republic, Tunisia, Turkey |
| *Erugosquilla massavensis* | Turkey |
| *Metapenaeus stebbingi* | Israel, Turkey |
| *Penaeus semisulcatus* | Israel, Syrian Arab Republic, Turkey |
| *Penaeus hathor* | Turkey |
| *Metapenaeopsis aegyptia* | Turkey |

### 3.3.4 Evaluation of Lessepsian mollusc catch

Little data are available on commercial mollusc species. However, Salman (2002) and Lefkaditou, Corsini-Foka and Kondilatos (2009) have reported that the bigfin reef squid (*Sepioteuthis lessoniana*) (Plate 21), has a commercial value in Turkey and Greece. Tzomos *et al.* (2010) have also noted that 38 mollusc species from Cyprus are well established and available at Cypriot fish markets, particularly *S. lessoniana*, a Lessepsian cephalopod species. Additionally, Bariche (2012) studied the species *Chama pacifica* and *Spondylus spinosus*, though they are of lower economic importance.

The conch, *Conomurex persicus*, is served frequently in seafood restaurants in Greece and Israel, while *Pinctada radiata* and *Magallana gigas* are of minor commercial value in Greece (Katsanevakis *et al.*, 2008). In Tunisian waters, *P. radiata* is caught occasionally with bottom trawls and dredges and consumed mainly in Kerkennah and Djerba, where its demand is growing (B. Öztürk, unpublished data).

# 4. Harmful non-indigenous invasive species and their impacts

## 4.1 Poisonous and toxic non-indigenous fish species

The important issue of venomous fish species requires thorough investigation due to public health concerns and threats to fishers' safety. A number of poisonous non-indigenous fish species are known from the eastern Mediterranean countries (Table 4), including Greece (Kasapidis *et al.*, 2007), Israel (Golani, 1996), the Syrian Arab Republic (Saad, 2005) and Turkey (Bilecenoğlu, 2003; Bilecenoğlu, Kaya and Akalin, 2006; Bariche, Torres and Azzurro, 2013; Akyol *et al.*, 2005); Sümen and Bilecenoğlu, 2019; Dimitriadis *et al.*, 2020; Saggiomo *et al.*, 2021). All six species in Table 4 are now present in the majority of eastern Mediterranean countries and it can be predicted that they will be commonly found in the entire eastern Mediterranean in the near future.

**TABLE 4 – Distribution of poisonous and toxic fish species in the eastern Mediterranean countries**

| Species | Cyprus | Egypt | Greece | Israel | Lebanon | Syrian Arab Rep. | Turkey |
|---|---|---|---|---|---|---|---|
| *Lagocephalus sceleratus* | + | + | + | + | + | + | + |
| *Lagocephalus spadiceus* | + | + | + | + | + | + | + |
| *Lagocephalus suezensis* | + | + | + | + | + | + | + |
| *Plotosus lineatus* | + | + | - | + | + | + | + |
| *Synanceia verrucosa* | + | - | - | + | + | + | + |
| *Pterois miles* | + | + | + | + | + | + | + |

Note: + = present; - = absent.

In Turkish waters, Başusta, Erdem and Kumlu (1998) reported the skate stingray (*Himantura uarnak*), while Bilecenoğlu (2011) noted the first sighting of the Red Sea-originated stonefish (*Synanceia verrucosa*). Tetraodontid species (pufferfish), namely *Lagocephalus sceleratus* (Plate 27), *Lagocephalus spadiceus* (Plate 28), *Lagocephalus suezensis* (Plate 29), *Torquigener flavimaculosus* (Plate 30) and *Plotosus lineatus* have invaded the Mediterranean Sea and caused severe problems for the local people who consumed them, due to the poison contained in the fishes' internal organs and flesh. Zaki (2004) reported eight fatalities in the seaport city of Suez from TTX poisoning caused by eating the striped catfish (*P. lineatus*). Galil (2007b) has stated that the silverstripe blaasop (*L. sceleratus*) and the striped catfish (*P. lineatus*) pose major health hazards in Israel as well. Similarly,

**PLATE 27**
*Lagocephalus sceleratus*

©Murat Draman

**PLATE 28**
*Lagocephalus spadiceus*

©Elif Özgür-Özbek

Eisenman *et al.* (2008) and Kheifets *et al.* (2012) emphasized the potential impact on humans of *L. sceleratus* in the Mediterranean Sea. In Turkish waters, Doğdu *et al.* (2016) observed the catfish *Plotosus lineatus*, while Draman (2016) observed tetraodontid species from the Kaş marine protected area. Given the presence of the neurotoxin TTX in the edible muscles, liver, intestines and ovaries of *L. sceleratus* (Kırımer *et al.*, 2016), and reports of population explosions, such as that of *L. sceleratus* over the last few years in Cypriot waters or occurrences of *Lagocephalus* spp. in the Adriatic Sea (Dulčić and Dragicevic, 2017) and elsewhere, careful monitoring is required, as well as the ready availability of medical treatments. Sümen and Bilecenoğlu (2019) reported a traumatic finger amputation caused by *L. sceleratus* bite from Turkey, while Shakman *et al.* (2019) reported that *L. sceleratus* had caused the death of three fishermen who had eaten the gonads of this species.

**PLATE 29**
*Lagocephalus suezensis*

©GFCM/Murat Draman

**PLATE 30**
*Torquigener flavimaculosus* eating fish flesh

©Bayram Öztürk

**PLATE 31**
*Pterois miles*

©Bayram Öztürk

Public awareness campaigns advertised through posters or leaflets have been undertaken in some countries, such as Cyprus, Greece, Israel, Turkey and others to educate on the threats of venomous fish species. Recently, the devil firefish (*Pterois miles*) has spread to all eastern Mediterranean coasts (e.g. Özgür-Özbek *et al.*, 2017), though information is still lacking on its abundance. Indeed, the devil firefish is one of the most invasive species the world over (Schofield, 2010; Poursanidis, 2015; Johnston and Purkis, 2014; Morris *et al.*, 2009). These same authors noted that twenty-two years after its first recorded sighting off the Israeli coast (Golani and Sonin, 1992), the Indo-Pacific devil firefish (*P. miles*) (Plate 31), had been reported from the waters of Cyprus, Greece, Italy, Lebanon, Tunisia and Turkey (Iglésias and Frotté, 2015; Bariche, Torres and Azzurro, 2013; Turan *et al.*, 2014; Crocetta *et al.*, 2015; Oray *et al.*, 2015, Azzurro *et al.*, 2017b) (Figure 17).

Frazer *et al.* (2012) and Morris *et al.* (2009) summarized the impacts of the devil firefish (*P. miles*) on non-native ecosystems as: i) predation on a wide range of native organisms, consuming up to 2.5–6 percent of its body weight per day; ii) occupation of native species' key habitats, especially for juveniles, iii) increases in algal dominance due to a reduced abundance of herbivorous fish,

**FIGURE 17**
Location records of *Pterois miles* in the Mediterranean Sea


1. Golani and Sonin (1992); 2. Bariche *et al.* (2013); 3. Turan *et al.* (2014); 4. Oray *et al.* (2015); 5. Iglésias and Frotté (2015); 6. Turan and Öztürk (2015); 7. Crocetta *et al.* (2015); 8. Mytilineou *et al.* (2016); 9 and 10. Dailianis *et al.* (2016); 11. Özgür-Özbek *et al.* (2017); 12. Azzuro *et al.* (2017b).

Source: UN, 2021 modified from Özgür-Özbek *et al.* 2017.

iv) predation on, and competition with, commercial and endangered fish that may negatively impact fisheries and related incomes, and v) humans being injected with venom. They additionally emphasized the importance of management approaches and protective measures. Arias-Gonzáles *et al.* (2011) also clearly showed that devil firefishes produce marked direct and indirect effects on the food web in terms of predation and competition.

Therefore, it is important to create public awareness in order to convince people that when hunting, trading, or cooking devil firefish, they are actually helping native ecosystems to recover (e.g. Azzurro and Bariche, 2017). As such recognition increases, *P. miles* could become part of the fisheries industry like other invasive non-indigenous species, such as *Saurida lessepsianus* in the Mediterranean Sea. Alvarez (2014) proposed an action plan to combat devil firefish that includes an import ban and an app to mitigate population and limit distribution.

## 4.2 Harmful non-indigenous jellyfish and impacts

In recent years, large populations of more non-indigenous jellyfish species have been observed along Mediterranean coasts, causing anxiety among fishers and tourists alike. Even though some jellyfish species are harmless and native to the Mediterranean Sea, their distribution ranges have expanded. At the same time, jellyfish species of Indo-Pacific origin, like *Rhopilema nomadica* (Plate 32) and *Cassiopea andromeda*, have established themselves in the eastern Mediterranean Sea and damaged local economies to some extent, by becoming entangled in fishing nets or being stranded on beaches and frightening visitors.

The upside-down jellyfish, *Cassiopea andromeda,* is frequently encountered in the eastern Mediterranean Sea (Bilecenoğlu, 2002). Özgür and Öztürk (2008) have reported that the distribution of this stinging species has extended from the south further north, up to Fethiye on

**PLATE 32**
*Rhopilema nomadica* (left) and *R. nomadica* (right) among fish catch on the deck

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the Turkish Mediterranean coast. Kideys and Gücü (1995), and Avşar (1999) observed that the proliferation of *R. nomadica* off the eastern Mediterranean coast of Turkey poses a potential risk to human health, tourism and fisheries. In August 1995, many swimmers were stung and sought medical treatment. Local fishers claimed that the catch from gillnet fisheries had decreased and that the jellyfish entangled in their nets presented a major nuisance. In Iskenderun, due to mass jellyfish blooms, fish farmers could not lift their nets to the surface when they wished to extract fish from cages. Although *R. nomadica* do not move actively, they have penetrated the Levant Sea like other Lessepsian species, carried by the Levantine current system. According to Schembri, Deidun and Vella (2010), *C. andromeda* has also been observed off the Maltese islands in the central Mediterranean Sea.

Galil (2008) reported as early as the summer of 1987 cases of severe jellyfish envenomations requiring hospitalization in Israel. Since then, Öztürk *et al.* (2019) noted that jellyfish envenomation was increasing in both the eastern Mediterranean and the Black Sea. Öztürk and İşinibilir (2010) reported that 815 incidents related to *Rhopilema nomadica* requiring hospitalization had been recorded in five areas in Turkey. This species has also been noted from the Syrian coast (İkhtiyar and Durgham, 2002), Maltese waters (Deidun, Arrigo and Piraino, 2011), Greece (Frangou-Siokou, Sarantakos and Christou, 2006), and the Tunisian coast (Daly Yahia *et al.*, 2013). Yoffe and Baruchin (2004) reported massive jellyfish swarms stretching 100 km along the Israeli coast.

Besides *Cassiopea andromeda* and *Rhopilema nomadica*, Deidun *et al.* (2017) reported at least three other non-indigenous jellyfish species of Indo-Pacific origin from the Mediterranean basin: *Phyllorhiza punctata*, *Marivagia stellata* and *Cotylorhiza erythraea*. Çevik *et al.* (2011), Uysal and Boz (2018) have also reported *Phyllorhiza punctata* in Turkish waters and fear that a population proliferation of this species would pose a risk for tourism and fisheries.

## 4.3 Impacts on fisheries

Marttin *et al.* (2006) have widely examined the fisheries sector in the eastern Mediterranean under the framework of the MedFisis project[5] and have found that the abundance of some native

5 The regional project on a Mediterranean Fishery Statistics and Information System (MedFisis), jointly financed by FAO and the European Union, was implemented from 2004 to 2011 and aimed to develop a regionwide integrated system to collect and compile fishery statistics and information in reliable and timely manner.

species has declined while the abundance of some Lessepsian species has increased. Competition for the same ecological niches and direct interference could provide possible explanations for successful colonization (Goren and Galil, 2005). It has been suggested that increasing exploitation of non-native species has caused a spatial shift in trawl fishing grounds towards shallower waters, where the biomass density of those species is highest (i.e. at bottom depths of up to 50 m) and a consequent increase in the ratio of non-native to native species in Levantine trawl landings (Pisanty and Grofit, 1991). Nader, Indary and Boustany (2012) investigated the pufferfish (*Lagocephalus sceleratus*) in the eastern Mediterranean and proposed that the TTX obtained from its tissues might be potentially useful in the pharmaceutical industry. Detailed impact studies of pufferfish on fisheries have also been carried out by Nader, Indary and Boustany (2012), who have documented frequent complaints from local fishers in Cyprus, Greece, Lebanon and Turkey. Generally, these complaints are linked to the destruction of nets due to entangled *L. sceleratus* or predation by this species on already entangled fish. *Lagocephalus sceleratus* is considered a major nuisance by fishers not only due to the damage it can do to fishing gear when attacking fish caught in nets and lines, but also because of the reduction in local stocks of squids and octopus it effects through predation.

Vella and Vella (2017) have reported that among the non-indigenous species being investigated in the Mediterranean, there are various toxic and dangerous species, including the blunthead puffer fish (*Sphoeroides pachygaster*), which has extended its range from the Atlantic to the Mediterranean since 1980 and was recorded as far east as Maltese waters in 1999. Through interviews with Maltese fishers dating back to the 2000s, it was possible to determine not only the abundance and distribution of this species, but also its various impacts on artisanal fishing activities. Ünal and Bodur (2017) also highlighted studies from Turkish waters, which showed that the financial loss to fishers caused by the silver-cheeked toadfish (*Lagocephalus sceleratus*) in 2013–2014 approached approximately EUR five million, more than doubling from the 2011–2012 period and demonstrating that attention must be directed toward the issue by the relevant fisheries authorities.

Moreover, mostly in the eastern Mediterranean Sea, several non-indigenous species have serious impacts on fisheries, including the alga *Caulerpa taxifolia*, which fouls on fishing nets, and the crab *Callinectes sapidus* (Plate 5), which damages nets through entanglement and cutting.

The impacts of non-indigenous species should be considered with regard to the management of fisheries as well, as traditional mesh sizes of fishing nets may need to be changed due to the presence of new commercial species, mostly in the eastern Mediterranean. This process involves extra costs for fishers and fishing communities, which may be difficult to cover, particularly for traditional fishers with limited financial resources. Another issue is that fish inspectors may easily confuse native and non-indigenous fishes when examining catches on fishing boats (e.g. the native mullet species *Mullus barbatus* and *M. surmuletus* with the non-indigenous mullet *Upeneus moluccensis*).

## 4.4 Impacts on tourism, human health and other socio-economic activities

Non-indigenous jellyfish species can present a threat to seaside tourism, with cases of hospitalization having occasionally been reported after contact with certain species in the eastern Mediterranean Sea. Some people can be highly allergic to jellyfish, though the most important factor in the severity of the injury is the amount of poison that ends up in the bloodstream.

Though the stings are rarely fatal, other effects are observed, such as itching, severe poisoning, muscle cramps, abdominal rigidity, decreases in touch sensation, nausea, vomiting, serious back pain, speech difficulties, involuntary muscle contractions and breathing difficulties (Mariottini, Giacco and Pane, 2008). Certain venomous jellyfish species can therefore pose a serious threat to tourism (Spanier and Galil, 1991).

Table 5 summarizes the impacts of some non-indigenous species on tourism and human health either from stings and/or consumption

**Table 5 – Some harmful non-indigenous species and their impacts on tourism and human health in the Mediterranean Sea**

| Alien species | Target groups | Results |
|---|---|---|
| *Rhopilema nomadica, Cassiopea andromeda, Phyllorhiza punctata* | Tourists, fishers, divers, sailors | Injury, hospitalization |
| *Macrorhynchia phillipina* | Tourists, divers, fishers | Injury, hospitalization |
| *Diadema setosum* | Tourists, divers, fishers | Injury |
| *Lagocephalus* spp. | Everyone | Hospitalization |
| *Plotosus lineatus* | Everyone | Hospitalization |
| *Synanceia verrucosa* | Everyone | Hospitalization |
| *Pterois miles* | Everyone | Hospitalization |
| *Himantura uarnak* | Everyone | Hospitalization |

The white stinger, *Macrorhynchia philippina* (Plate 33), is a common circumtropical hydroid (Watson, 2002). This Lessepsian migrant occurs along the coast of Lebanon at depths of 0–40 m (Bitar and Bitar-Kouli, 1995; Zibrowius and Bitar, 2003) and is also abundant in Turkish waters near Mersin and Iskenderun (Çınar *et al.*, 2006), where 10–15 cm-high colonies are frequently found on rocks at depths of 1–2 m. They are also distributed in the coastal waters of Cyprus, Israel and the Syrian Arab Republic. Dense populations of this species in shallow water may pose a risk for tourism, as *M.philippina* causes a painful and itchy sting.

**PLATE 33**
*Macrorhynchia philippina*

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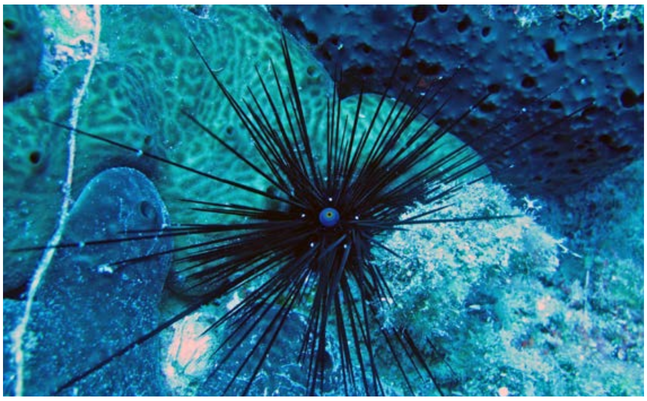
A non-indigenous echinoderm species described by Yokes and Galil (2006b), the needle-spined urchin, *Diadema setosum* (Plate 34), also poses a threat to bathers in shallow water due to its spines. Additionally, Ergen *et al.* (2002) reported that fouling serpulid worms, a common nuisance in ports and harbours in the Mediterranean Sea, are causing economic losses, by, for example, encrusting boats, blocking pipes and boring into structures.

Further examples include reports from Bentur *et al.* (2008) of clinical TTX poisoning following consumption of the Lessepsian immigrant fish *Lagocephalus sceleratus* caught on the Israeli coast and reports from Çınar *et al.* (2005) of the polychaete *Eurythoe complanata* leading to poisoning in fishers catching fish in their nets.

It is difficult to assess the economic damage collectively caused by the range of non-indigenous species versus the new economic benefits gained. However, as a real-world economic example, Galil (2007b) noted that jellyfish-blocked water intake pipes can pose a threat to the cooling systems of port-bound vessels and coastal power plants. Given this concern, the Israel Electric Corporation removed tonnes of jellyfish from its seawater intake pipes at its two largest power plants in the summer of 2001, at an estimated cost of USD 50 000.

**PLATE 34**
*Diadema setosum*



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## 4.5 Impacts on biodiversity

Although non-indigenous species have increased marine biodiversity in both the Mediterranean and the Black Sea, they may have altered the evolutionary pathways of native species through competitive exclusion, niche displacement, predation and other ecological and genetic mechanisms (Mooney and Cleland, 2001). Furthermore, in the eastern Mediterranean Sea, it should be noted that most ecological changes take place in the first 50 m below the surface. Bitar, Ocana and Ramos-Espla (2007) have pointed out that due to non-indigenous Red Sea species and tropicalization in the Mediterranean, particularly in the infralittoral (30–50 m) zone and, to a lesser degree, in the circalittoral zone, some important changes are already occurring in the Levant Sea in terms of species distribution.

In the case of the Levant Sea, the rapid reduction in the abundance of the herbivorous fish *Salpa salpa*, a common species in the rest of the Mediterranean, appears to be closely linked to the establishment of a competitor, *Siganus rivulatus,* a Lessepsian migrant recorded in the Levant Sea since the early 1990s (Bariche, Letourneur and Harmelin-Vivien, 2004). UNEP-MAP-RAC/SPA (2011a) has reported that if a non-indigenous species exploits the same trophic resources as a native species, the resulting competition between the two species can cause the native species to decline or disappear over the long term.

According to Harmelin-Vivien *et al.* (2005), Lessepsian migrants represent 13 percent of the species richness and 19 percent of the total abundance of individuals along the rocky coasts of Lebanon. Furthermore, they reported that Lessepsian fish species occupied either similar ecological niches in the Red Sea and in the Levant Sea (such as *Pempheris rhomboidea* and *Sargocentron rubrum*) or have expanded their distribution towards deeper water in their new environment (such as *Siganus luridus* and *S. rivulatus*).

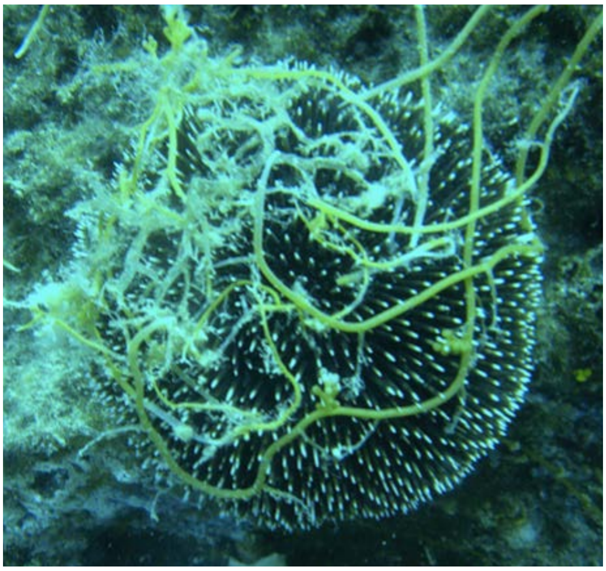
Furthermore, Sala *et al.* (2011) have reported that the two non-indigenous herbivorous fish from the Red Sea *Siganus luridus* and *S. rivulatus* have a significant impact on benthic communities, which show extremely low biomass on the Turkish coast. The shift from well-developed native algal assemblages to "barrens" implies a dramatic decline in biogenic habitat complexity, biodiversity and biomass. A targeted *Siganus* fishery could therefore help to restore the macroalgal beds of the rocky infralittoral zone on the Turkish coast.

Additionally, Bariche (2012) has reported that non-indigenous species compete intensively with indigenous species for space and various resources, often leading to local displacement or elimination of the indigenous species from the invaded area. Indeed, the presence of the Lessepsian *Upeneus moluccensis* has resulted in a decline of the indigenous *Mullus barbatus* in shallower waters. Similarly, the invasion of *Saurida lessepsianus* has impacted the indigenous *Merluccius merluccius* population in shallower waters. Currently, *M. barbatus* and *Synodus saurus* are found mainly in deeper waters along the Levantine coast (Oren, 1957; Golani and Ben-Tuvia, 1995). Further studies have indicated that the narrow-barred Spanish mackerel, *Scomberomorus commerson*, is in the process of replacing the native *Argyrosomus regius* in the Levant Sea (Galil, 2006). Meanwhile, the indigenous prawn *Melicertus kerathurus* has been outcompeted by various Lessepsian prawns in the Levant basin and in Turkish waters (Udekem d'Acoz, 1999). Likewise, the very common Lessepsian bivalves *Pinctada radiata, Chama pacifica* and *Spondylus spinosus* have replaced native Mediterranean bivalves.

Trophic relationships between non-indigenous and native species represent one of the important issues to be more fully examined in order to better understand the ecological balances of the newly emerging ecosystems in the Mediterranean Sea. For example, Bariche (2006) found that the non-indigenous *Siganus luridus* occasionally ingests the toxic non-indigenous macrophyte *Caulerpa cylindracea* (Plate 35). Additionally, a number of non-indigenous fish species, including *Siganus rivulatus, S. luridus*, *Aphanius dispar* and *Pranesus pinguis* continue to host their monogenean Lessepsian ectoparasites upon arrival in the Mediterranean (Paperna, 1972). Interestingly, Shakman *et al.* (2009) have also recorded two native ectoparasitic cymothid isopod species from *S. luridus* and *S. rivulatus* in Libyan waters. All of these parasite species should be monitored in order to assess their potential impacts to fisheries, human health and biodiversity.

Bardamaskos *et al.* (2009) have observed that among the Lessepsian fish migrants, *Siganus luridus* has established a permanent population in the southeastern Ionian Sea. Crocetta, Renda and Colamonaco (2009) have also reported that two Lessepsian mollusc species, *Cerithium scabridum* and *Fulvia fragilis*, are well established along the Italian coast with large and stable populations.

**PLATE 35**
*Caulerpa cylindracea* (left) and sea urchin covered by *C. cylindracea* (right)



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Therefore, the major impact of non-indigenous species, beyond the harmful effects on human activities mentioned above, is on local biodiversity. For example, the native barrel jellyfish, *Rhizostoma pulmo*, has been replaced by the non-indigenous nomad jellyfish, *Rhopilema nomadica*, in the eastern Mediterranean Sea (Boudouresque, 1999). Swarms of *R. nomadica* may actually be contributing, however, to growing populations of the commercially important non-indigenous carangid fish, *Alepes djedaba*, whose juveniles shelter among the jellyfish tentacles (Galil, Spanier and Ferguson, 1990).

Galil, Shoval and Goren (2009) have noted the reappearance of the non-indigenous floating bell jellyfish (*Phyllorhiza punctata*), along the Israeli coast. Meanwhile, *Mnemiopsis leidyi*, a non-indigenous species of comb jelly, has been reported from many parts of the Mediterranean Sea, from Turkey to Italy (Uysal and Mutlu, 1993; Kideys and Niermann, 1994; Shiganova and Malej, 2009; Galil, Kress and Shiganova, 2009; Shiganova *et al*., 2001; Boero *et al*., 2008). This species is known as one of the "100 world's worst invaders" by Lowe, Browne and Boudjelas (2000). While *M. leidyi* has been the cause of hypoxia and fisheries collapse in the Black Sea fisheries, it is uncertain to what extent it will impact the Mediterranean Sea. As *M. leidyi* and anchovies generally inhabit the upper mixed layer, with their spawning peaks coinciding in time and space, and as both favour higher water temperatures and consume mainly the same prey, *M. leidyi* presents potential risks to native Mediterranean Sea ecosystems. *Mnemiopsis leidyi*'s continuing spread, to, for example, the northern Aegean Sea (İşinibilir and Tarkan, 2002), the Adriatic Sea and Sicily (Faris, 2009), represents a concerning dynamic.

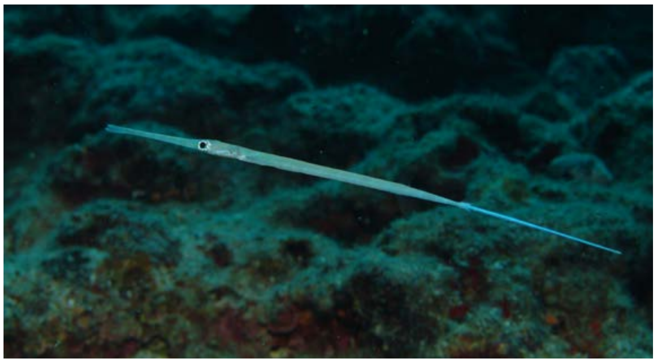
While *M. leidyi* and *Beroe ovata* have only recently invaded the central Mediterranean, it should be remembered that these species have been found in the eastern Mediterranean (Uysal and Mutlu, 1993) and in the Aegean Sea (Shiganova, Christou and Frangou-Siokou, 2007) since 1992, without having had any major impact on pelagic fisheries, such as those of anchovy, Atlantic horse mackerel and sardine, at least in Turkish waters. Moreover, the Mediterranean ecosystem differs from the Black Sea's in terms of the number of species, biodiversity, competition and currents, among many other aspects. Furthermore, the presence of comb jellies such as *Beroe forskalii*, *B. cucumis* and *Bolinopsis vitrea*, which are native to the Mediterranean and prey on *M. leidyi,* and the presence of *Beroe ovata*, a competitor for *M. leidyi* in the Mediterranean, may provide a buffer against the threat of *M. leidyi*. Nevertheless, as climate change likely represents the main reason for the jellyfish invasion, it seems that a shift from fish- to jellyfish-dominated ecosystems may be underway in the Mediterranean Sea. This process is certainly accelerated by overfishing and by non-indigenous species being able to easily find empty niches to establish themselves in the new environment. More research is needed to better understand, for example, the impacts of *M. leidyi* on fisheries, local communities and Mediterranean biota.

Azzurro *et al.* (2013a) have showed that the bluespotted cornet fish (*Fistularia commersonii*), a Lessepsian fish species, (Plate 36) has dispersed from the eastern to the western Mediterranean Sea (Figure 18) with a speed unequalled by any other Mediterranean non-indigenous fish (Ben Rais Lasram, Guilhaumon and Mouillot, 2010). Indeed, *F. commersonii* has colonized almost the entire Mediterranean region within only seven years of its first sighting.

Galil (2007b) and Popper and Gunderman (1975) have reported that *Fistularia commersonii* feeds on the fry of siganids. However, Bariche *et al.* (2009) have also noted that the diet of *F. commersonii* includes a large variety of prey (41 taxa), with *Spicara smaris* and *Boops boops* among the most important.

Rilov, Benayahu and Gasith (2004) have noted a massive build-up of the Red Sea mussel, *Brachidontes pharaonis*, along the Israeli coast due to a recent shift in habitat conditions, while the same species has also spread to Sicily and established dense populations there (Sará, Romano and Mazzola, 2006). Çinar (2006) has also recorded the presence of 16 serpulid species, nine of which are considered to be non-indigenous species.

**PLATE 36**
*Fistularia commersonii*



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Occhipinti-Ambrogi (2002) has observed that *Anadara inaequivalvis* has replaced the native *Cerastoderma glaucum* in the northern Adriatic Sea. Streftaris and Zenetos (2006) described non-indigenous species according to their impacts on socio-economic conditions (fisheries/aquaculture, health and sanitation, infrastructure and buildings) and documented 43 species in the Mediterranean Sea. Putting these examples within a broader perspective, Savini *et al.* (2010) assessed the negative ecological impacts of the 27 most utilized non-indigenous aquaculture species in Europe following their escape from aquaculture facilities, evaluating: (i) their distribution across Europe (including non-European Union Member States); (ii) evidence of their environmental impact in the wild; and (iii) evidence of their being vectors of non-target non-indigenous species and other hitchhikers (e.g. pathogens).

Çinar *et al.* (2006) reported that a non-indigenous tunicate, *Phallusia nigra*, found in Alanya harbour, may cause economic loss by fouling ship hulls or water intake pipes. Conversely, some ship-transported species can become economically important in new areas, such as the blue crab (*Callinectes sapidus*). Meanwhile, the impacts of the only successful colonizer of the eastern

**FIGURE 18**
Chronological reconstruction of *Fistularia commersonii* invasion in the Mediterranean Sea


Source: UN, 2021 modified from Azzurro *et al.*, 2013a.

Mediterranean from among the Red Sea sea grasses, *Halophila stipulacea* (Plate 37), are not yet known.

**PLATE 37**
*Halophila stipulacea*

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Özcan, Ateş and Katağan (2008) have reported that the native mantis shrimp, *Squilla mantis*, has been displaced by a non-indigenous mantis species, *Erugosquilla massavensis,* along the Levantine coast of Turkey. Similarly, *Trachysalambria palaestinensis*, a non-indigenous prawn species in Tunisia, has been displaced by *Penaeus kerathurus* in the Gulf of Gabès (Jarboui and Ghorbel, 1995).

**PLATE 38**
*Synaptula reciprocans*

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The Indo-Pacific holothurian, *Synaptula reciprocans* (Plate 38), has also been reported in Greek, Israeli and Turkish waters (Çınar *et al.* 2006; Galil, 2007a; Antoniadou and Vafidis, 2009).

Rius, Pascal and Turon (2008) indicated that the ascidian, *Microcosmus squamiger*, considered to be native to Australia, and having spread worldwide via transoceanic vessels, has successfully invaded both artificial and natural habitats in the Mediterranean, where it has become both a pest and economic threat requiring controls to prevent it from outcompeting the local biota. Likewise, Shenkar and Loya (2009) have reported seven non-indigenous ascidian species along the Mediterranean coast of Israel and indicated that the negative impacts on local species and habitats necessitate long-term studies and monitoring of this group.

In a recent study, Mannino, Balistreri and Deidun (2017) reported ten invasive species in the Mediterranean known worldwide for their environmental and economic impacts, which can be divided into the following groups: Phytobenthos (*Caulerpa cylindracea*, *Sargassum muticum*, *Undaria pinnatifida* and *Womersleyella setacea*); Zooplankton/Cnidaria (*Rhopilema nomadica*); Zoobenthos/Mollusca (*rachidontes pharaonis*); Zoobenthos/Crustacea (*Percnon gibbesi*), and Zoobenthos/Fish (*Fistularia commersonii*, *Lagocephalus sceleratus* and *Pterois volitans*).

The positive impacts of non-indigenous species have probably been underestimated, as emphasized by Katsanevakis *et al.* (2014a), as a perceived bias against non-indigenous species persists even when they contribute positively. Among the species herein assessed as high-impact species, 17 had only negative impacts and seven had only positive impacts; for the majority (63 species), both negative and positive impacts were reported, and therefore the overall balance was often unknown. Katsanevakis, Tempera and Teixeira (2016) have developed a standardized, quantitative method to map the cumulative impacts of 60 invasive non-indigenous species on marine ecosystems, which revealed the significance of the impacts. Turan (2018) has reported that dusky grouper feeds on devil firefish in Iskenderun Bay, Turkey, potentially helping to control devil firefish in the area. Jimenez *et al.* (2018) reported periodical removal events at various locations in Cyprus. Stern *et al.* (2018) emphasized the importance of genetic studies to understand population demography of the invasive devil firefish (*P. miles*) in the Levant basin of the eastern Mediterranean. Turan *et al*, (2018) also reported 101 non-indigenous fish species, of which 89 were bony, 11 cartilaginous and one jawless.

Sea water temperature rise due to climate change is likely to have a major influence on Lessepsian species distribution in the future. It is known that the Mediterranean Sea is undergoing a tropicalization process and that an expansion of warm-water species to the western part has already occurred. Some species of Atlantic origin have also migrated to the Mediterranean Sea or expanded their range. So far, there have been no extinctions or total disappearances of local species in the Mediterranean and the Black Sea due to non-indigenous invasive species, but greater displacement of native species is expected in the near future. Parravicini *et al.* (2015) have documented an increase in warm-water native and non-indigenous species richness during warming phases in the Mediterranean Sea.

The majority of the marine flora and fauna of the Mediterranean Sea belong to the Atlanto–Mediterranean Province. Recently, however, with the establishment of numerous species of Indo-Pacific origin, the Mediterranean Sea has acquired certain tropical and subtropical biological characteristics, particularly in the Levant Sea. Some non-indigenous species may thrive in their new environments by displacing native species. Therefore, the number of non-indigenous species establishing themselves in the Mediterranean will probably increase in the future, while the gene pool of the recipient regions will also change.

Furthermore, some of the non-indigenous invasive species in the Mediterranean, such as the comb jelly, *Mnemiopsis leidyi*, are among the top 100 of the World's Worst Invasive Alien Species, as listed by the Global Invasive Species Database (ISSG, 2021), and recognized globally as a major threat to biodiversity. Nevertheless, it is difficult to identify 100 invasive species from the Mediterranean Sea that really are "worse" than any others, as species and their interactions with ecosystems are highly complex. Some species, for example, may have invaded only a limited area to date but show a high probability of expanding and causing further damage.

From an economic point of view, assessment of the damage caused by non-indigenous species is not an easy task. An appropriate economic model is required and should not be based solely on market-based costs; it should include indirect and non-use value costs, as already mentioned by Born, Rauschmayer and Brauer (2005).

# 5. Intentionally and unintentionally introduced species in the Mediterranean Sea and their impacts

Aquaculture remains one of the most prominent reasons behind the intentional introduction of non-indigenous species. In the Mediterranean Sea, aquaculture is one of the fastest-growing sectors, due to high seafood demand, which is expected to increase further in the future. However, in bad weather conditions, for example, seabass and seabream cages have been known to break, allowing fish to escape into Cypriot, Greek, Israeli and Turkish waters (UNEP/MAP/MEDPOL, 2004), where genetic hybridization is of concern, especially between closely related native and non-indigenous species.

In the Mediterranean Sea, two commercial bivalve species, namely the Pacific cupped oyster *Magallana gigas* and the Japanese carpet shell (*Ruditapes philippinarum*), were introduced intentionally for aquaculture purposes during the 1960s and 1970s and have since escaped. The methods employed to subsequently catch *R. philippinarum* in the Venice Lagoon have been damaging to the benthic community (Pranovi *et al*., 2006), while populations of *M. gigas* introduced into French waters have been hit by successive disease outbreaks (Petton *et al*., 2015). Therefore, intentional introductions may cause unexpected problems and pose risks both to human health and marine biodiversity. Elsewhere, species like the rayed pearl oyster, *Pinctada radiata,* introduced intentionally into Greek waters for aquaculture purposes, has since established thriving populations in the wild (Serbetis, 1963). Verlaque, Boudouresque and Mineur (2007) also reported that, in the 34 Mediterranean coastal lagoons investigated, 67 non-indigenous macrophyte species were observed and that oyster transfer represented the most efficient vector for macrophyte introduction into the Mediterranean Sea. Additionally, at least 30 introduced species have been recorded in the Venice Lagoon, many of which have established large populations and supplanted native species (Occhipinti-Ambrogi, 2000). Impacts were reported on the coastal lagoon ecosystems from *Undaria pinnatifida, Desmarestia viridis* and *Antithamnion nipponicum*.

Accidental introductions include, for example, the seaweed *Caulerpa taxifolia*, which originally escaped from aquaria (Jousson *et al*., 1998), was introduced into Monaco´s waters in 1984, subsequently spread into French, Italian, Spanish and Croatian waters, and was eventually reported off Turkey, with an entirely different variety than was observed in the Gulf of Iskenderun (Çevik *et al*., 2007). Meinesz and Boudouresque (1996) cited *C. taxifolia* as a major threat to the coastal ecosystem of the Mediterranean Sea. Meinesz (1997) noted that *C. taxifolia* initially formed a dense cover of more than one hectare in five years (1984–1989), yet 13 years after its first observation in the Mediterranean, by spreading via vegetative reproduction, it had impacted more or less 3 000 hectares. Montefalcone *et al.* (2015) described the two non-indigenous green algae *C. taxifolia* and *C. cylindracea* (formerly *Caulerpa racemosa* var. *cylindracea*) as among the most notorious and threatening invasive species in the Mediterranean Sea. Nevertheless, the same authors have also noted that the abundance of *C. taxifolia* has steeply declined as it disappeared from most of the areas where it was previously found. In contrast, *C. cylindracea* has exhibited an impressive and constant expansion since its first appearance in the Ligurian Sea and is still increasing its range and habitat occupancy. The environmental impacts have been mixed, however: for example, the number of native polychaeta and especially amphipod species has decreased in

*C. taxifolia* meadows, with fish and sea urchin populations also affected, while molluscan diversity has increased (Boudouresque *et al.*, 1995). Meanwhile, Francour *et al.* (1995) have reported that *C. taxifolia* meadows seem to offer favourable environments for fish recruitment of certain species of Labridae (*Coris julis, Symphodus ocellatus*), Sparidae (*Diplodus annularis*) and Serranidae (*Serranus cabrilla*) in autumn.

Apart from *C. taxifolia,* the closely related non-indigenous alga *C. cylindracea* has also dispersed widely in the Mediterranean, successfully colonizing the waters of Albania, Croatia, Cyprus, France, Greece, Italy, Lebanon, Libya, Malta, Montenegro, Spain, the Syrian Arab Republic and Turkey, (Cirik and Öztürk, 1991; Lakkis and Lakkis-Novel, 2001; Macic and Kascelan, 2007; Verlaque *et al.*, 2003). Akçali and Cirik (2007) have also reported that *C. cylindracea* and the invasive tropical seagrass *Halophila stipulacea* have both impacted the biota of the Turkish coasts.

Katsanevakis *et al.* (2014b) have identified the Thau and Venice lagoons as two important hotspots for aquaculture-introduced species, from which non-indigenous species have been able to spread into the Mediterranean. For example, the invasive seaweed *Sargassum* is no longer confined to these sites and it inhibits the recruitment and growth of other local algal species. Furthermore, the seaweed species *Saccharina japonica* and *Asparagopsis armata* also show invasive characteristics in the Mediterranean Sea, while *Womersleyella setacea* and *Acrothamnion preissii* are similarly invasive on the Italian coast, clogging fishing nets and impacting fishing (Verlaque, 1989). Boudouresque and Ribera (1994) estimated that in 2050, between 250 and 1 000 non-indigenous marine macroalgal species may be found in the Mediterranean Sea and, should this estimate be accurate, indigenous species and non-indigenous species would then be almost equally represented.

# 6. Conclusions and recommendations

Coll *et al.* (2010) have concluded that climate change favours the introduction of Red Sea species into the southeastern Mediterranean and their rapid spread northwards and westwards. Monitoring is therefore required to predict the colonization and distribution patterns of non-indigenous species. The threat comes not only from the Suez Canal: it is expected that more non-indigenous species will enter the Mediterranean from the Atlantic Ocean and the Black Sea as well. Lasram-Rais and Mouillot (2009) have already recorded 62 species of Atlantic origin in the Mediterranean Sea. As in other bodies of water around the world, the Mediterranean Sea is becoming warmer: in the last 30 to 40 years, the temperatures of the western Mediterranean have been rising both at depth (Bethoux *et al.*, 1990) and at the surface (Diaz Almela, Marba and Duarte, 2007), which is reflected in the increased presence of thermophilic marine species. Nevertheless, in the Mediterranean Sea, the largest number of species are from Atlantic origin (Tortonese, 1964), where tropical species are still rare. Nevertheless, the species composition of the Levant Sea has also been drastically altered after the opening of the Suez Canal. Bianchi (2007) also pointed out that climate change combined with Atlantic influx, Lessepsian migration and human introduction of exotic species facilitated the entrance and establishment of warm-water species, whether exotic or native, in the Mediterranean Sea.

Katsanevakis *et al.* (2013) reported that within the framework of the European Alien Species Information Network (Joint Research Centre, 2021), the countries with the highest reported numbers of marine alien species were Egypt, France, Greece, Israel, Italy and Turkey, i.e. those of the eastern Mediterranean and closest to the Suez Canal.

Galil *et al.* (2015) have stated that the expansion of the Suez Canal in 2015 is expected to continue to facilitate the arrival of Lessepsian migrants. Conversely, Zenetos (2017) has observed that the rate of bio-invasions via the Suez Canal has not "doubled" as anticipated, but rather decreased in relation to previous years. Regardless, this ongoing process requires monitoring. Moreover, several reports suggest that the general trend in the number of non-indigenous species in the Mediterranean Sea is increasing (Evans *et al.*, 2015; Ounifi-Ben Amor *et al.*, 2016; Zenetos *et al.*, 2017; Shakman *et al.*, 2019; Çınar *et al.*, 2021).

More regional cooperation is therefore increasingly necessary, particularly capacity building for experts in the impacted countries, as it seems impossible to stop Lessepsian species from entering via the Suez Canal. However, slowing down the transit of non-indigenous species to the Mediterranean Sea should be prioritized and thoroughly investigated in terms of biodiversity, human health and in order to protect the natural heritage of the entire Mediterranean and Black Sea.

In terms of monitoring, Azzurro *et al.* (2014, 2018, 2019) have shown that fishery-generated sources of information such as local ecological knowledge and underwater observations, made, for example, by divers, can produce useful geo-referenced information and significantly contribute to detecting the early geographical and demographic expansion of non-indigenous species. These participatory approaches should be carefully considered in planning large-scale monitoring programmes.

Molecular approaches may provide additional and more precise information on the origins of non-indigenous species and the relationships between species, including regional variations. Bariche *et al.* (2015) have stressed the importance of establishing a barcode library for non-indigenous Mediterranean fishes of probable Red Sea origin and initiating analyses of their invasion dynamics. In fact, it will be critical to study the genotypic and phenotypic changes of these newly established populations, as they continue to be driven by natural selection through interactions with indigenous populations and in response to new abiotic environments. Undoubtedly, genetic information on a number of Lessepsian species would also provide a more comprehensive understanding of migration patterns.

The Gibraltar, Sicily, Turkish and Kerch Straits, as hotspots and ecological corridors, require special monitoring to supply early warning information about non-indigenous species in the Mediterranean and the Black Sea. Additionally, as the Suez Canal, and even the Red Sea, represent transition zones, studies on speciation and population genetics should be pursued. Furthermore, the impact on the genetic structure of taxonomically closely related native species by Lessepsian or other introduced species would be essential to investigate.

In the eastern Mediterranean Sea, the current dispersal of non-indigenous species to the central Mediterranean Sea or further north already demonstrates the ease of environmental adaptation for these species. Lessepsian species are more successful at shallower depths and Por (2010) considers temperature to be the most important factor for their colonization in the intermediate layers at depths of about 20–40 m with relatively higher and stable temperatures.

According to the Protocol concerning Specially Protected Areas and Biological Diversity in the Mediterranean Sea (SPA/BD Protocol), contracting parties should take all appropriate measures to regulate the intentional or accidental introduction of non-indigenous species. In addition, according to the Convention on Biological Diversity (CBD), the obligatory top-priority response for all states is to prevent invasive species, after which later eradication efforts may be necessary. Thus, as non-indigenous species represent a transboundary concern, concerted action and tight cooperation are essential to tackle the issue.

The Convention on the Control and Management of Ships Ballast Water and Sediment came into force on September 2017 and is expected to mitigate the effects of ballast-transported non-indigenous species. UNEP/MAP-RAC/SPA (2008) have also underlined the importance of collecting reliable data concerning maritime traffic and ballast water uptake and discharge in the Mediterranean and the Black Sea. The International Convention on the Control of Harmful Anti-fouling Systems in Ships (AFS Convention) works to combat hull-fouling, which is another important factor in the dispersal of non-indigenous biota. According to this convention, paints including organotin compounds are forbidden and more environmentally-friendly coating materials are recommended, such as silicone and others. However, non-toxic coating materials still do not impede hull-fouling from some sessile species. Recently, the GEF-UNDP-IMO GloFouling Partnerships Project was launched to bring together key partners to respond to a global environmental problem, namely invasive aquatic species introduced via biofouling (GloFouling, 2019).

The coordination of institutional efforts between relevant international organizations, such as the GFCM, the Barcelona Convention, CIESM, RAC/SPA, the International Union for Conservation of Nature IUCN), IMO, CBD, the Intergovernmental Oceanographic Commission

(IOC), the European Union and other organizations in the Mediterranean and the Black Sea, is essential. In particular, the CIESM *Atlas of Exotic Species in the Mediterranean* (CIESM, 2002–2015) has been very useful in many ways to identify and evaluate the current status of non-indigenous species. Additionally, UNEP/MAP-RAC/SPA (2008) has published a helpful booklet and CD for invasive species in the Mediterranean Sea.

Better catch statistics, including discard ratios, are required for most commercial non-indigenous fish species and should be considered as a main priority for the GFCM. The recent implementation of monitoring programmes, such as regional surveys or the presence of observers on board, as well as the employment of standardized methods (FAO 2019a; FAO 2019b; Carpentieri, Bonanno, and Scarcella, 2020), is expected to provide important information on non-indigenous species. A permanent expert group should also be established by the GFCM to monitor impacts on fisheries and biodiversity. Investigations of the impacts of non-indigenous species on small-scale fishing activities and of the contribution of non-indigenous species to catch totals are likewise required as the majority of Mediterranean and Black Sea fishing fleets are composed of minor craft operating mostly locally.

A special early warning system, database and networks are furthermore needed, especially for poisonous fish and other species like jellyfish and hydroids. Even though some very useful global, regional or national databases and networks do exist, such as the GloBallast Partnership Programme, the East and South European Network for Invasive Alien Species (ESENIAS), the Global Invasive Species Database, the FAO Database on Introductions of Aquatic Species (DIAS), the CIESM Atlas, United States National Marine and Estuarine Invasions Database, etc., they are not adequate for fishers and local people in general. In the meantime, data on non-indigenous species require standardisation to avoid confusing species, as well as routine updates. A regional warning system is also necessary for toxic phytoplankton species, in case they are transferred in ship ballast water to the Mediterranean or Black Sea, in order to mitigate negative impacts on fish, mussel and oyster farming and human health. Using satellite images may help facilitate the implementation of these systems. Efforts such as the one initiated by the GFCM and the United Nations Environment Programme/Mediterranean Action Plan (UNEP/MAP) to establish a subregional joint pilot study monitoring non-indigenous species in the eastern Mediterranean should therefore be continued (GFCM, 2018).

Public awareness campaigns and educational materials are vital for identifying non-indigenous species, disseminating information on their distribution and highlighting the risks posed to all stakeholders, including fishers, harbour authorities, divers, tour operators and fisheries cooperatives. Updated information can also be deployed from the GFCM website and elsewhere. Citizen science is similarly important to obtain rapid and cost-efficient data on non-indigenous species; for example, Azzurro *et al.* (2013b) reported that *Abudefduf saxatilis*, a species of damselfish, was first detected by a recreational diver.

While eradication, or complete removal, of non-indigenous species from the ecosystem is almost impossible, their control or containment may be possible for some species, like *Caulerpa taxifolia* or *C. cylindracea*. Control campaigns targeted at lionfishes and other heavily impacting species may be helpful. For example, the Governments of Cyprus and Turkey have started to provide compensations for small-scale fishers facing damages caused by pufferfish. Consuming edible non-indigenous species may also help controlling those species, taking however all necessary precautions to avoid poisonous species.

Regarding the introduction and transfer of non-indigenous species, some Mediterranean countries already boast legislation and guidelines, but their regional enforcement is weak and a regional updated code of practice for the transfer or introduction of non-indigenous species is needed.

A healthy, pristine ecosystem may defend the native fauna and flora of the Mediterranean and the Black Sea, while stresses to the marine environment, such as climate change, facilitate the spread of non-indigenous species. Therefore, key species and key habitats, for example, *Posidonia* meadows, should be protected, as only through the maintenance of healthy ecosystems can they be expected to combat non-indigenous species invasion. Indeed, the Mediterranean Sea may represent only 0.82 percent of the global oceanic surface area, yet it holds 4–18 percent of all known marine species (17 000) and shows high levels of endemism.

UNEP-MAP-RAC/SPA (2011a) has suggested that by creating more marine protected areas (MPAs) in the Mediterranean, the expansion of non-indigenous species could be restricted and even potentially prevented, while strengthening the resilience of biodiversity.

Risk assessments of non-indigenous species should be a priority and is in fact required by the European Union Regulation No 1143/2014 on the prevention and management of the introduction and spread of invasive alien species. Galanidi *et al.* (2019) reported that risk assessment is important for invasive non-indigenous species such as venomous *P. lineatus* in the Mediterranean Sea.

Preventing the introduction of non-indigenous species is the cheapest and most effective option, which can be assisted by early detection and regional cooperation. As shown, once non-indigenous are established, eradication efforts are not only very costly, but mostly ineffective.

As the economic damage caused by non-indigenous invasive species in the Mediterranean is not fully known, credible assessment methods should be developed to understand more accurately the economic and environmental costs incurred. According to the European Union, non-indigenous species have caused significant economic losses, estimated at EUR 12 billion annually, which includes both the damage and the control measures applied (European Commission, 2013). As some non-indigenous species are potentially of commercial importance and underexploited, additional studies could be carried out to promote the consumption of targeted non-indigenous species to reduce their impact and allow native species and their ecosystems to recover.

According to the Marine Strategy Framework Directive, non-indigenous species represent one of the core issues among eleven descriptors for determining good environmental status. Even for some of the countries which are not European Union members around the Mediterranean and the Black Sea, this directive and descriptor can be one of the targets for harmonizing efforts to halt non-indigenous invasive species in both basins.

Concerning marine mammals, Özbilgin, Kalecik and Gücü (2018) reported that two to four humpback dolphins (*Sousa plumbea*) were recorded for the first time in Turkish coastal waters. This development may be linked to the 2015 expansion of the Suez Canal, with the individuals having strayed from the Indian Ocean, as also suggested by Bishop (2016).

Resolution GFCM/40/2016/2 for a mid-term strategy (2017–2020) towards the sustainability of Mediterranean and Black Sea fisheries (GFCM, 2019) reported that among non-indigenous invasives in the region, the two toxic species posing risks to humans in the Mediterranean, the devil firefish *Pterois miles* and the pufferfish *Lagocephalus sceleratus*, and the predatory sea snail *Rapana venosa*, a threat to native bivalve fauna in the Black Sea, have been chosen as target species for monitoring. This decision marks significant progress at the regional level in the pursuit of halting non-indigenous species. Recently, *L. sceleratus* has also been reported from the Black Sea (Bilecenoğlu and Öztürk, 2018).

In conclusion, all Black Sea and Mediterranean countries should be constantly prepared for all kinds of marine invasive species, with national and regional coordination mechanisms in place, strategies developed for defining issues and risks, and backed up by appropriate action plans and public awareness campaigns. UNEP-MAP-RAC/SPA (2011b) has already developed a feasibility study for such purposes. Needless to say, prevention measures against such invasion of alien species should be elaborated where possible, although they may not be feasible in many cases. Once a species is introduced, it may be too late to employ any measures to reduce its impacts.

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# 8. Glossary

**Ballast water:** Water carried by a vessel to adjust its overall weight and internal distribution in order to improve stability.

**Biofouling:** Accumulation of waterborne organisms, such as bacteria, protozoa or other marine life, on the surface of mechanical structures in water, that contributes to corrosion or deformation of the structures or damages them.

**Casual species:** Non-indigenous species that do not form self-sustaining populations in the invaded region.

**CPUE:** Catch per unit effort, i.e. the amount of catch taken per unit of fishing effort, is an important performance indicator of fishery health and an index of abundance.

**Cryptogenic species:** A species whose origins are unknown. In ecology, a cryptogenic species is one that may be either a native species or an introduced species, as long as clear evidence of its origin is absent.

**Ctenophora:** A phylum of marine gelatinous invertebrates, commonly know as comb jellies. Most species have eight comb rows of fused cilia arranged along the sides of the animal.

**Euryhaline species:** A species tolerant to a wide range of salinity.

**Eurythermal species:** A species tolerant to a wide range of ambient termperatures.

**Fouling:** An assemblage of organisms growing on the surface of floating or submerged man-made objects (such as pilings or ship hulls).

**Macrophyte**: An aquatic plant large enough to be seen by the naked eye.

**Non-indigenous species:** A non-indigenous species (exotic, non-native, alien) is a species, a sub-species or a lower taxon occurring outside of its natural range (past or present) and dispersal potential (i.e. outside the range it occupies naturally or could occupy without direct or indirect introduction or care by humans). It includes any part, gametes or propagule of such species that might survive and subsequently reproduce.

**Non-indigenous invasive species:** A non-indigenous species that becomes established in natural or semi-natural ecosystems or habitat, is an agent of change, and threatens native biological diversity. However, there is no uniform terminology for a non-indigenous species and some organizations use different definitions, such as the United Nations Environmental Programme (UNEP), the International Council for the Exploration of the Sea (ICES) or the United States Environmental Protection Agency (EPA).

**Lessepsian migrant:** Term used for the first time by Por (1969, 1971) to define marine species of Indo-Pacific origin that have passed through the Suez Canal and settled in the eastern Mediterranean.

**Lessepsian migration:** Unidirectional migration of marine species of Indo-Pacific origin passing from the Red Sea to the Mediterranean via the Suez Canal.

# NON-INDIGENOUS SPECIES IN THE MEDITERRANEAN AND THE BLACK SEA

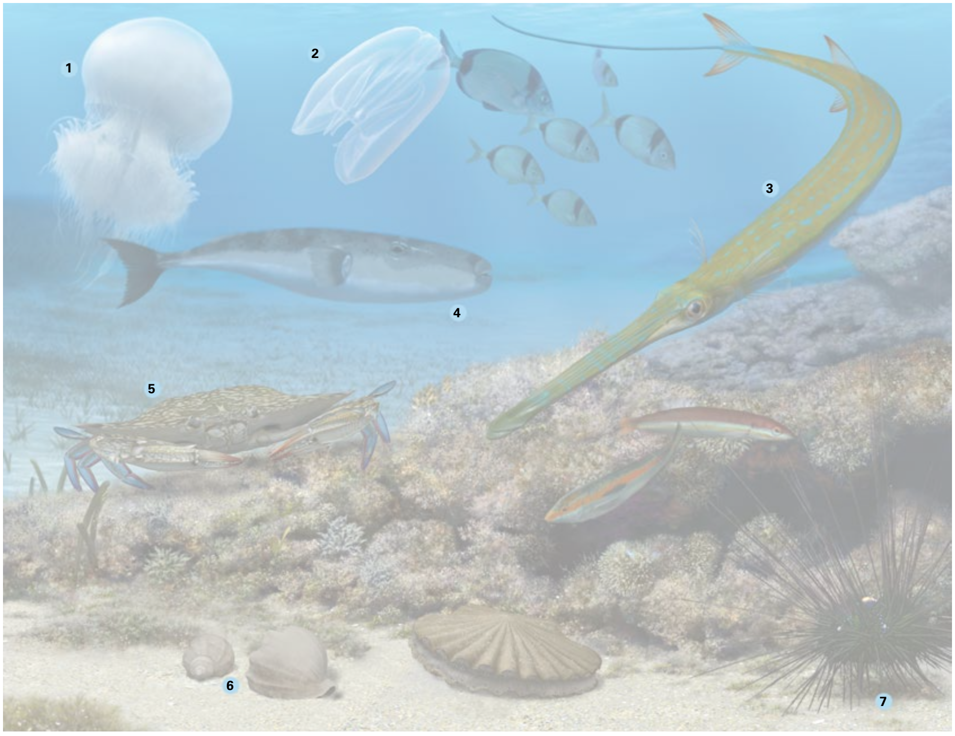
Recent decades have seen significant changes in the biota of the Mediterranean and the Black Sea due to the introduction of non-indigenous species. Reliable scientific data on the dynamics of their distribution and abundance are essential to understand their ecological and economic effects. This review – in addition to providing images and descriptions of relevant species to aid in identification – presents a unique historical and regional perspective on these species' impacts, based off many years' worth of research.

The Black Sea's primary invaders come from the Mediterranean. Species like the comb jelly *Mnemiopsis leidyi* have caused major declines in biodiversity in the region by crippling key segments of the food chain. Similar results have been noted in the Marmara Sea, a crucial water exchange point located between the Aegean Sea and the Black Sea.

Infiltration into the Mediterranean comes from both the east and west – with Lessepsian species passing through the Suez Canal and fish and invertebrate species originating from the Atlantic expanding their ranges. As of the publication of this review, over 900 non-indigenous species have been reported in the Mediterranean and almost 300 in the Black Sea, with these numbers expected to rise in the future.

Numerous Lessepsian fishes are commercially relevant and have been absorbed into local markets, particularly in the eastern Mediterranean region. While these species are targeted through various fishing techniques, many others are simply discarded due to a lack of value and there are even some, such as lionfishes, pufferfishes and several species of jellyfishes, that present immediate dangers to human health.

Stewardship of native species, regional cooperation on the enforcement of legal measures, increased public awareness and the creation of marine protected areas are thus essential to minimize and reduce the impacts of non-indigenous species both in the Mediterranean and the Black Sea.




1. *Rhopilema nomadica*; 2. *Mnemiopsis leidyi*; 3. *Fistularia commersonii*; 4. *Lagocephalus sceleratus*;
5. *Calinectes sapidus*; 6. *Rapana venosa*; 7. *Diadoma setosum.*

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