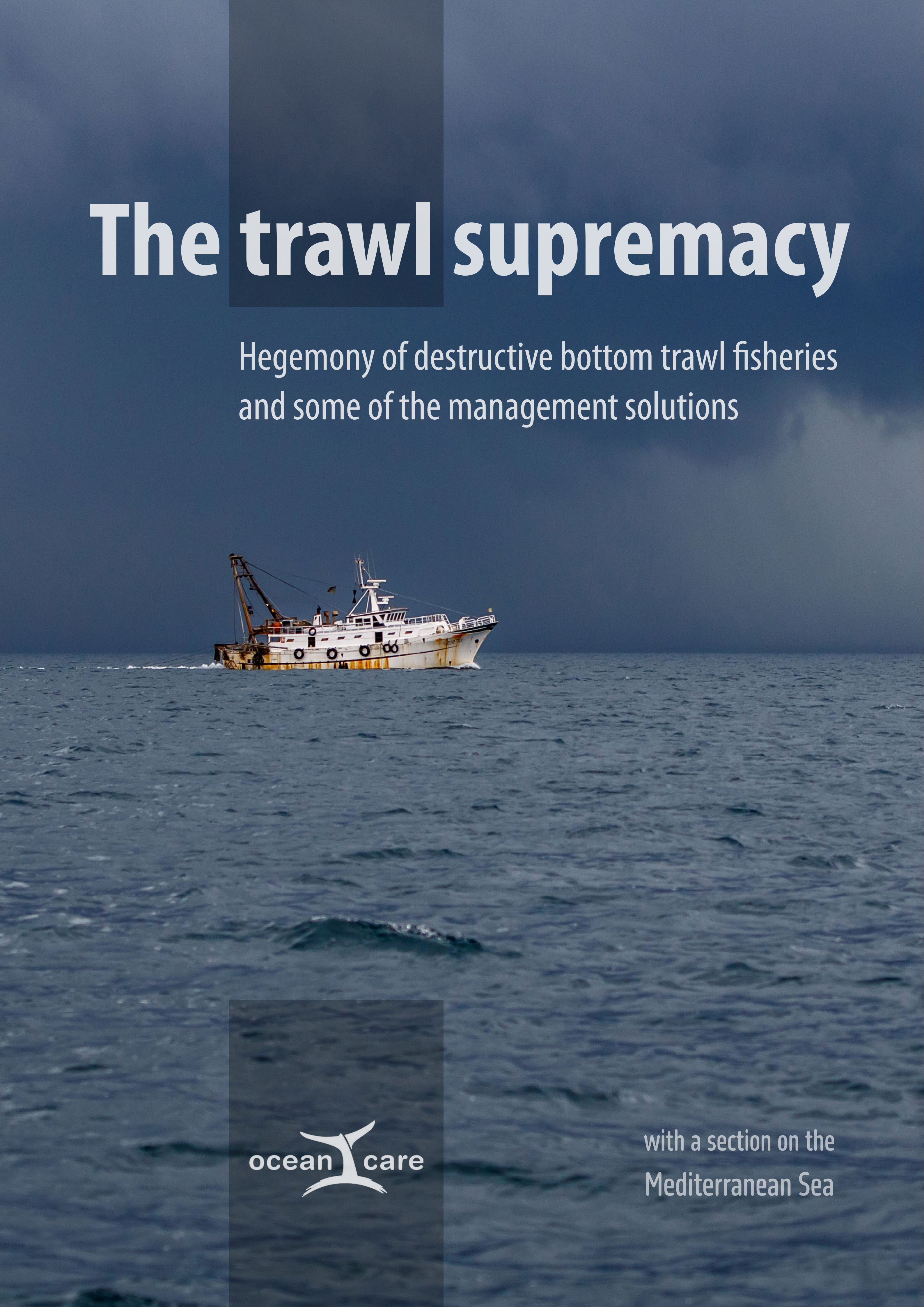
The trawl supremacy

Hegemony of destructive bottom trawl fisheries and some of the management solutions



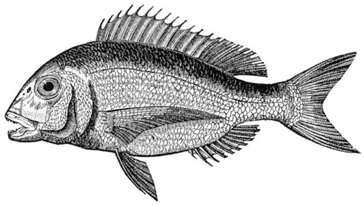
ocean care

with a section on the Mediterranean Sea

Authors

Giovanni Bearzi[1,2], Silvia Bonizzoni[1,2], Randall R. Reeves[3,4]

[1] Dolphin Biology and Conservation, Italy
[2] OceanCare, Switzerland
[3] US Marine Mammal Commission, USA
[4] IUCN-SSC Cetacean Specialist Group and Okapi Wildlife Associates, Canada



Acknowledgements

We are grateful to Jason Baker, Rohan Currey, Nicolas Entrup, Mariana Ferreira Da Silveira, Pingguo He, James Kerry, Fabienne McLellan, Chiara Piroddi, Tommaso Russo, Mark P. Simmonds and an anonymous reviewer, all of whom provided edits, suggestions or feedback to improve this report or parts thereof (contribution does not imply endorsement of the final text). We also thank Sigrid Lüber and everyone at OceanCare. Vladimir Burkanov and Marta Picciulin contributed useful information.

This report benefited from several literature reviews and summaries compiled by other authors. Of special importance were reviews on the characteristics of fishing gear (He et al. 2021), the best practices for managing impacts of trawl fishing (McConnaughey et al. 2020), the scale, context and impacts of bottom trawling (Steadman et al. 2021), and the sustainability and environmental impacts of trawling compared to other food production systems (Hilborn et al. 2023).

Credits

Front cover: a beam "rapido" trawler in the northern Adriatic Sea. Back cover: a bottom otter trawler in the northern Adriatic Sea. Both photos © Silvia Bonizzoni / Dolphin Biology and Conservation

Drawings of trawl gear from Seafish https://seafish.dash.app

Drawings of marine organisms adapted from Harter (1979) courtesy of Dover Publications

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ISBN 978-3-9525671-3-5

Required citation

Bearzi, G., Bonizzoni, S., Reeves, R.R. 2024. The trawl supremacy: hegemony of destructive bottom trawl fisheries and some of the management solutions. OceanCare, Wädenswil, Switzerland. 134 pp.

The trawl supremacy

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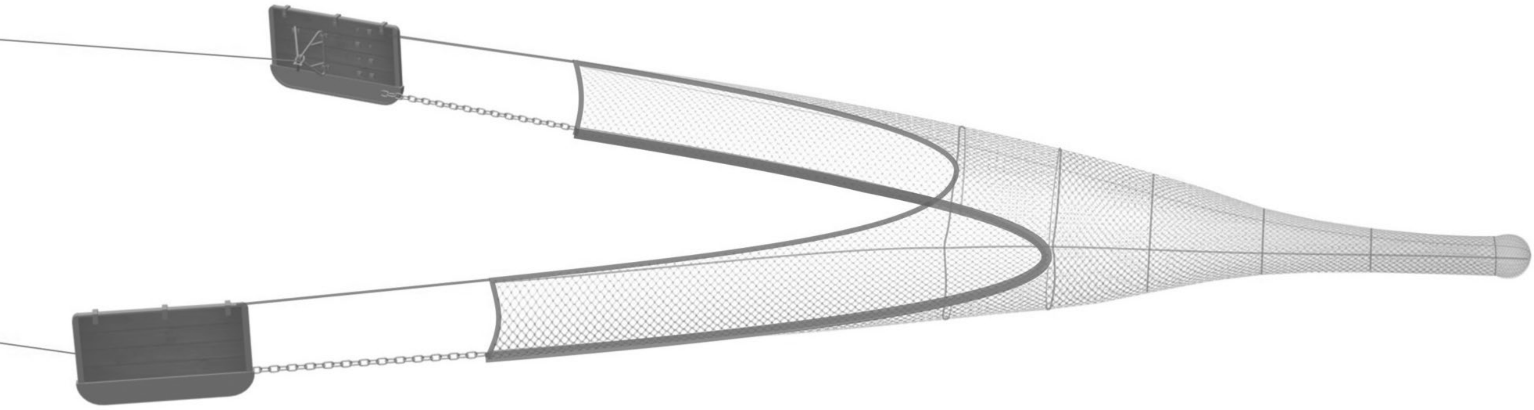

ISBN 978-3-9525671-3-5

OceanCare, Gerbestrasse 6, P.O. Box 372, CH-8820 Wädenswil, Switzerland

www.oceancare.org

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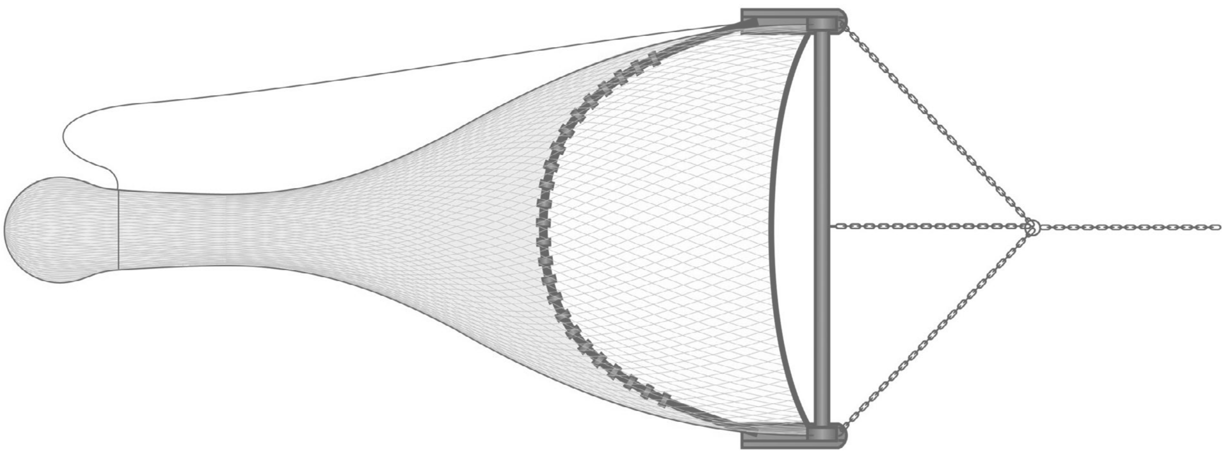
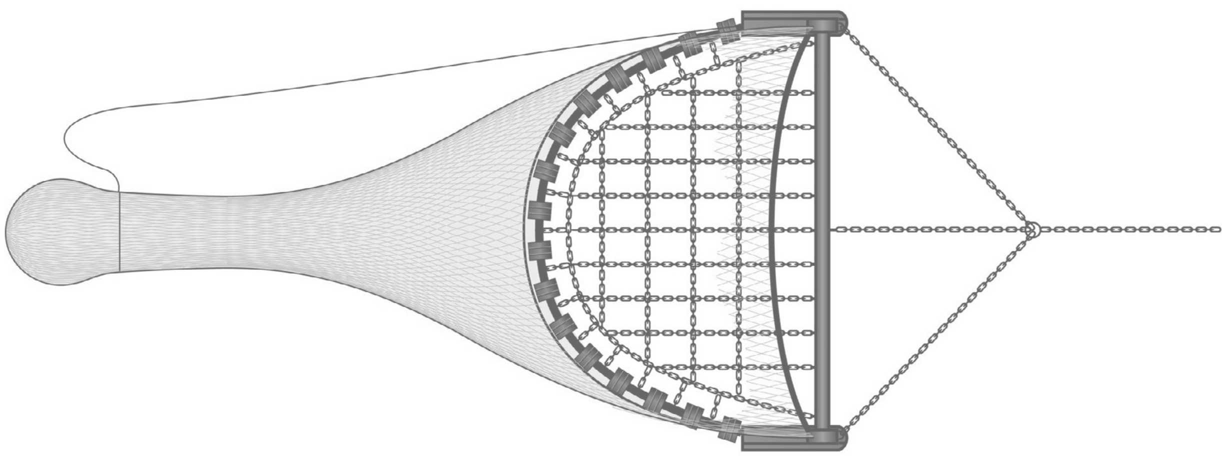




Foreword

by Fabienne McLellan, Managing Director, OceanCare

Our relationship with the Blue Planet is intimate and complex. We have the privilege of benefiting from a spectacular diversity of forms of life across seas and along coastlines. The ocean stabilises our climate, supplies food, medicines, energy and other resources. It supports people and national economies. The ocean is also an endless source of inspiration, recreation, rejuvenation and discovery, and it is a core component of the cultural heritage of many societies. Yet, our ocean is wounded and under threat.

The international community has been calling for collaboration and integrated governance to address our collective impacts on marine biodiversity. The United Nations aptly coined the term "triple planetary crisis" with reference to climate change, pollution and biodiversity loss. From an ocean conservation perspective, key pillars for addressing this crisis include, among others, the Paris Agreement, the Agenda 2030 and its Sustainable Development Goals, the Kunming-Montreal Global Biodiversity Framework, and the UN Agreement towards the conservation of Biodiversity Beyond National Jurisdiction. To heal our ocean's wounds, and turn the tide back towards a healthy and thriving ocean, wise management of human activities, including fisheries, is critical.

The destructive nature of bottom trawling has been extensively documented, with cumulative evidence of massive damage of many kinds. Regulations to reduce or altogether ban destructive fishing gear and practices such as bottom trawling within vulnerable marine ecosystems have been in place for quite some time. These range from policy work by the UN General Assembly to address the impacts of such gear and practices to the European Union's Action Plan to protect and restore marine ecosystems for sustainable and resilient fisheries. The latter points to a growing consensus within the European Union that bottom trawling has major and unacceptable environmental and climate impacts. Conservation and management measures aimed at closing certain areas to bottom trawling and prohibiting destructive fishing gear and practices have also been adopted by Regional Fisheries Management Organisations across the world.

Nonetheless, bottom trawling remains a primary way of catching fish in many parts of the world. It is indeed a global phenomenon and even occurs inside some Marine Protected Areas in different regions. The continuation of this practice, its scale, and the poor enforcement associated with it are among the main reasons that several civil society organisations have decided to take legal action to mitigate the environmental damage caused by bottom trawling.

OceanCare has long been willing to clarify and critically review the various impacts of bottom trawling worldwide. To that extent, we commissioned a broad study into the origins of this practice, its evolution, its distribution and scope throughout the world's ocean, as well as its operations, its targeted fish and other organisms, and its impacts. While the report resulting from this study should not be regarded by readers as a truly exhaustive and comprehensive account of bottom trawling, we hope it will provide a useful compendium that can serve as a basis for reflecting upon, and perhaps reconsidering, our relationship with the Blue Planet.

With the support of this report, OceanCare is determined to raise awareness about destructive fishing practices and their impacts, which extend far beyond marine species and ecosystems. For instance, the carbon footprint of bottom trawling must be taken into account when addressing the global problem of greenhouse gas emissions.

We invite you to read this report with an open mind, and we encourage you to support us in building momentum to end destructive fisheries, prevent further damage, and strive for meaningful recovery and restoration of vulnerable marine ecosystems.

Fabienne McLellan

Executive summary

Trawling is a type of fishing characterized by the active towing of nets by a moving boat. Trawl nets vary greatly in size and shape, and they target a wide variety of species, including bottom-dwelling fish, crustaceans and molluscs, pelagic and semi-pelagic schooling fish, and deep-water fauna. In this report, we provide a general overview on towed gear, but we focus more specifically on **bottom trawling: the towing of nets along the seabed**.

Bottom trawling has become a cornerstone of global food supplies, accounting for **more than one quarter of global fishery landings**. In 2016, this equated to over 30 million tonnes of seafood. In several European and African countries, half of fishery landings come from bottom trawling.

Bottom trawling, however, has long been known to be detrimental to marine life. It was regarded as a **destructive fishing method** since the early 14th century, and was often vocally opposed by communities of fishers who saw it as a threat to marine resources and their own livelihoods. The introduction of steam and diesel engines (in the 1830s and 1930s, respectively) marked the modern era of trawling. Engine-powered trawling increased rapidly during the 1960s, and by the 1980s large fleets of trawlers were combing the global oceans. Today's bottom trawlers can operate virtually anywhere, from shallow inland channels and rivers to deep offshore waters.

Countless scientific studies, encompassing decades of fishery research, have documented the harmful nature of bottom trawling, with substantial cumulative evidence of **damage to marine species and ecosystems**. Bottom trawling reduces the biomass, diversity and complexity of benthic communities, and the action of trawl gear on the seabed causes dramatic mechanical and chemical alterations, compromising the seabed's functionality and productivity. In addition to the target species, most types of trawl gear take unwanted species, such as threatened elasmobranchs, sea turtles, seabirds and marine mammals. Apart from these biological impacts, recent studies indicate that bottom trawling has a considerable **carbon footprint**, with high direct and indirect greenhouse gas emissions contributing to climate disruption.

Information on the harmful effects of bottom trawling has resulted in **public and institutional awareness of environmental damage**, and in restrictions that have sometimes included complete bans. Trawling is often prohibited in the most coastal and shallow waters. However, regulations and enforcement levels vary greatly across areas, and environmental protection measures are often ineffective—to the point that the intensity of bottom trawling can be higher inside than outside some Marine Protected Areas.

In this report, we review the evidence of **how bottom trawling affects marine life and human life**. We also summarize some of the primary **management approaches** that could help mitigate the harmful effects of trawling—consistent with international commitments to protect the marine environment.

We conclude that the amount of seafood produced by bottom trawling can no longer justify or excuse the pervasive damage caused to marine ecosystems and communities of small-scale fishers, and we advocate the use of **less destructive fishing gear**, combined with the creation of areas protected from harmful fishing practices, and more sustainable strategies to "feed the world".

A glance at global fisheries

1

Global fishery trends

Global capture fisheries (the harvesting of naturally-occurring organisms in both marine and freshwater environments) and aquaculture production have greatly expanded in the past seven decades, rising from 19 million tonnes in 1950 to 185 million tonnes in 2022 (excluding algae; FAO 2024; Fig. 1). Since the late 1980s, aquaculture has been a major driver of this growth. In terms of animal production, in 2022 aquaculture surpassed capture fisheries with 94 million tonnes, representing 51% of the world total and 57% of the production destined for human consumption (FAO 2024).

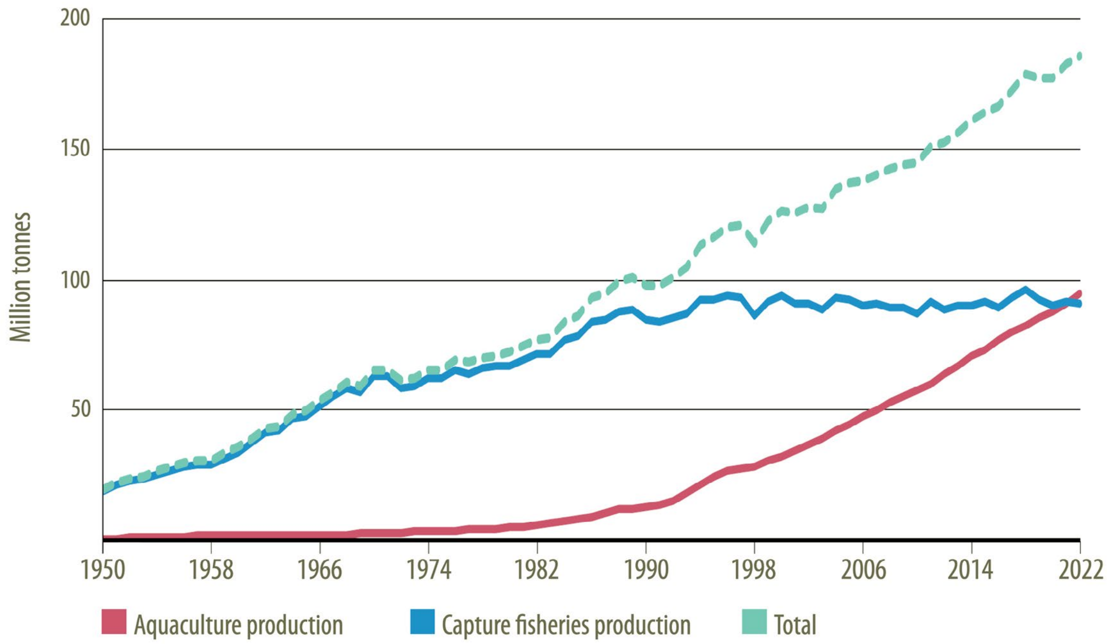


Fig. 1. Estimated global fishery production (including both marine and inland waters) and aquaculture production (excluding algae), 1950–2022 (adapted from FAO 2024)

Of the total aquatic animal production, 89% was used for human consumption. The remaining 11% of aquatic animal production went to “non-food uses”, mostly fishmeal and fish oil (FAO 2024). The global consumption of aquatic animal foods was approximately 163 million tonnes in 2021, with an average annual growth of 3% per year since 1961. Asia accounted for 71% of this consumption, followed by Europe (10%), Africa (8%), North America including Mexico (5%), Latin America and the Caribbean (4%), and Oceania (1%).

Globally, per capita consumption of aquatic animal foods increased from 9 kg per year in 1961 to 21 kg per year in 2021. However, the combined per capita consumption in Europe, Japan and the United States decreased (from 47 to 18% of the total consumption), whereas the combined per capita consumption in China, Indonesia and India increased (from 17 to 51%, with China alone accounting for 36%; FAO 2024).

In 2021, aquatic animal foods contributed at least 20% of the per capita protein supply from all animal sources to the world’s 3.2 billion people. Generally, non-high-income countries relied more heavily on protein from aquatic animal foods than high-income countries (FAO 2024).

Capture fisheries in marine waters have long been a major source of production, accounting for 87% of total aquatic animal production in 1950–1980, and 43% of total aquatic animal production in 2022 (FAO 2022b, 2024). In 2022, capture fisheries produced 91 million tonnes of aquatic animals (80 million tonnes caught in marine areas and 11 million tonnes in inland waters; FAO 2024). China was the top capture-fishery producer (14%), followed by Indonesia (8%), India and Peru (6% each), the Russian Federation and the United States (5% each), Vietnam (4%) and Japan (3%).

Of the 80 million tonnes of marine animals caught in 2022, approximately 85% was finfish, mainly anchoveta (4.9 million tonnes), Alaska pollock (3.4 million tonnes) and skipjack tuna (3.1 million tonnes). Catches of valuable species groups continued to increase, reaching a record 8.3 million tonnes for tunas and tuna-like species, 3.9 million tonnes for cephalopods, and 3.3 million tonnes for shrimps and lobsters (FAO 2024).

In 2022, the global fishing fleet was estimated at approximately 5 million vessels, of which two-thirds were motorized. Asia deployed the world's largest aggregate fishing fleet (71% of the total), followed by Africa (19%), Latin America and the Caribbean (5%), North America (2%), Europe (2%) and Oceania (<1%). Asia deployed the largest fleets of motorized (80%) and non-motorized (54%) vessels; (FAO 2024).

The majority of fish[1] populations considered by FAO to be fished "sustainably" are currently fished at maximum sustainable levels ("maximally sustainably fished" in Fig. 2). In 2021, 38% of populations were "overfished", 51% "maximally sustainably fished", and 12% "underfished" (FAO 2024). Overall, the percentage of populations fished at biologically or ecologically "sustainable" levels has been declining, whereas the percentage of populations fished at "unsustainable" levels has been increasing since the late 1970s (from 10% in 1974 to 38% in 2021).

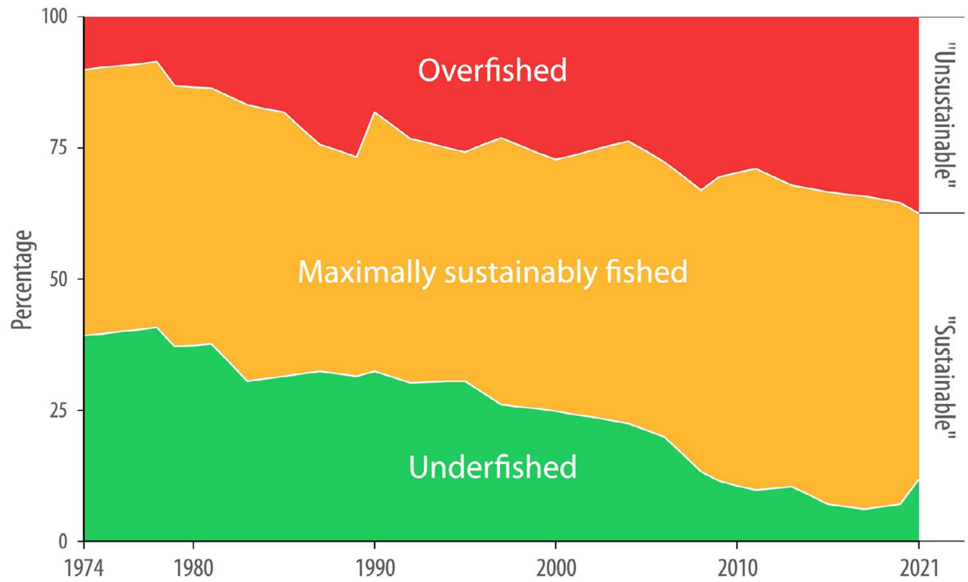


Fig. 2. Global trends in the state of the world's marine fish "stocks", 1974–2021 (adapted from FAO 2024); the graph shows a general pattern of worsening status trends encompassing four decades, with increasing proportions of overfished populations

The degree of exploitation, and the status of fish populations, vary greatly among areas (Fig. 3), with the Southern Pacific Ocean (FAO area 87) and Mediterranean and Black Sea (FAO area 37) having the highest proportions of populations being overfished (i.e. fished unsustainably), worldwide. In 2021, approximately 67% and 63% of the fish populations were fished unsustainably in these areas, respectively (FAO 2024).

1 In this report we often use "fish" in a generic way, meant to encompass "seafood" generally (particularly cartilaginous and bony fishes, crustaceans and molluscs).

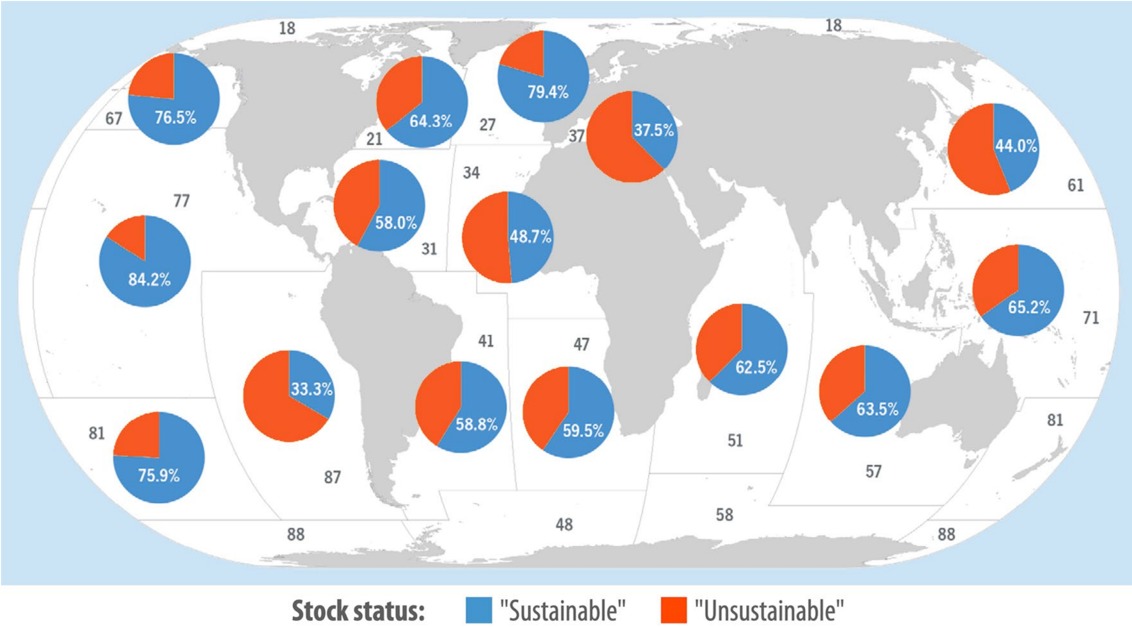


Fig. 3. Percentages, by FAO major fishing area, of biologically "sustainable" and "unsustainable" fish "stocks" in 2021 (adapted from FAO 2024)

Natural constraints

Fish resources were long thought to be inexhaustible because of the vastness of the oceans. The eminent 19th century biologist Thomas Huxley provided one of the most striking examples when he argued for easing fishery regulations, complaining that there was no scientific basis for them and that they unnecessarily hurt fishers (Smith 2002). Huxley went as far as making his notorious, influential and disgraceful proclamation on the infinite bounty of the sea:

> *I believe, then, that the cod fishery, the herring fishery, the pilchard fishery, the mackerel fishery, and probably all the great sea fisheries, are inexhaustible; that is to say, that nothing we do seriously affects the number of the fish. And any attempt to regulate these fisheries seems consequently, from the nature of the case, to be useless* (Huxley 1885).

Since then, fishing effort and technology have advanced, with fleets spreading across the world oceans and hunting marine species at ever lower trophic levels, in waters farther offshore (Pauly et al. 2003; Essington et al. 2006; Roberts 2007). It has become abundantly clear that no marine resource is inexhaustible, and regulating fisheries is far from useless.

Seafood is vital to global food security, and its importance is expected to grow (Watson and Tidd 2018; FAO 2024). However, in marine systems, seafood production is limited by natural constraints including the biology and demography of target species and the vulnerability of ecosystems to damage and overexploitation (Pauly 2019). Global fishery catches and economic performances are also constrained by primary production (Pauly and Christensen 1995; Chassot et al. 2010; Watson et al. 2013, 2015; Marshak and Link 2021) and oxygen concentration (Breitburg et al. 2018).

Natural constraints, natural variability, human-caused variability (including fishing pressure and climate change), and the cumulative impact of the extraction and production systems, all need to be taken into account in the management of modern fisheries. It is obviously an enormously complex challenge, which should be addressed with an appropriate degree of precaution, and with attitudes contrary to those expressed by Huxley.

Positive trends?

The United Nations Sustainable Development Goal #14 ("Conserve and sustainably use the oceans, seas and marine resources for sustainable development") set Target 14.4 in order to end overfishing by 2020. According to FAO (2024), however, world fisheries have diverged from this target, with overfishing having increased in recent years.

According to FAO's latest global assessment of fished "stocks"[2], trends in abundance and catch rates of marine capture fisheries have remained relatively stable since the late 1980s (FAO 2022b, 2024). Conversely, Pauly and Zeller (2017) argued that the vaunted stability of catches from wild capture fisheries (or "production" in FAO terminology) is a myth, and that global catches have actually been declining since peak global catches in the mid-1990s. Such criticism has led to additional and ongoing technical debates (e.g. Ye et al. 2017).

Hilborn et al. (2020) assessed 635 fished stocks worldwide and reported that, on average, stocks were increasing, reversing previous declines. The study showed an increase in fishing pressure accompanied by a decline in biomass until 1995, after which fishing pressure began to decrease. By 2005, the average biomass of some stocks had started to increase, and by 2016 the biomass across all stocks reviewed was, on average, higher than the estimated maximum sustainable yield (MSY)[3], while fishing pressure was lower than that which would be expected to result in MSY. However, such trend was not observed across all stocks assessed, and increases were not evident across all depleted fishery stocks (FAO 2022b).

While studies such as that of Hilborn et al. (2020) are indicative of some improvement in the management of global fisheries, information on the status and trends of a large share of the world's fish stocks is poor, and fishery management is deficient in many areas (FAO 2022b). A better understanding of the status of global fish populations is needed, with the greatest challenge being represented by unassessed fisheries, often in tropical and subtropical regions, where diverse fisheries support some of the world's communities that are most dependent on fish (FAO 2022b).

Irrespective of the uncertainty surrounding fishery trends, and how these trends are being interpreted, it must be emphasized that fishery catches are imperfect proxies for ecosystem status and health (Sguotti et al. 2022). Increased landings of target species, in particular, may tell nothing about the degree of damage to a marine ecosystem (see Chapter 5: Impact on target species). As described in several chapters of this report, bottom trawling causes impacts that go well beyond those on target species and populations.

Effective fishery management requires accurate estimates of stock biomass and reliable assessment of trends. However, assumptions in stock assessment models almost always bring high levels of uncertainty and error. Edgar et al. (2024) considered 230 fisheries worldwide, and found that reports of increasing trends for overfished stocks were often inaccurate. After accounting for biases, Edgar et al. concluded that 85% more stocks than the currently recognized number had likely collapsed to levels below 10% of their maximum historical biomass. The high degree of uncertainty and bias in modelled stock assessments indicates a need for greater precaution in the management of fisheries (Edgar et al. 2024).

And finally, there is much disagreement among fishery scientists and marine ecologists as to what "sustainability" actually means, with great differences in both perception and definition of the concept (Standal and Utne 2011; Hilborn et al. 2015). For instance, according to Pauly et al. (2002) the concept of sustainability upon which most quantitative fishery management is based is fundamentally flawed, and fisheries that destroy the habitat of the species upon which they rely are inherently unsustainable.

2 The use of the term "stock" with reference to wild animal populations has been questioned, with "population", "unit to conserve" and "community" being some of the possible replacements, depending on context. In this report, we use "stock" for clarity, when citing specific studies that adopt that term.

3 FAO (2024) defines maximum sustainable yield (MSY) as "The highest theoretical equilibrium yield that can be continuously taken (on average) from a stock under existing environmental conditions without significantly affecting the reproduction process. It is estimated using surplus production models (e.g. the Schaefer model) and other methods. In practice, however, MSY and the level of effort needed to reach it are difficult to assess." Another definition of maximum sustainable yield is "the highest average catch that can be continuously taken from an exploited population (= a stock) under average environmental conditions" (Tsikliras and Froese 2019). Another definition of maximum sustainable yield is "the highest average catch that can be continuously taken from an exploited population (= a stock) under average environmental conditions" (Tsikliras and Froese 2019).

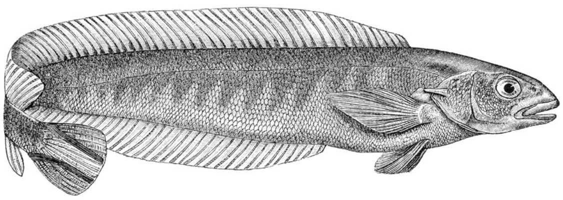
The diversity of trawling

2

What is trawling?

The term trawling generally refers to the active towing of a fishing net (a trawl) by one or two vessels called trawlers. Trawlers can tow fishing gear that touches the seabed (bottom trawlers) or mostly remains in the water column (midwater trawlers). A single trawler may tow one, two or more nets. In the case of pair trawling, the nets are towed by two boats. Trawl nets can be used to catch a variety of species, with vessel speed and mesh size being important determinants of the size and species of organisms caught. Specially designed netting panels, escape openings, or devices such as grids can be incorporated into a trawl net to reduce bycatch and enhance species and size selectivity (He 2007; He et al. 2021).

In this report, we provide a general overview on all trawl gear, but focus specifically on bottom trawls. Midwater trawls, which normally do not touch the seabed, are treated as a separate category: when specific information on midwater trawls is included, we normally qualify it as referring to this gear. However, we note that the term "midwater" may be misleading, considering that some midwater trawls do not fish only in midwater. For instance, the pollock fishery in Alaska uses midwater gear, but is estimated to be in contact with the seabed roughly half the time (Hilborn et al. 2023; Stratton and Wilson 2023). Dredges and seines are also described here, considering that these gears have impacts on the seabed similar to those of bottom trawls.



For images of trawl gear see
Trawl gear examples (pp. 89-94)

Types of trawls

Trawlers can range from small boats fishing in shallow coastal waters to large factory ships up to 150 m long, with fleets that can operate in oceanic waters thousands of metres deep and stay at sea for weeks or months (Bakkala et al. 1979; Clark et al. 2016; FAO 2021a). To work at sea for long periods of time, operating trawlers may be supported by "mother ships" and vessels including base ships that provide logistical support, factory ships for processing catches, refrigerated transport ships to replenish stores and to receive, freeze, and transport catches to home ports, as well as oil tankers, passenger ships, patrol vessels, and even hospital ships (Chitwood 1969; Bakkala et al. 1979; Kose 2010). While the smallest trawlers may only catch enough fish to feed a family, large factory freezer trawlers such as the Abel Tasman (formerly Margiris) or the Annelies Ilena (formerly Atlantic Dawn) can process up to 350 tonnes of fish a day and store up to 7,000 tonnes of catch (Kose 2010).

Trawl types can be subdivided into two general categories: bottom (or "demersal") trawls, and midwater (or "pelagic") trawls, and these categories include a variety of different gears. Semipelagic trawls are otter trawls that have either the trawl net or the otter boards touching the seabed, but not both (He et al. 2021). Discriminating among gears is important, because the type of gear (e.g. bottom versus midwater trawls) is one of the primary factors that determine the environmental impact of trawling. Much of the following summary is based on the FAO report by He et al. (2021).

Bottom trawls

Bottom trawls are towed gears that make sustained contact with the seabed, either touching, raking or intruding into it. They can operate at depths ranging from a few metres to more than 1,000 m. A bottom trawl is normally a cone-shaped net with a codend (i.e. the end of a trawl net that retains the catch), sometimes attached to a rigid structure, towed along the seabed to catch various bottom-dwelling and semi-pelagic organisms, including cartilaginous and bony fishes, molluscs and crustaceans.

The classification of bottom trawls depends primarily on the method to keep the net open (beams or otter boards), the number of nets deployed (single, twin or multiple), and the number of vessels towing the nets (one or two).

Bottom trawls often have components and "groundgear" that include heavy-duty ropes, "rockhoppers", discs, bobbins, chains and weights to ensure that seabed contact is maintained during fishing and damage to the net is minimized (He et al. 2021). The net's mouth may be kept open by a pair of otter boards (otter trawls), by a rigid beam (beam trawls), or by towing the net between two boats (pair trawls). Floats and weights, or a rigid frame, may also help keep the net mouth open. Some trawlers may tow two or more nets simultaneously (twin or multi-rig trawls), and one trawl net may have more than one codend to split the catch, reduce fish damage and facilitate handling large catches. Bottom trawls may be towed from a vessel's stern or from outriggers; in the latter case, an even number of trawls are towed to balance the load.

Bottom otter trawls

The net of a bottom otter trawl is kept open by a pair of otter boards, also called "doors", which are made of metal, wood, or a combination of these. Otter boards are towed ahead of the net and fixed with towing warps, so that the water force pushes them outward. Otter board design varies greatly, from simple rectangular wooden boards to sophisticated cambered metal or composite boards used to improve stability, efficiency or robustness.

The groundgear may include heavy metal bobbins for rough seabed, rubber discs of various sizes for softer seabed, light chains wrapped on the fishing line, or bare fishing lines with no protection when used over smooth seabed (He et al. 2021). Bottom otter trawlers may tow single or multiple nets: "twin" bottom otter trawlers have two otter trawl nets towed by one boat, whereas "multiple" bottom otter trawlers have more than two otter trawl nets towed by one boat. In addition, bottom otter trawls may be operated by either one or two boats.

The **single boat bottom otter trawl** is the most common type of trawl and it may be called "bottom otter trawl", "otter trawl", "bottom trawl", or "demersal trawl". A cone-shaped trawl is towed along the seabed by one boat, with its horizontal spread maintained by a pair of otter boards, and its vertical spread by floats along the headrope (also called headline), or simply by the height of the otter boards, while ground (seabed) contact is maintained by weighted groundgear which also protects the net from damage. Fish and other marine organisms may be herded into the net by the otter boards and by the sweeps and bridles. Therefore, the fished (or swept) area can be larger than the area swept by the net alone. Towing speed is generally between 3.5 and 7.5 km per hour, depending on target species.

A **twin bottom otter trawl** consists of two trawl nets towed along the seabed by one boat. These trawls may be called "twin trawls" and can be rigged as a two-warp system using a bridle arrangement, or as a three-warp system. Each trawl net is rigged between a central clump weight and one of the two otter boards. Similar to single trawls, sweeps and bridles connected to the otter boards or clump weight can help herd fish into the nets. In some cases, especially in shallow waters, one vessel may use outriggers to tow two separate trawls, each with its own pair of otter boards.

A **multiple bottom otter trawl** consists of more than two trawl nets towed along the seabed by one boat. These trawls may be called "multi-rig trawls" and can be rigged between two or more otter boards and clump weights or sledges. For instance, a triple-rig trawl typically uses two otter boards and two sledges, which open the three nets horizontally. A quad-rig system typically uses two pairs of nets, rigged as a three-warp system. These trawls may be towed from the stern or from outriggers.

Bottom pair trawls

A bottom pair trawl is a trawl towed over the seabed by two boats—normally vessels of similar power and size, with each vessel towing one cable and maintaining a steady distance from the other vessel. This provides the horizontal force needed to keep the net open without using otter boards (Selby and Evans 1997; Grieve et al. 2014; He et al. 2021). The trawl net is similar to single boat bottom otter trawls, but it is typically larger because of the higher power available from two boats.

Bottom pair trawls may be confused with pair seines, but the former catch fish through long-duration towing, whereas the latter catch fish mainly through encircling the fish while towing for a short time. Additionally, the shape of a pair trawl remains largely stable after the net starts fishing, whereas the shape of a pair seine changes constantly (He et al. 2021).

Beam trawls

The net of a beam trawl is kept open horizontally by a rigid beam across the net mouth. The beam is usually made of metal, and has steel frames (called "shoes") to let the beam slide over the seabed. Consistent contact with the seabed is ensured by the heavy weight of beam and shoes, as well as by chains (He et al. 2021). These chains run ahead of the groundgear and act as tickler chains to stir up fish burrowed into the seabed, such as flatfish, shrimp and demersal molluscs. Some beam trawls are towed at high speeds of up to 13 km per hour, to reduce the chance of fish escaping from the sides of the net and to exploit wider areas.

Beam trawlers can tow a single beam trawl from the stern of the vessel, or they can use outriggers to deploy multiple trawls simultaneously (Martín et al. 2014). A single vessel may tow up to 18 small beam trawls (Selby and Evans 1997). Contrary to bottom otter trawls (where single, twin and multiple trawls are classified separately), beam trawls are normally included in a single category, irrespective of how many nets are being towed (He et al. 2021).

A special type of beam trawler used in the Adriatic Sea has outriggers and tows up to four nets simultaneously at relatively high speeds of 13–15 km per hour (thence their name "rapido", meaning "quick" in Italian). In this case, trawl gear consists of a heavy metal frame, about 4 m wide and 30–40 cm high, with metal teeth along the lower leading edge and a cone-shaped net bag to collect the catch (Pranovi et al. 2000, 2001; Lucchetti and Sala 2012). Rapido trawls dig deep into the sediment, making furrows up to 10–13 cm deep (Lucchetti and Sala 2012).

Another special category is that of "skimmer" and "hang" trawlers, which use L-shaped or triangular frames (also functioning as outriggers) that allow the simultaneous towing of separate nets at both sides of a fishing vessel (Hein and Meier 1995). Skimmer trawls often fish the entire water column, from surface to bottom. While skimmer trawls may resemble boat-mounted stow nets (normally classified as "traps"), the former are towed on the side of a fishing vessel which moves forward, whereas stow nets are normally operated from an anchored boat (He et al. 2021).

Beam trawls using electric stimulation

Most beam trawls use mechanical stimulation (e.g. tickler chains) to flush or chase fish up from the seabed and concentrate them in the mouth of the net. While tickler chains improve catch efficiency, they also increase adverse impacts on benthic ecosystems, as well as friction and fuel consumption. To reduce these effects, tickler chains can be replaced by electrodes. This method, known as "pulse trawling", sends electric signals through a set of wires in front of the net, which stun and startle fish away from the seabed before they are scooped up in the net (Penca 2022). Electric stimulation evokes a startle or cramp response that prevents target species from escaping the approaching gear. However, the cramp response may lead to fractures and haemorrhages in both target and non-target organisms (particularly fish; e.g. de Haan et al. 2016).

Pulse trawling is intended to reduce fuel costs by reducing the friction caused by dragging metal parts (tickler chains or bobbins) along the seabed, while also reducing seabed penetration, mechanic disturbance, sediment mobilization and bycatch of benthic organisms (Soetaert et al. 2015; Sala et al. 2023; and see Chapter 10: Electric pulse trawling).

Midwater trawls

Midwater trawls are cone-shaped nets used to target epipelagic, pelagic and semi-demersal species. A single net may be towed through the water column by a single vessel or by two vessels of similar power and size. In the case of two vessels, there is a higher capacity to tow gear faster than single trawlers, and target fast-swimming fish. While midwater trawls are sometimes called "pelagic" trawls, they may operate in coastal and relatively shallow waters (making the name "midwater" somewhat ambiguous and potentially misleading). While the net can be towed very close to the surface (FAO 2021d), in shallow-water areas it may fish close to, or come into contact with, the seabed (Sala et al. 2018a).

In the Adriatic Sea, for instance, benthic taxa were caught in 12% of 496 midwater trawl hauls, indicating net proximity with the seabed (Casale et al. 2004). Midwater trawls targeting pollock in Alaska, in waters that are not particularly shallow, were estimated to be in contact with the seabed between 40% and 80% of the time, with contact rates of up to 100% on factory trawlers (Stratton and Wilson 2023), despite regulations intended to limit seabed contact to 10% of the time (McConnaughey et al. 2020). This practice has been characterized as "the myth of 'mid-water'" (Stratton and Wilson 2023). When midwater trawls are towed along the seabed, whether or not doing so is deliberate, the effects may resemble those of bottom trawling.

The nets of midwater trawls are typically larger than those of bottom trawls, and have high vertical openings. Towing speeds often range between 5 and 10 km per hour, though greater speeds may be required to catch fast-swimming species (He et al. 2021).

A **single-boat midwater otter trawl** is a trawl towed in the water column by one vessel using otter boards to spread the net horizontally (He et al. 2021). These trawls are used to catch pelagic schooling fish. For instance, important midwater trawl fisheries in the Northeast Atlantic (particularly in the Baltic Sea) target Atlantic herring *Clupea harengus* and Atlantic mackerel *Scomber scombrus*.

A **midwater pair trawl** is a net towed in the water column by two vessels, whose distance apart determines and maintains the horizontal spread of the net (He et al. 2021). The trawl's vertical opening is achieved using weights and floats, whereas net depth depends on towing speed and the length of towing warps. While midwater pair trawls target the same species as single-boat midwater trawls, the former are particularly suitable for catching epipelagic species (Thomson 1978; He et al. 2021). For instance, an important midwater trawl fishery in the Adriatic Sea targets European anchovy *Engraulis encrasicolus* and European pilchard *Sardina pilchardus*.

Semipelagic trawls

Semipelagic trawls can have two configurations: either the otter boards or the trawl net can touch the seabed, but not both simultaneously. If the system is configured with the boards off the seabed, efficient semipelagic otter boards with a high aspect ratio are often used. These consist of spreading otter boards that "fly" (or "jump") over the seabed, thus reducing gear penetration and therefore drag and sediment resuspension. If the system is designed to have the trawl net off the seabed while the otter boards are on the bottom, the net's groundgear may be lightened. Apart from these differences, the trawl net and otter boards of a semipelagic trawl resemble those of a single bottom otter trawl (He 2007; He et al. 2021; and see Chapter 10: Semipelagic otter boards).

A note on "target species trawls"

In the scientific literature, there is frequent reference to trawl categories other than those described above, with trawl gears being referred to according to their primary target species. Examples include shrimp (or prawn) trawls, hake trawls, haddock trawls, as well as broad characterizations such as haddock trawl fishery, Alaska pollock fishery, or Alaska groundfish fishery. In the absence of more accurate descriptions or first-hand experience with those gears and fisheries, it may be difficult to determine the precise type of trawl gear that is being discussed. In some cases, it can be a combination of different trawl gears deployed by a variety of fishing vessels.

Additionally, reference to a single target species can be misleading, because many other species may be caught, and target species may vary over time. In this report, we refer to standard trawl gear categories whenever possible but, in some cases, we adopt the gear designation of the original article to avoid misinterpretations.

A bottom otter trawler in the semi-enclosed Northern Evoikos Gulf, Greece; photo by G. Bearzi

Fishing gears similar to trawls

Demersal fishing gears other than trawls can be partly towed, and are designed to touch or intrude into the seabed. Dredges and seines, in particular, can produce effects similar to those of bottom trawls, and some of these fishing gears may be classified in the same category as bottom trawls. For instance, Hilborn et al. (2023) considered "shellfish dredges" as bottom trawls, and Danish seines (see the description below) are classified as bottom trawls under European legislation.

Dredges

A dredge is a cage-like structure often equipped with a scraper blade or with teeth on its lower part, either pulled or towed along the seabed to dig marine organisms out of the substrate and lift them into the cage or bag. Dredges can create furrows 15–16 cm deep (Lucchetti and Sala 2012; Hiddink et al. 2017), causing serious and extensive impacts on benthic fauna and seabed subsurface (Eigaard et al. 2016; Hiddink et al. 2017).

Several studies have described the impact of dredging (Caddy 1973; Gaspar and Chícharo 2007). For instance, dredging can have lethal impacts on fish eggs and larvae (Wenger et al. 2017), and adverse effects on seagrass meadows (Erftemeijers and Lewis 2006), maërl habitats (Hall-Spencer and Moore 2000) and corals (Erftemeijers et al. 2012). Molluscs struck or disturbed by dredges experience high levels of mortality (McLoughlin et al. 1991; Moschino et al. 2003; Ragnarsson et al. 2015). Dredges, however, are much less widespread than bottom trawls (Steadman et al. 2021), and their spatial footprint is consequently lower—though it may be higher at a local level. For a comprehensive review on dredge fisheries and technologies see Beentjes and Baird (2004).

Towed dredges

Towed dredges have cage-like structures made of robust metal frames that are towed behind a boat. Dredges may have teeth along the lower edge of the frame, as well as wheels to help them move along the seabed. The basket-like bag, attached to the metal frame, may be made of interlocking metal rings, chains, nets, or synthetic netting. The number of dredges may vary from a single dredge towed behind the boat, to four dredges towed from each side of the stern, or up to as many as 18 dredges towed from outriggers. Common targets include clams, oysters and scallops (He et al. 2021).

Mechanized dredges

Mechanized dredges, also called “hydraulic dredges”, are large metal cages used in combination with high-pressure hydraulic pumps to siphon marine organisms off the sediment and into the cage (He et al. 2021). Common target species include molluscs such as mussels, oysters, scallops and clams, that are scooped up by the dredge and delivered onboard either via a conveyer belt, by using a suction pump, or by bringing the dredge itself to the surface. Dredges may be towed at slow speeds during fishing, especially with larger vessels. In some cases, a large anchor may be set and the dredge fishes around the anchoring position.

Hand dredges

Hand dredges are small, light, hand-operated cages with a handle and a metal frame. The frame may have teeth on its lower edge, attached to a bag made of synthetic netting or wire mesh. The hand dredge may be pulled manually by wading, or from a small boat in shallow waters. Typical target species are clams, oysters and mussels. Hand dredging may be confused with raking, with the main differences being the steadier motion of dredging and the attachment of the bag to the hand dredge’s frame (He et al. 2021).

Seines

Seines can be either cone-shaped nets with long wings and a codend, or long pieces of net without a codend. They catch fish by encircling and herding. A seine net is usually framed by a headrope along its upper edge and a weighted footrope along its lower edge. The wings are often elongated and used in conjunction with long ropes. The bunt is typically in the centre of the net and can consist of a netting bag similar to a trawl codend, but some seines do not have a bag. Seines can be set from one or two boats (boat seines) or from shore (beach seines).

Boat seines

A boat seine is a cone-shaped net with elongated wings, seine ropes and a codend, which captures fish by encircling and herding. Compared to trawl nets, seine nets usually have longer wings and use long heavy ropes that extend from the wings through a pair of bridles to increase the area over which fish are herded. Seine nets also differ from one another in the way they are operated. The path swept by a seine’s ropes and net changes constantly during fishing. Conversely, operating trawls maintain a largely consistent shape. In addition, boat seines are towed more slowly and for a shorter time than trawls. Boat seines can be operated by either one or two boats (pair seining), and are used to catch demersal species including flounder, sole or cod (He et al. 2021).

Two variations of one-boat seines are the Danish seine and the Scottish seine, widely used around the world. Danish seines, also known as “Danish anchor seines”, are a type of active demersal fishing gear that does the hauling, in the same way as a trawl, but while the vessel is at anchor (Eigaard et al. 2016). Under European legislation, these seines are assigned to the same category as demersal trawls (Council Regulation EC 850/98; Noack et al. 2019). Scottish seines, also known as “fly-shooting” seines, are a hybrid method between anchor seining and demersal otter trawling (Eigaard et al. 2016): they do not use an anchor and the vessel moves forward during the retrieval phase (Noack et al. 2019). Compared to Danish seiners, Scottish seiners are usually larger and use larger nets (Noack et al. 2019).

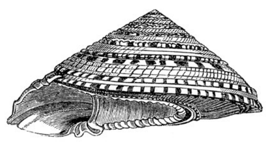
When seining is done by two vessels (pair seining), the second vessel picks up the highflyer buoy dropped by the main (shooting) vessel, or receives the end of the seine rope from the shooting vessel. Then the main vessel encircles a large area, and the two vessels slowly tow the net while gradually closing it. Towards the end of the haul, the rope in the second vessel is passed back to the main vessel where the codend is hauled onboard (He et al. 2021). Pair seining is widely used in Finnish inland waters, primarily for vendace *Coregonus albula*. In Europe and North America, pair seining evolved from Scottish seining in the 1960s, and primarily targets flounders (Walsh and Winger 2011). The gear, operation and variations of seine fisheries were extensively described by Thomson (1978, 1981). For more information on the history, operation and environmental impacts of boat seining see Walsh and Winger (2011).

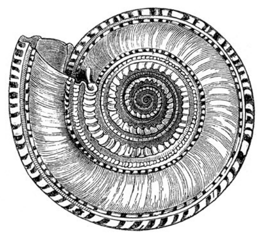
Beach seines

A beach seine is a long-winged net with or without a codend, that encircles fish in shallow waters, typically from a beach or from a boat anchored near (or on) the shore. The seine may be set from a small boat or by hand (wading) and hauled to the beach or onboard the anchored boat. The net usually extends from the surface to the seabed, preventing fish from escaping from the area surrounded by the net. Beach seines may be hauled by hand, by vehicles on land, or by winches or other machinery installed on shore or on the boat. Historically, beach seines could be hauled by horses or other animals. Beach seines are widely used in small-scale fisheries. For a review of beach seine fisheries see Tietze et al. (2011).

Other seines

Purse seines and other types of surrounding nets normally do not touch the seabed (although they might when used in shallow waters). These nets are often lumped together in a different category, i.e. “surrounding nets” (He et al. 2021), and they are not considered in this report.





Industrial and small-scale fisheries

The terminology of capture fisheries is often simplified and subdivided into “industrial” (or large-scale) and “small-scale” (or artisanal) fisheries (Thomson 1980; Jacquet and Pauly 2008). While these two broad categories may be presumed to be distinct and mutually exclusive, exactly where to draw the line has long been debated (Pauly and Charles 2015; Rousseau et al. 2019), and the variants are best understood as elements “along a continuum rather than as belonging to hard and fast categories” (Smith and Basurto 2019, p. 4). Such variants may include the size and type of boat, engine horsepower, equipment type, time commitment, catch rates, disposal (or use) of catches, significance of fishing as a livelihood, and marginality (Smith and Basurto 2019).

Generally, however, industrial fisheries have more powerful engines, larger crews and relatively heavy, mechanized gear, whereas small-scale fisheries (including the artisanal, subsistence and recreational sectors) tend to involve smaller vessels and crews, and have a more local, community-level orientation (Smith and Basurto 2019). Bottom trawl fisheries are normally viewed as industrial and are referred to as such in this report.

Footprint, impact and baselines

Bottom trawling physically modifies the seabed and disturbs benthic communities, producing a distinct *footprint*. A footprint is defined here as “the area of seabed trawled at least once in a specified region and time period, with area trawled determined from gear dimensions and tow locations” (Amoroso et al. 2018). Such footprints depend on factors including the distribution and catchability of target organisms, the technical capacity of the trawl fleet, production costs and market prices, the seabed’s ruggedness and composition, other environmental conditions (e.g. weather), the degree of fishery development and any prevailing management measures (McConnaughey et al. 2020). As these factors vary in space and time, the local or regional footprint of trawling can vary from year to year, although it tends to be consistent at broader geographic scales (Amoroso et al. 2018; McConnaughey et al. 2020).

The actual *environmental impact* of bottom trawl gear depends on the gear type and design, how the gear is operated, the penetration depth of the gear, the frequency and intensity of trawling, the sensitivity of the affected ecosystem and benthic community, the susceptibility of the affected species to mortality and injury, and the ability of affected populations to recover (Rijnsdorp et al. 2016; Eigaard et al. 2017; McConnaughey et al. 2020).

To assess the environmental impact of bottom trawling within a given area, one should be able to compare the situation in the present with a pre-trawling *environmental baseline* in the past. Defining such a baseline, however, requires arbitrary decisions, e.g. deciding how far back in time the baseline should be set. As some bottom trawl fisheries have continued for centuries, whereas others date back only a few decades, baselines normally need to be fishery- and area-specific. Additionally, and perhaps more importantly, baseline data on seabed habitats and demersal communities are typically scant and often biased, challenging the process of defining impacts and recovery standards (Klein and Thurstan 2016; Schijns and Pauly 2022).

Considering that the first passes of towed demersal gear over a given area tend to cause the most seabed damage (Cook et al. 2013; Sciberras et al. 2018; Currie et al. 2020; Mazor et al. 2021), the majority of the damage may have occurred in the early years of the trawling. Intensive trawling for decades may have erased vulnerable species from the area, damaging their habitat to a point of no return. Research in the fields of historical ecology and environmental history (Thurstan et al. 2010, 2014; Thurstan 2022) can provide credible historical baselines of marine biodiversity and environmental conditions, which can aid in assessments of the effects of trawling and the setting of meaningful conservation goals.

Trawlers in the port of Pila, northern Adriatic Sea; photo by G. Bearzi

The hegemony of trawling

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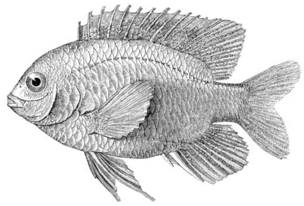


The rise of trawling

The first clear historical references to fishing gear comparable to present-day beam trawls and dredges date back to seven centuries ago. The earliest records come from Italy, France and Great Britain in the 14th century (Roberts 2007; Pitcher and Lam 2015; De Nicolò 2016, 2018) and from the Netherlands and Japan in later centuries (de Groot 1984; Engelhard 2008). In the Mediterranean Sea, hand-operated towed gears resembling trawls are described in reports dating back even farther (De Nicolò 2018).

Early trawlers were wooden vessels equipped with sails or oars, operated relatively close to shore (Robinson 1998; De Nicolò 2016). These types of trawlers became progressively widespread (Coull 1994; Pitcher and Lam 2015; Jones 2018) until the advent of the first steam-powered trawlers, that were built in France in the 1830s (Martín et al. 2014). Steam engines transformed the trawling industry and triggered its rapid expansion across the globe (Coull 1994; Roberts 2007). Driven by the industrial revolution in Europe and North America, steam-powered bottom trawlers started being designed to operate at industrial scales, and in waters ever farther away from port (Engelhard 2008).

In the early 1880s, steam trawlers spread across the North Atlantic and North Pacific, leading to massive increases in catches and expansion into previously unfished areas where target fish such as herring, flatfish and cod could be easily caught (Chitwood 1969; Bakkala et al. 1979; Pitcher and Lam 2015). By the early 1900s, the main fishing ports of Britain, Germany and the Netherlands possessed large fleets of steam trawlers (Robinson 1998), and in the following decades these trawlers became increasingly common around the world (Bakkala et al. 1979; Pitcher and Lam 2015; Novaglio et al. 2020). For instance, by the 1940s more than a thousand British trawlers roamed the North Sea, with a peak of about 1,500 vessels (15-tons gross and over) in 1920 (Engelhard 2005). Otter trawls were developed from beam trawls in the 1890s, replacing the rigid beam with otter boards that allowed for a wider spread of the net's mouth (Graham 2006).

From the mid-1930s, steam-powered trawlers were progressively replaced by trawlers with diesel engines, and in the 1940s and 1950s technological improvements increased the catching potential and the mobility of global trawl fleets (Engelhard 2008; Pitcher and Lam 2015). While the two world wars slowed down the rise of industrial trawling in some areas, post-war inventions and innovations resulted in further global development (van Hoof et al. 2020). Stern trawlers with an "A frame" to lift the trawl net were invented in Europe just after World War II, and the first freezer trawler—a prototype of modern factory freezer trawlers (Engelhard 2008)—began operations in the late 1950s (Pitcher and Lam 2015).

Trawling increased rapidly in the 1960s, and by the 1980s large fleets of trawlers were combing the global oceans (Bakkala et al. 1979; Watson et al. 2006). The geographic range of global industrial fleets has continued to expand, with bottom trawl catches growing from less than eight million tonnes per year in the 1950s to a peak of 36.5 million tonnes in 1989 (Steadman et al. 2021).

Bottom trawling played a major role in the industrialization and globalization of fisheries, becoming a cornerstone of fishing economies in Asia, Europe, North America and West Africa (Pitcher and Lam 2015). Modern bottom trawlers can operate virtually anywhere, and have been reported in areas as diverse as Australian lake and river systems (Liggins and Kennelly 1996), the shallow inland channels and estuaries in Galveston Bay, Texas (Dellapenna et al. 2006), and oceanic offshore waters up to 3,000–4,000 m deep (Clark et al. 2016).

Global trawl catches

Humans have exploited fish populations for millennia, but it was only since the 1950s that pressure by industrial fisheries aided by technological innovations turned fish and other marine organisms into global commodities (Pitcher and Cheung 2013; Pitcher and Lam 2015). Bottom trawl catches increased steadily between the 1950s and the late 1980s, when catches started to decline (Watson et al. 2006; Watson and Tidd 2018; Steadman et al. 2021; Fig. 4).

The vast majority of catches by bottom trawlers have come from the Exclusive Economic Zones (EEZs) of coastal countries (Steadman et al. 2021). According to FAO data, bottom trawl landings between 2011 and 2013 averaged 19–20 million tonnes per year, corresponding to 23–24% of total annual marine wild-capture landings (Amoroso et al. 2018). Consistently, analyses of reconstructed global catches based on the Sea Around Us dataset (seaaroundus.org) for the years 2007–2016 indicated that bottom trawling accounted for 26% of global catches, with significant variations across countries and regions. In 2016 (the latest year of reconstructed catches), bottom trawl catches were approximately 31 million tonnes (Steadman et al. 2021). Based on these estimates, the total amount of seafood caught annually by bottom and midwater trawls appears to exceed the total seafood caught by all the world's small-scale fisheries (Steadman et al. 2021).

Main bottom trawling countries

Based on Sea Around Us reconstructed global catch data from 2007–2016, Steadman et al. (2021) estimated that more than 60% of global bottom-trawled fish was caught in Asia. In Africa, Asia and North America, approximately 21–29% of the total catch came from bottom trawling. In contrast, only a small proportion of the total catch in South America came from bottom trawling (4%), reflecting the region's reliance on small pelagic fish caught with other gear (Steadman et al. 2021).

The top ten bottom trawling countries contributed 64% of the global bottom trawling catch. Seven of these countries are in Asia: China, Vietnam, Indonesia, India, Japan, Republic of Korea and Malaysia. The three non-Asian countries are Morocco, United States and Argentina. China was responsible for the largest bottom trawl catch, accounting for 15% of the total bottom trawl reconstructed catch (Steadman et al. 2021). The Chinese catch increased from 1.4 million tonnes in 1985 to 5.2 million tonnes in 2015. Vietnam, which shares a border with China, had the second largest catch and the largest trawl fleet in Southeast Asia (approximately 20,000 vessels). Vietnam has experienced the fastest growth in bottom trawling since the 1970s, while both India and Myanmar have seen a fourfold growth (Suuronen et al. 2020; Steadman et al. 2021).

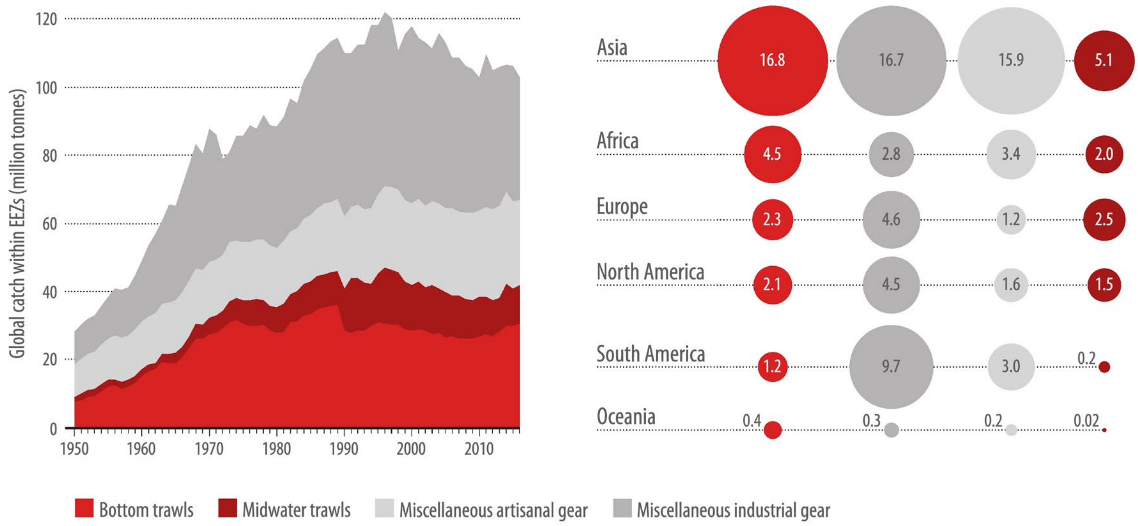


Fig. 4. Estimated global fish catches (million tonnes) within EEZs between 1950 and 2016, subdivided by gear type and main geographic area; numbers in the right figure indicate million tonnes per year (adapted from Steadman et al. 2021)

Bottom trawling has continued to grow rapidly in Asia, but in most other parts of the world it has been declining or staying constant. In Europe, the percentage of overall catch produced by bottom trawling has decreased relative to midwater trawling, with the latter having become more prominent since the late 1990s (Steadman et al. 2021).

Reliance on bottom trawling

In many countries, including several in Europe and Africa, more than half of the total fish catch comes from bottom trawling, indicating a high reliance on this practice. The Table below, from Steadman et al. (2021), shows the ten countries with the highest proportion of total catch from bottom trawling within their EEZ (only countries with at least 10,000 tonnes/year are included).

Country	Average bottom trawl catch (2007–2016) in millions of tonnes	Percentage of seafood catch from bottom trawling (2007–2016)
The Netherlands	0.09	65
Morocco	1.8	65
Somalia	0.1	64
Vietnam	2.3	59
Guinea	0.5	59
Côte d'Ivoire	0.1	56
Germany	0.9	55
Republic of the Congo	0.05	54
Guyana	0.03	53
New Zealand	0.4	53

While some countries rely heavily on bottom trawling, approximately half of all coastal countries have little or no trawling. Steadman et al. (2021) estimated that of 156 coastal countries, 73 (47%) had less than 10,000 tonnes per year caught in their EEZs through bottom trawling. These included South and Central American countries such as Colombia, Venezuela, French Guiana, Honduras, and Belize (which banned the practice in 2010).

Bottom trawling by distant countries

Several countries have bottom trawling fleets that operate well beyond their own EEZs, and into the waters of other countries. In particular, highly mobile Asian and European fleets have moved to ever more distant productive waters since the 1970s (Watson et al. 2013). These fleet movements have put stress on additional marine areas, including the EEZs of developing countries, often depleting local resources and competing with local fleets (see Chapter 7: Foreign fleet effects on local communities, and Chapter 8: The problem with subsidies). Specifically, Steadman et al. (2021) estimated that distant-water fishing fleets were responsible for 22% of all the catches by bottom trawlers within EEZs. These distant-water fleets operated primarily in the EEZs of Africa and Oceania, with more than half of bottom trawl landings in these continents being by vessels with Asian or European flags. In 34 countries, mostly in Africa, over 90% of the bottom trawl catch was by foreign vessels.

With few exceptions, the countries with the highest bottom trawl catches in their own waters, such as China, Vietnam, Indonesia, and India, also have high catches by their fleets operating in foreign waters. China has the largest distant-water fishing fleet in the world in terms of number of vessels (Kularatne 2020). The expansion of its fishing fleet has led to significant bottom trawling by Chinese vessels in most African countries, as well as in Japan, Vietnam, Philippines, and Republic of Korea (Xue 2006; Kim 2012; Mallory 2013; Pauly et al. 2014; Steadman et al. 2021). For instance, up to 75% of licensed bottom trawlers operating in West African waters in 2017 were registered in, or owned by, China (Virdin et al. 2022). Europe also relies heavily on catches by its own distant-water fleets, which operate predominantly in Africa (Belhabib et al. 2015; Virdin et al. 2019; Lin et al. 2021; Fig. 5).

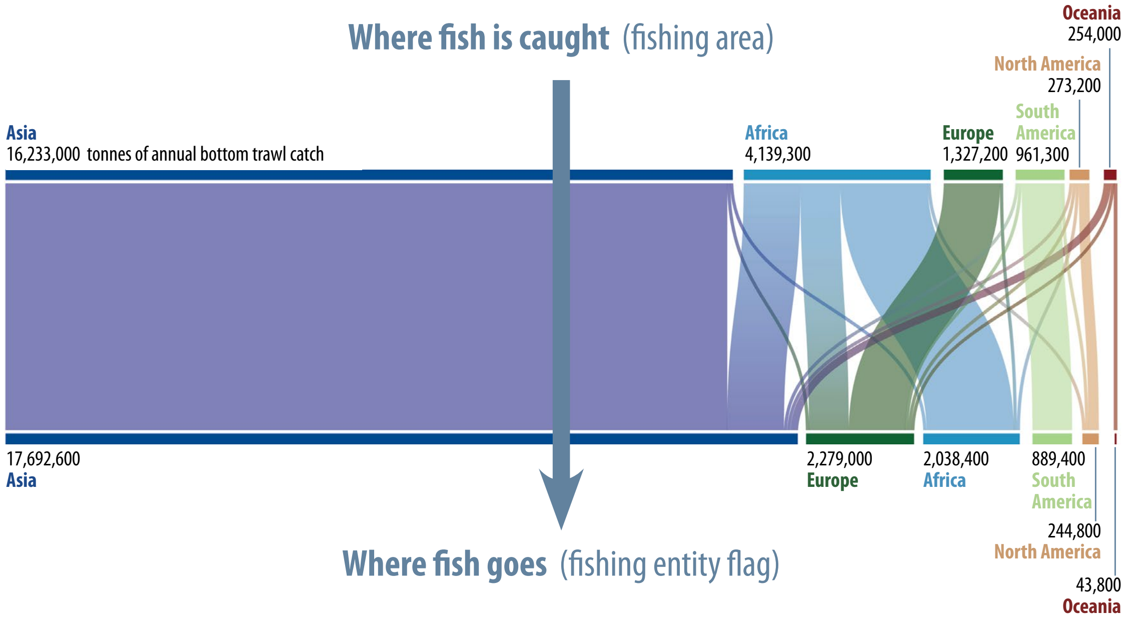


Fig. 5. Flow of annual bottom trawl catch, in tonnes, showing the catch displaced by foreign-flagged vessels (adapted from Figure 8 of Steadman et al. 2021; underlying data apparently apply to the period 2007–2016)

Geographic intensity and impacts of trawling

The effects of bottom trawling on marine life and marine habitats depend more on the intensity of trawling than on the size of the catch. Intensity can be estimated by dividing the total catch within a given area of sea by the size of that area. Steadman et al. (2021) reported the highest intensities of bottom trawling within the EEZs of Guinea, Guinea-Bissau, Morocco, Republic of Korea, Cambodia, Thailand, and Cameroon, suggesting that the seabed and benthos of these nations have been strongly impacted by trawling.

Another index of bottom trawling intensity is the swept area ratio, defined as “the total area swept by trawl gear over a defined time period (usually one year) divided by the total seabed area at a defined spatial scale (usually from grid cell to region)” (Amoroso et al. 2018). Based on this index, several European regions had the highest intensity of trawling of 24 regions included in the analyses. These intensively trawled regions included Adriatic Sea, waters west of Spain, Skagerrak and Kattegat (waters surrounded by Norway, Sweden and Denmark), Tyrrhenian Sea, Irish Sea and North Sea. The western Baltic Sea, Aegean Sea and waters west of Scotland also ranked high in this list (Amoroso et al. 2018), indicating a generally high intensity of bottom trawling in Mediterranean and Northeast Atlantic waters. Information from the Adriatic Sea, the region with the highest regional intensity, indicated that more than 70% of the seabed had been trawled, and more than half of the seabed area was trawled at least once per year (Amoroso et al. 2018).

Pitcher et al. (2022) used a quantitative indicator of biological state of seabed sedimentary habitats in 24 regions exposed to bottom trawling. Seabed status differed greatly among regions, indicating different intensities of trawling and fishery sustainability. Similar to the finding in Amoroso et al. (2018), eight European regions had the worst seabed status: Adriatic Sea, waters west of Spain, Skagerrak and Kattegat, Tyrrhenian Sea, western Baltic Sea, North Sea, Aegean Sea and Irish Sea. The Adriatic Sea had by far the poorest seabed status (Pitcher et al. 2022; Hilborn et al. 2023).

While levels of bottom trawling may not be particularly high in some areas, the degree of environmental damage caused by trawl gear upon a single passing (Cook et al. 2013; Sciberras et al. 2018; Currie et al. 2020; Mazor et al. 2021) implies that even low intensities or short durations of trawling can have large and lasting impacts (see Chapter 5: Ecosystem recovery after trawling).

The problem with trawling

4

A destructive fishing method

Bottom trawling has often been described as a non-selective fishing method that disrupts the marine ecosystem not only by removing target and bycaught species, but also by causing broad-spectrum collateral damage to the seabed and benthos. There is ample scientific literature providing evidence that bottom trawling is the fishing practice with the greatest negative impact on marine ecosystems (see Chapter 5). Trawl nets can mow down corals, biogenic reefs, sponges, seagrass meadows and other vulnerable habitats, leaving behind a bare trail. Mechanical impact of trawl gear on the seabed causes dramatic chemical alterations and reduces the biomass and diversity of benthic communities, compromising their functionality, productivity and complexity.

In many coastal areas, the impact of bottom trawling on marine life dates back decades or centuries, and it has resulted in substantial ecosystem changes and even regime shifts. For example, sensitive systems such as oyster reefs—once a dominant three-dimensional feature of European coastal waters—were essentially wiped out by trawling and dredging in historical times, with their loss resulting in seabed flattening. In the North Sea, enormous oyster reefs that extended widely two centuries ago were destroyed following the introduction of industrialized trawling and dredging between the late 19th century and the beginning of the 20th (Thurstan et al. 2013, 2014, 2024; Bennema et al. 2020). Hall-Spencer et al. (2002) documented widespread damage caused by trawling to cold-water coral reefs at depths of 800–1300 m off Ireland and Norway. Bycatch included large pieces (up to 1 m^2) of coral that had been broken from reefs, as well as a variety of coral-associated benthos. The trawled coral matrix damaged by trawling was at least 4550 years old (Hall-Spencer et al. 2002). In intensively-trawled areas, such as the Adriatic Sea (Pitcher et al. 2022; Hilborn et al. 2023), marine communities have experienced alterations and declines (Fortibuoni et al. 2010, 2017; Lotze et al. 2011; Barausse et al. 2021; Sguotti et al. 2022). For instance, elasmobranchs declined by >94% across 60 years, and 11 of 33 studied species could no longer be detected (Ferretti et al. 2013).

Because of impacts such as those above, and many others of the same type (see Chapter 5), bottom trawling has often been characterized as a "destructive" practice in academic, media and policy output—though use of the term destructive has been inconsistent and controversial (Willer et al. 2022). For instance, Watling and Norse (1998) compared bottom trawling with forest clearcutting[4]. Halpern et al. (2008; Table S4, p. 41) categorised bottom trawls as "demersal, destructive", along with dredges, whereas they categorised midwater trawls as "pelagic, high bycatch". In a subsequent article, bottom trawling was classified as being destructive and having high bycatch (Table S3, p. 10 in Halpern et al. 2019). A 2009 review of FAO's Code of Conduct for Responsible Fisheries, discussing progress on the prohibition of "destructive fishing methods and practices", referred to bottom trawls as "implicitly covered by the measure", and a gear "rubbishing benthic habitats worldwide" (Hosch 2009, p. 22). Hiddink et al. (2017) described bottom trawling as "the most widespread source of anthropogenic physical disturbance to global seabed habitats".

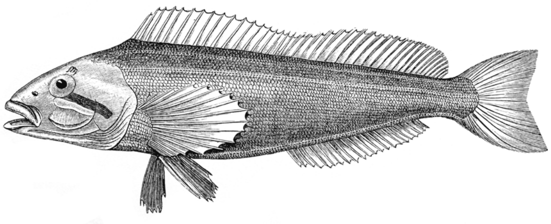
This understanding is increasingly embedded in legal documents, governmental reports and legislation. For instance, a recent proposal to phase out bottom trawling from EU Marine Protected Areas stated that "fishing using certain mobile bottom-contact gear (mobile bottom fishing), in particular bottom trawling, is among the most widespread and damaging activities to the seabed and its associated habitats" (EC 2023a).

4 Hilborn et al. (2023, p. 1572) noted that "The obvious flaw in this analogy is that, for the most part, the same areas are trawled each year, and indeed, in some cases, many times each year, but you cannot clearcut the same area twice". However, in their analogy with "forest clearcutting", Watling and Norse (1998, p. 1180) clarified that "structures in marine benthic communities are generally much smaller than those in forests, but structural complexity is no less important to their biodiversity". They also pointed out that "recovery after disturbance is often slow because recruitment is patchy and growth to maturity takes years, decades, or more for some structure-forming species". We note that after a portion of seabed has been trawled, demersal organisms start growing again, and with time can revert to a status resembling pre-trawling—if a trawler did not pass over them again and again. In that sense, the same area can be "clearcut" twice, or more than twice.

Early opposition to bottom trawling

For centuries, bottom trawling was perceived as being detrimental to other fisheries, and concerns were raised about its effects on the marine environment and the viability of the trawled fish populations. Reports of trawl bans date as far back as 1307 (De Nicolò 2016, 2018). Many of the historical references to trawling are actually vehement calls to restrict it (Pitcher and Lam 2015). For instance, the environmental damage and waste caused by early trawls with fine-meshed netting was so alarming to coastal communities that they petitioned the 1376 Parliament of King Edward III of England, and after subsequent inquiry a royal commission banned the use of this “new and destructive fishing method” (Roberts 2007).

In the following centuries, similar concerns about depletions of fish populations, often resulting in trawl bans, have been documented in European countries including at least Belgium, England, France, Italy, Scotland and the Netherlands (de Groot 1984; Coull 1994; Roberts 2007; Lentini 2010; Thurstan et al. 2014; Bennema and Rijnsdorp 2015; De Nicolò 2016, 2018; Jones 2018). In England, concerns culminated in a British Act of Parliament in 1714, under which illegal trawls were to be burned (Pitcher and Lam 2015).



The trawling debate

Bottom trawling has been attracting opposition and controversy for seven centuries, and the magnitude of impacts from bottom trawling remains the subject of intense scientific debate (Hilborn et al. 2023). Concerns about the impacts of trawling have also fuelled strong public campaigns, which have resulted in restrictions or bans of trawl gear in some countries and regions (McConnaughey et al. 2020).

In Europe, there seems to be a growing consensus that bottom trawling has major and unacceptable environmental, social, and climate impacts. For instance, several campaigns have been conducted in recent years to ban the practice within protected areas and inshore zones[5]. The degree of support for these campaigns suggests that at least some sectors of civil society are strongly opposed to bottom trawling, whereas powerful fishery lobbies demand that trawling continues to be supported and subsidized.

In the realm of fishery science and management, academic debates and polarised views are the norm. For instance, a 2003 survey revealed a broad consensus among U.S. fishery stakeholders that bottom trawling has the most severe impact in terms of bycatch and habitat damage (Chuenpagdee et al. 2003). However, Hilborn et al. (2023) contended that “the concerns about trawling impacts can be significantly mitigated when existing technical gear and management measures (e.g. gear design changes and spatial controls) are adopted by industry and regulatory bodies and the race-to-fish eliminated”.

These hopes of mitigation rely on the assumption that high standards of management can actually be met in trawl fisheries around the world. Such achievement would be far from straightforward in many or most countries; eliminating the race-to-fish will always be a challenge. Even if the necessary measures are adopted in principle by industry and regulatory bodies, and assuming such measures can indeed “significantly mitigate” trawling impacts, problems with compliance and enforcement are bound to remain. Unless and until the necessary measures are no longer merely aspirational high-level goals of government, and are accepted as rules of the game by fishing entrepreneurs, fishers and fishing communities, there will continue to be “concerns” about the impacts of trawling.

5 E.g. see eu.patagonia.com/gb/en/eu-marine-protected-areas.html and act.wemove.eu/campaigns/bottom-trawling

Steadman at al. (2021) provided a useful summary of the competing arguments in the current debate, exemplified by the Table below (the original text has been slightly amended).

Common anti-trawling arguments	Common pro-trawling counterpoints
Bottom trawls are unselective fishing gears that catch seafood indiscriminately.	There is no other way to catch these fish at a scale demanded by the seafood market. Bottom trawl fisheries can be managed to reduce environmental impacts.
Bottom trawling has negative impacts on the livelihoods, cultural practices, and food security of small-scale fishers.	Bottom trawling is an efficient way to catch seafood in order to meet market demand.
Bottom trawling causes widespread and often irreversible harm to marine seabed ecosystems.	Bottom trawling does cause damage, but the impacts are often not as bad or as widespread as is claimed. Certain areas that are already heavily trawled cannot recover and are viable locations for continued bottom trawling.

Sustainable bottom trawling?

One other point made by advocates of trawling relies on the fact that some trawl fisheries have been certified by the Marine Stewardship Council (MSC) as being "sustainable". Specifically, the meeting of MSC standards for 83 bottom trawl fisheries has been presented as evidence that all bottom trawling is potentially sustainable, if properly managed (Hilborn et al. 2023).

Evaluation of such sustainability requires an understanding of certification mechanisms. MSC is the most widespread and prominent fishery certification programme that uses "eco-labelling" to identify fisheries with low environmental impacts (Gutiérrez et al. 2012, 2016; Bush et al. 2013; Agnew et al. 2014). To achieve MSC certification, fisheries are expected to conform to standards that are subject to regular review (Jardim and Currey 2023). Some analyses have suggested that MSC certification promotes community development (Pérez-Ramírez et al. 2012b; Field et al. 2013), emphasizes scientific evidence (Butterworth 2016), and ensures environmental sustainability (Martin et al. 2012). For instance, Gutiérrez et al. (2012) compared the trends in status and abundance of 45 stocks targeted by fisheries certified by MSC as being sustainable with the trends of 179 stocks targeted by fisheries that were not certified, and found that 74% of the former were above biomass levels that would produce MSY, compared with 44% of the latter. On average, the biomass of stocks targeted by certified fisheries had increased by 46% over the previous 10 years, whereas the stocks targeted by uncertified fisheries had increased by 9%. Stocks targeted by certified fisheries also had lower mean exploitation rates. Gutiérrez et al. (2012) concluded that MSC-certified seafood was 3–5 times less likely to come from harmful fishing than uncertified seafood, and that MSC certification accurately identified healthy fish stocks and conveyed reliable information on stock status to seafood consumers.

However, entirely different results were reported in a study by Opitz et al. (2016). These authors examined the status and exploitation level of 31 northern European stocks targeted by fisheries certified by MSC as being sustainable and well managed, and found that about half of those stocks with available data were at or above the MSY level, and four were outside of safe biological limits in the first year of certification. No significant changes in fishing pressure or stock size were detected 1–10 years (average 4 years) after certification, for 28 stocks with available data. In the last certified year with available data, 44% of the stocks were subject to overfishing and five were outside of safe biological limits. Opitz et al. (2016) suggested that MSC rules should be changed so that 1) "overfishing" (defined as fishing pressure above that which can produce the MSY) or "unsafe" stock size (defined as stock size above that which can produce the MSY) leads to immediate suspension of certification, and 2) no certification is issued for a stock that is already in such a situation.

More recently, Melnychuk et al. (2022) compared published biomass estimates of fish stocks targeted by fisheries certified by MSC as being sustainable with those of stocks targeted by fisheries that were not certified. Individual stocks from both groups were highly variable in terms of relative biomass trends over the previous two decades. In the years 2014–2018, however, stocks targeted by fisheries certified by MSC as being sustainable had, on average, greater biomass than uncertified ones. The former were also less frequently overfished than the latter. Melnychuk et al. (2022) recognized that MSC had previously certified eight fisheries targeting overfished stocks as being sustainable, based on the scientific advice available at the time of certification. When revised stock assessments estimated the biomass to be lower than previously thought, fisheries for those stocks were suspended from certification. Melnychuk et al. (2022) concluded that MSC certification is associated with credible sustainability designations and, as such, provides a useful way of distinguishing between seafood that was “sustainably caught” and seafood that was not.

Apparent discrepancies in the above-mentioned assessments (and in others; e.g. Froese and Proelss 2012 vs Agnew et al. 2013) are consistent with the lively debate that has been surrounding MSC and other sustainability certification programmes for two decades. Specifically, the environmental standards of MSC certification have been repeatedly questioned (Jacquet and Pauly 2007; Ward 2008; Jacquet et al. 2010a, 2010b; Christian et al. 2013; Opitz et al. 2016). MSC has also been criticized for favouring large-scale industrial fisheries and failing to involve fisheries in developing countries, with large-scale industrial fisheries in the United States, Canada and the U.K. dominating MSC certifications (Pérez-Ramírez et al. 2012a, 2016; Ponte 2012; Wakamatsu and Wakamatsu 2017). In response, several environmental groups have resolved to distance themselves from the MSC programme and sought to advance fishery sustainability through their own market strategies, such as seafood buyer guides (Gulbrandsen and Auld 2016).

Jacquet et al. (2009) conceded that market-based strategies can raise awareness among consumers and encourage suppliers to adopt better practices. However, the labelling of selected trawl fisheries in certain areas (predominantly in the Global North; Jones et al. 2023) falls far short of eliminating the global impacts of these fisheries in the foreseeable future, and “certification alone is unlikely to arrest the decline of fish stocks” (Gulbrandsen 2009).

As noted previously, Hilborn et al. (2023) pointed out that 83 bottom-trawl fisheries were certified by MSC as sustainable, and “many” were recommended by the Seafood Watch programme of the Monterey Bay Aquarium. Those authors contended that “these are the two best-known international standards for fisheries sustainability, and the fact that bottom-trawl fisheries meet their standards is evidence that bottom-trawl fishing can be sustainable”.

At present, however, it cannot be assumed that certification, or any other form of eco-labelling, to encompass global trawl fisheries would be a convincing strategy for making all, or even most of them, truly sustainable. The biomass of a targeted stock cannot be considered a reliable indicator of the overall “sustainability” of a fishery: many other factors must be taken into account. In addition, sectors of civil society (including small-scale fishers) find it difficult to accept that certification of any fishery that deploys large (50 m or longer) factory trawlers can be characterized as sustainable (Tracey et al. 2013; Murphy-Gregory 2018). The ongoing certification of fisheries that deploy factory freezer trawlers between 50 and 140-m long cannot convey ideas of sustainability that are universally, or even just widely, shared—given the unavoidable ecological impacts that accompany the technology itself when applied at such a scale.

Impacts on marine life

5

Overview of trawl impacts

Countless studies, encompassing decades of fishery research, have documented the harmful nature of trawling. These studies are so numerous and detailed, and they refer to so many areas and ecosystems, that it is virtually impossible to come close to reviewing all of them, or even identifying the most compelling. This report does little more than scratch the surface of a huge body of information.

In the following Table, we describe some of the main direct and indirect impacts of trawling discussed in this chapter and list key references.

Impact	References
Damage to deep-water coral ecosystems and deep-sea organisms that are exceptionally long-lived and grow extremely slowly; recovery after fishing impact has ceased may take decades to centuries	Koslow et al. 2000; Hall-Spencer et al. 2002; Reed et al. 2007; Althaus et al. 2009; Clark et al. 2019; Williams et al. 2020a
Damage to hard corals, soft corals and sponges	Fosså et al. 2002; Buhl-Mortensen et al. 2016; Pierdomenico et al. 2018; Morrison et al. 2020
Large-scale degradation and frequently complete annihilation of oyster reefs	Thurstan et al. 2013, 2024; Bennema et al. 2020
Degradation, decline and disappearance of seagrass meadows	Ardizzone and Pelusi 1983; Sánchez-Jerez and Esplá 1996; González-Correa et al. 2005; Boudouresque et al. 2009; Cunha et al. 2013; Krause-Jensen et al. 2021
Reduced biomass and species richness of benthic invertebrate fauna	Collie et al. 1997, 2005; Pitcher et al. 2000; Duplisea et al. 2002; Olsgard et al. 2008; Hinz et al. 2009; Pusceddu et al. 2014; Hiddink et al. 2019
Mortality of benthic invertebrates in the path of the gear	Mensink et al. 2000; de Juan et al. 2007a; Hiddink et al. 2011; Hammond et al. 2013; Collie et al. 2017
Mortality of non-edible organisms such as Echinodermata, Cnidaria, Mollusca, Arthropoda and Porifera, including species of conservation concern such as threatened syngnathiform fishes (e.g. seahorses, pipefishes and seadragons)	Ramsay et al. 2001; Maynou and Cartes 2011; Cook et al. 2013; Lawson et al. 2017; Lakshmanan et al. 2021; Pollom et al. 2021
Depletion of target species	Thurstan and Roberts 2010; Thurstan et al. 2010; Foley et al. 2011; Hiddink et al. 2011; Dransfeld et al. 2013; Johnson et al. 2015; Novaglio et al. 2020
Bycatch of elasmobranchs	Bonfil 1994; Stobutzki et al. 2002; Tamini et al. 2006; Zeeberg et al. 2006; Coelho and Erzini 2008; Damalas and Vassilopoulou 2011; Hsu et al. 2012; Molina and Cooke 2012; Barausse et al. 2014; Oliver et al. 2015; Bonanomi et al. 2018; Gray and Kennelly 2018; White et al. 2019; Francis and Duffy 2022

Bycatch of sea turtles	Álvarez de Quevedo et al. 2010; Wallace et al. 2010, 2013; Finkbeiner et al. 2011; Lucchetti et al. 2017; Casale et al. 2018; Parga et al. 2020
Bycatch of seabirds	Weimerskirch et al. 2000; González-Zevallos and Yorio 2006; Sullivan et al. 2006; Baird 2008; Varty et al. 2008; Watkins et al. 2008; Waugh et al. 2008; Abraham et al. 2009; Bull 2009; Favero et al. 2011; Marinao and Yorio 2011; Marinao et al. 2014; Tamini et al. 2015, 2023; Crawford et al. 2017; Pott and Wiedenfeld 2017; Dias et al. 2019; Phillips et al. 2024
Bycatch of cetaceans	Clausen and Andersen 1988; Waring et al. 1990; Perez and Loughlin 1991; Maigret 1994; Couperus 1997; Crespo et al. 1997, 2000, 2017; Dans et al. 1997, 2003a, 2003b; Fertl and Leatherwood 1997; Starr and Langley 2000; Perez 2003, 2006; Ross and Isaac 2004; Zollett and Rosenberg 2005; Read et al. 2006; Slooten et al. 2006; Zeeberg et al. 2006; Du Fresne et al. 2007; Young and Iudicello 2007; Andersen et al. 2008; Fernández-Contreras et al. 2010; Jaiteh et al. 2013, 2014; Thompson et al. 2013; Greenman and McFee 2014; Wakefield et al. 2014, 2017; Lyssikatos 2015; Gearhart and Hataway 2018; Santana-Garcon et al. 2018; Busson et al. 2019; Franco-Trecu et al. 2019; Hayes et al. 2021; Jiménez et al. 2021; Kuit and Ponnampalam 2021; Peltier et al. 2021, 2024; Rouby et al. 2022; Bolling et al. 2023; Constantine 2023
Bycatch of pinnipeds	Fowler 1987; Fowler et al. 1990; Perez and Loughlin 1991; Pemberton et al. 1992; Wickens 1995; Crespo et al. 1997; Boren et al. 2006; Chilvers 2008, 2015, 2018; Thompson et al. 2013; Hofmeyr 2015; Young et al. 2023
Changes in marine communities, trophic web structure and function, often reducing the environmental carrying capacity for many species and sometimes including regime shifts	Kaiser and Ramsay 1997; Jennings et al. 2001; Tillin et al. 2006; Callaway et al. 2007; de Juan et al. 2007b; Shephard et al. 2010; Thurstan and Roberts 2010; Thurstan et al. 2010; Strain et al. 2012; Pusceddu et al. 2015; van Denderen et al. 2015; Hiddink et al. 2016; Currie et al. 2020; Sguotti et al. 2022
Changes in the size, diet and body condition of marine organisms	de Juan et al. 2007a; Smith et al. 2013; Johnson et al. 2015; Collie et al. 2017; Hinz et al. 2017
Changes in the movements, distribution, diet, group size, social behaviour and social organization of populations that have adapted to foraging and scavenging behind trawlers	Jackson 1988; Hill and Wassenberg 1990; Waring et al. 1990; Crawford et al. 1991; Berruti et al. 1993; Oro et al. 1995, 1999; Arcos and Oro 1996; Oro and Ruiz 1997; Chilvers and Corkeron 2001; Hamer and Goldsworthy 2006; Louzao et al. 2006; Navarro et al. 2009; Bartumeus et al. 2010; Camphuysen 2011; Ferretti et al. 2013; Bodey et al. 2014; Lyle et al. 2016; Allen et al. 2017; Karris et al. 2018 ; Mitchell et al. 2018; Bearzi et al. 2019; Giménez et al. 2021; Tixier et al. 2021; Bonizzoni et al. 2022, 2023; Picariello et al. 2022
Degradation of physical habitat through resuspension of sediment, erosion, turbidity, changes in seabed organic matter, trophic state and morphology (sometimes resulting in localized pollution and toxicity)	Churchill 1989; Martín et al. 2008, 2014b; Bradshaw et al. 2012, 2021; Puig et al. 2012, 2015; Pusceddu et al. 2014, 2015; Oberle et al. 2016a; Hale et al. 2017 ; Tiano et al. 2019, 2024; Paradis et al. 2021; Palanques et al. 2022; Bruns et al. 2023; Durán et al. 2023

Factors affecting trawl impacts

The impact of bottom trawling on demersal ecosystems depends on the degree of contact with the seabed, and intrusion into it, which varies according to the specific trawl gear. The magnitude of trawl disturbance also depends on the intensity of trawling, the rate of depletion per trawl pass, and the recovery rates of organisms and communities exposed to trawling (Hiddink et al. 2017; Hilborn et al. 2021).

Gear and intensity of trawling

Bottom-towed fishing gears such as bottom trawls and dredges typically kill between 20 and 50% of the benthic invertebrates in their path, depending on gear type, substrate, and vulnerability of the local biological communities (Collie et al. 2017). Each pass of towed gear can reduce benthic invertebrate abundance and species richness, with gears that penetrate deeper into the sediment having a significantly larger impact than those that penetrate less deep (Kaiser et al. 2006; Sciberras et al. 2018).

As noted earlier, the first pass of towed demersal gear tends to cause the most seabed damage (Sciberras et al. 2018; Currie et al. 2020; Mazor et al. 2021). In a study by Cook et al. (2013), the first pass of a bottom otter trawl on shellfish reefs reduced the total number of epifaunal organisms by 90%, with declines in anthozoans, hydrozoans, bivalves, echinoderms and ascidians accounting for most of the change. The amount of damage to seabed communities becomes lower once much of the original epifauna has been removed by earlier trawling. At that stage, between 5 and 15% of marine organisms may be removed at each pass of the trawl net, with a degree of removal that depends on the type of gear (beam trawls being more intrusive and harmful than otter trawls), and on the footprint and intensity of trawling (Hiddink et al. 2017; Sciberras et al. 2018). Midwater trawls have no impact on the seabed as long as fishing occurs in the water column, but in some fisheries rates of contact with the seabed are high (Casale et al. 2004; Sala et al. 2018a), and in those cases the effects can resemble those of bottom trawling (e.g. Stratton and Wilson 2023).

Seabed type and biological community

Certain seabed habitats are of greater concern than others. For instance, slow-growing coral communities in deep-water areas are more sensitive to disturbance than areas of naturally mobile sediments. Bottom trawling in deep-water seabed is especially harmful, and it typically causes a rapid and substantial decline of most of the fished (including non-target) populations. Because many deep-sea organisms are exceptionally long-lived and grow extremely slowly, recovery after trawling ceases may take decades to centuries (Jones 1992; Koslow et al. 2000; Roberts 2002; Clark et al. 2019).

Some studies on the impact of bottom trawling on the seabed have short timeframes and fail to account for the serial, sometimes multi-century bouts of seabed contact that may have preceded the baseline years of such studies (Jones 1992). Clearly, the perceived degree of impact may become lower after bottom trawling has turned a vulnerable and rich three-dimensional marine ecosystem into a two-dimensional system characterized by low diversity and a predominance of resilient species. By that time, there may be a perception that there is “little or nothing left to save”, and such a perception can affect management decision.

The effects of trawling on sensitive habitats such as seagrass meadows, oyster reefs, sponge gardens or coral and biogenic reefs are much more severe than those on sedimentary habitats—such as unconsolidated muddy and sandy seabed (Collie et al. 2000; Kaiser et al. 2002, 2018; Sciberras et al. 2018; Hilborn et al. 2023). Epifaunal species that stabilize the sediment, and provide habitat for benthic invertebrates, are especially vulnerable to trawling (Collie et al. 2017). Impacts tend to be more severe in gravel or mud than in sand, and degree of impact also depends on the life history of the affected species (Hiddink et al. 2019; Pitcher et al. 2022). Considering that areas with complex seabed habitats can be heavily impacted by trawling, and impacts can be long-lasting (Parker et al. 2009; Clark et al. 2019; Williams et al. 2020a), the best practice should be to close such areas to towed gear before the damage is done (Goode et al. 2020; McConnaughey et al. 2020).

Currents, waves and storms can erode seabed sediments, cause resuspension of organic matter and affect the settlement of new recruits (Hunt and Scheibling 1997; Morris and Howarth 1998). Such effects can promote opportunistic species that are adapted to and can withstand natural disturbance, and these species may therefore be comparatively more resilient to trawl disturbance (Jennings and Kaiser 1998; Kaiser 1998). van Denderen et al. (2015) noted that benthic responses to bottom trawling may be limited or even undetectable in areas exposed to high levels of natural disturbance, suggesting that trawl disturbance and the mechanical effects of strong currents, waves and storms can affect benthic communities in a similar way. However, the degree to which the effects of trawling are masked by (or made indistinguishable from) the effects of severe natural perturbations would depend on the intensity of trawling, as well as on the type of gear and its penetration into the seabed. Also, natural perturbations normally do not remove biomass to the extent that trawling does.

Physical and chemical alteration of the seabed

Apart from the removal of benthic organisms, the scraping and ploughing of the seabed modifies the physical and chemical properties of sediments, increasing resuspension, erosion, near-bottom turbidity and sediment flux (Churchill 1989; Durrieu De Madron et al. 2005; Martín et al. 2008, 2014a; Puig et al. 2012, 2015; Palanques et al. 2014; O'Neill and Ivanović 2016; Paradis et al. 2017). The resuspension of sediments increases the concentration of organic particles, which can influence the trophic state of benthic systems (Pusceddu et al. 2015; Linders et al. 2018; Breimann et al. 2022; Bilan et al. 2023). Trawling-induced resuspension may increase with water depth, because of the rapid decay of wave effects (Mengual et al. 2016).

Where bottom trawling is frequent, sediment biogeochemistry may not have time to "recover" between disturbance events, and elevated turbidity may persist even outside the trawled area (Bradshaw et al. 2021). A bottom otter trawler operating on a muddy Baltic seabed produced a 36 m-wide trawl track, with parallel furrows and sediment piles caused by the otter boards and shallower grooves from the groundgear. The trawl gear displaced an estimated 1,000 m^3 (500 tonnes) of sediment and suspended 9.5 tonnes of sediment per kilometre of track. Otter boards had less effect than the rest of the gear in terms of total sediment mass, but the boards had five times the displacement and twice the suspension effect per square metre, due to their greater penetration and hydrodynamic drag (Bradshaw et al. 2021). The suspended sediment spread >1 km away over the following 3–4 days, creating a 5–10 m thick layer of turbid water. Twenty hours after trawling, turbidity measured 550 metres away from the track was still high (Bradshaw et al. 2021). Dissolved methane concentrations were elevated in the water for at least 20 hours, and two hours after trawling there was a pulse of dissolved N, P, and Mn to a height of 10 m above the seabed within a few hundred metres of the track. In the track of otter boards, sediment biogeochemistry was still perturbed after 48 hours, with a decreased oxygen penetration depth (Bradshaw et al. 2021).

The stirring, mixing, erosion and oxygenation of sediments induced by recurrent trawling changes the physical properties of sediments (grain size, cohesiveness, density) and promotes a general homogenization (Puig et al. 2012). Decades-long bottom trawling can produce large-scale erosion and deep excavation of the seafloor, leading to permanent large-scale changes of seabed morphology and long recovery periods after the cessation of trawling activities (Martín et al. 2014; Durán et al. 2023). For instance, intense bottom trawling (240–750 hauls per ha during 2005–2017) off Spain's continental shelf (northwestern Mediterranean Sea) produced seabed depressions of up to 1.2 m (Durán et al. 2023).

A study off northwestern Spain (Atlantic Ocean) indicated a mean mass of sediment resuspended by bottom trawling of 13.5 megatonnes per year: a sixfold increase in off-shelf sediment transport when compared to natural resuspension mechanisms (Oberle et al. 2016a). By combining the worldwide distribution of soft sediments on continental shelves with estimates of bottom trawling intensity, those authors estimated that the mass of resuspended sediments induced by bottom trawling amounted to approximately 22 gigatonnes per year—approximately equivalent to the aggregate mass of all sediments supplied to the continental shelves by the world's rivers (Oberle et al. 2016a).

Bottom trawling also alters the release of oxygen and nutrients, and disrupts the cycles of chemicals that allow for the proper functioning of marine ecosystems (Ferguson et al. 2020). Pusceddu et al. (2014) compared the sediments of trawled and untrawled areas in the Mediterranean Sea, and found that sediments in trawled areas had approximately 50% lower organic content, 40% slower organic carbon turnover, 80% lower abundance of meiofauna, 50% lower biodiversity, and 25% lower richness of nematode species.

Impact on target species

Bottom trawling is the primary method to target demersal and semi-demersal organisms, also known as "groundfish" in the jargon of fishery science. Excessive trawling has often resulted in the decline of groundfish and other target and non-target species (Thurstan and Roberts 2010; Thurstan et al. 2010, 2024; Foley et al. 2011; Hiddink et al. 2011; Dransfeld et al. 2013; Barausse et al. 2014; Johnson et al. 2015; Novaglio et al. 2020). However, a review of groundfish populations based on published assessments of 349 individual "stocks", representing 90% of global groundfish catch, found that, on average, stock abundance was increasing (Hilborn et al. 2021), and these stocks were reportedly "above the target levels for sustainable exploitation" (Hilborn et al. 2023). Regions with the most depressed groundfish populations included the Northwest Atlantic and the Pacific coast of South America, while populations from the Northeast and Eastern Central Pacific, Northeast Atlantic, Southeast Atlantic and Southwest Pacific tended to have a greater average of fish abundance in relation to MSY. Based on their assessment, Hilborn et al. (2021) suggested that there is "modest opportunity to increase catch of global groundfish fisheries by reducing overfishing on some stocks, but more by increasing harvest on others", but they noted that "there may be other reasons not to fully exploit these stocks".

Global trends in groundfish population abundance, and evidence of increasing trends for some populations (Fernandes and Cook 2013; Zimmermann and Werner 2019; Hilborn et al. 2021), suggest that well-managed bottom trawl fisheries don't necessarily lead to the decline of target species. Bottom trawling, however, causes impacts that go well beyond those on target species and populations (as reviewed in several chapters of this report).

While trends in regional and global catches provide essential information (Pauly 2019, p. 89), they can obscure and even distract attention from impacts at a regional, national, or sub-national level (Steadman et al. 2021). Global landings of target species, in particular, are imperfect proxies for environmental impact as a whole. For instance, areas with low fish densities can be trawled more intensively in order to achieve catch levels comparable to those obtained in areas with high fish densities (Halpern et al. 2019, p. S11). Low catches may indicate that an area that was historically trawled has been depleted (Steadman et al. 2021), and stable or increasing catch trends per se may tell nothing about the pre-impact status of, or the degree of damage to, a marine ecosystem. Indeed, relatively stable catches may be made after major ecosystem damage has already led to dramatic regime shifts (Sguotti et al. 2022), and catch trends may remain stable, or increase, after a few species highly resilient to trawl impacts have replaced a formerly rich and diverse ecological community.

Bycatch

In a FAO technical paper, Alverson et al. (1994) defined bycatch as the sum of discarded catch (the "portion of the catch returned to the sea as a result of economic, legal, or personal considerations") and incidental catch (the "retained catch of non-targeted species)." Another definition of bycatch refers to the catch "of organisms that are not targeted, including organisms that are outside legal-size limits, over-quotas, threatened, endangered and protected species, and discarded for whatever other reasons, as well as nontargeted organisms that are retained and then sold or consumed" (Pérez Roda et al. 2019). This bycatch is then subdivided into non-target organisms retained to be sold or eaten ("landed bycatch") and animals alive or dead thrown back into the sea ("discards"; Pérez Roda et al. 2019).

Outside the realms of fishery science and management, the term bycatch tends to be applied mainly to the incidental (i.e. unintentional) capture of megafaunal species, notably elasmobranchs, sea turtles, seabirds and marine mammals (particularly cetaceans and pinnipeds), which often are considered threatened and are sometimes legally protected. The incidental mortality and injury of these groups of marine megafauna is summarized in the following sections.

Elasmobranchs

The bycatch of elasmobranchs (including sharks, rays, skates and sawfish) in trawl gear is often a conservation concern (Fischer et al. 2012). These animals tend to have low reproductive rates, long gestation periods and lifespans, and late sexual maturity, limiting the ability of their populations to withstand or recover from fishery-related mortality (Dulvy et al. 2008). Several populations of elasmobranchs have experienced dramatic declines as a result of fishing impacts, and some species have been eradicated (Brander 1981; Casey and Meyers 1998; Shepherd and Myers 2005; Ferretti et al. 2013).

Catches of elasmobranchs occur in virtually all types of trawl gear, including bottom otter trawls (Shepherd and Myers 2005; Tamini et al. 2006; Damalas and Vassilopoulou 2011), beam trawls (Rogers and Ellis 2000; Barausse et al. 2014; Silva and Ellis 2019), and midwaters trawls (Zeeberg et al. 2006; Bonanomi et al. 2018). Most of these catches, however, are not reported at species level, if they are reported at all (Clarke et al. 2006; Worm et al. 2013; Grey and Kennelly 2018). Information on elasmobranch bycatch in trawl fisheries comes largely from the North Atlantic, while information from other regions is limited (Molina and Cooke 2012; Oliver et al. 2015; Gray and Kennelly 2018), making bycatch rates and impacts difficult to assess at a global level.

While some elasmobranchs may be retained and sold, in trawl fisheries most of them are discarded at sea (Bonfil 1994; Coelho and Erzini 2008). For example, of the three most common sharks in Mediterranean and Black Sea trawl catches (*Galeus melastomus*, *Scyliorhinus canicula* and *Etmopterus spinax*), more than 70% are discarded (FAO 2022a). Similarly, almost 93% of the elasmobranchs caught in bottom otter trawls in the Aegean Sea (eastern Mediterranean) are discarded (Damalas and Vassilopoulou 2011).

In other areas, the discard rate of some elasmobranch species may reach 100%. That is the case of a number of species bycaught in bottom otter trawls off Argentina (including *Discopyge tschudii*, *Psammobatis extenta*, *Psammobatis bergi*, *Zapteryx brevirostris*, *Myliobatis goodei*, and *Dasyatis pastinaca*; Tamini et al. 2006). Deep-water trawls targeting crustaceans such as lobsters and shrimps off Portugal caught small-sized and mostly immature elasmobranch specimens (Coelho and Erzini 2008). Similarly, studies on shrimp trawl fisheries in Australia and Papua New Guinea showed that for most bycaught elasmobranch species, the majority of individuals taken were immature (Stobutzki et al. 2002; White et al. 2019). Midwater pair trawling in the Adriatic Sea also has high catches of immature elasmobranchs; for instance, an average of 70% of smooth-hound *Mustelus mustelus* and 92% of picked dogfish *Squalus acanthias* were immature (Bonanomi et al. 2018).

Charismatic filter-feeding elasmobranchs such as manta and devil rays (family Mobulidae), basking sharks *Cetorhinus maximus*, and whale sharks *Rhincodon typus*, are not immune from bycatch in trawl gear (e.g. Zeeberg et al. 2006; Hsu et al. 2012; Oliver et al. 2015; Francis and Duffy 2022).

Sea turtles

The total reported bycatch of sea turtles in global fisheries throughout the period 1990 and 2008 was about 85,000 individuals (Wallace et al. 2010). However, as acknowledged by those authors, such a figure "likely [grossly] underestimates the true total by at least two orders of magnitude", meaning that the actual bycatch over that 19-year period could have been as high as 8,500,000, or higher. Only a very small percentage (almost certainly no more than 5% and possibly less than 1%) of global industrial fishing effort involves onboard bycatch observation and reporting. Moreover, according to Wallace et al. (2010), reported bycatch data "underrepresent small-scale fishing activities, which have been documented to have large cumulative bycatch impacts." About 20% of the reported sea turtle bycatch was in bottom and midwater trawls, but data did not allow estimation of mortality associated with the bycatch (Wallace et al. 2010).

Bycatch in trawl gear affects all of the seven sea turtle species (loggerhead turtle *Caretta caretta*, green turtle *Chelonia mydas*, leatherback turtle *Dermochelys coriacea*, hawksbill turtle *Eretmochelys imbricata*, Kemp's ridley turtle *Lepidochelys kempii*, olive ridley turtle *Lepidochelys olivacea*, and flatback sea turtle *Natator depressus*; Wallace et al. 2013). Mortality in trawl gear was also reported for a freshwater turtle species that can adapt to the marine environment, the Nile soft-shelled turtle *Trionyx triunguis*, frequently caught in bottom otter trawls and midwater trawls operating in Turkish waters, with at least 437 individuals being caught by 12 vessels in the 1996–1997 fishing season (Oruç 2001; Taskavak and Akcinar 2009).

Turtles are especially vulnerable to bycatch by bottom trawlers when they rest quiescently on the seabed, sometimes for several hours (Casale et al. 2018). Mortality may be caused by drowning, decompression sickness, or post-capture release by fishers of comatose individuals that haven't fully recovered (Casale et al. 2018; Parga et al. 2020). Between 1990 and 2007, the shrimp trawl fishery in the southeastern United States and northern Gulf of Mexico alone was thought to be responsible for more than 80% of all sea turtle mortality in United States fishing gear (Finkbeiner et al. 2011). About 69,300 sea turtles were estimated to have died annually in the shrimp fishery before 2003, but following the enlargement of openings in turtle exclusion devices to expel all sizes of sea turtles, together with a reduction of fishing effort in the Gulf, this number was reduced to around 3,700 deaths per year (Finkbeiner et al. 2011). Between 2003 and 2007, U.S. shrimp trawlers in the southeastern United States and Gulf of Mexico were responsible for the deaths of about 2,700 Kemp's ridley turtles, 650 loggerhead turtles, 320 green turtles and 15 leatherback turtles (Finkbeiner et al. 2011). Although the turtle bycatch estimates in this fishery may appear complete and rigorous, Finkbeiner et al. (2011) acknowledged that the numbers are "fraught with high uncertainty due to lack of [other than minimal] observer coverage".

In the Mediterranean Sea, a FAO report on fishery bycatch between 2000 and 2020 indicated that bottom trawling caused the highest sea turtle mortality (37.1%) after longlining (42.7%; FAO 2020a). Bottom trawling was reportedly responsible for 172,100 deaths, and midwater trawling for an additional 10,200. The greatest proportion of fishery bycatch of sea turtles in the Adriatic Sea and central Mediterranean was caused by bottom trawls: 76% and 50%, respectively (FAO 2020a).

A study based on interviews with fishers in Catalonia, Spain, between June 2003 and July 2004 estimated an annual bycatch of approximately 480 loggerhead turtles, of which 52% were killed in bottom trawls and 26% in drifting longlines (Álvarez de Quevedo et al. 2010). Another study based on fisher interviews, conducted in 2014 and covering about 6% of the Italian fishing fleet, estimated that some 4,000 loggerhead turtles could have died every year due to trawl nets; of all fishing gears, trawl nets were the main threat to sea turtles, with the Adriatic Sea being the main bycatch hotspot in the region (Lucchetti et al. 2017).

Seabirds

Seabirds have adapted to approach and forage near fishing vessels, feeding on discards, offal, bait or fish taken from the nets. This behaviour increases the risk of entanglement. While globally, seabird mortality in fishing gear has often been associated primarily with longlining and gillnetting (Pott and Wiedenfeld 2017), and much less with trawling, mortality in trawl gear is certainly underestimated (Weimerskirch et al. 2000; Sullivan et al. 2006; Phillips et al. 2024). A global assessment of threats to seabirds (Dias et al. 2019) indicated that, with reference to IUCN's Threats Classification Scheme (iucnredlist.org/resources/threat-classification-scheme), trawling had the greatest impact as a threat source in terms of both "scope" (proportion of total seabird numbers affected in a given population) and "severity" (overall decline of number of individuals in a population caused by the threat). Trawls, however, affected many fewer seabird species than gillnets, and about the same as longlines (Figure 6 in Dias et al. 2019).

Taxonomic groups of seabirds known to interact with trawlers and die in trawl gear include albatrosses, auks, murrelets, cormorants, petrels, gulls, penguins, fulmars, prions, shearwaters, gannets, skuas, and sulids (Pott and Wiedenfeld 2017; Phillips et al. 2024). A total of 41 seabird species have been reported to interact with bottom otter trawls and beam trawls, and 29 with midwater trawls; while "interactions" do not necessarily imply mortality or harm per se in all cases, they may be indicative of a species' vulnerability to trawl gear (Pott and Wiedenfeld 2017). According to a more recent review, at least 55 seabird species can die as a result of bycatch in trawl fisheries including bottom and midwater trawls (Phillips et al. 2024).

Seabird mortality in trawl fisheries can result from entanglement (either outside or inside the net) or be cable-related, resulting from strikes on trawl warps, on the net sonde cable (used to tow an acoustic sonde which transmits operational information), or on other cables (Bull 2009). Net-related mortality tends to be much lower than cable-related mortality (Watkins et al. 2008; Favero et al. 2011; Tamini et al. 2015). The latter can occur when birds are either flying or sitting on water, and it seems to depend mainly on the presence of fishery discards, as well as factors including wind strength and directionality (Weimerskirch et al. 2000; Sullivan et al. 2006; Watkins et al. 2008; Favero et al. 2011; Tamini et al. 2015). Mortality due to collision or entanglement with cables and warps is typically underestimated, as seabirds being harmed or dying out of board and outside of the net are not routinely counted.

A recent review of trawl fisheries estimated that approximately 44,000 birds died annually following bycatch in trawl gear in a total of 25 fisheries (Phillips et al. 2024), with global mortality likely much higher considering the many trawl fisheries that weren't monitored, as well as unobserved mortality (individuals killed or fatally injured but not landed or recorded; Phillips et al. 2024). The highest mortality rates in those 25 fisheries (more than 1,000 birds estimated to have died annually) apparently were of black-browed albatrosses *Thalassarche melanophris*, southern giant petrels *Macronectes giganteus*, northern giant petrels *Macronectes halli*, Magellanic penguins *Spheniscus magellanicus*, and Cape petrels *Daption capense*.

Seabird mortality observed on trawlers operating off New Zealand between 1998 and 2004 included 757 white-capped albatrosses *Thalassarche steadi*, 581 sooty shearwaters *Puffinus griseus*, 169 white-chinned petrels *Procellaria aequinoctialis*, 94 Salvin's albatrosses *Thalassarche salvini*, 83 Buller's albatrosses *Thalassarche bulleri,* and a few individuals of other albatross and petrel species (Waugh et al. 2008). In the Falkland (Malvinas) Islands, observers onboard seven bottom otter trawlers estimated that 1,529 seabirds (CI 1,075–1,983; largely black-browed albatrosses) were killed in one year, mainly due to warp cables (Sullivan et al. 2006). This estimate was based on a count of individuals hauled onboard (therefore excluding those that died out of board).

On the Patagonian Shelf off Argentina, annual estimates of dead and injured seabirds interacting with 33 stern factory trawlers using demersal nets included 13,548 (CI 8,000–19,673) black-browed albatrosses, 2,463 (CI 612–4,306) southern giant petrels, 1,847 (CI 61–3,689) northern giant petrels, and 1,232 (CI 0–3,077) Cape petrels (Tamini et al. 2015). From counts of birds killed or injured by cable collisions, the mortality rate of black-browed albatrosses was estimated as 0.24 birds per hour (Tamini et al. 2015).

Observers onboard bottom trawlers in South African waters reported high mortality of shy albatrosses *Thalassarche cauta* and black-browed albatrosses, and lower mortality of white-chinned petrels, Cape gannets *Morus capensis* and sooty shearwaters (Watkins et al. 2008). In South Africa, the estimated annual seabird mortality caused by 79 trawlers was 18,000 (CI 8,000–31,000); of these, 12,000 were albatrosses. During the "dumping of fishery wastes", the estimated mortality rate ranged between 0.2 and 0.6 birds per hour. These estimates were considered conservative (negatively biased) as they included only validated kills (Watkins et al. 2008).

A global review of penguin bycatch in fisheries indicated that gillnets, and to a lesser extent trawl nets, pose the greatest risk to these seabirds, as compared to purse seiners and longliners. Trawling seems to affect mainly Magellanic penguins, with only a few mortality records for gentoo *Pygoscelis papua* and king penguins *Aptenodytes patagonicus* (Crawford et al. 2017). Incidental catches of Magellanic penguins in trawl gear were reported by various studies off Argentina (Crawford et al. 2017), with mortality counts varying between 35 individuals in 2003 and 1,516 individuals in 2004 (González-Zevallos and Yorio 2006), and mean mortality rates varying between 0.003 and 0.087 birds per haul (Marinao and Yorio 2011; Marinao et al. 2014). Seabird bycatch observed from "over ten" coastal ice trawlers operating bottom trawl gear off Argentina in 2008–2009 and 2011–2012 included 203 Magellanic penguins; of these, 112 were released alive, but survival rates are unknown. Numbers of bycaught individuals varied between one and 20 per haul (Marinao et al. 2014).

Seabird mortality is also caused by midwater trawlers, either 1) during hauling and shooting, when the net is close to the surface and the meshes expand and contract due to wave action and winching, increasing the chances of entanglement (Varty et al. 2008), or 2) due to collisions with net sonde and warp cables (Abraham et al. 2009; Tamini et al. 2023). Observed deaths caused by midwater trawlers in New Zealand waters included white-capped albatrosses, Buller's albatrosses, Salvin's albatrosses, sooty shearwaters and white-chinned petrels (Baird 2008).

Cetaceans

Incidental mortality and injury of cetaceans (dolphins, porpoises and whales) in trawl gear has generally been perceived as less of a conservation problem than incidental mortality and injury of these animals in other fishing gears, notably gillnets, longlines and the vertical lines used to tether, mark and retrieve bottom-set traps and pots. However, like many seabirds, some dolphin species are attracted to and "interact with" working trawlers in ways that put them at high risk of being bycaught (Bonizzoni et al. 2022).

Bycatch in trawl gear has involved a number of odontocete species, worldwide (Crespo et al. 1997, 2000; Dans et al. 1997; Fertl and Leatherwood 1997; Ross and Isaac 2004; Zollett and Rosenberg 2005; Read et al. 2006; Du Fresne et al. 2007; Young and Iudicello 2007). Generally speaking, the risk of cetacean entanglement or engulfment into a trawl net may increase when 1) individuals enter the trawl net during hauling in; 2) individuals stay deep inside the net for long periods beyond the limits of their breath-holding capability; 3) fishing gear becomes unstable (e.g. due to slower vessel speed or abrupt turns); and 4) individuals rub themselves against or otherwise come into contact with trawl net, lazy line (a rope attached to the codend), or other gear (Greenman and McFee 2014; Wakefield et al. 2014; Gearhart and Hataway 2018; Santana-Garcon et al. 2018). While the large net opening of big trawlers may increase the likelihood of cetaceans being funnelled into the net, smaller trawlers can also represent a threat due to the frequency of hauls and net-collapsing events (Zeeberg et al. 2006).

The relative impact of trawling as a threat varies markedly across cetacean species and populations. Trawl fishing is not known to result in the injury or death of baleen whales other than occasionally. A detailed review of the extensive United States trawl fisheries concluded that "bycatch of a live [baleen] whale in a trawl is a very unlikely event, occurring extremely rarely at the scale of all U.S. trawl fisheries over the last couple of decades" (NMFS 2020, p. 55). Similarly, most porpoises (Phocoenidae), including the critically endangered vaquita *Phocoena sinus*, appear generally less prone to bycatch in trawls than in gillnets and other entangling nets (Rojas-Bracho et al. 2006). However, harbour porpoises *Phocoena phocoena* are known to follow, interact with, and become bycaught in midwater trawls in Danish waters (Clausen and Andersen 1988), small numbers of Dall's porpoises *Phocoenoides dalli* have been caught by trawlers in the Bering Sea and Gulf of Alaska (Perez and Loughlin 1991), and there are reports of Indo-Pacific finless porpoises *Neophocaena phocaenoides* being caught in trawls offshore of Matang, Peninsular Malaysia (Kuit and Ponnampalam 2021).

In contrast, delphinids, especially those in the genera *Delphinus* (Waring et al. 1990; Crespo et al. 2000; Fernández-Contreras et al. 2010; Thompson et al. 2013; Lyssikatos 2015; Hayes et al. 2021; Peltier et al. 2021, 2024; Rouby et al. 2022), *Lagenorhynchus* (Couperus 1997; Crespo et al. 1997, 2000; Dans et al. 2003a, 2003b; Lyssikatos 2015), *Tursiops* (Lyssikatos 2015) and *Cephalorhynchus* (Crespo et al. 1997, 2017; Dans et al. 2003a) that forage on schooling fish (e.g. anchovies, bass, hake), shrimp and squid in the water column and near or at the bottom, are susceptible to bycatch, often in large numbers (especially common dolphins), in both midwater trawls and bottom trawls. Risso's dolphins *Grampus griseus* and pilot whales *Globicephala* spp., both primarily squid eaters, are regularly killed in midwater trawls in at least some areas (Waring et al. 1990; Lyssikatos 2015). Also, franciscanas *Pontoporia blainvillei* are caught in significant numbers in the industrial pair trawl fishery for croakers that operates in the Argentina-Uruguay Common Fishing Zone (33 licensed vessels about 20 m long; Franco-Trecu et al. 2019; Jiménez et al. 2021).

Trawl impacts on highly endangered species or populations are a primary concern, considering that the loss of only a few individuals could have an outsized effect on population survival and recovery. Several cetaceans stand out in addition to the vaquita, mentioned above. Endangered Hector's dolphins *Cephalorhynchus hectori* are known to associate with trawlers and are bycaught at least occasionally (Starr and Langley 2000; Slooten et al. 2006). Instances of multiple individuals dying in a single trawl have been recorded, and industrial trawl fishing is regarded as a potential threat to the critically endangered subspecies of Hector's dolphin, *C. h. maui* (Constantine 2023).

In Alaska, bycatch in trawl nets represents the majority (54%) of all reported killer whale *Orcinus orca* entanglements (Bolling et al. 2023). Of 20 cases reported between 1991 and 2022, 13 were associated with demersal trawls targeting flatfish, and 7 with pelagic trawls targeting pollock. These entanglements can occur when killer whales interact with the net as they are feeding on offal or discards, or when the animals feed inside the net, or in front of it. Killer whales entangled in trawl nets in Alaska seem to have low survival rates: 15 animals died whereas all of the five animals released alive had serious injuries (Bolling et al. 2023). Given the social structure and demography of killer whales, their mortality in trawl nets is of increasing concern (Busson et al. 2019).

Among the most alarming cases of cetacean mortality in terms of the sheer numbers killed concern fisheries operating in the Bay of Biscay, off France and Spain (Peltier et al. 2021, 2024; Rouby et al. 2022). A recent report by ICES (2023) estimated that as many as 9,040 (95% CI 6,640–13,300) common dolphins *Delphinus delphis* die there per year in fishing gear including bottom and midwater trawls, whereas Peltier et al. (2024) estimated that 6,920 (95% CI 4,038–15,368) individuals were bycaught during the winter 2021/2022.

Another alarming situation arose off north-western Australia (Kimberley and Arafura seas) beginning in 1959 and possibly earlier, when trawl fleets from Japan, Taiwan, Republic of Korea, China and Thailand, sometimes under joint-venture arrangements with Australia, fished the area heavily – at a rate of more than 30,000 trawl hours per year (Ramm 1994). Catches (and apparently also abundance) of targeted fish such as breams and snappers declined rapidly in the 1980s (Ramm 1994; Sainsbury 1988). Although it has proven impossible to obtain cetacean bycatch data for most of the fishing effort, it was estimated from reports by independent onboard observers that nearly 13,500 cetaceans were caught by Taiwanese gillnet vessels and pair-trawlers from mid-1981 to mid-1986 (Wakefield et al. 2014). As noted by Wakefield et al. (2014, p. 3), given the great fishing effort and the limited information obtained on dolphin bycatch, it is likely that dolphin deaths caused by foreign trawlers were many orders of magnitude greater than reported during the four decades they operated (ca 1960s–1990s).

In some cases, dozens of dolphins become bycaught and die during a single trawl operation (Maigret 1994; Crespo et al. 2000; Fernández-Contreras et al. 2010). When investigating the population-level effects of mortality in trawl gear, it is important to understand the interaction dynamics and to focus on the most affected component or components of the population (Crespo et al. 1997). Mortality of breeding females is likely to have a much greater impact on a population than mortality of other age-sex classes. Several studies reported unbalanced ratios in the bycatch of females and males. For example, common dolphins bycaught in midwater pair trawlers off south-western England (de Boer et al. 2012; Murphy et al. 2013) and off northwestern Spain (Fernández-Contreras et al. 2010) were mainly juvenile and subadult males. Conversely, dusky dolphins *Lagenorhynchus obscurus* bycaught in midwater pair trawls off Patagonia, and pilot whales *Globicephala* spp. bycaught in bottom otter trawls and midwater trawls off the north-eastern USA, included significantly more females than males, and also included pregnant females and young individuals (Waring et al. 1990; Dans et al. 1997).

Odontocete cetaceans (largely delphinids) can suffocate or drown after having their rostrum caught in the trawl net while trying to pull out fish, becoming entangled around the tail stock, or being caught in a bycatch exclusion device (Fertl and Leatherwood 1997). Because dolphins cannot manoeuvre easily within the narrow confines of a trawl net, they may become distressed when venturing or being funnelled far into a net, and have difficulty locating and escaping through small openings (Zeeberg et al. 2006; Jaiteh et al. 2013, 2014; Wakefield et al. 2017). The sublethal effects of cetacean bycatch in fishing gear are almost invariably overlooked (Wilson et al. 2014).

Pinnipeds

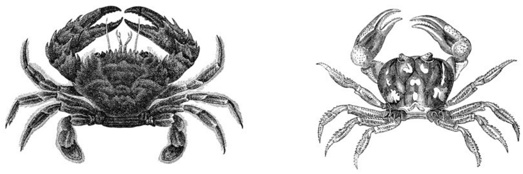
Some pinnipeds, particularly otariids (Otariidae, the sea lions and fur seals), are strongly attracted to trawlers and, given their unwary and bold behaviour, are exceptionally vulnerable to bycatch. Assessing the bycatch impacts of trawling on pinnipeds is often confounded by the difficulty of separating those impacts from the impacts of other factors—including the hunting of seals and sea lions for products, culling to reduce pinniped numbers (and benefit fisheries), entanglement in marine debris (including trawl net material; Fowler 1982), and reduced benthic and epibenthic prey due to fishing pressure.

The true impact of trawling, in terms of pinniped bycatch, is essentially incalculable. From the early 1950s through the 1970s, large fleets of stern trawlers, pair trawlers, side trawlers and factory trawlers (from the Soviet Union, Japan and the Republic of Korea) fished for flounder, pollock, cod, rockfish and other "groundfish" in the Bering Sea and along the Aleutian Islands, where Steller sea lions *Eumetopias jubatus* and northern fur seals *Callorhinus ursinus* were once extremely abundant (Bakkala et al. 1979; Wickens 1995). During most of that period, monitoring and reporting of marine mammal bycatch was either non-existent or sporadic.

It was not until 1973 (following passage of the U.S. Marine Mammal Protection Act) that observers were placed aboard a sample of U.S. and non-U.S. trawlers to record the marine mammal bycatch (Loughlin and Nelson 1986; Perez and Loughlin 1991). Of all incidental mortality of marine mammals reported by observers on trawlers in the Gulf of Alaska and eastern Bering Sea between 1973 and 1988, 87% were Steller sea lions: 1,000–2,000 per year in the U.S. EEZ from the early 1970s to 1982 (Perez and Loughlin 1991), declining to fewer than 100 per year since then (Young et al. 2023). Cumulative mortality in trawl fisheries likely contributed to observed regional declines in sea lion abundance between the 1950s and 1980s (Perez and Loughlin 1991), though other factors were apparently also driving the declines (Merrick et al. 1987; Loughlin and York 2000).

Other instructive examples of otariid populations subject to relatively high trawl mortality include: New Zealand sea lions *Phocarctos hookeri* bycaught in a squid trawl fishery and other trawl fisheries around the Auckland and Campbell Islands, regarded as the most serious and immediate threat to that endangered species (Chilvers 2008, 2015, 2018; Thompson et al. 2013); hundreds of New Zealand fur seals *Arctocephalus forsteri* dying in trawl fisheries around New Zealand in most years (Thompson et al. 2013); Australian fur seals *Arctocephalus pusillus doriferus* "subject to significant and ongoing bycatch mortality associated with demersal and mid-water trawling operations", constituting "a significant bycatch in the mid-water trawl sector of the small pelagic fishery" (Hofmeyr 2015 and contained references); and around 600 South American sea lions *Otaria byronia* bycaught per year by a large Argentine fleet of midwater trawlers fishing on the Patagonian shelf, mainly for hake and shrimp, during the early 1990s (Crespo et al. 1997).

Other pinnipeds bycaught in trawl nets in the northern North Pacific have included small numbers of walruses *Odobenus rosmarus*, ringed seals *Pusa hispida*, spotted seals *Phoca largha*, and an occasional northern elephant seal *Mirounga angustirostris* (Perez and Loughlin 1991). Many other phocids besides ringed, spotted and elephant seals (the Phocidae, the "true" or "earless" seals) have been bycaught, and doubtless continue to be bycaught, in trawl nets, though in relatively small numbers (Wickens 1995).



Discards

A FAO technical paper defined "discarded catch" (or "discards") as the "portion of the catch returned to the sea as a result of economic, legal, or personal considerations" (Alverson et al. 1994). More recently, Pérez Roda et al. (2019) subdivided "bycatch" into non-target organisms that are retained to be sold or eaten ("landed bycatch"), and "discards" which are those organisms thrown back (alive or dead) into the sea.

These definitions, however, seem inconsistent with the recent advent of bans on discarding (often referred to as landing obligations), and the emergence of legal mandates in some countries or regions to land (i.e. retain) all catches. Landing obligations have therefore blurred the distinction between catches and discards. For the purposes of this report, we consider discards as the unwanted and unmarketed portion of the catch, which can either be discarded at sea, or landed (and possibly discarded afterwards).

Amount of discards

Recording and accounting for discards is critical for assessing the environmental effects and sustainability of fisheries (Hall et al. 2020). It was estimated that, globally, about 11% of all catches are discarded, accounting for 9.1 million tonnes annually (CI 6.7–16.1; Pérez Roda et al. 2019).

Bottom trawls have the highest discard rate of all fishing gears, averaging 46% of the total annual discards by global fisheries (i.e. 4.2 million tonnes; Pérez Roda et al. 2019). For example, Borges et al. (2008) estimated that pelagic freezer-trawlers operating in the North Sea and western waters of the British Isles discarded about 30,000 tonnes of fish annually.

On average, bottom trawlers discard 22% of their catch (Pérez Roda et al. 2019), though rates in some shrimp trawl fisheries may be as high as 80–90% (Ramsay et al. 1997). As a comparison, gillnet fisheries produced 0.8 million tonnes of discards annually, with an average discard rate of about 10% (Pérez Roda et al. 2019). Trawl discard rates vary greatly depending on whether the trawling occurs on the bottom or in midwater, with bottom trawls having much higher rates than midwater trawls (Hilborn et al. 2023).

In the Mediterranean Sea, fishery discards average 19%, but those in bottom trawls average 33%; those in "shrimp trawls", 43% (Tsagarakis et al. 2014). FAO (2023) reported bottom trawl discards between 15 and 40% in all Mediterranean subregions (33% in the Adriatic Sea). Discard rates of around 40% were reported from the eastern Mediterranean (Stergiou et al. 1998; Machias et al. 2001; FAO 2022a), with a high occurrence of alien species (Acarli et al. 2022; Cerim et al. 2022).

Trawl discard rates can vary according to regulations, the size and market value of the catch, and other factors (Snyder and Erbaugh 2020). In some instances, landing obligations can require trawl fishers to keep the entire catch onboard, and land everything caught to allow for discard assessments (Damalas 2015; Prellezo et al. 2016; Damalas et al. 2018). For instance, a landing obligation was initiated in 2015 in the expectation that this would help to reduce unwanted catches in EU fisheries (Uhlmann et al. 2012; Damalas 2015). Such regulations, however, are typically difficult to enforce, are not readily accepted by fishers, and have considerable downsides (Celić et al. 2018; Maynou et al. 2018; Onofri and Maynou 2020; Borges 2021).

Impact of discards

Discarding has adverse ecological impacts on marine ecosystems (many of them indirect and not immediately obvious), changing the structure of trophic webs, affecting biodiversity, and playing an important role in the depletion and simplification of marine communities (Harrington et al. 2005; Bellido et al. 2011). Discarding is also an economically wasteful use of valuable fishery resources (Alverson et al. 1994), and the waste of animal lives has been (and should be) condemned on moral grounds (Hall et al. 2000).

The large majority of organisms discarded from trawlers do not survive, with survival depending on species, size, environmental conditions (e.g. air temperature), handling practices (e.g. sorting time), and haul duration and depth (Broadhurst et al. 2006; Tsagarakis et al. 2018; Zeller et al. 2018). There is limited availability of baseline data on non-edible organisms discarded from trawlers, hampering understanding of the ecological damage (Lakshmanan et al. 2021).

A study on the diversity and biomass of non-edible marine fauna discarded from bottom trawlers in the Gulf of Mannar Marine Biosphere Reserve, India, found phyla including Echinodermata (48%), Cnidaria (16%), Mollusca (13%), Arthropoda (10%) and Porifera (5%), with the biomass being dominated by sea stars (26%), sea urchins (17%), jellyfish (16%), gastropods (8%), stomatopods (7%) and sponges (5%). Echinoderms and molluscs showed the highest number of species (Lakshmanan et al. 2021). Discarded species often include species of conservation concern, such as threatened syngnathiform fishes (e.g. seahorses, pipefishes and seadragons; Lawson et al. 2017; Pollom et al. 2021) or corals and sea pens (Fosså et al. 2002; Pierdomenico et al. 2018).

Discard trends

A declining trend in discards of global fisheries became apparent starting in the late 1980s (Hilborn et al. 2023). Specifically, Zeller et al. (2018) found that global annual discards peaked at around 19 million tonnes in 1989, and gradually declined to under 10 million tonnes by 2014. This trend was interpreted as being a result of declining industrial landings, management efforts to reduce discards, and greater retention and use of previously unused catches (Zeller et al. 2018). Such greater retention, in turn, is thought to be driven by a combination of increased gear selectivity and the rising market value of organisms that are landed for use as animal feed and fishmeal (Cashion et al. 2017; Watson and Tidd 2018; Zhang et al. 2020), largely due to a growing aquaculture sector (Tacon and Metian 2015).

An additional reason behind the decline in discards may be that the unwanted and discarded species are declining. As noted by Zeller et al. (2018), fisheries may appear better able to manage and reduce discards not only because of improved fishing practices or gear technology, but also (or rather) because of reduced abundance of the species that are incidentally caught and then discarded. This would mean that even as discards are decreasing, exploitation rates remain high (Zeller et al. 2018).

The global trends described above relate to fisheries generally and are not specific to trawl discards. However, declining discard trends have also been reported for trawl fisheries around the world. In Southeast Asia, for instance, the profitability of trawl fisheries has declined over time due to decreasing catches of valuable species and the increasing costs of fishing (Suuronen et al. 2020). The industry has attempted to compensate for lower revenues by targeting larger quantities of small-sized and low-value species, known as “trash fish” (Suuronen et al. 2020). These organisms are used mainly as feed in the growing aquaculture sector, but they also provide a source of cheap human food for domestic markets (Funge-Smith et al. 2012). Therefore, low-value organisms that were previously discarded have become economically important for bottom trawl fisheries (Suuronen et al. 2020).

In summary, trawl discards are increasingly marketed in response to declines in the marine animal populations that formerly represented the bulk of fishery landings, and a growing demand for all types of seafood and animal feed. Lower discard rates or less discards overall, however, do not simply translate into lower impacts on marine ecosystems. When organisms that were formerly discarded at sea are landed (either to be marketed or because of landing obligations), the net result is further biomass impoverishment (Celić et al. 2018; Watson and Tidd 2018). These negative effects are in addition to those already resulting from exploitation of target species.

Pollution

Pollution related to trawl vessels includes indirect inputs such as toxic contaminants buried in the seabed being mobilized by towed gear, as well as more direct inputs such as engine exhaust, fishing gear being lost or discarded at sea, and the dislodgement of static gear from non-trawl fisheries. The global contribution of trawl fisheries in the form of engine exhaust and other pollutants is unknown; the direct input in terms of greenhouse gas emissions is described in Chapter 6.

Resuspension of contaminants

Seabed sediment acts as a sink for anthropogenic contaminants, and contaminated sediments can have a range of toxicological effects on benthic fauna and associated species. Bottom trawls can mobilize and resuspend large amounts of sediment buried in the seabed; particulate-bound contaminants thus can re-enter the marine food web and become bioavailable to an additional assemblage of species (Robert 2012). For instance, metals including mercury, lead, copper, zinc and cadmium, associated with urbanisation and industrial activities in the Palermo area of Italy were resuspended from sediment by bottom trawls, that mixed heavy contaminated sediment with less contaminated sediment, while also transferring the contaminated sediment to deeper areas (Palanques et al. 2022).

This type of contamination can have health impacts on long-lived species such as marine mammals and other apex predators (humans included) that can accumulate xenobiotic compounds via biomagnification (Desforges et al. 2016; Green and Larson 2016; Jepson et al. 2016; Hall et al. 2017). For example, in a contaminated Norwegian fjord, a single 1.8 km-long trawl pass created a 3–5 million m^3 sediment plume containing around 9 tonnes of contaminated sediment. Substantial amounts of polychlorinated dibenzo-p-dioxins and -furans (PCDD/Fs) and non-ortho polychlorinated biphenyls (PCBs) were released from the sediment, likely causing a semi-permanent layer of contaminated sediment in the bottom waters (Bradshaw et al. 2012). PCDD/Fs from the sediment were also taken up by mussels which, during one month, accumulated to levels above the EU maximum advised concentration for human consumption (Bradshaw et al. 2012).

Lost and discarded fishing gear

Fishing gear comprises a substantial proportion of the global input of plastic waste to marine waters, which results from the disposal and loss at sea of non-biodegradable nets and other synthetic fishing gear such as floaters, polystyrene boxes and ropes (Lebreton et al. 2018; Lambert et al. 2020; Morales-Caselles et al. 2021). Fishing gear that is abandoned, lost or discarded at sea has environmental, social and economic impacts that go beyond those of plastic pollution, as lost gear ("ghost nets") often continues to catch marine organisms, while occasionally also posing a navigation and safety hazard (Stelfox et al. 2016; Jepsen and de Bruyn 2019; FAO 2020b; Gilman et al. 2021; Syversen et al. 2022).

Mortality of marine organisms in lost trawl netting may be lower than that in lost gillnetting, considering that trawl nets are composed of thick synthetic multifilament twine believed to be more visible and detectable by fish (Brown and Macfadyen 2007; Macfadyen et al. 2009; Lively and Good 2019).

Richardson et al. (2022) estimated that nearly 220 km^2 of trawl netting is lost at sea every year. Bottom trawlers lose much more netting than midwater trawlers, with annual point estimates of 215 km^2 for the former and 3 km^2 for the latter (Richardson et al. 2022). The amount of trawl netting lost at sea is small when compared to estimates of more than 75,000 km² of purse seine netting, nearly 3,000 km² of gillnetting and close to 750,000 km of longline lost annually (Richardson et al. 2022). However, in some areas derelict trawl nets are the main type of fishing gear found on beaches and seabed. For example, Donohue et al. (2001) reported that trawl nets accounted for 86–91% of all derelict nets found at two sites in Hawaii. A study of derelict fishing gears found at 11 coastal sites in northern Australia indicated that the majority (71%) were trawl nets, while gillnets accounted for 23% (Edyvane and Penny 2017). Lost trawl netting may become entangled with other nets and litter on the seabed, with bundles that can weigh as much as 1–4 tonnes (Stolte et al. 2022).

As the precise type of netting involved in entanglements of marine fauna is often unreported, or merged with other fishing gear (Stelfox et al. 2016; Jepsen and de Bruyn 2019), the relative impact of trawl netting can be difficult to assess. Laist et al. (1999) provided an extensive global review of "marine debris pollution", citing a study in Alaska by Fowler (1982), who "postulated that in the late 1970s, 50,000 fur seals were entangled and killed annually at sea by marine debris". In the Bering Sea, 10–17% of trawl net fragments examined contained at least one entangled seal (Fowler 1987). As a consequence of additional drag, entangled seals spend longer periods foraging than non-entangled seals (Bengtson et al. 1989; Fowler et al. 1990). In the years 1981–1988, between 15 and 36% of juvenile male northern fur seals were entangled annually in trawl net fragments (Fowler et al. 1990). As noted by Laist et al. (1999), citing Yoshida et al. (1985) and Bengtson et al. (1988), "captive juvenile fur seals—but not the older, larger animals—exhibit a penchant for approaching and entangling themselves in floating trawl net fragments". Although Fowler's early work was strongly challenged, it spurred "vigorous debate and precipitated investigations of interactions involving other species" (Laist et al. 1999, p. 346) not only in North America but more widely (Page et al. 2004).

In the Kaikoura region of New Zealand, 42% of fur seal entanglements were in trawl netting (Boren et al. 2006). Trawl netting was also the most common source of entanglement for Australian fur seals in Tasmanian waters between 1979 and 1991, accounting for 40% of neck collars (Pemberton et al. 1992). Despite seasonal cleaning of hazardous items from their haulout beaches, Hawaiian monk seals *Neomonachus schauinslandi* continued to become entangled in marine debris including trawl netting in the Northwestern Hawaiian Islands (Henderson 2001).

Dislodgement of static fishing gear

In addition to their direct contribution to plastic waste, trawlers are often blamed for damage caused to other fishing gear, including static nets, pots, and buoys that are snagged, dislodged and towed away. The gear displaced or damaged by trawlers is normally not recovered or reused, thus shortening the life cycle of materials and adding to environmental pollution.

For instance, fishers in parts of Norway, Sweden, Britain and Spain reported that trawlers were the main reason behind the loss of gillnets (MacMullen et al. 2003). Fishers in Iran reported that trawlers were the principal cause of the loss of pots (Figure 9 in Haghighatjou et al. 2002). Loss of fishing gear due to trawling was also reported by small-scale fishers in Senegal (Kebe and Ndiaye 1993). Artisanal nets in northern Sri Lanka are regularly damaged by Indian trawlers that cross the International Maritime Boundary to fish illegally during the night, without lights (Menon et al. 2016; Scholtens 2016; Kularatne 2020). In India, artisanal fishers receive regular compensation for the damage to their fishing gear caused by trawlers (Bavinck 2005).

A note on "fishing for litter"

In recent decades, several initiatives have involved the use of trawl nets to collect plastic debris lying on the seabed and bringing it ashore, therefore contributing to clean-up efforts and reducing the quantity of debris that breaks down into microplastics, sinks to the deep seabed or is stranded on the shoreline (Ronchi et al. 2019). The marine litter caught in trawl nets may also be used for scientific research on the amount and type of litter lying on the seabed (Neves et al. 2015; Balcells et al. 2023).

In Europe, "fishing for litter" initiatives developed in the early 2000s, with the support or direct involvement of various stakeholders, organisations, and national governments (Ronchi et al. 2019; Mannaart and Bentley 2022). Attempts to remove plastic from the marine environment are laudable, and may be important for awareness and education as well as for reducing the amount of plastic in particular areas for particular reasons. Generally speaking, however, fishing for litter does not decrease the net global amount of plastic pollution. Its main effect is to move litter from the seabed to ports, and eventually to landfills or incinerators. The litter recovered by trawlers participating in fishing-for-litter programmes is typically not recycled or reused because of its mixed composition and high processing cost (Weißbach et al. 2022). The processes leading to plastic production and consumption (which include industry lobbying, consumer preferences and ease of handling and disposal) remain unaffected.

One fishing-for-litter initiative in the northwestern Adriatic Sea had bottom trawlers display a large sticker reading "This boat takes care of the sea." The owners of these vessels had agreed to land some of the litter caught in their nets instead of discarding it routinely at sea (Ronchi et al. 2019). The sticker was given to trawl fishers "as a recognition of their commitment to protect and preserve the marine environment", in the context of an EU-funded project (Bearzi 2020). However, promoting trawling as a way to "protect and preserve" the marine environment in the northwestern Adriatic Sea—one of the most intensively trawled areas in the world (Eigaard et al. 2017; Gissi et al. 2017; Amoroso et al. 2018; Ferrà et al. 2018; Sguotti et al. 2022) and consequently one with truly deplorable seabed condition (Pitcher et al. 2022)—sends a confusing message to the public.

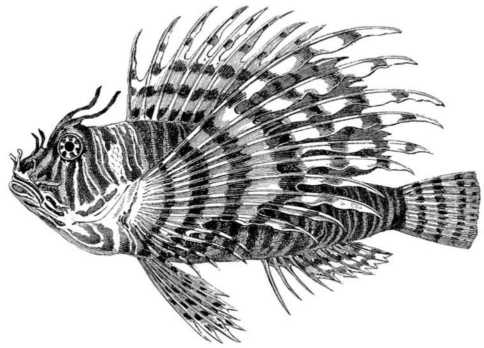
Noise

Anthropogenic noise has a number of negative impacts on marine life (Duarte et al. 2021). For instance, loud noise can influence fish movements, behaviour and physiology, impair hearing, mask communication, and alter predator-prey dynamics (Schwarz 1985; De Robertis and Handegard 2013; Simpson et al. 2016; Popper and Hawkins 2019). Changes in behaviour and movements in response to noise were also observed in shark species (Chapuis et al. 2019). Cetaceans are especially sensitive to sound, and anthropogenic noise can reduce their communication capabilities due to masking effects, modify their communication strategy (e.g. alter the source level, frequency and repetition rate of their calls), and affect their ability to detect prey and predators (Weilgart 2007; Clark et al. 2009; Erbe et al. 2016; Williams et al. 2020b).

Trawling represents a significant source of noise pollution in some areas (Daly and White 2021), with impacts on fish behaviour including avoidance of approaching trawlers (Ona and Chruickshank 1986; Ona and Godø 1990; De Robertis and Wilson 2006). As reviewed in De Robertis and Handegard (2013), fish reactions are not only related to the pressure-based radiated noise, or to sound pressure level alone, but also to changes in the propeller pitch and noise produced by trawl warps. Marine predators that spend considerable time foraging and scavenging behind operating trawlers, such as several species of odontocete cetaceans and pinnipeds (Tixier et al. 2021; Bonizzoni et al. 2022), are more exposed to the noise generated by engines and trawl gear, and the source levels of this noise may exceed hearing-damage thresholds of odontocetes (Daly and White 2021).

While the harmful effects of onboard noise are relatively well documented (Zytoon 2013; Levin et al. 2016; Lee et al. 2021), there is scant and methodologically inconsistent information on the underwater noise produced by trawlers. Bottom trawlers in Croatia, 10–30 m long and powered by 220–320 HP engines, recorded at distances of approximately 25 m, produced "equivalent continuous" sound pressure levels (SPLs) of 143 dB re 1 µPa and a maximum "instantaneous" SPL of 149 dB re 1 µPa (Rako et al. 2013). The activity of these trawlers at the time was not specified. Bottom trawlers off Ireland, 17–20m long and actively fishing, produced noise with a peak in "root mean square" SPL of 127 dB re 1 µPa (Daly and White 2021). A 66 m stern trawler powered by a 1,800 HP engine and towing a pelagic trawl produced "radiated noise" of up to about 160 dB re 1 µPa (De Robertis and Wilson 2006). Noise estimated by acoustic modelling for three bottom otter trawlers, 16, 20 and 25 m long, actively trawling in the Jabuka/Pomo Pit area of the central Adriatic Sea, produced continuous noise levels of 171, 173 and 175 dB re 1 µPa, respectively (GFCM/FAO 2021).

The levels of underwater noise caused by trawlers in a given area depend on various factors, including engine type and power, activity, number of trawlers and sea bottom depth, substrate and morphology (Daly and White 2021; GFCM/FAO 2021). Engine noise is especially loud when a net is being towed (e.g. Figure 6 in Lee et al. 2021), and gear drag and friction generate additional underwater noise. For instance, trawl gear such as shackles and chains produce rattling noise, and ground ropes and otter boards produce low-frequency sound (Hawkins and Chapman 2000; Daly and White 2021).



Behavioural effects on marine megafauna

As fisheries expanded in coastal and pelagic habitats, large marine predators came into contact with fishing gear that concentrated prey predictably and made it is easier to catch. Some of these predators have responded by modifying their behaviour to take advantage of the foraging opportunities provided by fishing gear—including trawl nets (Bearzi et al. 2019). Species and populations of marine vertebrates known to have become accustomed to foraging and scavenging in the wake of trawlers include elasmobranchs (Ferretti et al. 2013; Mitchell et al. 2018), seabirds (Oro and Ruiz 1997; Karris et al. 2018), odontocete cetaceans (Bonizzoni et al. 2022) and pinnipeds (Tixier et al. 2021).

Sharks are known to take fish from fishing gear (Mitchell et al. 2018), and they can follow trawlers to feed on discards (Mitchell et al. 2023). For instance, sharks followed prawn trawlers in Australia during both day and night to feed on discards (Hill and Wassenberg 1990). Fishers onboard shrimp trawlers in the south-eastern United States reported increasing observations of sharks taking fish from the nets (Picariello et al. 2022).

Seabirds, including gulls, albatrosses, petrels, cormorants, gannets, shearwaters, and penguins, often approach trawlers to feed on discards and take fish from the nets. This can influence the birds' movement pattern, group size, diet and breeding success (Oro et al. 1995, 1999; Arcos and Oro 1996; Louzao et al. 2006; Navarro et al. 2009; Bartumeus et al. 2010; Camphuysen 2011; Bodey et al. 2014; Giménez et al. 2021). For instance, discards from the deep-water hake fishery make up more than half the diet of white-chinned petrels (Jackson 1988; Crawford et al. 1991), and are important for black-browed and shy albatrosses and Cape gannets (Crawford et al. 1991; Berruti et al. 1993).

At least 19 odontocete cetacean species are known to feed in the proximity of trawlers in many parts of the world (Bonizzoni et al. 2022). Interactions with trawlers affect the movement, distribution, diet, group size, social behaviour and social organization of the populations involved (Waring et al. 1990; Chilvers and Corkeron 2001; Allen et al. 2017; Bonizzoni et al. 2022, 2023). Similarly, at least seven species of pinnipeds are known to forage in the proximity of trawl nets (Tixier et al. 2021), and operating trawlers can affect their distribution and group size (Hamer and Goldsworthy 2006; Lyle et al. 2016).

In some cases, the trawl gear that provides extra calories to the animals is also responsible for degrading their habitat (Bonizzoni et al. 2023). While animals that forage behind trawlers may increase their food intake and reduce energy expenditure, any advantage gained may be partially or entirely offset by the risk of injury and death resulting from proximity to trawl gear (see the Bycatch section), as well as by the negative impacts of exposure to noise and depletion of their natural prey. Therefore, foraging behind trawlers can either benefit the predators involved and help them survive, or ultimately prove maladaptive by increasing mortality and contributing to population decline. Due to the complexity of marine food webs (as well as the effects of trawling on food webs and the general environmental degradation caused by trawling), it is often impossible to disentangle causes and effects, determine whether the benefits to the animals outweigh the risks, and assess potential population-level impacts of trawler-associated foraging. Adaptation to foraging behind trawlers may, at least in some contexts, reflect a lack of options: the animals may be simply exploiting the resources that remain available to them (Bearzi et al. 2019).

Ecosystem recovery after trawling

Ecosystem recovery dynamics are often complex, and the time required for full or partial recovery depends on a number of physical, environmental and biological variables. These include factors influencing the recruitment and growth of benthic species (e.g. local hydrodynamics, temperature, food availability, reproductive mode, larval development, availability of substrate for settlement), habitat features, structure of the affected communities, connectivity to undisturbed or less disturbed populations, and the magnitude and spatial extent of fishing disturbance (Goode et al. 2020). While some degree of recovery can be expected in the years or decades following trawl bans (McConnaughey et al. 2020), the extent and pattern of such recovery are often hard to predict.

Sessile and low-mobility biota with longer life-spans such as soft corals, sponges and bivalves require longer recovery times than mobile biota with shorter life-spans such as polychaetes and malacostracans (Kaiser et al. 2006; Sciberras et al. 2018). For example, model-based estimates of ecosystem recovery following a 67% biomass removal by bottom trawling indicated that sponges may recover to 80% of their original biomass after 20 years, while corals may recover to 80% of their original biomass after 34 years (Rooper et al. 2011).

One study suggested recovery of the mantis shrimp (stomatopod) assemblage 3.5 years after a ban on bottom trawling in Hong Kong waters (Tao et al. 2018). In the same area, comparisons of survey data before and after the ban revealed higher organic content in sediment, lower loads of suspended particles, significant increases in species richness and functional diversity, higher mean trophic level of the fish community, and greater abundance and biomass of fish after the ban, suggesting that trawl bans can help restore and conserve biodiversity in tropical coastal waters (Mak et al. 2021; Wang et al. 2021).

In waters off Sicily, Italy, four years after a trawl ban there was an impressive increase in biomass of red mullet *Mullus barbatus* and other finfish species, including a roughly 500-fold increase in large-scaled gurnard *Lepidotrigla cavillone* (Pipitone et al. 2000; Fiorentino et al. 2008; Agnetta et al. 2019). The only recorded decrease was for horned octopus *Eledone cirrhosa* (Pipitone et al. 2000).

However, the long-term consequences of bottom trawling, and the time needed for benthic communities to recover, are often underestimated. In a study involving active restoration of former oyster reef habitat following disturbance caused by bottom fishing (including dredging), Hemraj et al. (2022) noticed a rapid increase in biodiversity and abundance of reef-associated species within the first two years, but increase rates slowed substantially afterwards, leaving a shortfall in recovery of 35% below the pre-disturbed status, with the time to achieve full recovery remaining unquantified. In the case of major regime shifts caused by long-lasting and intensive trawling, full recovery to ecosystem conditions resembling those pre-trawling may be either impossible or would require a very long time (Sguotti et al. 2022).

A study on the recovery of seagrass *Posidonia oceanica* meadows degraded by bottom trawling estimated that reaching the status of control meadows unaffected by trawling could take up to 100 years, due to low rates of vegetative growth (González-Correa et al. 2005). Benthic communities on deep-water seamounts affected by bottom trawling showed little sign of recovery in the years following a trawl ban (Althaus et al. 2009; Clark et al. 2019). Based on a literature review, Goode et al. (2020) concluded that trawl impacts on seamount benthic communities are not reversible within several decades, and suggested that the ideal conservation approach would be to protect seamounts before any fishing occurs. However, those authors noted that protection measures of some kind should be applied to both pristine and disturbed areas, because protecting the latter to at least some degree increases the chances for long-term recovery of benthic communities and restoration of ecosystem services.

Impacts on climate

6

The carbon footprint of global fisheries

Industrial fisheries rely heavily on fossil fuels. Based on a 2011 estimate, global fisheries burned about 40 billion litres of fuel annually and generated approximately 180 million tonnes of carbon dioxide equivalent (CO_2eq) greenhouse gas emissions (GHG), accounting for approximately 4% of global food production emissions (Parker et al. 2018). Global fishery emissions increased by 28% between 1990 and 2011. Over the same period, emissions per tonne of landings increased by 21%. The overall increase in emissions was driven primarily by fuel-intensive crustacean fisheries as well as fisheries for demersal and reef fish (Parker et al. 2018).

Fuel consumption accounted for 60–90% of GHG emissions up to the point of landing (Tyedmers 2004; Parker and Tyedmers 2015). When emissions across the seafood supply chain were considered (including vessel construction and maintenance, and post-landing activities such as processing, packaging and transport), vessel fuel use still remained the primary source of emissions (Vázquez-Rowe et al. 2012; Parker and Tyedmers 2015).

The median fuel use intensity of global fisheries was estimated as 639 litres per tonne of landed fish (Parker and Tyedmers 2015), with approximately 2.2 kg CO_2eq produced for each landed kilogram (Parker et al. 2018). GHG emissions, however, varied markedly among fishery sectors and fleets, largely reflecting differences in the amount of fuel required for fishing different target species (Parker et al. 2018).

For instance, between 1990 and 2011, crustacean fisheries that accounted for only 6% of landings contributed over 22% of CO_2eq emissions (Parker et al. 2018). The least energy-efficient fisheries, globally, are those targeting shrimps and lobsters. These fisheries catch relatively small quantities per trip compared to those targeting finfish, and can use 10,000 litres per landed tonne, or more (Parker and Tyedmers 2015; Parker et al. 2018). Conversely, fisheries for pelagic species (typically under 30 cm in length), which accounted for about 20% of landings, contributed only 2% of global fishery emissions (Parker et al. 2018; Fig. 6).

A beam “rapido” trawler in the northern Adriatic Sea; photo by S. Bonizzoni

Between 1990 and 2011, the fishing fleets with the largest overall GHG emissions were based in China, Indonesia, Vietnam, the United States and Japan. In 2011, these countries accounted for 37% of landings and 49% of total emissions, producing 81 million tonnes CO_2eq (Parker et al. 2018). The high GHG contribution by Asia reflects its extensive fishing fleets. The Chinese fleet, alone, emitted 50 million tonnes CO_2eq: approximately one quarter of total global fishery emissions and more than the combined CO_2eq of all fisheries in Europe and the Americas. Countries that targeted primarily crustaceans, such as Saudi Arabia and Australia, had the most carbon-intensive fleets. Conversely, the west coast of South America accounted for 15% of global fishery production by landed weight in 2011, but contributed only 3% of CO_2eq emissions—largely due to a high proportion of the landings consisting of small pelagic fish, whose capture tends to be relatively “fuel-efficient” (Parker et al. 2018).

The carbon footprints of bottom trawling

In recent years, bottom trawling has been the subject of considerable media attention and scientific debate because of its high direct and indirect contributions to global GHG emissions—and thus climate impacts. The carbon footprint of bottom trawling consists primarily of two things. First, bottom trawlers are among the least fuel-efficient fishing vessels, and vessel operations contribute most of the fishing industry’s direct GHG emissions. Second, bottom trawling resuspends biogenic carbon stored in marine sediments, resulting in carbon release into the sea and the atmosphere. The GHG footprint of seafood caught by bottom trawls is consequently high.

Direct carbon footprint from fuel use

Fuel burning is by far the most significant contributor to GHG emissions of fisheries, and this is particularly true for fuel-intensive vessels such as bottom trawlers. Bottom trawlers rank among the highest emitters in terms of fuel consumption per kilogram of fish when compared to all major gear types used in global fisheries (Thrane 2004; Schau et al. 2009; Parker and Tyedmers 2015; Jafarzadeh et al. 2016; Bastardie et al. 2022). Clark and Tilman (2017) estimated that, on average, bottom trawl fisheries emit 2.8 times more GHG than non-trawl fisheries, due to the fuel required for dragging a heavy net across the seabed. For instance, European beam trawlers longer than 40 m could emit up to 9.5 kg of CO_2 per kg of fish (Cheilari et al. 2013). Mohiuddin et al. (2024) investigated the emissions of CO_2, CO, CH_4, NO_x, SO_x, and PM during the 21-day voyage of a 40-m long factory freezer trawler equipped with midwater trawl gear. The study estimated engine emissions ranging between 213 and 275 tons, with carbon dioxide emissions accounting for approximately 97% of total emissions.

Overall, there is high variability in fuel use and GHG footprint within and among trawl fisheries. Trawlers targeting well-managed and abundant finfish populations tend to require less fuel per landed catch, as do newer and more fuel-efficient vessels (Parker et al. 2018; Hilborn et al. 2023). Bottom trawlers targeting crustaceans tend to be the least fuel-efficient. For example, trawlers targeting brown tiger prawns *Penaeus esculentus* in Australia, and bottom trawls targeting Norway lobsters *Nephrops norvegicus* in Sweden, could consume more than 11,000 and 17,000 litres per landed tonne, respectively (Parker and Tyedmers 2015): 11 and 17 litres of fuel per kg of landed seafood.

In a comparison of trawlers operating in the Adriatic Sea, beam “rapido” trawlers targeting common sole *Solea solea* and purple dye murex *Bolinus brandaris* were the least fuel-efficient, while midwater pair trawlers targeting European anchovy and European pilchard were the most fuel-efficient. Specifically, beam “rapido” trawlers required 13.6 litres (95% CI 10.5–16.6) of fuel to obtain 1 kg of common sole (Sala et al. 2022).

Larger trawl vessels tend to have higher landing rates, but burn more fuel per unit of effort than small vessels (Sala et al. 2022). Fuel costs can represent 50% of the operational costs for large bottom trawlers, while it can be only 5% for small-scale fishing vessels operating static gears (Cheilari et al. 2013). However, trap fisheries targeting crustaceans such as lobster can require substantial amounts of fuel (Parker and Tyedmers 2014; Hilborn et al. 2018).

Seafood is often credited for being a “sustainable” dietary choice with regards to climate impacts. However, seafood caught by bottom trawls ranks among the most GHG-intensive foods, with footprints exceeding those of poultry and pork (Clark and Tilman 2017).

Hilborn et al. (2023) compared the carbon footprint of processed products from life cycle assessment of crops, livestock, and capture fisheries: the average footprint for bottom trawl fisheries was higher than all other foods listed except beef (and much higher than plant-based foods). They argued, however, that a few well-managed bottom trawl fisheries had carbon footprints below chicken and pork.

A bottom otter trawler in the semi-enclosed Northern Evoikos Gulf, Greece; photo by G. Bearzi

Indirect carbon footprint from seabed disturbance

Marine ecosystems absorb CO_2 from the atmosphere, with the biological pump assimilating inorganic carbon into organic compounds. This process of carbon capture is an important sink for CO_2 released into the atmosphere as a result of human activities (Khatiwala et al. 2009; Gruber et al. 2019; Watson et al. 2020).

Marine sediment is one of the most important long-term carbon stores (Epstein et al. 2022; Atwood et al. 2024). Once buried into the seabed, organic carbon can remain unmineralized for millennia (Burdige 2007; LaRowe et al. 2020). Seabed sediment, however, is subjected to physical disturbance from a variety of human activities (Halpern et al. 2019; Levin et al. 2020; O'Hara et al. 2021). Bottom trawling (together with dredging) is by far the most widespread single cause of seabed disturbance (Epstein et al. 2022). The mechanical action of bottom trawl gear can disturb carbon that took millennia to accumulate into marine sediment. Through mixing, resuspension and oxidation of the sediment, bottom trawling increases the remineralization of organic carbon while also limiting its burial by inhibiting the settlement and consolidation of sediment (Martín et al. 2014b; Oberle et al. 2016a; Keil 2017; Luisetti et al. 2019; De Borger et al. 2021). This disturbance can increase the concentration of inorganic carbon, lower the ocean's buffering capacity, and slow the rate of CO_2 uptake from the atmosphere, while also contributing to ocean acidification and releasing oceanic CO_2 into the atmosphere (Bauer et al. 2013; Keil 2017; Luisetti et al. 2019; LaRowe et al. 2020; Epstein et al. 2022).

Seabed disturbance by trawling, however, triggers multiple, sometimes conflicting, mechanisms, and the net effects on carbon stocks and fluxes remain uncertain due to the complexity of the involved processes (Legge et al. 2020). Specifically, the cycling and storage of organic carbon is influenced by factors including the occurrence and activity of benthic organisms, seabed features such as lithology and granulometry, and the chemistry, hydrology and biology of the surrounding water column (Epstein et al. 2022). All these factors are affected by positive and negative feedback mechanisms, and identifying ("teasing out") the effects of bottom trawling on the net storage of organic carbon has proved challenging (Keil 2017; Snelgrove et al. 2018; LaRowe et al. 2020; Rühl et al. 2020; Epstein et al. 2022).

As the amount of carbon stored in sediment varies among regions, the extent of GHG emissions caused by bottom trawling disturbance can depend on locality (Diesing et al. 2021). Epstein et al. (2022) evaluated 49 studies on the effects of bottom trawling on seabed carbon, and found that 61% showed no significant effects, 29% reported lower organic carbon after trawling, and 10% reported higher organic carbon after trawling. Those authors noted that “more evidence is urgently needed to accurately quantify the impact of anthropogenic physical disturbance on seabed carbon in different environmental settings and to incorporate full evidence-based carbon considerations into global seabed management” (Epstein et al. 2022). Hilborn et al. (2023) concluded that “there is little evidence that trawling increases sediment carbon mineralization significantly, even less that it impacts atmospheric CO_2 levels, but uncertainty certainly remains”. Conversely, a study by Atwood et al. (2024), based on satellite information on fishing and carbon cycle models, indicated that between 1996 and 2020 global bottom trawling could have released up to 370,000,000, but more likely in the range of 20,000,000 to 210,000,000, metric tonnes of CO_2 into the atmosphere while also altering water pH in some semi-enclosed and heavily trawled seas. Those authors also estimated that 55–60% of the CO_2 released into the water column by bottom trawling will have entered the atmosphere within nine years after a trawling event. They suggested that reduction of bottom trawling would help “close the emissions gap to limit global temperature increases to 1.5°C” and, as such, be “an effective ocean-based climate solution.”

Overall, the impact of demersal fisheries (primarily bottom trawling and dredging) on seabed carbon is a topic of growing concern, yet “existing literature presents inconsistencies leaving experts divided on the topic” (Tiano et al. 2024). Despite limitations in the information currently available, Tiano et al. (2024) highlighted the strong potential of demersal fisheries to disrupt sediment biogeochemistry, and noted that “without urgent management strategies, these effects will likely affect the global carbon footprint, ultimately impairing the functioning of marine ecosystems and their ability to provide goods and services to humanity”.

Impacts on people and society

7

The human dimension of trawling

In addition to its effects on marine life and the climate, bottom trawling has impacts on people and on society generally (Teh et al. 2019; Kularatne 2020; Seto et al. 2023). Literature on the social and economic effects of trawling is limited, and much of the evidence of conflict, safety and health issues, and human rights violations is anecdotal.

More systematic, rigorous social science studies and investigative reports on the human dimensions of industrial fisheries are needed. An improved understanding of these dimensions may not only result in legislative and management actions that benefit the victims of abuse, but possibly also contribute to more effective environmental conservation outcomes (Bennett et al. 2017). In this chapter, we touch briefly on some of the problems that trawl and other industrial fisheries can bring to coastal communities as well as to the trawl workers themselves.

Spatial overlap and conflict with small-scale fisheries

Small-scale fisheries contribute an estimated 40% of the global catch and support 90% of the capture fisheries workforce, with women representing 40% of all those engaged in the aquatic value chain (FAO 2024). According to FAO, approximately 500 million people rely on small-scale fisheries for their livelihoods, including 53 million involved in subsistence fishing (45% of whom are women; FAO 2024).

Bottom trawling can overlap with areas of intensive fishing by small-scale fleets, particularly in continental shelf waters. Such overlap can result in competition for local resources. In the worst cases, trawl fleets deplete and jeopardize the marine resources that sustain small-scale, community-based coastal fisheries that are usually short-range and rely on static fishing gear such as gillnets, trammel nets, longlines and traps. Depletion of local resources inevitably fuels social and economic unrest, usually with loss of revenue and jobs, hence emigration. Additional economic damage, and occasionally conflict, can come from trawl nets displacing, removing or damaging the static gear used by local fishers (Seto et al. 2023).

These types of conflict can lead to trawl bans in areas used by inshore fisheries (Hart et al. 2002; Belhabib et al. 2020; de la Puente et al. 2020; McConnaughey et al. 2020). In Scotland's Firth of Clyde, bottom trawling was banned in coastal waters in 1889 to protect small-scale fisheries. Nearly a century later (in 1984), repeal of the ban resulted in collapse of inshore fisheries, conflict between small-scale and industrial fishers, and a loss of economic opportunities for coastal communities (Thurstan and Roberts 2010).

Many small communities that rely on fishing for income, employment and food security regard competition and conflict with industrial fleets as one of the greatest threats to their welfare and way of life (Salayo et al. 2006; JALA 2007; Sumaila 2018; EJF 2021). Conflicts tend to be particularly acute in the coastal waters of tropical developing countries, where trawlers out-compete small-scale fishers and deplete the resources historically consumed by local communities (World Bank 2012; Visser 2015; Glaser et al. 2019). Such conflicts can involve verbal altercations, physical confrontation, destruction of boats, assault, kidnapping and even murder (Seto et al. 2023).

In countries such as India, where the vast majority of small-scale fishers live below the poverty line, conflict or competition with trawl fleets resulted in less food, lower incomes and forced migration (Jacob and Rao 2016). A long history of violence and conflict between Sri Lankan and Indian fishers and the bottom trawl fleet has been particularly well documented, with heightened tensions leading to violent confrontation, physical injuries, increased patrolling and arrests (Stephen 2014; Menon et al. 2016; Scholtens 2016; Stephen and Menon 2016; Ravi Krishnan and Pichaandy 2017; Kularatne 2020).

In Senegal, loss of static fishing gear due to trawling—often carried out illegally in coastal waters, during the night and without lights—caused social and economic unrest for artisanal fishers who fished with canoes (Kebe and Ndiaye 1993). In a rare case documented in Papua New Guinea, large portions of trawl bycatch were discarded, but in inshore areas close to major towns arrangements were made to give a portion of the unwanted trawl catch to local villagers, and this form of almsgiving reportedly mitigated the degree of hostility towards trawling (White et al. 2019, p. 3).

Foreign fleet effects on local communities

Coastal countries often engage in agreements that confer fishing access to foreign bottom trawlers. These agreements allow governments to gain short-term revenue in exchange for local fish resources. For instance, in the West African states of Ghana, Guinea and Sierra Leone, the revenue from access agreements with foreign bottom trawl fisheries in 2015 was between 2 and 8% of the estimated value of the foreign trawlers' landings (Virdin et al. 2019). In 2017, fleets of Chinese bottom trawlers operating in the coastal waters of West Africa generated little or no net economic benefit to the "host" country in terms of rent paid (Virdin et al. 2022). Any such benefit that does go to the host country tends to stay within the central government and is not reinvested into livelihoods and the long-term economic improvement of the affected local communities. In general, access agreements ultimately deprive coastal communities of critical resources, threaten local economies, and sometimes lead to social unrest and emigration (Virdin et al. 2019, 2022).

The social effects of foreign bottom trawling on local communities appear to be under-investigated globally, but some regions have a long and relatively well-documented history of conflict. For instance, social unrest, violence and food insecurity in Mauritania were related to foreign fishmeal industries that were primarily sustained by foreign trawl fleets, with many trawlers operating within areas reserved for small-scale fishers (Corten et al. 2017; Steadman et al. 2021). In Somalia, bottom trawling by foreign fleets fuelled public anger and social conflict because of direct competition with the domestic fishery, illegal trawling near shore, issues around vessel licensing, and links to piracy, jeopardizing long-term livelihood security (Glaser et al. 2019). In Sri Lankan waters, where bottom trawling is banned, conflict with illegal bottom trawlers from India led to adverse socio-economic impacts as well as violent conflict with local communities (Kularatne 2020).

Abuse of human rights

The nature of working at sea makes it difficult to monitor the labour conditions of fishing crews. Industrial fishing vessels can remain at sea for months, and during that time crew members may be unable to disembark and subjected to working conditions beyond the oversight of regulators (Tickler et al. 2018). In cases of abuse, it is also often unclear in which country a crew member can seek redress, given the jurisdictional complexities and the use of flags of convenience (Österblom et al. 2010; Miller and Sumaila 2014; Lindley and Techera 2017). These factors can enable the use of exploitative practices that reduce employer costs at the expense of employee salaries, safety and freedom (Marschke and Vandergeest 2016; Tickler et al. 2018).

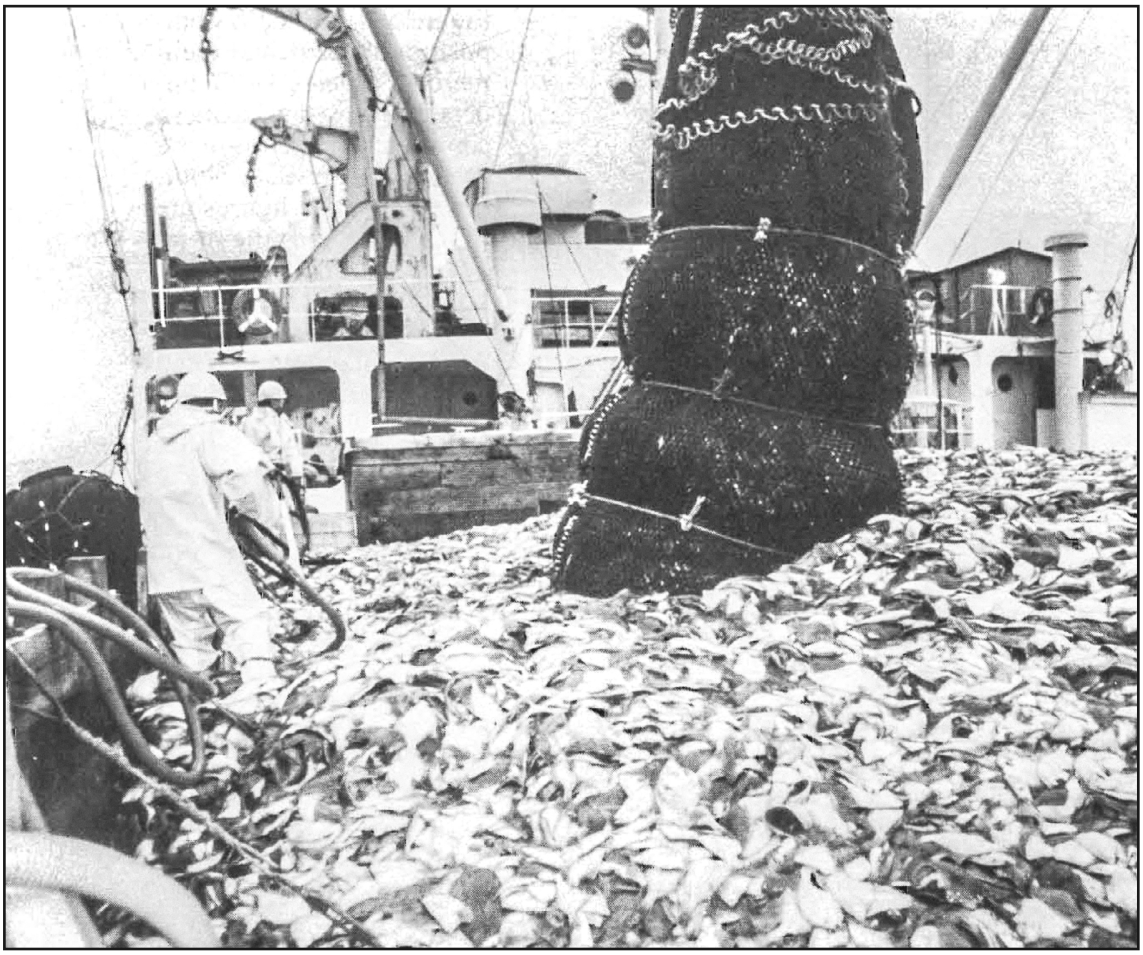
Reliance on forced labour in industrial fisheries, including in some trawl fisheries, is widespread and serious, in many cases meeting the definition of modern slavery (Tickler et al. 2018; Teh et al. 2019). Reported cases include evidence of forced confinement, physical abuse, human trafficking, appalling violations of human rights, and murder (EJF 2010, 2015; UNODC 2011; ILO 2013; IOM 2016; Pocock et al. 2016; Tickler et al. 2018; Teh et al. 2019; Selig et al. 2022). Gruesome cases involving South Asian trawl fleets and other fisheries have been well documented in media articles (e.g. Hodal and Kelly 2014; Urbina 2015a, 2015b, 2022). However, violations and crimes on industrial fishing vessels are not confined to developing countries, areas of weaker jurisdictions, or the high seas—cases have been reported in Europe, New Zealand and the United States (Murray 2014; Simmons and Stringer 2014; Mendoza and Mason 2016; Lawrence and McSweeney 2017; McSweeney and Lawrence 2017).

Human safety and health

Fishing has always been and continues to be one of the most hazardous occupations. Labour conditions on industrial fishing vessels can be challenging even when the operations are well managed. In 1996, reported fatality rates for fishers around the world could be up to 20–40 times higher than the national averages (Petursdottir et al. 2001). Conservative global estimates over the past two decades have put human fatalities at between 24,000 and 32,000 per year, but recent research revealed that the true rates could be three or four times higher (Willis et al. 2023).

Specific information on occupational health and safety impacts onboard bottom trawlers is scant, but reports on trawlers generally suggest that trauma and disability affect roughly 7% of workers (Shapovalov 2017). In Norway, where industrial fishing is the occupation with the most fatal and non-fatal accidents, the trawl fleet had the highest injury rate, accounting for 37% of all reported injuries across the entire Norwegian fishing fleet (McGuinness et al. 2013). High mortality and injury rates were also found in Egypt, where use of trawl and gillnet or trammel gears resulted in higher rates of injuries and health disorders than use of purse seines (Zytoon 2012).

Exposure to fuel exhaust onboard trawlers can result in respiratory and other health impairment. In a study of trawl fisheries in India, fishers exposed to fuel exhaust reported more than double the number of respiratory symptoms compared to unexposed fishers. They also had higher chances of experiencing chronic cough and phlegm, wheezing, and other symptoms of respiratory disease (Moitra et al. 2015). Workers on trawl and other industrial fishing vessels are also routinely exposed to high, prolonged and potentially harmful noise levels (Zytoon 2013; Levin et al. 2016; Lee et al. 2021), with effects ranging from tinnitus to noise-induced hearing loss.



Catch of yellowfin sole *Limanda aspera* being unloaded on the deck; from Bakkala et al. (1979)

Fishery subsidies

8

What are fishery subsidies?

There are many formal definitions of what constitutes a subsidy (Cox and Sumaila 2010). Within the fishery sector, the FAO defined subsidies as "government actions or inactions outside of normal practices that modify—by increasing or decreasing—the potential profits by the fisheries industry in the short-, medium- or long-term" (Westlund 2004). More simply, Sumaila et al. (2016) defined fishery subsidies as "financial payments from public entities to the fishing sector, which help the sector make more profit than it would otherwise."

Fishery subsidies can include fiscal incentives, loans, or the provision of services, as well as payments by governments that are labelled as economic assistance, support programmes, financial support, or financial transfers (Sumaila et al. 2016). "Fuel subsidies" (i.e. discounted prices for fuel, including fuel-specific tax exemptions) represent the most common and greatest proportion of fishery subsidies, both globally (Sumaila et al. 2019; Schuhbauer et al. 2020) and locally (Carvalho and Guillen 2022; Shen and Chen 2022; Vaughan et al. 2023).

The classification of subsidies

FAO classified fishery subsidies into four categories, based on the type of government intervention (Westlund 2004): 1) direct financial transfers, 2) services and indirect financial transfers, 3) regulations and 4) lack of intervention.

Sumaila et al. (2010a) provided an alternative and more practical classification, often used as a reference in the scientific literature, by classifying subsidies based on their potential impact on the fishery resource:

a) "beneficial" subsidies that lead to investment in natural capital assets, including fishery management programmes and services, fishery research and development, and marine protected areas;

b) "capacity-enhancing" subsidies that encourage fishing capacity to develop beyond maximum sustainable yield (resulting in overexploitation) and may include fuel subsidies, boat construction, renewal and modernization programmes, fishing port construction and renovation programmes, price and marketing support, processing and storage infrastructure programmes, fishery development projects and support services, and foreign access agreements;

c) "ambiguous" subsidies that have undetermined impacts and may lead to either lower or higher fishing pressure, including controversial fisher assistance initiatives such as vessel buyback programmes and rural fishing community development programmes (Sumaila et al. 2010a, 2016, 2019).

The scale and allocation of subsidies

Total monetary value of global fishery subsidies was estimated to be in the order of USD 35.4 billion in 2018, with the greatest proportion (USD 22.2 billion) being represented by capacity-enhancing subsidies (Sumaila et al. 2019). The total monetary value of capacity-enhancing subsidies tended to be larger than that of beneficial and ambiguous subsidies in all regions except North America and Oceania, where the value of beneficial subsidies tended to be larger (Sumaila et al. 2019).

Asia (especially China) was by far the greatest subsidizing region (55% of the total), followed by Europe (18%) and North America (13%). In terms of individual countries, China provided the highest proportion of global subsidies (21%), followed by United States and Republic of Korea (10% and 9% of the total, respectively). EU Member States collectively provided 11% of global fishery subsidies (USD 3.8 billion); of these, 54% of the EU estimate consisted of capacity-enhancing subsidies and 40% of beneficial subsidies (Sumaila et al. 2019; Figs. 6 and 7).

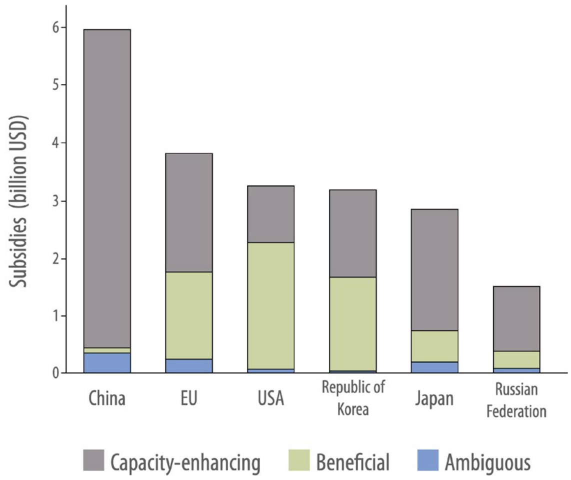


Fig. 6. Fishery subsidies by major fishing countries and political entities in 2018 (adapted from Sumaila et al. 2019)

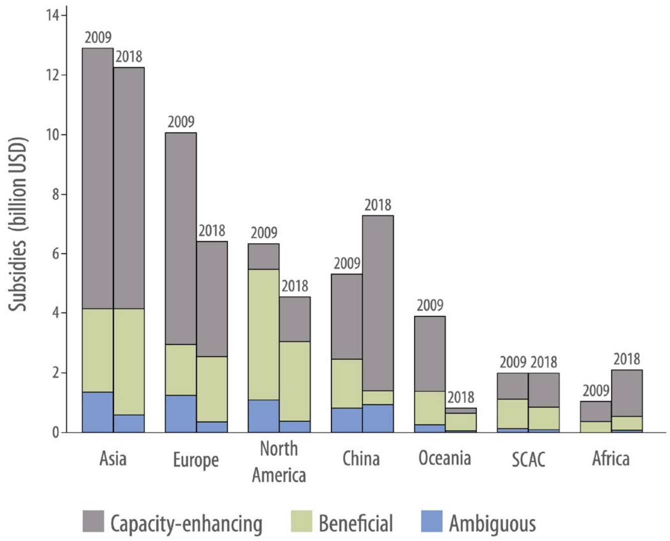


Fig. 7. Fishery subsidies by nation or region in 2009 and 2018; SCAC = South America, Central America and Caribbean (adapted from Sumaila et al. 2019); differences between the 2009 and 2018 estimates can be due in part to improved estimation methods and increased data collection effort (Sumaila et al. 2019)

The allocation of fishery subsidies typically favours industrial fisheries (Schuhbauer et al. 2017; Villasante et al. 2022). Of the USD 35.4 billion of global fishery subsidies provided in 2018, approximately 19% went to the small-scale sector, and 80% to the industrial sector (Schuhbauer et al. 2020). Between 2009 and 2018, capacity-enhancing subsidies increased for both fishery sectors, but the increment was 18% for industrial fisheries and 3% for small-scale fisheries (Schuhbauer et al. 2020).

Fuel subsidies constituted the majority (22%) of global subsidies, whereas subsidies for fishery management accounted for 19% and (non-fuel) tax exemptions for 10% (Sumaila et al. 2019). Fuel subsidies to the fishing sector were estimated to be in the range of USD 4.2–8.5 billion per year, globally (Sumaila et al. 2008). These subsidies are considered to be the most directly linked to overfishing (Sumaila et al. 2019), and are especially important for trawl fisheries, where fuel is the main operational cost (Cheilari et al. 2013; Sala et al. 2023).

The problem with subsidies

It is widely acknowledged that the overcapitalization of global fisheries has contributed to the depletion of fish populations (Sumaila et al. 2016; Pauly et al. 2022). Capacity-enhancing subsidies increase profit and fishing effort, and drive the build-up of excessive fishing capacity, thereby threatening marine resources and the livelihoods that depend on them (Clark et al. 2005; Sumaila et al. 2010a, 2016; Harper et al. 2012). While capacity-enhancing subsidies have historically contributed to overcapacity and overfishing, they continue to be allocated to maintain the profitability of industrial fisheries (Villasante et al. 2022).

Fuel subsidies, in particular, perpetuate fuel-inefficient technologies and help industrial fisheries stay in business even when the true operating costs outweigh fishing revenues (Sumaila et al. 2010b; Schuhbauer et al. 2020). These and other subsidies also give industrial fisheries a significant and unfair advantage over small-scale fisheries (Schuhbauer et al. 2020).

Without capacity-enhancing subsidies, many industrial fisheries would be economically unviable (Sumaila et al. 2010b; Sala et al. 2018b; Vaughan et al. 2023). For example, Sala et al. (2018b) noted that the economic balance of deep-sea bottom trawlers operating in the Northeast Atlantic was mostly negative, and only became positive when capacity-enhancing subsidies were considered. An analysis of the economics of the U.K. fishing fleet indicated that bottom trawling and dredging would become unprofitable or only marginally profitable if fuel subsidies were eliminated (Vaughan et al. 2023).

Capacity-enhancing subsidies are sometimes used to support trawl fisheries that have been shown to have severe environmental impacts. For instance, a part of the fishery subsidies the European Maritime and Fisheries Fund (EMFF) allocated in 2016 and 2017 were used by France to fund the purchase of midwater trawlers operating in the Bay of Biscay (Seas at Risk 2020), where high numbers of common dolphins are bycaught in midwater trawls (Rouby et al. 2022). This means that EC public funds have been used to support the purchase of trawlers that are directly responsible for the deaths of thousands of dolphins while, at the same time, the EC filed an infringement procedure against France for its lack of action in protecting cetaceans from bycatch in fishing gear (Seas at Risk 2020; and see ec.europa.eu/commission/presscorner/detail/en/inf_20_1212).

Globally, industrial fishing is dominated by wealthy nations, and the fleets of these nations have expanded to encompass the territorial waters of many lower-income nations (see Chapter 3: Bottom trawling by distant countries). Most of the industrial fishing within EEZs of lower-income nations is conducted by the distant-water fleets of high-income nations (McCauley et al. 2018), and much of this effort relies on a capacity which has been augmented by subsidies. Skerritt et al. (2023) estimated that between 20% and 37% of the total monetary value of all capacity-enhancing subsidies supported fishing outside the jurisdictions of the countries providing the subsidies. Specifically, between 17% and 30% of the subsidy value was to support fishing in foreign nations, and between 3% and 7% was to support fishing in the high seas.

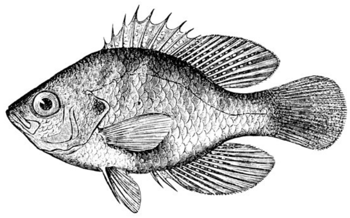
Capacity-enhancing subsidies used by "developed" nations to enhance their own fishing fleets in foreign waters have large, albeit indirect, impacts on "developing" nations with low fishery-management capacity and productive but vulnerable fish stocks. Skerritt et al. (2023) documented that Asian, European and North American nations are "net subsidy sources" (i.e. providers of "harmful" subsidies[6] to promote their distant-water fishing fleets), whereas nations in Africa, Oceania, Central-South America and the Caribbean are "net subsidy sinks" (i.e. foreign fishing in their EEZ waters is supported by harmful subsidies).

For example, fishing in Oceanian waters was supported by more than three times the aggregate dollar value of subsidies than that of subsidies provided by Oceanian nations themselves, with the vast majority of the foreign subsidies originating in Asia. Similarly, fishing in African waters was supported by substantial harmful subsidies originating in Asia and Europe. Skerritt et al. (2023) contended that such use of subsidies is unjust, and that subsidies for trade agreements conferring fishing access to distant-water fishing fleets should be prohibited. As noted previously, capacity-enhancing subsidies are not only environmentally harmful, but they also deprive coastal communities of critical resources (Kaczynski and Fluharty 2002; Virdin et al. 2019, 2022; and see Chapter 7: Foreign fleet effects on local communities).

6 Skerritt et al. (2023) define a harmful subsidy as any one that "artificially increases revenue or reduces the costs of fishing and include support for vessel construction, tax exemptions, fuel subsidies, and investment in marketing and processing infrastructure."

Between 1994 and 2006 the EU spent approximately EUR 165 million annually to encourage developing countries to sign agreements and trade their fishing rights to EU fleets (Cordón Lagares and García Ordaz 2014). This means that EU fishery funds which could have been used to support fishing in EU waters (subject to EU regulation and oversight) were instead spent to support what was very likely unsustainable fishing elsewhere (Le Manach et al. 2013; Schuhbauer et al. 2020).

In some cases, eliminating capacity-enhancing subsidies could bring both environmental and economic advantages. A model-based simulation meant to assess the impact of capacity-enhancing subsidies on trawl fisheries in the North Sea over the past 20 years suggested that although removing those subsidies might reduce the total catch and revenue, it would increase the overall profitability of the fisheries and the total biomass of commercially important species (Heymans et al. 2011).



Rethinking fishery subsidies

It has long been contended that subsidies to enhance the capacity and effort of trawling fleets should be eliminated, and replaced by beneficial subsidies, e.g. convert fleets and ensure that fishers are provided with viable alternatives (Cullis-Suzuki and Pauly 2010; Heymans et al. 2011; Sumaila et al. 2016, 2021; Schuhbauer et al. 2020; Villasante et al. 2022; Skerritt et al. 2023).

The harm caused by fishery subsidies is increasingly recognized by national agencies, inter-governmental organizations and regional organizations, worldwide (Sumaila et al. 2016). Fishery subsidies started being questioned in the early 1990s, when FAO noted that their provision could result in excess fishing capacity (FAO 1993). By the 2000s, several other bodies started questioning the assumed benefits of these subsidies and looking more closely at their effects on fishery management (Munro and Sumaila 2002; Sakai et al. 2019). For instance, the United Nations repeatedly pointed to the problem of subsidies that have led to IUU fishing and overcapacity (Sumaila et al. 2016), and UNEP's Millennium Ecosystem Assessment highlighted the need to eliminate "perverse" subsidies that promote "excessive use of ecosystem services" and "affect the management of resources and their sustainable use by encouraging overexploitation of the resource, thereby worsening the common property problem present in fisheries" (Millennium Ecosystem Assessment 2005). In 2015, the United Nations' Sustainable Development Goals stipulated that certain forms of capacity-enhancing fishery subsidies must be prohibited. The harmful impacts of subsidies on developing countries, notably in relation to fishing access agreements and food sufficiency (Kaczynski and Fluharty 2002), have also become widely acknowledged.

In the EU, some capacity-enhancing subsidies were removed, but many remain (Skerrit et al. 2020). Attempts to rebuild fish stocks and end overfishing are often undermined by contradictory programmes, such as favouring the reduction of vessel numbers but simultaneously subsidizing the construction of new vessels or the modernization of other vessels (Skerritt et al. 2020). In Italy, recent subsidies were used at least in part to promote the reduction rather than the development and expansion of trawl fleets. Specifically, 83% of the funds allocated for the "permanent cessation of fishing activities" were directed at otter trawls and beam trawls, as compared to 1% directed at static fishing gears (Gambino et al. 2022). Between 2014 and 2019, the number of bottom trawlers decreased by 11%, although that trend could not be conclusively linked to the allocation of subsidies (Gambino et al. 2022).

World Trade Organization (WTO) Members have long been negotiating to limit harmful fishery subsidies, including those "contributing to IUU fishing, to overfishing, to overcapacity, and affecting overfished stocks" (wto.org/english/thewto_e/minist_e/mc11_e/briefing_notes_e/bffish_e.htm), but so far those negotiations have borne little fruit (Skerritt et al. 2020, 2023; Villasante et al. 2022). In 2022 the WTO adopted an Agreement on Fisheries Subsidies that would prohibit those considered harmful. However, such Agreement would only enter into force if two-thirds of the 164 WTO members completed their domestic ratification processes. As of October 2024, approximately 60 WTO Members had accepted the Agreement (https://www.wto.org/english/tratop_e/rulesneg_e/fish_e/fish_acceptances_e.htm).

Monitoring trawling

9

The importance of monitoring

Monitoring the spatial and temporal dimensions of trawl fisheries, i.e. where and when a given trawler operates, is critical for management. Many vessels do not broadcast their location or are not detected by public monitoring systems, resulting in an incomplete understanding of fishing footprints. Until recent times, the global footprint and impacts of trawl fisheries could only be inferred from self-reporting (e.g. logbooks), reports by onboard observers, disaggregated catch data and various other monitoring means that tended to produce heterogeneous data which were neither publicly available nor global in coverage (Kroodsma et al. 2018).

Remote tracking systems now make it possible to investigate the behaviour of fishing fleets down to individual vessels (McCauley et al. 2016) and the use of either remote tracking, radar and optical imagery, or combinations of any of those technologies, makes it possible to map and quantify the global extent of industrial fisheries (Kroodsma et al. 2018; Longépé et al. 2018; Li et al. 2021; Kerry et al. 2022; Paolo et al. 2024), including trawl fleets (Amoroso et al. 2018). Such monitoring can also reveal the extent of trawling within areas where trawl gear is prohibited (Ferrà and Scarcella 2023), as well as the extent of unreported trawling (Coro et al. 2023).

A different type of electronic monitoring, based on technology onboard fishing vessels, has increasingly become viewed as a viable solution to the problem of documenting catch as well as bycatch and discards (Mangi et al. 2015), and it can contribute to assessments of environmental impacts and economic performance.

In this chapter, we provide a quick overview on some of the technological advances toward improved monitoring of trawl and other fisheries, with an emphasis on monitoring vessel distribution, movements and activity patterns.

Monitoring vessels

Fishing activities can be monitored through cooperative and non-cooperative systems. Cooperative systems rely on information transmitted from vessels, whereas non-cooperative systems employ optical remote sensing and radar systems. Technologies such as Automatic Identification System (AIS) and Vessel Monitoring System (VMS) use radio or Global Positioning System (GPS) devices to transmit vessel positions, and can be used to track the location and activity of fishing vessels equipped with the necessary onboard instrumentation (Kroodsma et al. 2018; Coro et al. 2023). Conversely, Synthetic Aperture Radar (SAR) systems use radar imagery and do not require onboard instrumentation. The combined use of cooperative and non-cooperative systems is an effective way of monitoring and evaluating fishing effort (Park et al. 2020).

Monitoring via AIS

AIS, a form of vessel tracking, was introduced in the 1990s by the International Maritime Organisation (IMO) to broadcast a ship’s position, so that other ships are aware of its location, with the aim of preventing collisions and improving maritime safety. AIS regulations vary greatly depending on the country (McCauley et al. 2016). For instance, in the United States and Canada, fishing vessels are required to carry AIS if they are longer than 19.8 m. In the EU, since 2014 all fishing vessels above 15 m are required to carry AIS (EC 2009, 2011; Natale et al. 2015).

AIS takes advantage of both land-based and satellite systems, which can be complemented to improve spatial coverage (Fernandez Arguedas et al. 2018). Early AIS was designed as a very high frequency (VHF) radio-based tool to communicate among vessels in line of sight, as well as with land-based receivers. Since 2018, however, AIS receivers have been placed on low-orbit satellites, and this has greatly increased their coverage, while ensuring that AIS signals can be detected from vessels operating offshore. AIS information may include static, dynamic and voyage-related data: the first group includes non-alterable vessel identity information (plate, size, length etc.), the second includes information relying on onboard sensors (vessel geographic position, heading, fishing activity etc.) and the third includes data that can be entered manually (vessel destination, route plan, etc.; IMO 2015).

All this information has become extremely valuable in fishery research and management (e.g. Woodil et al. 2021; Ferrà and Scarcella 2023). Data are unencrypted and generally available to the public (with exceptions; Paolo et al. 2024). AIS encompasses both nearshore and high-seas vessel monitoring, and the transmission is more or less continuous. AIS data can help determine whether a given vessel is a fishing vessel, and whether that fishing vessel is a trawler. On a finer analytical scale, AIS data can be processed via analytical routines and algorithms to determine the most likely type of trawl gear (e.g. whether a trawler has beam or otter trawl gear—which may be used interchangeably), and whether the trawler is actually fishing (as opposed to being in transit or engaged in other non-fishing activities)—sometimes with high levels of accuracy (e.g. Bonizzoni et al. 2023).

The following drawbacks may apply to AIS: it tends to underestimate fishing activities conducted offshore, it covers only a fraction of global fishing vessels, its coverage varies among regions, and in areas of high vessel density AIS signals can interfere with each other (Russo et al. 2016; Kroodsma et al. 2018; Shepperson et al. 2018; Taconet et al. 2019).

Monitoring via VMS

Similar to AIS, VMS (Bland 1980) determines the vessel's location at set intervals, and transmits this information to a monitoring station on land. VMS can track fishing activity at large spatial scales, but historically access has been restricted to government regulators or other fishery authorities (though several countries make their VMS data public). VMS is used by management authorities to check whether a vessel is fishing at a time when, and in an area where, it is allowed to fish, and it can alert enforcement aircraft or vessels to known or suspected infringements (Lee et al. 2010).

In the EU, VMS has been mandatory for all vessels longer than 12 m since 2009 (EC 2009). Starting from January 2028, the use of a monitoring system allowing automatic localization and identification of the vessel at sea will also be required on smaller vessels (EC 2023b; https://oceans-and-fisheries.ec.europa.eu/fisheries/rules/enforcing-rules/inspections-monitoring-and-surveillance_en).

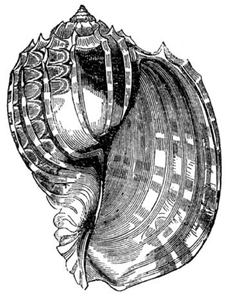
VMS has been used effectively in studies of trawl fisheries, e.g. to estimate trawling distribution and intensity, assess environmental and seabed impacts, or estimate fishery yields (Deng et al. 2005; Mills et al. 2007; Witt and Godley 2007; Russo et al. 2011; Hintzen et al. 2012; Gerritsen et al. 2013; Liu et al. 2023). While in some cases VMS offers better coverage than AIS (Russo et al. 2016; Shepperson et al. 2018), its main drawbacks are that the transmitted data do not indicate a vessel's activity (e.g. whether fishing or underway), and position records are transmitted too infrequently to infer fishing activity or follow a vessel's fine-scale movements (Skaar et al. 2011; Lambert et al. 2012; Russo et al. 2016; Zhao et al. 2024). AIS and VMS datasets can be integrated to provide more accurate information (Thoya et al. 2021). For instance, when vessels broadcast both AIS and VMS, information from the two systems can be combined to obtain higher-resolution vessel tracks and investigate the distribution and pattern of fishing effort (Russo et al. 2016).

Limitations of vessel monitoring via AIS and VMS

Both of the technologies described above have important data gaps. AIS/VMS devices must be installed on board and actually broadcast in order for signals to be detected: vessels that are not equipped with these devices or have non-functional systems cannot be included in analyses. In addition, an AIS system can be intentionally switched off whenever a vessel operator does not want to be "seen", and AIS or VMS positions sent by vessels can be deliberately manipulated (McCauley et al. 2016).

When AIS/VMS signals are regularly transmitted, the accuracy of information depends on the signals having adequate quality and being correctly received. For example, signal quality can be reduced or signals may get lost because of adverse meteorological conditions, limited range coverage of terrestrial receivers (for radio systems), satellite communication hindrances, saturated transmission bands and other technical issues (Shepperson et al. 2018; Taconet et al. 2019; Emmens et al. 2021). Bottom trawlers in many areas do not have AIS/VMS onboard, or operate in areas lacking AIS coverage: these trawlers cannot be monitored and their movements mapped (e.g. Taconet et al. 2019).

McCauley et al. (2016) made the following four recommendations to improve the monitoring of fishing vessels: 1) the IMO should enforce stricter rules and require that all fishing vessels ≥15 m, independent of country of origin, are equipped with AIS; 2) all fishing vessels should have a permanent IMO identification number that must be included in all AIS data; 3) having onboard AIS, but failing to use it properly (e.g. switching off the system or altering transmitted data), should no longer be considered as legal compliance; and 4) AIS data should be publicly available. More recently, other authors (e.g. James et al. 2018; Tassetti et al. 2022) have advocated that the use of AIS and other remote tracking systems be extended to smaller fishing vessels.



Monitoring via radar and optical imagery

Another way to track fishing activities, which does not require instrumentation onboard the tracked vessels, is by using radar (e.g. Cope et al. 2022) or optical imagery. Such imagery can be combined with deep-learning models, e.g. to produce maps of fishing effort or monitor illegal fishing (Rowlands et al. 2019). Optical data are images collected by sensors which do not emit their own radiation but measure solar radiation reflected into the atmosphere by objects on the Earth's surface. Optical imagery can be captured by sensors mounted on either satellites or aircraft and drones. Such imagery needs light, and is therefore limited at night. It is also limited by cloud coverage.

In recent years, SAR has emerged as one of the most promising technologies to monitor fishing vessels. SAR imaging is produced by a radar system mounted on satellites or aircraft, that can create two-dimensional images using radio waves. As it relies on radar signals, SAR imagery is independent from cloud cover and sunlight (Asiyabi et al. 2023). Its main disadvantage is the low frequency of transmitted data, e.g. one image every one or more days. Other disadvantages include speckle noise (granular noise texture) degrading image quality, limited accuracy in the estimation of vessel length and width, technical challenges in data analysis and interpretation, and difficult detection of small vessels and wooden vessels (Stastny et al. 2014; Devi and Sharma 2016; Stasolla et al. 2016; Park et al. 2020; Asiyabi et al. 2023). However, novel image processing techniques and algorithms have been implemented to overcome these drawbacks, with encouraging results (Devi and Sharma 2016; Stasolla et al. 2016; Bua et al. 2024). Optical and SAR imagery can be paired to optimize image quality.

Combining SAR information with machine learning is an efficient way of accelerating and improving the processing of large datasets (Yasir et al. 2023; Paolo et al. 2024). Information can also be enhanced by combining SAR with AIS or VMS data (Dechesne et al. 2019; Galdelli et al. 2021).

SAR can help assess the proportion of fishing vessels that do not transmit AIS or VMS information, thus revealing the degree of compliance (Dechesne et al. 2019; Park et al. 2020; Paolo et al. 2024). Recent SAR analyses of global datasets suggested that about 75% of all industrial fishing vessels are not publicly tracked via AIS, with the proportion of untracked vessels being particularly high in South and Southeast Asia and North Africa (Paolo et al. 2024).

A SAR-based study of illegal fishing detected between 700 and 900 Chinese vessels fishing illegally in North Korean waters, for a total estimated catch of over 164,000 tonnes of Japanese flying squid *Todarodes pacificus* (a catch similar to that of Japanese and South Korean fleets combined), which was worth more than USD 440 million (Park et al. 2020). The same study also detected about 3,000 North Korean vessels fishing, most of them illegally, in Russian waters (Park et al. 2020).

Monitoring catch and bycatch

Information on the activities and catches of fishing vessels has traditionally come from self-reporting by the fishers, e.g. through logbooks compiled on board. This is an inexpensive means of collecting data, and can provide vast amounts of information. However, the accuracy of self-reported data is typically difficult to assess, and it has important limitations and biases (Cotter and Pilling 2007; Lordan et al. 2011; Sampson 2011; Mangi et al. 2015; Suuronen and Gilman 2020; Basran and Sigurðsson 2021; Clegg et al. 2022).

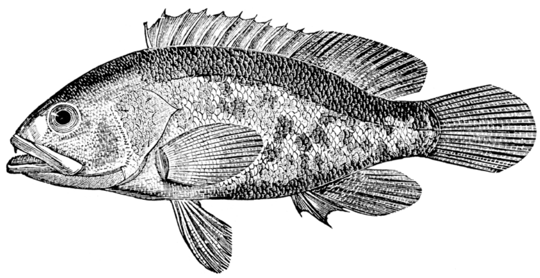
Attempts to mitigate bias and produce higher-quality information have included observer programmes, where trained personnel stationed on fishing vessels collect data on catches, discards and bycatch (Liggins et al. 1997; Jaiteh et al. 2014; Snyder and Erbaugh 2020; Teye et al. 2020). The presence of observers onboard is not always practicable (e.g. because of limited space or safety concerns), and observer schemes can be expensive and not entirely devoid of bias (Benoît and Allard 2009; Mangi et al. 2015; Cahalan et al. 2016; Suuronen and Gilman 2020).

In recent decades, electronic monitoring (EM) has emerged as a valuable tool to supplement existing monitoring programmes (Mangi et al. 2015; Bartholomew et al. 2018; Gilman et al. 2019). EM systems typically consist of cameras, sensors, and data storage devices installed on fishing vessels to monitor onboard activities and fully document catches (Ames et al. 2007; Kindt-Larsen et al. 2011; van Helmond et al. 2015, 2017, 2020; Plet-Hansen et al. 2019). Alternatively, EM can be used to audit or validate catch data self-reported by fishers (Stanley et al. 2009, 2011). High-resolution imagery enables identification and quantification of different species caught and discarded during fishing operations (Wallace et al. 2015a; Vilas et al. 2020), or to monitor non-fishing variables such as seabird interactions and mortality (McElderry et al. 2004). Artificial intelligence and machine learning algorithms are increasingly integrated into EM systems to automate data processing, species identification, and event detection.

The advantages of EM systems over traditional monitoring methods include cost-effectiveness, increased coverage and intensity of monitoring, enhanced data quality and accuracy, and reduced burden and risks to human observers, by minimizing their exposure to potentially hazardous fishing conditions. Challenges include regulatory frameworks, data privacy and security concerns, and stakeholder engagement and acceptance (van Helmond et al. 2015).

Specifically, fishers may perceive EM as an intrusion upon their private workspace (Plet-Hansen et al. 2017) and camera surveillance may be interpreted as a form of mistrust (Mangi et al. 2015). EM requires expertise for data analysis and validation, and it can be computationally intensive (van Helmond et al. 2015; Plet-Hansen et al. 2017; Snyder and Erbaugh 2020). Finally, current EM does not allow the collection of certain types of data (e.g. on fish biology), and automatic species identification may be incorrect (Mangi et al. 2015; Needle et al. 2015; Suuronen and Gilman 2020; van Helmond et al. 2020).

Still, EM can enhance transparency and accountability in fishing practices by accurately documenting catch composition and discards. EM data can be integrated with other monitoring tools, such as electronic logbooks and onboard observers, to enhance data verification and validation (van Helmond et al. 2015).



The management of trawling

10

Main management approaches

Fishery managers often need to reconcile conflicting economic, social and environmental priorities. One frequently stated management objective is to achieve "sustainable exploitation" of the targeted resources, based on striking a balance in terms of e.g. economic interests, catch quotas, fishing gear allowed, time and space constraints, and an "acceptable" degree of ecosystem damage.

Measures that aim to protect ecosystems and vulnerable species while ensuring the long-term availability of fishery resources typically constrain exploitation in some way. Therefore, a major challenge of fishery management is to balance economic needs or aspirations with the inherent limits of marine populations. Generally speaking, management approaches have a greater chance of success if they enhance or adjust, rather than completely overhaul, an existing management system or arrangement (McConnaughey et al. 2020). However, efforts that rely on collaboration with industrial interests, or align closely with existing socio-political views, often fail to change the status quo, and that is also true when it comes to fishery management (Steadman et al. 2021). Effective conservation action typically requires imaginative strategies that challenge business-as-usual resolutely.

Management efforts to eliminate or mitigate the impacts of bottom trawling can be subdivided into four broad categories: 1) measures to mitigate harmful impacts on the seabed, 2) measures to reduce the bycatch of certain species or age/size categories, 3) setting limits on the practice or banning it altogether, and 4) transition away from trawling, e.g. use of other fishing gear, or other forms of employment. These categories are largely intertwined. Efforts to promote the consumption of food that has less negative impact on the environment than the seafood caught by trawl nets may be seen as an additional category of management, but one that is not entirely within the realm of fisheries.

In this chapter, we describe and discuss some of the most common approaches, while recognizing that all these management scenarios are bound to be especially challenging in areas or contexts where the trawling industry is powerful and established, which is normally the case.

Mitigation of seabed impacts

Strategies to reduce harmful environmental impacts on the seabed have included modifications of fishing gear and the use of alternative gear components and devices, as well as "impact quotas" and "move on" protocols (He 2007; McConnaughey et al. 2020). Fishers generally have four main motivations to update gear and technologies (Eigaard et al. 2014; Kennelly and Broadhurst 2021): 1) increasing their revenues by catching more fish, 2) increasing their revenues by raising the value of the catch, 3) reducing the costs of fishing, and 4) enhancing onboard comfort and safety.

In some cases, the goal of increasing revenues and reducing the costs of trawling may be consistent with that of reducing harmful environmental impacts, because these impacts are partly related to the dragging of towed gear, and the more drag, friction and contact with the seabed, the higher the fuel consumption and net abrasion. Some fishers also have an interest in reducing the unintended environmental effects of fishing and improving the sustainability of their activities (e.g. to be granted some type of environmental certification or eco-label), and this may be seen as a fifth motivation (Sala et al. 2023).

The design and operation of trawl gear can be modified to improve functionality or maintain acceptable levels of performance while also attempting to reduce physical contact and penetration depth of gear within the seabed (ICES 2019). In their review of best practices for managing the environmental impacts of trawl fishing, McConnaughey et al. (2020) described numerous technical modifications of traditional trawl gear, operational changes, and designs that can reduce seabed impacts, bycatch, gear wear, and fuel consumption, while increasing catch efficiency. Some of these modifications, however, may have offsetting effects that go undetected or are difficult to quantify.

Two of the most relevant technological innovations developed in recent decades to reduce seabed contact and penetration depth, while maintaining or increasing the catchability of target species, are the use of “semipelagic otter boards” (otter boards that are not meant to touch the bottom) and “electric pulse trawling” (the use of electric shocks that make fish convulse and flip upwards into a beam trawl that does not scrape the seabed).

Semipelagic otter boards

There is a strong relationship between the penetration depth of towed fishing gear and the depletion of benthic biota (Hiddink et al. 2017). In the case of bottom otter trawls, ground contact of otter boards towed across the seabed increases the damage to benthic ecosystems, as well as drag, gear wear and fuel consumption. The furrows produced by otter boards are often the most evident physical effect of trawling. The energy transmitted to the bottom by standard otter boards is higher than that by any other part of the gear, and otter boards can penetrate deep into the seabed, digging a trench or furrow up to 35 cm deep and transferring large amounts of sediment on either side of their path (Lucchetti and Sala 2012; Eigaard et al. 2016; Bradshaw et al. 2021).

An alternative to standard otter boards consists of boards that “fly” (or “jump”) over the seabed. Semipelagic otter boards (sometimes referred to as “near-bottom otter boards”, or “flying doors”) are reported to have better hydrodynamic performance compared to standard otter boards, resulting in less gear penetration, less sediment resuspension, lower trawling speed, reduced seabed-drag, less power required for trawling and less fuel consumption, while catches do not appear to be significantly affected (Mengual et al. 2016; Sala et al. 2023). However, it should be noted that while otter boards cause the most severe impacts, they represent a small proportion of the trawl gear in contact with the seabed, and have the narrowest track (Eigaard et al. 2016). Also, even if semipelagic otter boards do not touch the seabed, the groundgear still does.

Considering that fuel consumption is the main operating cost for trawl fishing, the adoption of semipelagic otter boards may prove to be cost-effective and more resilient to fuel price fluctuations. While semipelagic otter boards may cost about twice as much as standard otter boards, the fuel savings reportedly can give a payback time in the order of 100 days (Sala et al. 2023).

Downsides for fishers include the more erratic behaviour of semipelagic otter boards as compared to standard otter boards, and the advanced skills and onboard technology required to operate semipelagic boards correctly (Sala et al. 2023). The main operational challenge is keeping a nearly constant distance between the seabed and semipelagic otter boards; this requires acoustic instrumentation to monitor their depth and optimise their distance from the seabed, e.g. by altering towing speed and warp length. Without proper control of their position in the water column, semipelagic boards will plough into or “jump” on the seabed, particularly when the seabed contour is irregular or in stormy seas. Finally, semipelagic otter boards may not be optimal in all fishing situations, and are used mainly to target organisms, such as shrimps and prawns, that are not herded along the bottom by the otter boards and the sweeps/bridles (Sala et al. 2023).

Novel designs aim to improve the vertical stability of semipelagic otter boards, to ease the work of captains while reducing drag and improving the efficiency of trawling. For instance, “semipelagic self-adjusting otter boards” reportedly maximize hydrodynamic performance and minimize contact with the seabed, and have altimeters and adjustable flaps that are controlled by an active feedback system (Eighani et al. 2023). Fishery scientists are also working on modifications to other gear components, such as elevated sweeps, floating bridles and lighter groundgear. While some of these modifications may reduce the negative impacts on the seabed, damage to epifaunal organisms such as corals, sea pens and sponges would not be eliminated.

Electric pulse trawling

Beam trawls normally have a steel beam to keep the net open, and tickler chains in front of their nets. “Pulse trawls” replace tickler chains with electrodes that emit electric pulses (Soetaert et al. 2015) to flush target species (usually flatfish or shrimp) out of the sediment. The electric shock makes fish convulse and flip upwards, into the net. Commercial application of electric stimulation flourished in Chinese shrimp beam trawl fisheries in the 1990s, but since 2001 the practice has begun to be banned in China, due to the misuse of electric pulse parameters that caused damage to juvenile shrimp and other benthic organisms (Yu et al. 2007).

Commercial electric fishing has also been banned in EU waters since 1998 (EC 1998), but a study fleet received a temporary exemption. A large part of the Dutch beam trawl fleet switched to pulse fishing between 2009 and 2015, and by 2016 about 95% of the Dutch sole quota was caught with pulse trawls (Haasnoot et al. 2016; Hintzen et al. 2021). The transition from tickler chain beam trawls to pulse trawls was reported to increase catch efficiency at lower towing speeds, reduce mechanical disturbance of benthic ecosystems, reduce the mortality of non-target invertebrates, and reduce discard rates (van Marlen et al. 2014; Depestele et al. 2019; Bergman and Meesters 2020; Rijnsdorp et al. 2020). Pulse trawls could be towed at speeds of around four to five knots, as compared to six to seven knots for traditional beam trawls (Depestele et al. 2016, 2019), and they reportedly had considerably lower fuel consumption (-46%) and greenhouse gas emissions (Turenhout et al. 2016). Pulse trawling improved the catch efficiency for common sole, although the catch efficiency for European plaice *Pleuronectes platessa* and some other species was reduced (Poos et al. 2020).

Debates about the legitimacy of the EU ban are ongoing (de Haan et al. 2016; Penca 2022; Delaney et al. 2023), based in part on the limited understanding of how electricity affects marine organisms and benthic ecosystems (Soetaert et al. 2015). However, there has been significant recent research in this field (e.g. see Schram et al. 2022; Boute et al. 2024). Also, the greater efficiency of pulse trawls and their expanded use in new trawling grounds can result in conflict with fisheries that experience reduced catches on shared fishing grounds (Sys et al. 2016; McConnaughey et al. 2020; Schram et al. 2022).

Impact quotas

Apart from the modifications of fishing gear described above, there have been attempts to mitigate the damage to vulnerable invertebrate species such as structure-forming corals and sponges through “invertebrate bycatch quotas” (McConnaughey et al. 2020). In areas where the occurrence of vulnerable species has been precisely mapped, the trawl activity can be regulated with real-time reporting and closures. For instance, vessel limits to reduce benthic impacts were implemented in British Columbia, Canada, where quotas could be traded among vessels and combined with more traditional spatial closures in areas of high coral and sponge concentration (Wallace et al. 2015b). A “move-on” protocol required vessels to notify the fleet if catches of corals and sponges exceeded a set threshold (20 kg) in a single tow. In 2012–2013, following the implementation of this management measure, sponge and coral bycatch reportedly was the lowest recorded in 17 years and fell below the prescribed fleet-wide maximum target of 884 kg (Wallace et al. 2015b).

Groenbaek et al. (2023) suggested that move-on protocols combined with detailed maps of sensitive areas, tradable quotas, and onboard observers, would help reduce invertebrate bycatch without affecting overall fleet performance. McConnaughey et al. (2020) noted that although this mitigation measure was limited to the British Columbia trawl fishery, a similar approach could be implemented in other areas and fisheries. Such protocols, however, displace effort to similar areas, therefore expanding the overall footprint of trawling and its effects (Hilborn et al. 2023).

Another theoretical management approach would be to reduce damage to vulnerable benthic fauna by setting “habitat impact quotas” (McConnaughey et al. 2020). Habitat impact quotas would allow trawlers to fish for long periods in less sensitive habitats, or for short periods in more sensitive habitats. Such an approach would combine the remote tracking of vessel locations with detailed mapping of sensitive habitats, and attempt to monitor trawl impacts in relation to the impact quotas (Holland and Schnier 2006a, 2006b). A drawback is that monitoring based on remote tracking of fishing activity (rather than onboard observations) is inherently less precise. In addition, habitat sensitivity maps are expensive to create and update, and they require stakeholder validation (McConnaughey et al. 2020). Although habitat impact quotas have not been implemented in real fisheries, model results (Holland and Schnier 2006b) indicated that this approach may be more cost-effective for limiting the benthic impacts of trawling than both fixed and rotating fishing closures (though effectiveness would depend on characteristics of the fishery). Impact quotas such as those described above, however, would still allow sensitive seabed habitats to be disturbed and damaged.

Mitigation of bycatch

As mentioned earlier, reducing the mortality of non-target organisms in trawl nets has become a major challenge for fishery managers over the past half-century (Hall et al. 2000). To reduce bycatch and improve species and size selectivity, numerous gear modifications and practices have been developed and implemented, in combination with at-sea trials to assess their advantages and disadvantages (Kennelly and Broadhurst 2021).

A significant obstacle to bycatch mitigation has been the limited uptake by fishers of gear modifications and practices that they consider inconvenient and costly (Suuronen 2022). Reduced catch rates may be offset by reduced operating costs and become acceptable once the fisher's investment in lower-impact, energy-efficient gear has been recovered (McConnaughey et al. 2020). However, fishers need to be assured that there is no significant impact on catches of target species, no significant increase in gear drag and wear, and no impairment of onboard operations, before bycatch reduction devices are widely accepted and adopted (van Marlen et al. 2007). Below, we describe some of the devices that have been used to reduce bycatch in trawl gear.

Mechanical devices

A variety of grids, deflectors, panels of netting, separator panels, guiding funnels, net openings and specific-mesh netting have been deployed to reduce bycatch in trawl nets. These devices aim to facilitate the exclusion (or expulsion) of unwanted organisms, ideally alive and unharmed, before they enter the codend (Broadhurst 2000; Eayrs 2007). Organisms to be excluded may include non-target fish as well as sea turtles and marine mammals (Eayrs 2007; Graham 2010; Hamilton and Baker 2019). Studies have shown that some species can be expelled alive from a trawl net (Hamilton and Baker 2015; Lyle et al. 2016; Wakefield et al. 2017) without significantly reducing the catch of target organisms (Brewer et al. 1998; Broadhurst 2000; Dotson et al. 2010; Wakefield et al. 2017; Lucchetti et al. 2019).

Fish

Fish bycatch may need to be reduced to comply with restrictions on fish size (e.g. to protect juvenile and immature individuals), fish species (e.g. because of fishing quotas), or quantity of overall bycatch (e.g. to keep bycatch levels below a given proportion of total catch). Certain modifications of trawl nets aim to reduce the catch of unwanted fish that cannot be landed or marketed. There are two main types of such gear modifications: those that separate organisms based on their size, and those intended to separate species based on their response behaviour to the trawl net (Broadhurst 2000). In some cases, both types of modifications are applied simultaneously.

Graham (2010) reviewed the variety of gear modifications intended to reduce catches of unwanted fish in trawl nets. For example, grids with longitudinal bars were used to exclude finfish from shrimp trawls, while retaining the shrimps that passed through the bars and reached the codend. Grids were also used in finfish trawl fisheries to exclude fish of certain sizes and species. For example, they were used in Norway pout *Trisopterus esmarkii* fisheries to reduce the bycatch of larger gadoids (cod, haddock, hake, whiting and saithe; Kvalsvik et al. 2006), and in sole fisheries to reduce the bycatch of rockfish, sablefish, and halibut (Lomeli and Wakefield 2016).

In some cases, a larger codend mesh size was preferred over the use of sorting grids, e.g. to target fish above a certain size, while favouring the escape of juveniles and smaller fish (Brinkhof et al. 2022). Other options included the use of "cut-away" or "topless" trawl nets, with a mouth that had a reduced upper panel compared to the lower one, so that certain species could swim upwards and avoid entering the net; these nets could favour the escape of non-target fish and reportedly led to bycatch reductions of up to about 50%, with insignificant loss of target species (Revill et al. 2006; Eayrs et al. 2017).

Some studies have focused on the physical mechanisms underlying retention in trawl nets, and fish penetration into the mesh, especially at the codend level (O'Neill and Kynoch 1996; Herrmann et al. 2009). Other studies have focused on fish behaviour in front of, or inside, a trawl net, to assess whether such behaviour can be used, or modified, to retain the wanted species and sizes and exclude the unwanted ones during trawling (Krag et al. 2009, 2010, 2017; Kim and Whang 2010; Herrmann et al. 2015; Melli et al. 2018a, 2019; Karlsen et al. 2019).

Few gear modifications are specific for elasmobranchs, though in some cases the sea turtle exclusion devices described in the following section can have the added benefit of reducing the bycatch of at least some elasmobranchs (Brewer et al. 2006; Raborn et al. 2012; Garstin and Oxenford 2018). In parts of the Mediterranean Sea, specific grids were trialled to exclude "sharks" from bottom trawls targeting more valuable demersal species, but these trials were only partly successful (Brčić et al. 2015; De Santis et al. 2024).

Kennelly and Broadhurst (2021) reviewed 203 papers describing 28 broad categories of technical modifications intended to improve selectivity (for either species or size) and reduce mortality of organisms that are unintentionally caught in bottom otter trawls. While several of these modifications were reported to be successful (e.g. physical barriers and exclusion devices), the authors noted that "no fishery had completely resolved all bycatch problems while maintaining targeted catches at conventional levels". The authors provided recommendations to assess gear modifications and improve selectivity (Kennelly and Broadhurst 2021).

Sea turtles

One of the most widely used mitigation measure to reduce sea turtle mortality in trawl nets is based on grids that function as turtle exclusion (or excluder) devices (TEDs). TEDs normally consist of a rigid or semiflexible sorting grid made of bars, positioned anterior to the codend. When a sea turtle (or other large animal) enters the trawl net, the animal is redirected by the grid and expelled through an opening (covered by a netting panel to prevent the loss of target species) located near the grid, at the top or bottom of the net.

TEDs are generally successful in reducing sea turtle bycatch (Jenkins 2012; Wakefield et al. 2017; Lucchetti et al. 2019) and, while not all sea turtles can be expelled, the use of turtle exclusion devices can reduce bycatch (and perhaps mortality) by more than 90% (Epperly et al. 2002; Brewer et al. 2006; Jenkins 2012). TEDs do not appear to have significant downsides, as normally they do not decrease fish catches (Al-Baz and Chen 2015; Lucchetti et al. 2016, 2019; Vasapollo et al. 2019). However, a few studies have reported catch losses between approximately 6 and 16% (Robins and Gilvray 1999; Brewer et al. 2006; Price and Gearhart 2011) and fishers sometimes report unacceptable loss of time, difficulties in the use of TEDs, or simply perceive them as unnecessary (Rao 2011; Duarte et al. 2019). Still, TEDs must be viewed as important conservation tools, and their use has become mandatory in several trawl fisheries (Epperly 2003; Shiode and Tokai 2004). Involving fishers in TED design and trials is one way of encouraging and facilitating their adoption (Sweeney Tookes et al. 2023).

Cetaceans

Hard or flexible grids may prevent marine mammals from entering and becoming trapped in the trawl codend, by redirecting them towards either a top-opening or bottom-opening escape hatch (Northridge et al. 2011; Hamilton and Baker 2019). Pinnipeds have been observed to negotiate exit via exclusion devices (e.g. Lyle et al. 2016), but cetaceans do not appear to manoeuvre easily within the narrow confines of a trawl net and become distressed when unable to find an escape route (Zeeberg et al. 2006; Jaiteh et al. 2013, 2014; Wakefield et al. 2017).

Rope or mesh barriers positioned near the mouth of a trawl net, used in conjunction with escape holes, have been trialled to reduce the bycatch of odontocete cetaceans, but the results have been inconclusive (Hamilton and Baker 2019; Bonizzoni et al. 2022). Most barrier designs caused substantial reduction in target catch as well as increasing drag and were, therefore, considered unacceptable (van Marlen et al. 2007; Bord Iascaigh Mhara and University of St Andrews 2010; Northridge et al. 2011). Other tests focused on a tunnel barrier positioned in the mid part of the trawl net and, while the method did not seem to affect the catch, its effectiveness in releasing cetaceans was unclear (van Marlen et al. 2007). Different mesh barriers trialled in midwater pair trawls had questionable efficiency and/or operational disadvantages (van Marlen et al. 2007). Trials with grids and escape hatches in midwater pair trawls off the U.K. suggested that only a small number of common dolphins successfully exited via the escape hatch, whereas most dolphins appeared to detect the grid and attempted, unsuccessfully, to escape upwards in areas with no escape holes (Bord Iascaigh Mhara and University of St Andrews 2010; Northridge et al. 2011).

Dolphin interactions with the Pilbara bottom otter trawl fishery off western Australia offer a good example of the complexity of assessing the effectiveness of physical exclusion devices. Bycatch reduction devices (BRDs), with a semi-flexible grid angled to an escape opening have been mandatory since 2006 to reduce mortality of megafauna including sea turtles, sharks and dolphins (Stephenson et al. 2008; Allen et al. 2014). However, establishing the effectiveness of BRDs in reducing mortality of common bottlenose dolphins *Tursiops truncatus* has proved challenging: while bycatch rates were initially reduced following the deployment of BRDs, these reductions were not maintained over time. Dolphin interactions with trawlers remained high, and some of the animals that exited (or were ejected) via bottom-opening BRDs were dead (Stephenson et al. 2008; Allen et al. 2014; Jaiteh et al. 2014).

A variety of physical barriers and exclusion devices to prevent cetacean bycatch in trawl fisheries have been trialled since the 1990s (Browne et al. 2005; Northridge et al. 2005; Stephenson and Wells 2006; Zeeberg et al. 2006; van Marlen et al. 2007; Lyle and Wilcox 2008; Stephenson et al. 2008; Bord Iascaigh Mhara and University of St Andrews 2010; Dotson et al. 2010; Allen et al. 2014; Jaiteh et al. 2014; Lyle et al. 2016; Wakefield et al. 2017). Despite these long-term efforts, there is no conclusive evidence that physical barriers or exclusion devices are effective solutions to reduce bycatch of cetaceans in trawl nets (Hamilton and Baker 2019; Bonizzoni et al. 2022).

Pinnipeds

Devices to reduce pinniped mortality in trawl nets—sometimes called sea lion exclusion (or excluder) devices (SLEDs) are similar to those used for sea turtles and cetaceans: hard or flexible grids to divert the animals, combined with top or bottom net openings to exclude them or allow them to escape from the net (Hamilton and Baker 2019). While there has been some debate on “cryptic” mortality caused by SLEDs (e.g. Meyer et al. 2017 vs Roberts et al. 2018), these devices can be relatively effective. For example, trial tests with midwater trawls off Tasmania indicated that the presence of a grid and a bottom escape opening was crucial for the survival of Australian fur seals. Large escape openings of 1x2 m decreased seal mortality by a factor of five, as compared to smaller openings of 1x1 m (Lyle et al. 2016). However, mortality still occurred, and the authors suggested that a top-opening escape, rather than a bottom-opening one, could have reduced mortality even further (Lyle et al. 2016).

Other trials were performed in bottom otter trawlers targeting squid off the Falkland Islands, to reduce the mortality of South American fur seals *Arctocephalus australis* and South American sea lions (Iriarte et al. 2020). Grids associated with either a netting funnel or a small-mesh panel before the codend reduced pinniped mortality by 90% and 100%, while allowing for large catches of squid. Conversely, trials with only a net barrier (25 cm^2 mesh) across the mouth of the trawl net, to prevent pinnipeds from entering into it, were ineffective due to pinniped entanglements (Iriarte et al. 2020). Similar tests were performed in a trawl fishery targeting krill off South Georgia, with before and after comparisons indicating successful reduction of pinniped bycatch (Hooper et al. 2005).

Acoustic devices

Acoustic deterrent or harassment devices have been widely used to prevent marine mammals from approaching fishing gear—primarily static nets (Dawson et al. 2013). The effectiveness of acoustic deterrence varies depending on cetacean species, fishing gear, context, and type or pattern of sound produced by the device (Hamilton and Baker 2019; Kindt-Larsen et al. 2019; Tixier et al. 2021; Dolman et al. 2022). Generally, acoustic deterrence tends to be more successful when it involves gillnet fisheries and neophobic cetacean species with large home ranges (Dawson et al. 2013), as these animals are less likely to habituate to noise. For instance, these devices eliminated the bycatch of beaked whales in a gillnet fishery (Carretta et al. 2008; Carretta and Barlow 2011), and helped reduce the bycatch of common dolphins in the same fishery (Carretta and Barlow 2011). Acoustic sound emitters (or “pingers”; Dawson at al. 2013) can substantially reduce the bycatch of harbour porpoises in gillnets (Kraus et al. 1997; Trippel et al. 1999; Gearin et al. 2000; Palka et al. 2008; Read et al. 2013; Larsen and Eigaard 2014).

While some odontocete cetaceans can be deterred by means of acoustic devices (Gazo et al. 2008; Clay et al. 2019; Ceciarini et al. 2023), the effect is often temporary (Cox et al. 2003; Dawson et al. 2013; Amano et al. 2017; Buscaino et al. 2021), or there may be no apparent deterrence (Berrow et al. 2008; Soto et al. 2013). Some studies indicate that habituation to noise may occur and porpoises, for example, have been found to approach pingers more closely over time (Cox et al. 2001; Carlström et al. 2009). Conversely, Palka et al. (2008) and Omeyer et al. (2020) found no evidence of habituation.

A study by Kindt-Larsen et al. (2019) suggested that harbour porpoises habituate more readily to a constant acoustic signal compared to a varying one, but they acknowledged that the time span of their experiments could have been too short for the porpoises to habituate to the varying signal, and habituation could appear after a longer exposure time. In other cases, the failure of deterrence is more evident. For instance, Tixier et al. (2015) found that acoustic harassment devices were ineffective at deterring killer whales from removing fish from longline gear. Killer whales habituated rapidly, and appeared to "put up with" what were assumed to be harmful noise levels. Those authors recommended the use of other, non-acoustic methods of deterrence.

The duration of a study appears to be an important variable affecting its results, with deterrence more likely to be observed in short-term studies, and habituation more likely in longer-term studies (e.g. Buscaino et al. 2009 vs Buscaino et al. 2021). After an initial phase of fright or caution, dolphins and other cetaceans may even interpret the noise of acoustic deterrents as signals of the presence of a net where entangled fish can be found. Acoustic devices then become "bells announcing dinner" (or "dinner bells"; Cox et al. 2003; Carretta and Barlow 2011), which exacerbates the problem.

A recent review of odontocete cetaceans foraging behind trawlers, worldwide (Bonizzoni et al. 2022), indicated that the effectiveness of acoustic devices in reducing bycatch in trawl gear (or in mitigating foraging and scavenging in the proximity of trawlers) is, at best, controversial. For instance, in trials to reduce the bycatch of common bottlenose dolphins in midwater trawls, neither loud nor quieter pingers were effective in reducing "interactions" (Stephenson and Wells 2006; De Carlo et al. 2012; Sala et al. 2014; Santana-Garcon et al. 2018). Based on present knowledge, acoustic devices are ineffective in repelling common bottlenose dolphins during trawling operations (Hamilton and Baker 2019; Bonizzoni et al. 2022).

In trials to reduce the bycatch of common dolphins in midwater trawls, the use of loud acoustic devices had mixed effects (Table 4 in Bonizzoni et al. 2022). Specifically, either non-significant effects or effects of unreported significance were found using certain acoustic devices (models DDD 02F and Cetasaver #7; Morizur et al. 2008; Northridge and Kingston 2009, 2010; Northridge et al. 2011). Controlled experiments in the absence of the loud operational conditions of trawls indicated that acoustic devices (including model DDD 02F) may not provide a consistently effective deterrent for common dolphins (Berrow et al. 2008). However, significant effects were reported for acoustic device models DDD 03F (Northridge et al. 2011; Rimaud et al. 2019). A recent study by Puente et al. (2023) compared bycatch frequency and number of individual common dolphins bycaught by one pair of bottom trawlers operating in the Bay of Biscay, and found both metrics to be significantly lower in the net equipped with acoustic device models DDD 03F than in the net without devices. This study, however, reports in the Abstract that "one of the vessels in the pair operated with a set of DDD pingers whereas the other operated without them" and in the Methods that "pingers were used as the treatment on one boat, while the other boat served as the control without pingers", which is confusing considering that the bottom pair trawlers towed a single net (Fig. 1 in Puente et al. 2023). Puente et al. (2023) also noted that common dolphin bycatch in this trawl gear was related to factors such as fishing zone and depth. Based on the available evidence, it is difficult to draw firm conclusions regarding the effectiveness of acoustic deterrence in reducing common dolphin mortality in trawl nets—which is indeed a major conservation concern given the scale of bycatch in the Bay of Biscay (Peltier et al. 2021, 2024; Rouby et al. 2022; ICES 2023).

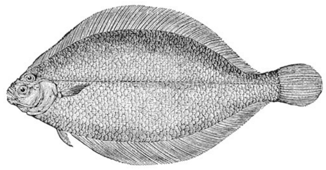
Acoustic deterrents deployed on trawl gear may be ineffective or less effective due to the loud noises associated with trawling operations (such as those produced by engine, cables and groundgear), that mask the sounds produced by the acoustic deterrents themselves (Morizur et al. 2008; Allen et al. 2014; Goetz et al. 2014). In any case, the additional noise produced by acoustic devices is unlikely to enhance detection of the trawl gear by marine mammals (Goetz et al. 2014).

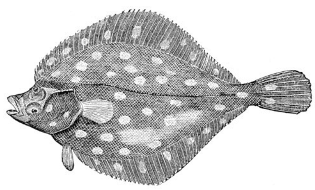
In the case of pinnipeds, acoustic deterrence tends to be largely ineffective, and in fact can significantly increase "interactions" of seals and sea lions with the fishing gear (Jacobs and Terhune 2002). Pinnipeds may either tolerate or habituate to high noise levels (likely as a result of food motivation) and may consequently suffer hearing damage, which would further reduce responsiveness (Götz and Janik 2013). The interactions can increase pinniped bycatch rates (Carretta and Barlow 2011) as well as damage to nets and loss of catch (Bordino et al. 2002). Loud sounds, including "cracker shells", "seal bombs", "bottle rockets", gunshots, broadcasted killer whale calls, harassment devices and direct harassment by boats, have been used experimentally to reduce pinniped interactions with static nets and marine aquaculture facilities. None of these approaches have proved successful in the long term (Jefferson and Curry 1996; Hamilton and Baker 2019; Tixier et al. 2021).

Deterring pinnipeds from approaching and entering trawl nets has proven especially challenging. Otariids (Otariidae, the sea lions and fur seals), in particular, are strongly attracted to trawlers and, given their unwary and bold behaviour, normally do not respond to attempts to scare them away. This makes them vulnerable not only to incidental mortality and harm in trawl gear but also to aggressive retaliation by fishers (including shooting; Tixier et al. 2021).

Tixier et al. (2021) concluded that acoustic deterrence is the most widely used approach, worldwide, but also the least successful in terms of minimizing “depredation” by marine mammals, and also the most ambiguous when it comes to reducing potentially harmful effects on them or other species. Still, acoustic deterrence remains the approach used most often—even when it comes to the deterrence of species that are highly opportunistic and adaptable (such as bottlenose dolphins or killer whales) and that, in the long run, often appear to tolerate acoustic and other disturbance as long as food is easily obtained (Bonizzoni et al. 2022).

Noisy and persistent acoustic devices can have negative side effects on cetaceans, including hearing damage and exclusion from critical habitat (Morton and Symonds 2002; Olesiuk et al. 2002; Kyhn et al. 2015; Tixier et al. 2015; van Beest et al. 2017; Findlay et al. 2021, 2024; Todd et al. 2021). For instance, harbour porpoises exposed to commercial acoustic harassment devices at distances of 1–7 km displayed fleeing, altered echolocation behaviour and unusual tachycardia. Moreover, during the 15-min exposures, half of the animals received cumulative sound doses close to published thresholds for temporary auditory threshold shifts (Elmegaard et al. 2023). Those authors concluded that exposure to acoustic harassment devices, kilometres away, can evoke cardiac responses which may impact blood-gas management, breath-hold capability, energy balance, stress level and risk of by-catch. Acoustic devices can also represent a significant source of chronic underwater noise, with negative effects on fish and other species (Goetz et al. 2015; Findlay et al. 2018).





Visual devices

Visual cues can affect the way fish and other marine organisms respond to fishing gear (Wardle 1993; Winger et al. 2010; Karlsen et al. 2021). In trawls, artificial light can illuminate portions of the gear and act as a visual cue (Hannah et al. 2015; Lomeli and Wakefield 2019; Grimaldo et al. 2018; Melli et al. 2018b; O’Neill and Summerbell 2019), sometimes helping to reduce the bycatch of certain species. For example, green LED lights illuminating the upper bridles and wings of the trawl helped reduce the bycatch of Pacific halibut *Hippoglossus stenolepis* in bottom trawls operating off Oregon (Lomeli et al. 2021). Green LED lights mounted on the groundgear of bottom otter trawls targeting ocean shrimp *Pandalus jordani* off Oregon reduced the bycatch of several unwanted species of fish (Hannah et al. 2015).

However, the use of artificial light for bycatch mitigation in trawl gear has not been straightforward, and many of the trials have been inconclusive (e.g. Larsen et al. 2018; Melli et al. 2018b). Fish reactions to light are inconsistent and depend on depth, light characteristics and intensity, species, size, and physiological state of the fish (Marchesan et al. 2005; Hannah et al. 2015; Southworth et al. 2020). The photosensitivity of commercial species is largely unknown (Karlsen et al. 2021), and artificial lighting is just one among many sensory stimuli when fish encounter and enter trawl gear (Hannah et al. 2015; Grimaldo et al. 2018; Melli et al. 2018b). While some species are attracted by bright lights, others tend to avoid them (Marchesan et al. 2005). In deep-water species adapted to dim light or darkness, bright lights may either temporarily impair vision or cause these species to swim away from the light source (Karlsen et al. 2021).

Luminous trawl netting (which does not require batteries or electronics) has recently been developed as an alternative to the use of artificial lights, primarily to increase the capture of fish, crustaceans or other marine organisms. Such netting might also be used to reduce unwanted catches (Karlsen et al. 2021).

Efforts to limit trawling

Management efforts to reduce trawling have included temporary (e.g. seasonal) and permanent fishing closures, habitat-based restrictions, inshore restrictions, designation of protected areas, deployment of anti-trawling structures, spatial confinement of trawling (i.e. "freezing the trawl footprint"; McConnaughey et al. 2020), and buyback initiatives (Squires 2010). These efforts have often been intended to benefit the non-trawl fishery sector, e.g. small-scale coastal fisheries using static gear, or to protect benthic habitat.

Efforts to limit trawling may bring unwanted effects (McConnaughey et al. 2020). For instance, when days at sea are reduced or an area becomes closed to trawling, fishers may respond by increasing vessel size and engine power (which partly compensates for reduced effort), or by increasing trawling effort outside of the area where trawling has been banned (which increases pressure on the adjacent resources and benthic habitats). In some cases, benthic impacts might increase despite management measures intended to reduce or remove fishing effort. For example, trawl fishers might use their buyback grants to invest in more fishing capacity and move their operations to more vulnerable habitats in distant fishing areas (McConnaughey et al. 2020).

Generally speaking, however, limiting the overall intensity of trawling should result in substantial ecosystem benefits, and in some cases these benefits may exceed those that would result from designating protected areas (Dinmore et al. 2003; Abbott and Haynie 2012; McConnaughey et al. 2020). Below, we describe some of the main management avenues to limit trawling effort.

Seasonal fishing closures

Temporal (as opposed to spatial) fishing closures require a complete cessation of trawling for a given period of time. Seasonal area closures are a common management measure, and normally aim to reduce mortality of target species or protect fish populations at vulnerable phases of their life cycle (e.g. during recruitment), for periods ranging between one and several months (Demestre et al. 2008).

Seasonal closures are assumed to offer relief to benthic communities impacted by bottom trawling. However, while the biomass of some species can actually increase after closures that last several years (Pipitone et al. 2000), closures that last only one or a few months are often too short to ensure meaningful or lasting benefits (Demestre et al. 2008). For instance, a study in Greece indicated that a four-month closure of bottom otter trawling did not allow for recovery of the benthic community (Smith et al. 2000).

Additionally, if trawling effort during temporal or seasonal closures is relocated to environmentally sensitive or previously unfished areas where trawling is still allowed, the negative effects on benthic communities would be merely displaced. In these cases, effort reductions or permanent area closures would be a better management option (Dinmore et al. 2003). Therefore, any attempt to assess the effects of temporal fishing closures should take into account not only the actual (rather than assumed) benefits of potential recovery during the closure periods, but also the negative effects caused by trawling in adjacent or nearby areas that result from displacing fishing activity into them (Hiddink et al. 2006).

Habitat-based restrictions

Bottom trawling is often prohibited in habitat types that are easily disturbed and slow to recover, such as coral and sponge reefs, seagrass meadows, and other seabed communities characterized by endemic or rare assemblages of sessile epifauna (Koslow et al. 2001; McConnaughey et al. 2020). Examples include prohibitions of trawling over seagrass, coralligenous and maërl habitats in several areas of the northern Mediterranean Sea, over mussel reefs and sand volcanos capped with cold-water corals in Scotland, and over glass-sponge reefs in western Canada (McConnaughey et al. 2020).

Habitat-based restrictions on towed fishing gear can provide effective protection, provided that prohibitions are introduced prior to significant physical disturbance, and that enforcement and compliance are adequate (Howell et al. 2010). However, the designated areas are often small, and their benefits to the ecosystem limited. At the same time, these small areas may benefit local economies that rely on small-scale fisheries and nature tourism (McConnaughey et al. 2020).

Inshore restrictions and zoning

The creation of fishing zones defined by depth or distance from shore can be a way to protect important nearshore habitats and minimize conflict between industrial and small-scale fishing fleets (McConnaughey et al. 2020). Industrial fishing, and trawling in particular, is often prohibited within Inshore Exclusion Zones (IEZs), which are normally reserved for small-scale fisheries. Fishing effort inside and outside of these zones may be partitioned based on either fishing gear or on vessel size and gross tonnage.

For example, many coastal countries in Africa and Asia have fishery restriction zones defined by depth or distance from shore, and these include regulations that limit or ban bottom trawling (Funge-Smith et al. 2013; Belhabib et al. 2020). The government of Liberia introduced a six nautical mile IEZ to protect the inshore small-scale fishery, which supports the livelihoods of an estimated 33,000 people (Steadman et al. 2021). In Croatia, trawling is prohibited within one nautical mile of the coast (McConnaughey et al. 2020), whereas in Italy it is prohibited within three nautical miles of the coast, or inside the 50 m isobath (Pranovi et al. 2015).

Inshore restrictions per se do not necessarily reduce overexploitation. For instance, small trawlers banned from a coastal area may be unable to fish in waters farther offshore, where trawling is still allowed, and consequently they would be forced to change their métier. Such a change could lead to impacts within the coastal habitat that the trawl ban was intended to protect (Pranovi et al. 2015). Therefore, inshore restrictions on trawling should be combined with management measures that prevent overfishing and do not lead to replacement of trawls by other gear that is as destructive as, or even more destructive than, trawls (e.g. towed and mechanized dredges).

Protected areas

Countless studies have demonstrated that setting aside areas that are either fully protected from fishing, or at least protected from the most destructive types of fishing, allows marine ecosystems to rebound and recover. Well-managed protected areas help preserve healthy food webs and protect vulnerable species (Mora et al. 2006; Guidetti and Sala 2007). When protection measures are implemented appropriately (including enforcement when necessary), fish biomass increases (Sala and Giakoumi 2017; Gill et al. 2024), fish and other organisms normally start colonizing nearby areas, and the biological spill-over effects can benefit the fishers operating in adjacent areas (Murawski et al. 2005; Roberts et al. 2005; Goñi et al. 2010; Kerwath et al. 2013; Marshall et al. 2019). Protected areas can also benefit other stakeholders, for instance in the tourism sector, offering opportunities for observation and education in a more natural environment (Angulo-Valdés and Hatcher 2010; Chae et al. 2012).

Designations that attribute importance to given sites vary greatly (e.g. Biosphere Reserve, Marine Park, Marine Reserve, Marine Sanctuary, National Park, Nature Reserve, Special Area of Conservation, Specially Protected Area), as do the management frameworks and protection measures applied to such sites. Generally speaking, however, as of 2006 only around 0.7% of the world's sea surface area (approximately 2.5 million km^2) was covered by some kind of "protective" regime, encompassing less than 2% of the total marine area within Exclusive Economic Zones (Wood et al. 2008). Only 0.1% of the world's oceans, and 0.2% of the total marine areas under national jurisdiction, were actually protected from commercial fishing (i.e. "no-take" areas; Wood et al. 2008), with the global distribution of protected areas being both uneven and unrepresentative, and only half of the world's marine protected areas (MPAs) being part of a coherent network. Global protection coverage has reportedly been increasing over time, but as of 2018 MPAs of all types were still covering a mere 3.6% of the world's sea surface area, with only approximately 2% being "strongly" or "fully" protected (i.e. no industrial fishing allowed; Lubchenco and Grorud-Colvert 2015; Sala et al. 2018c). A more recent assessment of the world's 100 largest MPAs by area, representing nearly 90% of reported global MPA coverage and 7.3% of the global ocean area, indicated that one third of the assessed MPA coverage was "incompatible with the conservation of nature" due to industrial activities, primarily industrial fishing; approximately 2.6% of the global ocean area was judged to be "highly" or "fully" protected (Pike et al. 2024).

The environmental benefits of protected areas depend on location and size of the area, connectivity with other protected areas (if part of a protection network), types of biological and ecological communities, staff capacity and budget, and the type and degree of realized, actual protection, which may range from "no access" to allowance for multiple uses (Edgar et al. 2014; Sciberras et al. 2015; Gill et al. 2017; Vrooman et al. 2022). Maximum conservation benefits are expected when sessile and habitat-forming species as well as slow-growing species with moderate dispersal capacity are protected from bottom trawling (Fulton et al. 2015; Kaiser et al. 2018). Similar to other area-based restrictions, a downside of protected areas is that they often displace trawling to adjacent areas (Dinmore et al. 2003; Hiddink et al. 2006; Sciberras et al. 2013).

While protected areas can effectively promote conservation (Wood et al. 2008; Erm et al. 2023), their raison d'être fails miserably when protection from bottom trawling is not included. In European waters, for instance, Dureuil et al. (2018) found that about 60% of 727 formally designated protected areas located in Atlantic European waters were commercially trawled (Dureuil et al. 2018). Various other European areas that are purportedly protected from harmful human impacts, including a vast network of "Natura 2000" sites, allow bottom trawling within their borders (Perry et al. 2022).

A positive example of place-based management in the Mediterranean Sea is the Fisheries Restricted Areas (FRAs) network created under the General Fisheries Commission for the Mediterranean (GFCM), which includes a large area (1,730,000 km^2) encompassing all Mediterranean and Black Sea seabed deeper than 1000 m, where the use of trawl nets and towed dredges is prohibited to protect deep-sea benthic habitats. Other examples of protection in waters that are mostly more than 1,000 m deep (McConnaughey et al. 2020) include: 1) the protection of nearly 1.8 million km^2 of benthic habitat from bottom trawling within United States marine areas, mostly in the Pacific (Hourigan 2009); 2) a 3.3 million km^2 network of protected areas in Australia, generally not including fishing grounds (Mazor et al. 2017); 3) a voluntary ban to protect benthic habitat in 11 deep-sea areas encompassing approximately 300,000 km^2 in the southern Indian Ocean, enacted by four fishing companies (McConnaughey et al. 2020); and 4) 17 areas in New Zealand's EEZ, encompassing 1.1 million km^2 and containing ten major seamounts and ten active hydrothermal vents, where bottom trawling and dredging are banned.

Anti-trawling structures

Underwater structures such as artificial reefs can be used to deter bottom trawlers from fishing in particular (usually fairly small) sites and therefore reduce the intensity of trawling (Fabi et al. 2015). By increasing the risk of gear entanglement and loss, these structures aim to dissuade trawl fishers from entering the area, while also reducing damage caused by trawlers to small-scale fisheries (Muñoz-Pérez 2008; Fabi et al. 2015). According to Ali et al. (2013), the construction and deployment of artificial reefs can "rejuvenate" fishery resources; and in addition, it would "curb encroachment of trawlers, reduce conflict between commercial and traditional fishers, and (...) increase opportunities for small-scale fisherfolk to improve their livelihood from fishing."

Artificial reefs used to be made of materials such as tires and derelict watercraft, but in recent decades they have been made mainly of concrete modules with metal rods or tines (Guillén et al. 1994; Muñoz-Pérez 2008; Ali et al. 2013). In a 30 km^2 coastal area of Cambodia, 40 concrete modules reportedly decreased illegal trawling within months (Strong et al. 2023). Concrete modules weighing 5–40 tons and measuring 2–4 m per side were reported to effectively deter illegal trawling in selected sites along the east coast of Peninsular Malaysia (Ali et al. 2013). In the coastal waters of northern Spain, anti-trawling reefs led to a progressive increase in total biomass, species richness and, to a lesser extent, maximum length and percentage of large fish. The main species profiting from the exclusion of trawlers included seabreams, catsharks and skates, red mullets, gurnards and John dory *Zeus faber* (Serrano et al. 2011). Anti-trawling reefs were installed in the Marine Reserve of Tabarca and in El Campello, Spain, to protect *Posidonia oceanica* seagrass meadows. These reefs, which consisted of approximately 400 concrete modules, reportedly eliminated illegal trawling and contributed to seagrass recovery (Guillén et al. 1994). Another example is the deployment of about 600 concrete modules, grouped in 11 barriers and placed in an area of 270 km^2 in the Gulf of Cadiz, Spain. This artificial reef reportedly eliminated illegal trawling within six months (Muñoz-Pérez et al. 2000). Examples from the Mediterranean Sea also include artificial reefs deployed off the Adriatic Sea coast of Italy to reduce conflict between trawl and artisanal fisheries, and those deployed in the Gulf of Gabès, Tunisia, to prevent illegal trawling near the coast, protect seagrass beds and provide shelter to marine organisms (Fabi et al. 2015).

Muñoz-Pérez et al. (2000) reported the cost of artificial reefs installed off southern Spain between 1989 and 1998: USD 63,000 to cover an area of 3 km^2 with 55 units of 4 tons each; USD 320,000 to protect 30 km^2 with about 500 units of 2–3 tons each; and USD 650,000 to cover 270 km^2 with 600+ units of 6 tons each. In Cambodia, USD 500 were required to construct and deploy a single anti-trawling concrete unit (2.5 m wide, 2–3 tons; Strong et al. 2023). According to Strong et al. (2023), the relatively modest time and costs involved in the construction and deployment of artificial reefs, and their long life-span, make them a reasonable investment as compared with alternatives such as decommissioning trawl vessels, diversifying fishing activities, or patrolling the area to prevent illegal trawling.

Spatial confinement of trawling

Studies based on remote tracking of vessels showed that trawl effort tends to be concentrated within particularly productive areas (Amoroso et al. 2018; Jennings and Lee 2012). One way to limit the overall ecological impact of trawling activities is by confining them to these high-effort areas. For example, untrawled and low-effort areas in the Great Barrier Reef region of Australia were temporarily closed to prevent further expansion of trawling (Pitcher et al. 2016). Similarly, bottom trawling in areas of Alaska, the Barents Sea and the North Atlantic was either temporarily or permanently confined to previously fished areas (McConnaughey et al. 2020).

An advantage over other forms of management is that "freezing the trawl footprint" (McConnaughey et al. 2020) avoids the negative effects of temporal and spatial bans, such as the displacement of fishing effort into previously untrawled areas (Dinmore et al. 2003; Hiddink et al. 2006; Sciberras et al. 2013). In addition, confining trawl effort within established fishing grounds may be more consistent with the expectations of trawl fishers (Gillis et al. 1993; Kaiser 2005). Such confinement, however, limits a fleet's adaptability to changing fish abundance and distribution (Hilborn et al. 2023), and it needs to be coupled with strict regulations of fishing effort and catches to prevent overexploitation of fish populations (McConnaughey et al. 2020).

Historical data indicate that the location of the most heavily trawled areas can vary over time, as fish populations decline or target-species preferences shift (Jennings et al. 2012). While geographic confinement may help prevent overexploitation, and justify place-based management action to maintain healthy fish resources, confined fleets won't be able to "follow the fish" if the distribution of target species changes because of climate and environmental shifts (McConnaughey et al. 2020).

Permanent prohibition of trawling

Efforts to constrain or prohibit bottom trawling have been contentious, and challenged by political, social, economic and cultural obstacles. Still, governments and managers may be interested in assessing the alternatives to a fishery that can be highly subsidized, economically inefficient, with an aging workforce, and which involves considerable environmental, climate, and social challenges (Steadman et al. 2021). In that sense, a permanent prohibition offers the most comprehensive protection of seabed habitats from the harmful effects of trawling (McConnaughey et al. 2020). Trawl prohibitions may be advocated for reasons other than environmental concerns, as they enhance fishing opportunities for small-scale and other fisheries. Still, it goes without saying that any transition away from trawling must avoid the demonization of trawl fishers, and ensure that they have access to suitable alternatives (see the following section).

Transitions away from trawling

Comprehensive trawl bans can have severe socio-economic consequences, and can cause significant unrest if alternative employment or fishing strategies are unavailable (Salim et al. 2010; McConnaughey et al. 2020). For instance, a trawl ban in Venezuela reportedly affected more than 260 trawl vessels, displacing approximately 6,500 industry workers, and indirectly affecting as many as 26,000 jobs, with supplies of cheap fish for the domestic market also being reduced as a result of the ban (McConnaughey et al. 2020).

In Indonesia, trawling was banned in response to protests by small-scale fishers in the Malacca Straits and off the northern coast of Java—two of the most important trawl grounds (Bailey 1997). Almost 25,000 trawl fishers became unemployed following the ban, while shrimp exports dropped by 22% during the first year of the ban. The ban also eliminated the supply of "trash fish" to fishmeal factories, and Indonesia began to import fishmeal as a result (McConnaughey et al. 2020). While the elimination of trawlers led to unemployment in that sector, small-scale demersal fisheries experienced a dramatic growth, with landing increases of up to 60% and new employment opportunities being quickly filled (Bailey 1997). Such growth generated significant new economic opportunities, but also raised new fishery management problems. The net effect of the trawl ban was to reallocate resource access to small-scale fishers, as the government recognized the importance of promoting a more "traditional" fishery (Bailey 1997). Overall, the Indonesian experience suggests that a ban on trawling can cause societal hardship in the short term, with potentially positive outcomes over a longer time period (McConnaughey et al. 2020).

Considering the societal challenges which arise with attempts to prohibit trawling, transitions away from trawling must consider access to alternative fishing gears or jobs for those affected by a trawl ban, as well as strategies to compensate for any significant decrease in fish catches. Shifting to alternative fishing gear may have undesired environmental effects (e.g. in terms of bycatch of vulnerable species in static gear), and these effects should also be considered.

Transition to alternative fishing gear

Trawl fisheries have evolved and boomed primarily because they offer higher catches and greater profitability, and because the environmental damage they cause is largely externalized (Seas at Risk and Oceana 2022). A common objective of trawl bans is to promote the use of alternative fishing gears such as static nets, traps/pots and longlines (Suuronen et al. 2012; Pham et al. 2014). Even if these alternative gears can catch some or all of the species previously caught by bottom trawls, a transition to static gears carries uncertainties and may be less profitable (at least in the short term). For instance, seabed degradation caused by trawling can contribute to a comparatively lower catch efficiency for newly-established fisheries, and a recovery plan may be required before a static gear fishery can be introduced successfully (Sala et al. 2023).

Moving from towed to static gears such as pots and longlines, however, can reduce ecological impacts of trawling on benthic habitat and species, while increasing selectivity of target species and decreasing bycatch and discards (Hintz et al. 2017; Hilborn et al. 2023; Sala et al. 2023). Static gears, and particularly pots, have been shown to be more environmentally benign and have limited effects on benthic species and habitats (Thomsen et al. 2010; Petetta et al. 2021).

While static gears tend to be less profitable than trawling (Sala et al. 2023), there can be exceptions. For instance, a trawl ban in the territorial waters of Venezuela led to increased catches by small-scale fishers, who supplied 70% of annual fishery production as compared to 6% by trawl fishers in previous years (McConnaughey et al. 2020). Transitions may also result in substantial societal benefits. For instance, in Qatar the number of small-scale fishers and their catch increased by 52% and 159%, respectively, after bottom trawling was banned (El Sayed 1996; Walton et al. 2018; McConnaughey et al. 2020). As described in the previous section, a major trawl ban in Indonesia led to substantially higher landings and new employment opportunities for small-scale fishers (Bailey 1997).

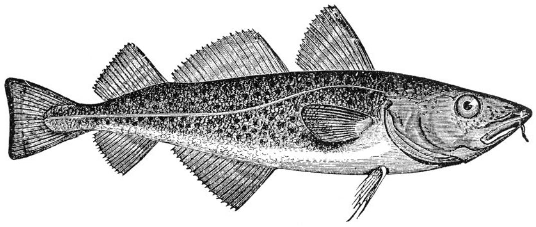
Switching to different gear types can also reduce greenhouse gas emissions from fuel use. For instance, Thrane (2006) found that fuel input per kg of caught flatfish would decrease by a factor of 15 by switching from beam trawl to Danish seine, and suggested that if all flatfish were caught by Danish seine or static fishing gear such as gillnets, it would be possible to save around 30,000 m^3 of fuel per year in the Danish fishery. Ziegler and Hansson (2003) found that fuel consumption per kg of cod would decrease by a factor of four by switching from trawls to gillnets. However, certain trap fisheries targeting crustaceans such as lobster require amounts of fuel per unit landed that can be higher than those of bottom trawlers (Parker and Tyedmers 2014; Hilborn et al. 2018).

Finally, when transitioning to fisheries using static gear, the risks must be considered and it is necessary to assess which gear type is the least harmful. For instance, gillnets have their own drawbacks in terms of bycatch (e.g. of small cetaceans; Read et al. 2006; Reeves et al. 2013; Brownell et al. 2019). In some areas, static gear making use of "vertical" lines connecting surface buoys to bottom gear can increase the risk of entanglement, injury and mortality of large whales. Striking examples are the entanglement of critically endangered North Atlantic right whales *Eubalaena glacialis* in lobster and crab pot lines in the eastern United States and Canada (Knowlton et al. 2022) and humpback whales *Megaptera novaeangliae* in crab pot lines off western North America (Riekkola et al. 2023).

Transition to alternative employment

Efforts to limit or ban trawling are challenging when they are likely to lead to unemployment for a large number of fishers. Simply enforcing bans and leaving fishers with no feasible alternatives can lead to protest, management failure or defiance of laws (McConnaughey et al. 2020). Therefore, attention must be paid to issues related to social justice and inclusion, whether these issues concern displaced workers in the trawl fishery sector, or the small-scale fisheries that are displaced or marginalized by the economic interests of industrial fisheries (Bennett 2018; Cohen et al. 2019).

"Just transitions" are strategies to move away from harmful extractive economies, and they aim to ensure that workers have access to new jobs and opportunities. These transitions normally include pathways to quality jobs, training, and worker transition funds (Steadman et al. 2021). For instance, a trawl ban in Hong Kong included compensation to the owners of trawlers that were forced to stop fishing in inshore waters, a voluntary buyout scheme, and payment of a one-off grant to help local deckhands (Morton 2011). The government paid USD 219 million to enforce the trawler buyout scheme, which included grants for all affected fishing crews (Loh and Jaafar 2015). The transition also included training and technical support to help fishers switch to other jobs (e.g. in the growing aquaculture sector; Morton 2011).



Transition to alternative foods

Strategic changes in the food extraction, production and consumption systems could help alleviate the negative effects of destructive fishing and agricultural practices, while also addressing climate and food security concerns (Krishna Bahadur et al. 2018). Altering diets is increasingly acknowledged to be an important element of any practical solution to the problem of feeding the world's growing human population (Parodi et al. 2018), and a plethora of food alternatives have been proposed in recent years.

The most-discussed alternative to business-as-usual is switching to a balanced plant-based diet to reduce the environmental impacts of meat and seafood production and consumption (Godfray et al. 2018; Poore and Nemecek 2018; Swain et al. 2018). Other alternatives to conventional animal sources of food and feed include edible microorganisms and precision fermentation (Ritala et al. 2017; Linder 2019a, 2019b; Augustin et al. 2023; Hilgendorf et al. 2024), cultured meat (Post et al. 2020; Zhang et al. 2020) and insect-based foods (Rumpold and Schlüter 2013; van Huis 2013; van Huis and Oonincx 2017; de Castro et al. 2018). Edible insects are, in fact, commonly consumed in Africa, Latin America and Asia (Raheem et al. 2018), with over 1,900 different species of insects being used as food, worldwide (FAO 2013). Within the fishing and aquaculture realms, alternatives include placing emphasis on the consumption of seafood that entails less harmful environmental impacts (e.g. farmed bivalves; Jacquet et al. 2017; Gawel et al. 2023; Willer and Aldridge 2019, 2020; Olivier et al. 2020; Willer et al. 2021).

Alternative diets for humans (and farm animals) hold promise. One study indicated that replacing animal-source foods in current diets with "novel foods" such as cultured meat, eggs, milk, plants, algae, bacteria, and fungi, could reduce environmental impacts by 80%, while providing nutritious options (Mazac et al. 2022). Alternative diets could also offer substantial health benefits and help prevent some diet-related chronic diseases (Tilman and Clark 2014).

Novel foods, however, require education and promotion, as well as market accessibility through appropriate supply chains and lower prices for consumers (Treich 2021; Pakseresht et al. 2022). Importantly, no single food—whether old or new—is going to solve what is a global problem: action is needed on all fronts to move towards diets and food systems that alleviate human impacts on ecosystems.

The Mediterranean scenario

11

Mediterranean Sea

The Mediterranean Sea is the world's largest (about 2,500,000 km^2) and deepest (average depth about 1,500 m, maximum 5,109 m) enclosed sea. It is connected to the Atlantic Ocean through the Strait of Gibraltar, to the Sea of Marmara and the Black Sea through the Dardanelles, and to the Red Sea through the Suez Canal. Overall, the Mediterranean Sea is considered oligotrophic (Basterretxea et al. 2018), though some subregions have high productivity—including the Adriatic Sea, the Gulf of Lions and large continental shelf areas between Tunisia and Sicily. Biological productivity decreases from north to south and from west to east, while salinity and temperature increase from northwest to southeast (Coll et al. 2010). The Mediterranean Sea has high marine species richness (about 17,000 species) and endemism, making it one of Earth's hotspots for marine biodiversity (Costello et al. 2010; Piroddi et al. 2020).

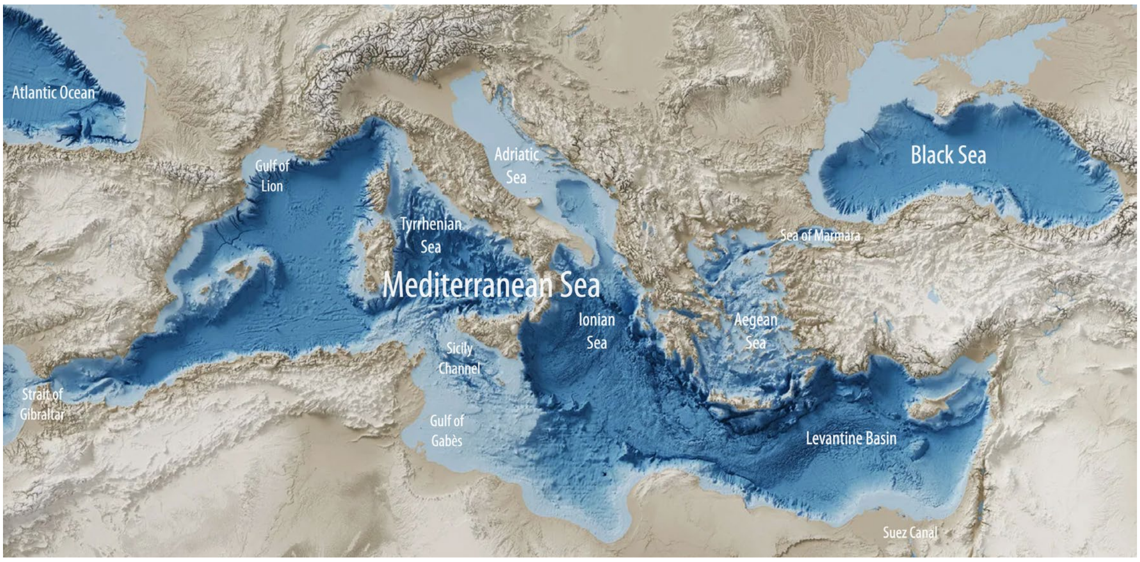


Fig. 8. The Mediterranean Sea, with some of the areas cited in the text (adapted from visualwallmaps.com)

Overview of Mediterranean fisheries

In this section we provide an overview of the current status of Mediterranean fisheries, with a focus on bottom trawling, based on relevant information contained in the latest regional reports by FAO (2022a, 2023), as well as other complementary information. We also discuss evidence of overfishing based on recent scientific literature from the Mediterranean region.

Some of the information refers to both the Mediterranean Sea and the Black Sea, because FAO and its regional fishery management organization GFCM (General Fisheries Commission for the Mediterranean) pool these two seas together within FAO Area 37 (FAO 2019). FAO Area 37 encompasses territorial waters of Albania, Algeria, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Egypt, France, Georgia, Gibraltar, Greece, Israel, Italy, Lebanon, Libya, Malta, Monaco, Montenegro, Morocco, Palestine, Romania, Russian Federation, Slovenia, Spain, Syria, Tunisia, Turkey and Ukraine. About half (53%) of the area is under national jurisdiction (including the entire Black Sea), while 47% is in areas beyond national jurisdiction.

Fishery landings and trends

Landings from Mediterranean capture fisheries increased sharply between the 1970s and 1980s, and continued to increase until 1994, with a peak of 1,087,100 tonnes, but have been declining over the past three decades, with annual landings down to 750,000 tonnes in 2015, and 665,053 tonnes in 2020 and 2021 (FAO 2023; Fig. 9). The drop in landings in 2020–2021 was likely exacerbated by COVID-19 restrictions (GFCM 2020; FAO 2022a, 2023).

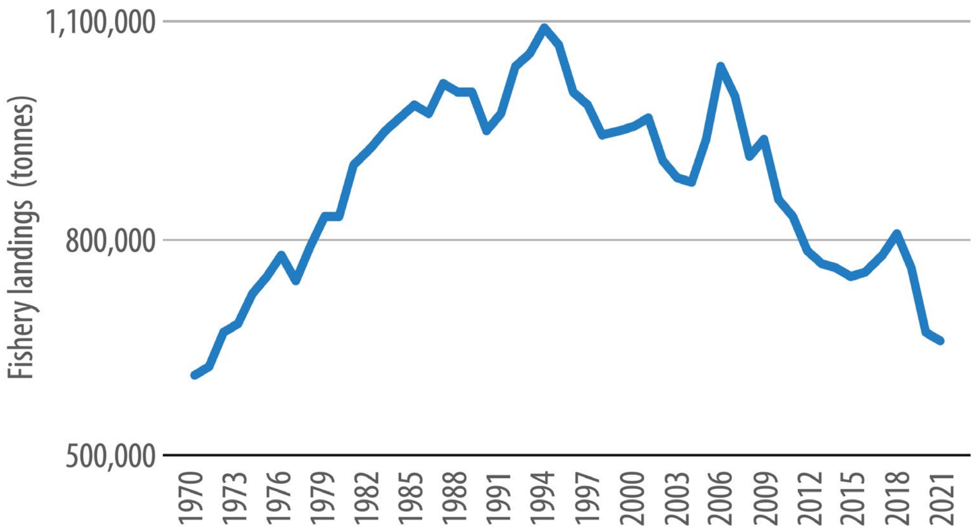


Fig. 9. Total reported fishery landings in the Mediterranean Sea, 1970–2021 (adapted from FAO 2023; the Black Sea trendline was removed from the original figure)

Of the 665,053 tonnes landed annually in 2020 and 2021, 196,695 (29.6%) came from the western Mediterranean, 159,986 (24.1%) from the central Mediterranean, 163,012 (24.5%) from the eastern Mediterranean, and 145,360 (21.9%) from the Adriatic Sea.

Of the four fleet segments or groups considered by FAO's GFCM, "purse seiners and pelagic trawlers" were responsible for the largest share (48%) of total landings (in the Adriatic Sea this figure was 59%). "Trawlers and beam trawlers" (i.e. bottom trawlers) had the second largest contribution (24%), whereas "small-scale vessels" contributed 20% of total landings. "Other fleet segments" (including dredgers, tuna seiners, longliners >12 m, and polyvalent vessels) contributed 9% of total landings (FAO 2023; Fig 10).

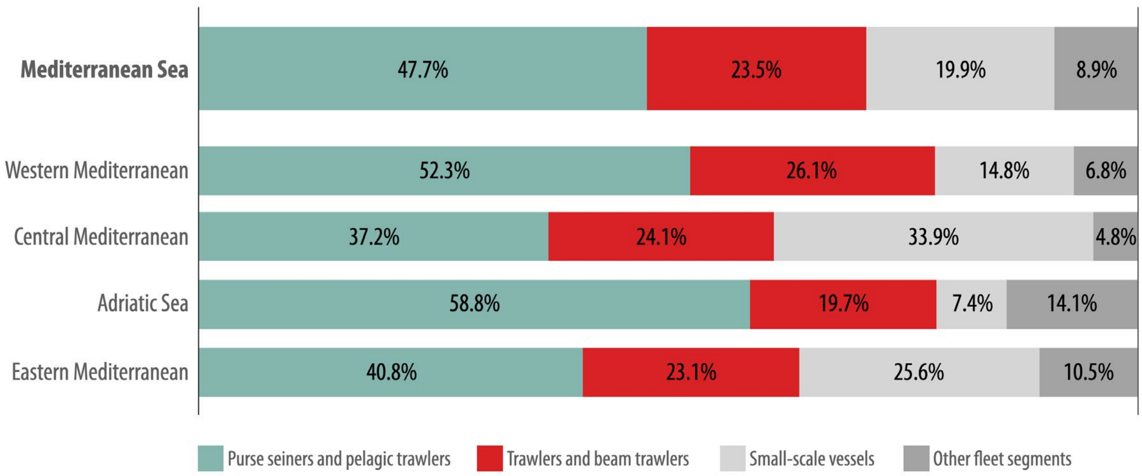


Fig. 10. Relative contribution of GFCM fleet segment groups to total reported landings in the Mediterranean Sea and its subregions, 2020–2021 (adapted from FAO 2023)

Mediterranean catches were dominated by small pelagic fish. In 2020–2021, sardines and anchovies were the two predominant species in the Adriatic Sea (20 and 47% of total landings, respectively), in the western Mediterranean (13 and 18%) and in the eastern Mediterranean (14 and 14%). However, landings of anchovies and sardines have shown declining trends since the 1980s, whereas landings of horse mackerel and whiting have declined since the early 1990s. Conversely, catches of deep-water rose shrimp *Parapenaeus longirostris*, cuttlefish (Sepiida), red mullet, and surmullet *Mullus surmuletus* have shown increasing trends in recent decades (FAO 2023). Details of annual (2018–2020) reported landings by main species are given in Fig. 11 (FAO 2022a).

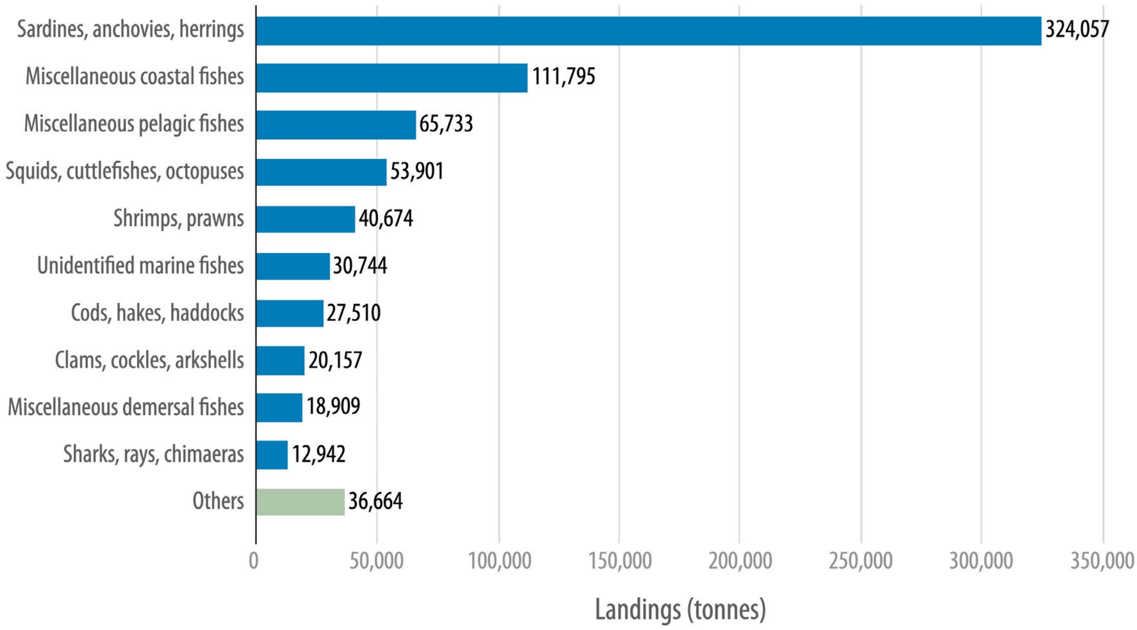


Fig. 11. Reported total landings of the main species caught in the Mediterranean Sea, 2018–2020 average (adapted from FAO 2022a)

Discard ratios varied widely depending on fishing method and geographic area (FAO 2022a). Trawlers showed by far the highest discard ratios, ranging from 34 to 44% of the catch. Discard ratios in small-scale fisheries ranged from 3 to 15%, whereas all other types of gear showed much lower ratios (below 7%). According to FAO (2022a), longliners and bottom trawlers were the two fleet segments with the highest recorded occurrence of incidental capture of vulnerable species, accounting for about 80% of reported individuals. Vulnerable species subject to bycatch included sea turtles (89% of the records), elasmobranchs (8%), cetaceans (2%) and seabirds (1%).

Fleet composition and revenue

In 2022, the fishing fleet operating in the Mediterranean totalled approximately 73,000 vessels. According to FAO (2023) "small-scale vessels" led the fleet composition in all subregions, and especially in the central and eastern Mediterranean, where they represented 85% of the operating fleet (19,600 and 18,800 vessels, respectively). In a previous assessment, FAO (2022a) estimated that 1,686 "trawlers and beam trawlers" were operating in the western Mediterranean, 1,381 in the Adriatic Sea, 1,303 in the central Mediterranean, and 1,298 in the eastern Mediterranean.

The total revenue from marine capture fisheries in the Mediterranean was estimated as USD 2.7 billion (FAO 2023), a figure which represents the value at first sale (prior to any processing or value-adding).

Figures encompassing both the Mediterranean and the Black Sea indicate that small-scale fisheries generated 61% of total employment and 26% of total revenue. "Trawlers and beam trawlers" generated 16.5% of total employment and 37% of total revenue, whereas "purse seiners and pelagic trawlers" generated 16.5% of total employment and 27% of total revenue (FAO 2023).

AIS-based monitoring of trawling

Using AIS data and Global Fishing Watch algorithms (Kroodsma et al. 2018), Merino et al. (2019) assessed the activity of different types of fishing vessels in the Mediterranean and Black Sea (FAO Area 37). A major challenge was AIS use and coverage: almost 100% of European vessels larger than 15 m in the northern Mediterranean transmitted AIS information, whereas almost no vessel from North African countries used this technology, and AIS reception was poor in southern and eastern Mediterranean areas. AIS captured a large fraction of trawlers and purse seiners, but failed to capture small vessels using set gillnets and other gears. Out of a total of 3,588 Mediterranean fishing vessels broadcasting AIS, 3,132 (73%) were matched to a registry including information on gear type (Merino et al. 2019).

In AIS registries, trawlers were the primary vessels by gear for all Mediterranean countries except Turkey, where purse seiners dominated. Of vessels over 15 m with AIS information, trawlers and purse seiners were dominant across Mediterranean subareas including the waters off northeast Spain, the Gulf of Lions, the waters off Liguria, and the Tyrrhenian and Adriatic seas (Merino et al. 2019).

Fig. 12 shows the spatial distribution of trawlers in 2017, obtained by Merino et al. (2019) based on AIS data. This pattern is largely consistent with those obtained in previous years by Ferrà et al. (2018, 2020). Trawling was intensive off the Mediterranean coasts of Spain, Italy and Greece, in the Gulf of Lions, in the Sicily Channel, and particularly in the Adriatic Sea. Poor or no AIS coverage in the Levantine Basin (where considerable trawling is known to occur) and on the continental shelves of North African countries (including in the Gulf of Gabès, Tunisia, where trawling is known to occur; Hattab et al. 2013) meant that trawling activity in those areas could not be analysed and illustrated.

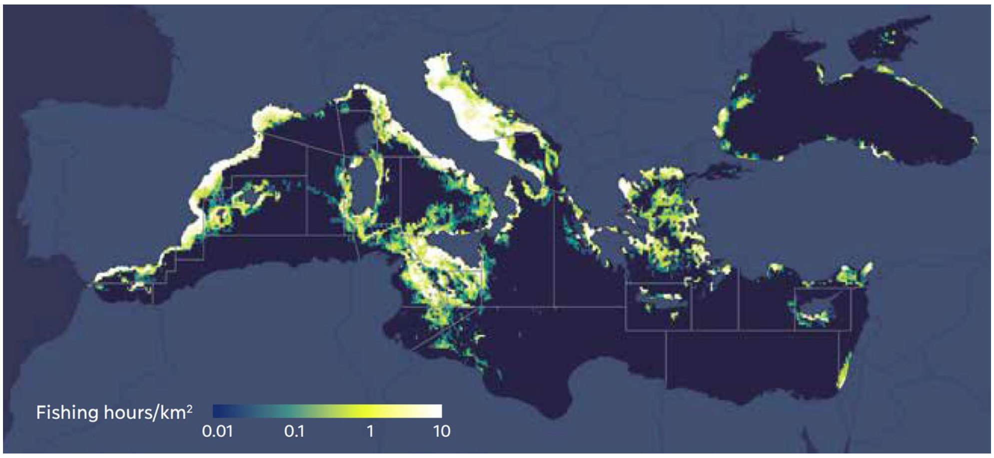


Fig. 12. AIS-based activity of trawlers in the Mediterranean and Black Sea (FAO Area 37), expressed as fishing hours per km^2 in 2017 (adapted from Merino et al. 2019); some of the data limitations are noted in the main text

Evidence of overfishing

As reported in Chapter 1 (and see Fig. 3), the Mediterranean and Black Sea (FAO area 37) is one of the world regions having the highest rate of fishery exploitation. In this region, 62.5% of the fish populations are fished "unsustainably" (FAO 2024), and fishing pressure is twice the level considered "sustainable" by FAO ($F/F_{MSY} = 2.25$; FAO 2022a). However, according to FAO, exploitation rates have been decreasing in the past decade, and that was interpreted as indicative of improved fishery management (FAO 2023).

Six years earlier, Colloca et al. (2017) had assessed the status of Mediterranean fish populations (thus excluding those in the Black Sea) and estimated that more than 90% were "out of safe biological limits". In areas that were reportedly exploited more "sustainably" (including the Gulf of Gabès, Eastern Ionian Sea and Aegean Sea), fishing pressure was characterized by either a low number of vessels per unit area or a prevalence of small-scale fisheries. Conversely, Western Mediterranean and Adriatic Sea areas had high fishing pressure and a large proportion of overfished populations (Colloca et al. 2017). Those authors noted that "the current level of fishing pressure in the Mediterranean basin, exerted by a large variety of fishing vessels and fishing gears, has impaired the productivity of commercial stocks, increased the extinction risks for vulnerable species, such as elasmobranchs, and contributed to disrupt the productivity and functions of the ecosystem". According to Colloca et al. (2017), this scenario was "exacerbated by high pressure from vessels using towed gears".

Piroddi et al. (2017) used a food-web modelling approach to assess how the historical (1950–2011) trends of various functional groups and species were impacted by changes in primary productivity and fishing pressure throughout the Mediterranean Sea. Those authors observed a reduction of approximately 34% in the abundance of "important" commercial and non-commercial fish species, and a decline of top predators of approximately 41%, whereas there was an increase of approximately 23% in the abundance of organisms at or near the "bottom" of the food web. Primary productivity was the strongest driver, and fishing pressure was the second most important driver affecting the dynamics of fish populations. Ecological indicators showed overall ecosystem degradation over time (Piroddi et al. 2017), and patterns of overexploitation were consistent with previous studies documenting the increasing impact of fishing (e.g. Vasilakopoulos et al. 2014; Tsikliras et al. 2015). The trophic level of catches indicated a clear "fishing down" (Pauly et al. 1998) effect in most Mediterranean subregions, with the exception of the Eastern Mediterranean where there was an apparent "fishing up" (Piroddi et al. 2017). Stergiou and Tsikliras (2011), however, noted that a false fishing-up effect can occur when the large fishes were depleted far in the past, whereas small pelagic fishes and invertebrates at low trophic levels are experiencing high present-day intensity of fishing.

Management framework

In the Mediterranean Sea, commitments to protect the marine environment and minimize the impacts of human activities include compliance with various international agreements, most notably the Convention on Biological Diversity (CBD), the Convention on Migratory Species (CMS), and the Barcelona Convention for the Protection of the Marine Environment and the Coastal Region of the Mediterranean. For instance, the Kunming-Montreal Global Biodiversity Framework, adopted in 2022 by parties to the CBD, refers to the need to "take urgent action to halt and reverse biodiversity loss" and "put nature on a path to recovery" (Convention on Biological Diversity 2022). Reference to this imperative is contained throughout the document, and is particularly emphasised under Target 2: "Ensure that by 2030 at least 30% of areas of degraded terrestrial, inland water, and coastal and marine ecosystems are under effective restoration, in order to enhance biodiversity and ecosystem functions and services, ecological integrity and connectivity".

In the EU, marine conservation action is required inter alia by Council Directives 92/43/EEC (the "Habitats Directive"; EC 1992), 2008/56/EC (the "Marine Strategy Framework Directive"; EC 2008) and 2014/89/EU (the "Maritime Spatial Planning Directive"; EC 2014). Strict enforcement of existing Council Directives would restrict bottom trawling, especially within protected areas or sensitive habitats (Seas at Risk and Oceana 2022). Until now, however, implementation of Council Directives has been an uphill process (e.g. Mackelworth et al. 2011), with regulatory frameworks operating at different scales and having uncoordinated objectives (Gissi et al. 2018). In addition, international agreements such as the CBD, the CMS, and the Barcelona Convention, and even the Agreement on the Conservation of Cetaceans of the Black Sea, Mediterranean Sea and Contiguous Atlantic (ACCOBAMS; www.accobams.org) prescribe measures that are, more often than not, ignored or met with insufficient enforcement and compliance by the very nations that agreed to them. As noted by Bearzi et al. (2024), "adopted resolutions often clash with the broad economic interests involved in the human uses of the marine environment, and these interests have invariably prevailed".

In 2023, the EC proposed an Action Plan to phase out all bottom trawling within MPAs and Natura 2000 sites, consistent with existing obligations and requirements, and called on EU Member States to impose national measures accordingly (EC 2023a). Member States with strong fishery lobbying groups, however, expressed strong opposition, and sought to water down any undertaking regarded as detrimental to bottom trawl fisheries.

Such opposition to even the most basic measures of protection is in sharp contrast to the declared global urgency of increasing and expanding MPAs and other area-based conservation measures—consistent with the United Nations commitment to "protect" 30% of global oceans by 2030 (UN 2023). Within the EU, opposition to the phasing out of bottom trawling within MPAs also defies the EU Biodiversity Strategy's objective to protect 30% of the EU's seas, as well as EU directives requiring

Member States to take measures to protect the seabed to achieve the “good environmental status” of EU waters (Directive 2008/56/EC), and achieve or maintain the “favourable conservation status” of certain seabed habitats in marine Natura 2000 sites.

In this context, the decision by Greece to ban bottom trawling in all of its Marine Protected Areas by 2030, starting from Marine National Parks by 2026, represents an important commitment, which may encourage other EU Member States to take similar action. The announcement was made at the 9th “Our Ocean” Conference (Athens, Greece, 16-17 April 2024; see https://www.ourocean2024.gov.gr/commitments).

MPAs encompass a small proportion of the Mediterranean Sea (Claudet et al. 2020; and see www.mapamed.org), and a trawl ban within those relatively small areas may not be a major management challenge. Conversely, Natura 2000 sites encompass a much larger proportion of the Mediterranean (see https://natura2000.eea.europa.eu/), and the proposed phasing out of trawling has been generating significant conflict with the interests of trawl fisheries that currently operate in those waters.

Bottom trawling in the Mediterranean is either restricted or banned within Fisheries Restricted Areas (FRAs) created under GFCM (fao.org/gfcm/data/maps/fras/en/). These geographically defined areas have been established to “protect vulnerable marine ecosystems (VMEs) or sensitive habitats from potentially significant adverse impacts (VME-FRAs) and to enhance the productivity of marine living resources by protecting essential fish habitats (EFH-FRAs)” (FAO 2023). In 2005, the GFCM established a large FRA (1,730,000 km^2) encompassing all Mediterranean and Black Sea seabed deeper than 1,000 m, where the use of trawl nets and towed dredges is prohibited to protect deep-sea benthic habitats. Nine other FRAs have been established since 2006. For instance, the Jabuka/Pomo Pit FRA, established in 2017 in the Adriatic Sea, is an important example of how spatial conservation management can benefit both the marine environment and fisheries. However, while compliance was high within the no-trawl area (as intended by the closure), fishers maintained similar overall effort by trawling elsewhere—likely to mitigate the economic consequences of the closure (Elahi et al. 2018).

Trawling is generally prohibited in shallow Mediterranean coastal waters. Under certain conditions (e.g. deep nearshore waters), trawling may be allowed offshore and to as close as 0.7 nautical miles (1.3 km) of the coast. In some shallow-water basins, such as the northern Adriatic Sea, trawling is banned within 3 nautical miles (5.6 km) of the coast. In addition, several Mediterranean areas are closed to trawling at certain times of the year (as mandated by EC Council Regulation 2019/1022, GFCM, or national legislation). Trawling may also be prohibited on certain days of the week. In EU waters, trawling is generally not allowed over seagrass beds of *Posidonia oceanica* and other marine phanerogams, as well as over coralligenous habitats and mäerl beds, and some Fishing Protected Areas (as defined by EC Council Regulation 1967/2006) include trawl bans.

Regarding the use of subsidies such as those provided by the EU’s European Maritime and Fisheries Fund and earlier funding schemes, forms of assistance intended to provide environmental benefits were often implemented in parallel with capacity-enhancing subsidies leading to overexploitation (Cordón Lagares and García Ordaz 2014).

Conclusions

12

This report summarises evidence that bottom trawling is a fundamentally destructive practice that damages marine habitats and depletes marine life. While it has been argued that "well-regulated bottom-trawl fisheries can avoid overfishing" (Hilborn et al. 2023), the direct consequences of towing gear along the seabed extend well beyond the amount of fishery landings. Measures of impact that rely solely or primarily on the biomass, total catches or landings of the targeted organisms fail to account for the kind and degree of damage caused to marine communities (both human and biotic) by bottom trawls.

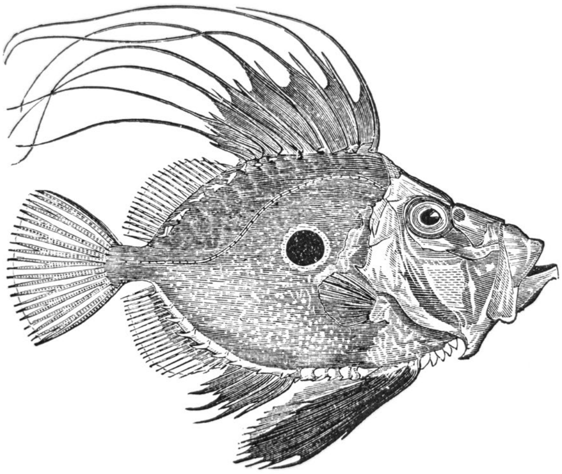
In a phenomenon aptly described as "the illusion of plenty" (Erisman et al. 2011), total landings—in terms of realized financial value, calories, protein, or just biomass—may be judged stable despite dramatic ecosystem shifts. Species turnover or changes in fishing behaviour and technology can help maintain what appear to be steady catch levels, or (perhaps more relevant) enable continued profitable returns on investment. Meanwhile, much of the former biodiversity may be annihilated, with coral, sponge and oyster reefs gone, large predators gone, and especially resilient species having replaced those that are most vulnerable to trawling.

For example, in the Adriatic Sea, one of the areas most exposed to bottom trawling, worldwide, and one with the worst seabed status (Pitcher et al. 2022; Hilborn et al. 2023), trawl catches have remained largely stable for decades (in terms of biomass) after damage caused primarily by trawling resulted in a major regime shift (Fortibuoni et al. 2017; Lotze et al. 2011; Sguotti et al. 2022). Vulnerable species have been lost (e.g. elasmobranchs; Fortibuoni et al. 2010; Ferretti et al. 2013), with rich three-dimensional habitats being turned into flattened plains that trawlers continue to exploit.

Advocates of bottom trawling emphasize the critical importance of this fishing métier and the massive amounts of seafood that trawlers provide for consumption by humans, farmed animals or other non-wild organisms (Suuronen et al. 2020; Naylor et al. 2021). Whether one bends toward the pro- or anti-trawl side, it is obvious and unquestionable that trawling provides food and employment for millions of people and, consequently, any attempt to mitigate its impacts must seriously consider the social and economic implications. The identification of alternative food sources that are healthy, accessible and palatable as well as economically viable, socially acceptable and environmentally sustainable is, indeed, one of the greatest challenges facing humanity (Tilman and Clark 2014; Krishna Bahadur et al. 2018).

Still, the quantity of seafood being produced can no longer justify or excuse the damage that bottom trawling causes to life-supporting systems. The use of fishing gear that is less destructive, combined with the creation of more areas where fishing is either not allowed or is at least effectively regulated, can bring longer-term benefits in the form of healthier, more diverse and resilient marine ecosystems that sustain local fishing communities and contribute to global biodiversity, sustainability and climate targets (Roberts et al. 2024).

Less destructive fisheries may not immediately produce as much seafood as bottom trawling. In the longer term, however, measures to "transform fisheries to protect marine life and support society" such as those proposed by Roberts et al. (2024) offer hope—and a variety of long-envisaged alternatives for "feeding the world" are quickly coming within reach.

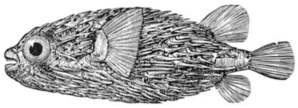


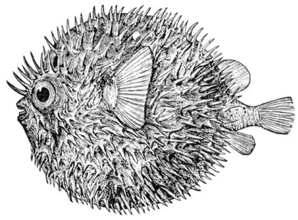
"... the great and long iron of the wondryechaun runs so heavily and hardly over the ground when fishing that it destroys the flowers of the land below the water there, and also the spat of oysters, mussels and other fish upon which the great fish are accustomed to be fed and nourished"

From a petition presented to King Edward III of England in 1376, which concerned something the petitioners called a wondryechaun—an object of amazement. It was one of the earliest beam trawls.

(Roberts 2007)

Glossary





Technical terms

Some of the technical terms used in this report, related to fisheries generally or trawl gear in particular, are described below.

A-frame	Rigid structure for net lifting installed at the stern
Aquaculture	The farming of aquatic organisms
Beam	Rigid beam which holds the mouth of the net open horizontally
Bobbins	Solid rubber wheels used as groundgear on some trawls
Bow	The front end of a ship or boat
Bridle	Wire connecting trawl doors to the net (= sweeps)
Bridle angle	Angle of the bridle compared to the direction of towing (sweep angle)
Bunt	The end of a net that retains the catch
Buoy	Float for marking the position of fishing gear
Bycatch	Caught organisms that are not the target species (or size)
Capture fisheries	The harvesting of naturally-occurring organisms in both marine and freshwater environments
Catch	Caught organisms that are the target species (or size)
Chain bridle	Chain on the leading edge of a beam trawl
Clump weight	Chain weight used in multi-rig and midwater trawling
Codend	The end of a trawl net that retains the catch
Demersal organisms	Organisms living on or near the seabed
Demersal gear	Fishing gear that is operated on or close to the seabed
Discards	The unwanted and unmarketed portion of the catch, which can either be discarded at sea, or landed (and possibly discarded afterwards)
Doors	See “otter boards”
Electric pulse trawling	See “pulse trawling”
Epipelagic organisms	Organisms living in the upper portion of the water column
Flying doors	See “semipelagic otter boards”
Footrope	Lower frame rope of a net (= groundrope)
Footprint	“The area of seabed trawled at least once in a specified region and time period, with area trawled determined from gear dimensions and tow locations” (Amoroso et al. 2018)
Gillnet	A sheet of thin netting, hung vertically in the water, to capture organisms by enmeshing them, usually by their gills
Groundfish	Collective term for target species of demersal organisms
Groundgear	Part of a net designed to be in contact with the seabed, to which the fishing line is attached
Groundrope	Lower frame rope of a net (= footrope)
Headrope	Upper frame rope of a net (= headline)

Headline	Upper frame rope of a net (= headrope)
Landings	The portion of the catch that is brought to shore
Lazy line	A rope attached to the codend, which helps bring the codend on board when full
Mesh	One of the enclosed spaces bounded by twine in a piece of netting
Mesh size	Distance between two opposite knots bounding the same mesh
Midwater	The intermediate portion of the water column, between seabed and surface (= mid-water)
Mobile gear	Fishing gear that is moved through the water by a vessel
Near-bottom otter boards	See "semipelagic otter boards"
Net sonde cable	Cable used to tow an acoustic sonde which transmits operational information on a trawl net
Otter boards	Shearing devices that hold open the mouth of a trawl net
Outriggers	Booms on either side of a beam trawler from which the nets are towed
Passive gear	See "static gear"
Pelagic fish	Fish living in midwater
Pot	Enclosure that attracts fish or other marine organisms through one or more entrances that allow entry but prevent or retard escape; pots are usually set on the seabed with bait, and connected by a rope to a floater
Pulse trawling	Trawling with gear using electric current to stun and startle fish away from the seabed and into the net
Rockhoppers	Solid and relatively large wheels used as groundgear on some trawls
Selectivity	How "selective" fishing gear is in terms of catching only the targeted species or size
Semipelagic otter boards	Otter boards that "fly" or "jump" over the seabed
Shoes	Steel frames used to support the beam on a beam trawl
Static gear	Fishing gear set in the water to wait for marine organisms to swim or move into it; in some static gear, marine organisms are enticed into the gear by using bait
Static net	Fishing net set in the water to wait for marine organisms to swim or move into it
Stern	The rearmost part of a ship or boat
Stern trawler	Trawler on which gear is handled over the stern
Sweep	Wire connecting trawl doors to the net (= bridle)
Sweep angle	Angle of sweep compared to the direction of towing (bridle angle)
Swept area ratio	"The total area swept by trawl gear over a defined time period (usually one year) divided by the total seabed area at a defined spatial scale (usually from grid cell to region)" (Amoroso et al. 2018)
Target species	Specific type of organism that gear is designed to catch
Tickler chain	Chain towed ahead of the ground gear to disturb marine organisms on the seabed
Towed gear	Fishing gear that is dragged through the water
Trammel net	A three-layer sheet of thin netting, hung vertically in the water, to capture marine organisms by enmeshing them
Trap	See "pot"
Trawl door	See "otter board"
Warp	Wire used for towing fishing gear
Wings	Ends of the trawl net, at both sides of the net mouth

Acronyms and abbreviations

AIS	Automatic Identification System
BRD	Bycatch Reduction Device
CBD	Convention on Biological Diversity
CMS	Convention on Migratory Species
CO_2eq	Carbon dioxide equivalent, a measure used to compare the emissions from various greenhouse gases on the basis of their global warming potential, by converting amounts of other gases to the equivalent amount of carbon dioxide with the same global warming potential
EC	European Commission
EEZ	Exclusive Economic Zone
EM	Electronic Monitoring
EU	European Union
FAD	Fish Aggregation Device
FAO	Food and Agriculture Organization of the United Nations
FRA	Fisheries Restricted Area
GFCM	General Fisheries Commission for the Mediterranean
GHG	Greenhouse Gases
GPS	Global Positioning System
ICES	International Council for the Exploration of the Sea
IEZ	Inshore Exclusion Zone
IMO	International Maritime Organisation
IUCN	International Union for Conservation of Nature
IUU	Illegal, Unreported, Unregulated (fishing)
IWC	International Whaling Commission
LED	Light-Emitting Diode
MPA	Marine Protected Area
MSC	Marine Stewardship Council
MSY	Maximum Sustainable Yield, “the highest average catch that can be continuously taken from an exploited population (= a stock) under average environmental conditions” (Tsikliras and Froese 2019)
NGO	Non-Governmental Organisation
NOAA	National Oceanic and Atmospheric Administration (US)
SAR	Synthetic Aperture Radar
SLED	Sea Lion Exclusion (or Excluder) Device
TAC	Total Allowable Catch
TED	Turtle Exclusion (or Excluder) Device
UN	United Nations
UNEP	United Nations Environment Program
VMS	Vessel Monitoring System
WTO	World Trade Organization

Species names

We list the common and scientific names of species mentioned in this report. In the main text, scientific names are indicated only when the species name appears for the first time.

Plants

Mediterranean seagrass	*Posidonia oceanica*
(also known as Mediterranean tapeweed, Neptune grass)	

Molluscs

horned octopus	*Eledone cirrhosa*
Japanese flying squid	*Todarodes pacificus*
purple dye murex	*Bolinus brandaris*

Crustaceans

Norway lobster	*Nephrops norvegicus*
brown tiger prawn	*Penaeus esculentus*
ocean shrimp	*Pandalus jordani*
deep-water rose shrimp	*Parapenaeus longirostris*

Osteichthyes

John dory	*Zeus faber*
large-scaled gurnard	*Lepidotrigla cavillone*
red mullet	*Mullus barbatus*
surmullet	*Mullus surmuletus*
Atlantic herring	*Clupea harengus*
Atlantic mackerel	*Scomber scombrus*
Norway pout	*Trisopterus esmarkii*
European anchovy	*Engraulis encrasicolus*
European pilchard	*Sardina pilchardus*
vendace	*Coregonus albula*
Pacific halibut	*Hippoglossus stenolepis*
yellowfin sole	*Limanda aspera*
common sole	*Solea solea*
European plaice	*Pleuronectes platessa*

Elasmobranchs

whale shark	*Rhincodon typus*
basking shark	*Cetorhinus maximus*
smooth-hound	*Mustelus mustelus*
picked dogfish	*Squalus acanthias*
blackmouth catshark	*Galeus melastomus*
lesser spotted dogfish	*Scyliorhinus canicula*
velvet belly	*Etmopterus spinax*
southern eagle ray	*Myliobatis goodei*
lesser guitarfish	*Zapteryx brevirostris*

blotched sand skate	*Psammobatis bergi*
common stingray	*Dasyatis pastinaca*
apron ray	*Discopyge tschudii*
zipper sand skate	*Psammobatis extenta*

Turtles

leatherback turtle	*Dermochelys coriacea*
loggerhead turtle	*Caretta caretta*
green turtle	*Chelonia mydas*
hawksbill turtle	*Eretmochelys imbricata*
Kemp's ridley turtle	*Lepidochelys kempii*
olive ridley turtle	*Lepidochelys olivacea*
flatback sea turtle	*Natator depressus*
Nile soft-shelled turtle	*Trionyx triunguis*

Birds

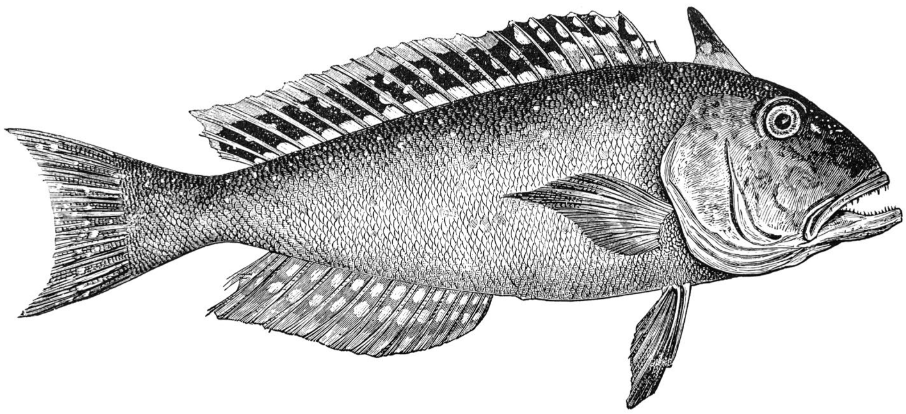
king penguin	*Aptenodytes patagonicus*
Magellanic penguin	*Spheniscus magellanicus*
gentoo	*Pygoscelis papua*
white-capped albatross	*Thalassarche steadi*
Salvin's albatross	*Thalassarche salvini*
Buller's albatross	*Thalassarche bulleri*
black-browed albatross	*Thalassarche melanophris*
shy albatross	*Thalassarche cauta*
white-chinned petrel	*Procellaria aequinoctialis*
southern giant petrel	*Macronectes giganteus*
northern giant petrel	*Macronectes halli*
Cape petrel	*Daption capense*
sooty shearwater	*Puffinus griseus*
Cape gannet	*Morus capensis*

Cetaceans

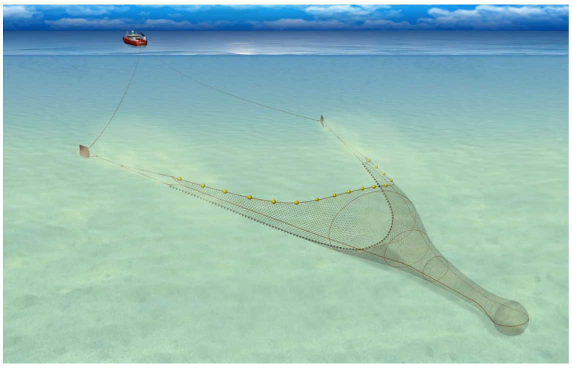
North Atlantic right whale	*Eubalaena glacialis*
humpback whale	*Megaptera novaeangliae*
killer whale	*Orcinus orca*
vaquita	*Phocoena sinus*
harbour porpoise	*Phocoena phocoena*
Dall's porpoise	*Phocoenoides dalli*
Indo-Pacific finless porpoise	*Neophocaena phocaenoides*
Risso's dolphin	*Grampus griseus*
pilot whales	*Globicephala* spp. (including *G. macrorhynchus* and *G. melas*)
franciscana	*Pontoporia blainvillei*
Hector's dolphin	*Cephalorhynchus hectori*
Māui dolphin	*Cephalorhynchus hectori maui*
common dolphin	*Delphinus delphis*
dusky dolphin	*Lagenorhynchus obscurus*
common bottlenose dolphin	*Tursiops truncatus*

Pinnipeds

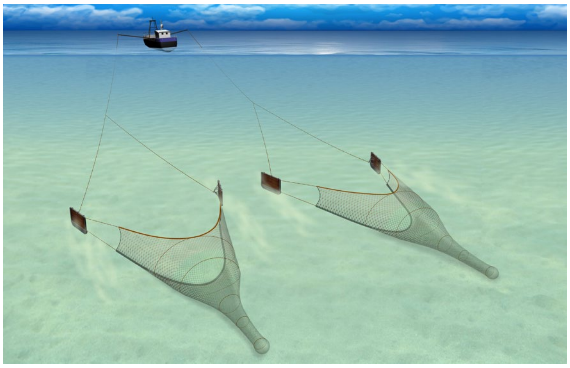
walrus	*Odobenus rosmarus*
northern elephant seal	*Mirounga angustirostris*
Steller sea lion	*Eumetopias jubatus*
New Zealand sea lion	*Phocarctos hookeri*
South American sea lion	*Otaria byronia* (= *O. flavescens*)
South American fur seal	*Arctocephalus australis*
Australian fur seal	*Arctocephalus pusillus doriferus*
New Zealand fur seal	*Arctocephalus forsteri*
northern fur seal	*Callorhinus ursinus*
spotted seal	*Phoca largha*
ringed seal	*Pusa hispida*
Hawaiian monk seal	*Neomonachus schauinslandi*



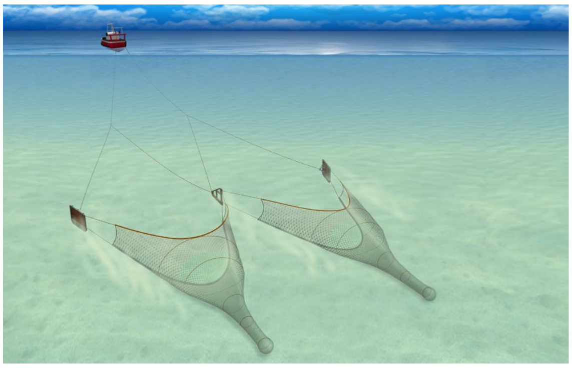
Trawl gear examples



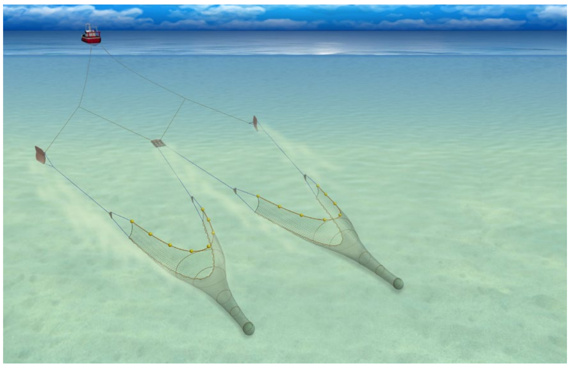
Bottom otter trawling



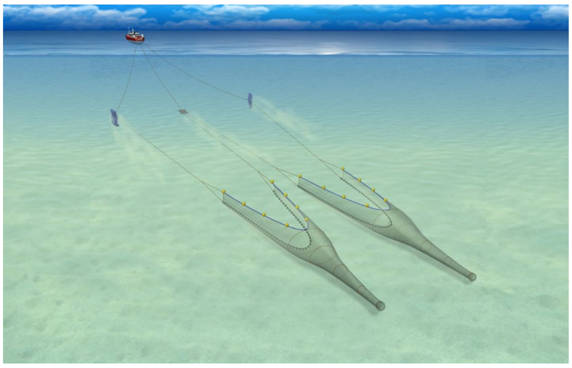
Bottom otter trawling (out-rig)



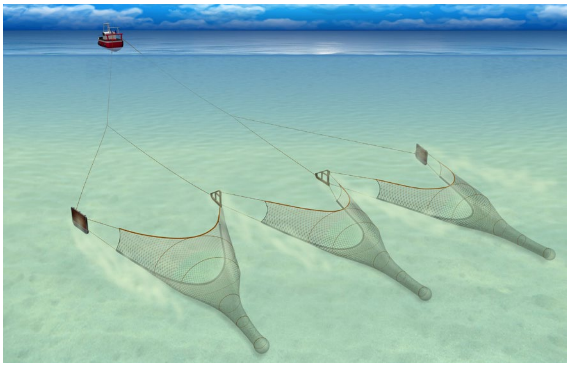
Bottom otter trawling (twin-rig)



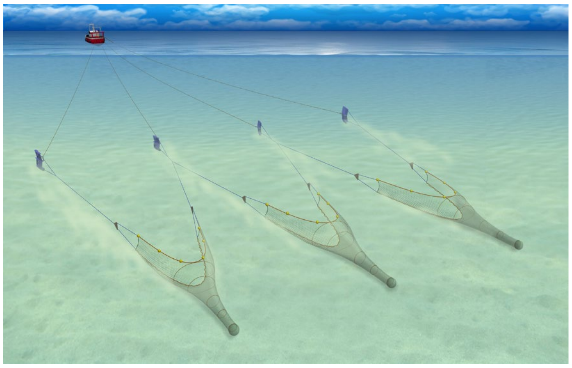
Bottom otter trawling (twin-rig, two warp)



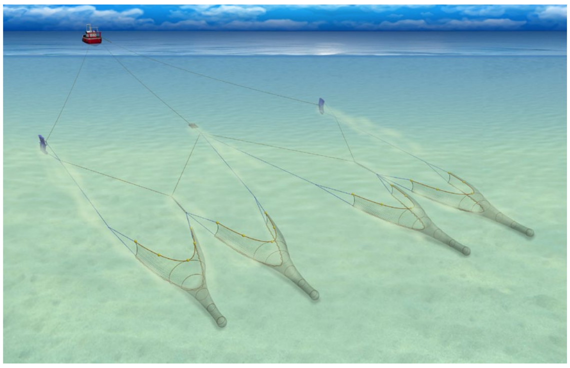
Bottom otter trawling (twin-rig, three warp)



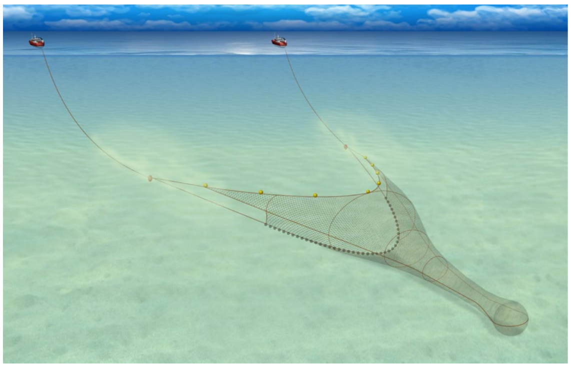
Bottom otter trawling (triple-rig)



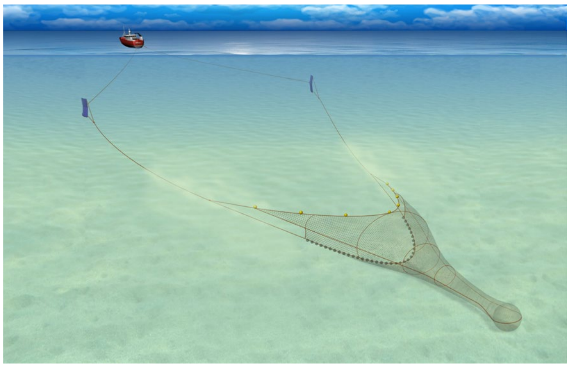
Bottom otter trawling (triple-rig, four warps)



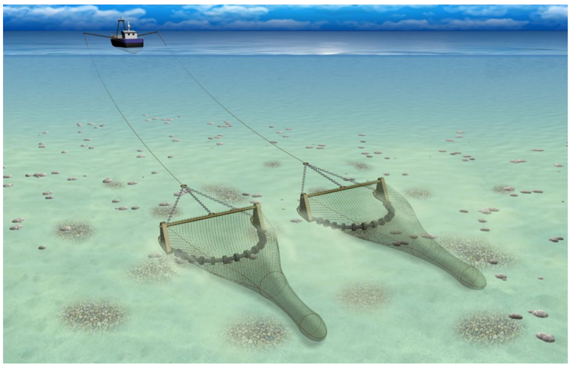
Bottom otter trawling (quad-rig)



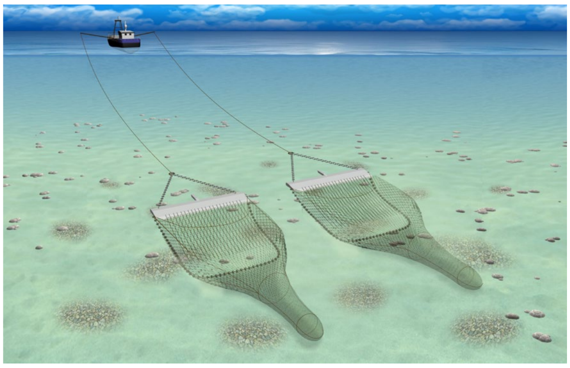
Bottom pair trawling



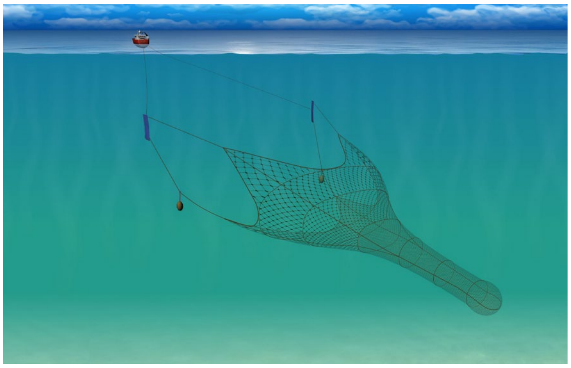
Semipelagic trawling



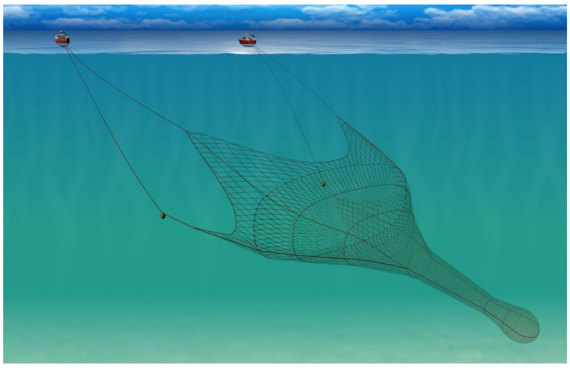
Beam trawling



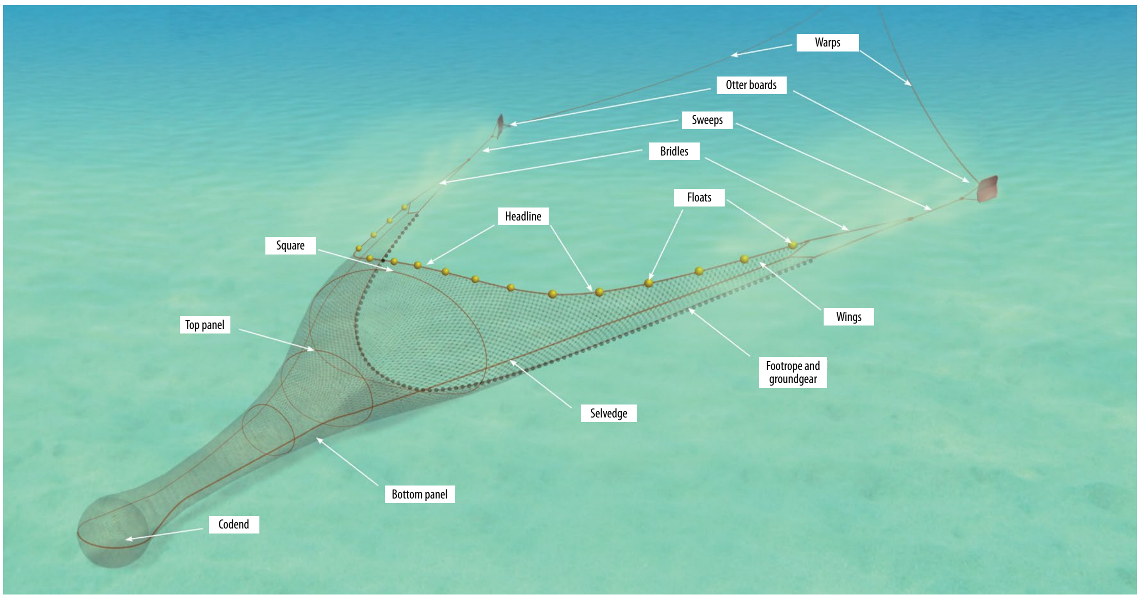
Pulse trawling



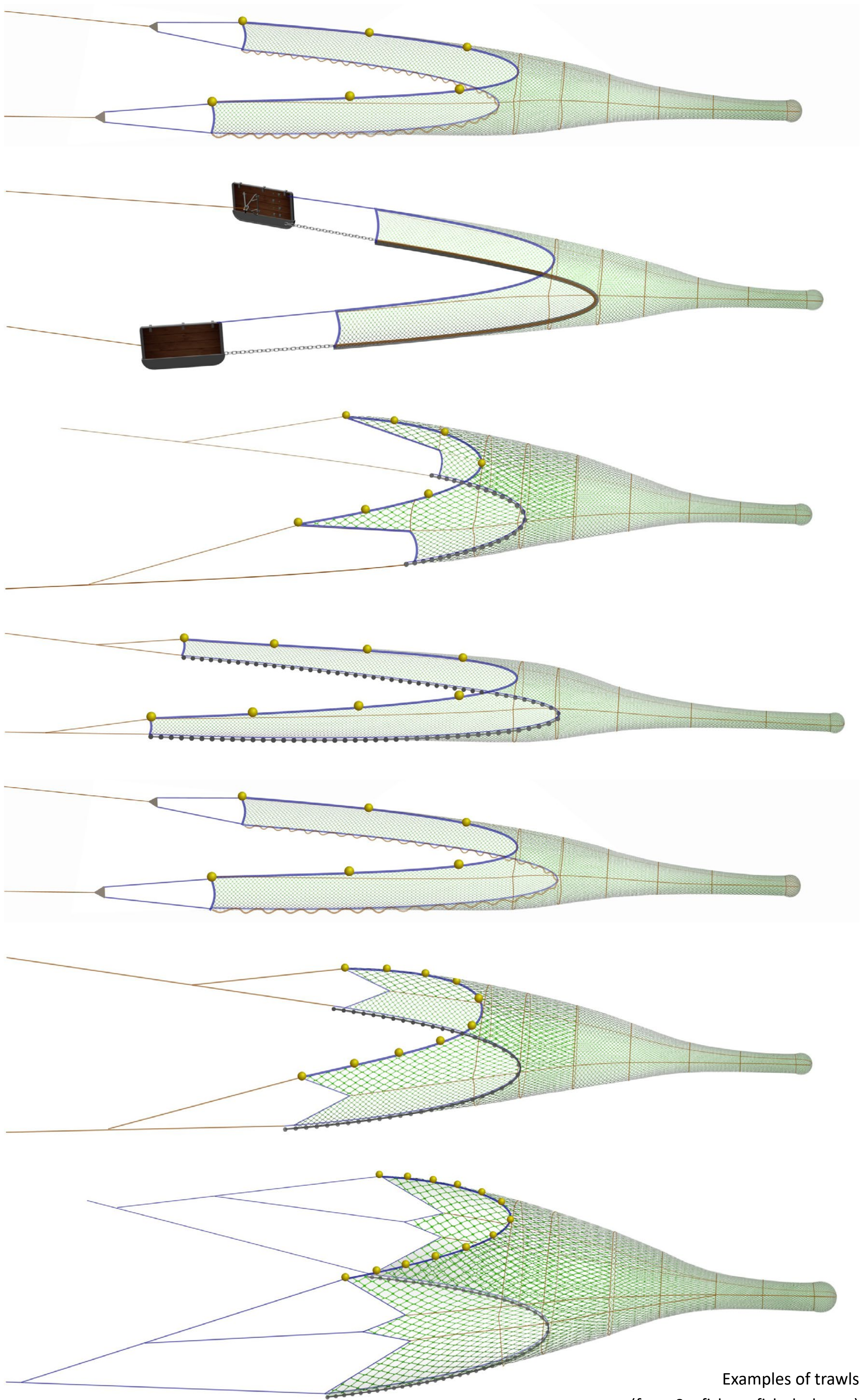
Single-boat midwater otter trawling



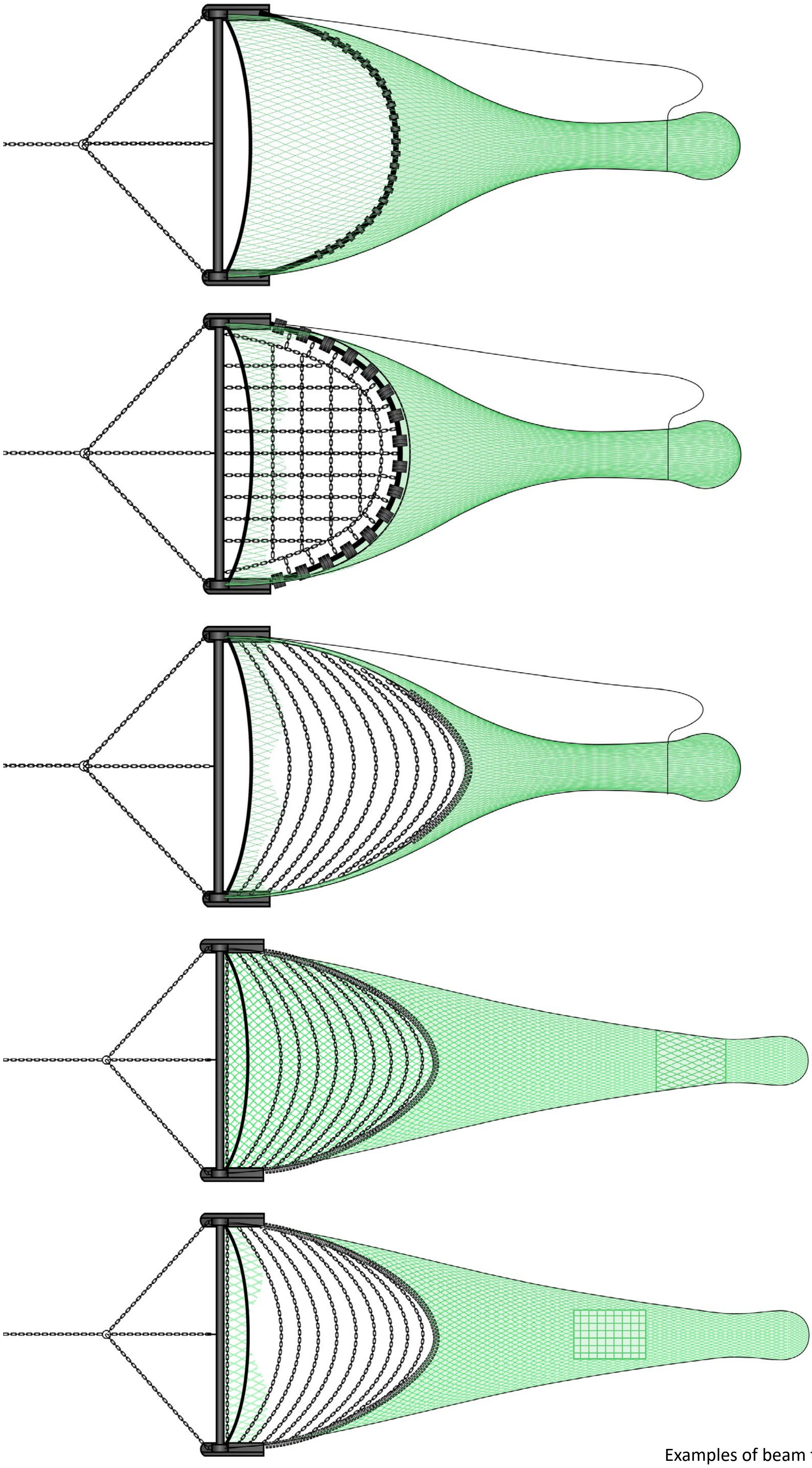
Midwater pair trawling



Trawl gear components (all images from Seafish seafish.dash.app)



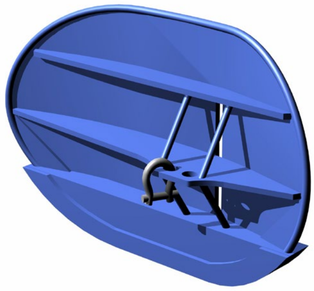
Examples of trawls
(from Seafish seafish.dash.app)



Examples of beam trawls
(from Seafish seafish.dash.app)

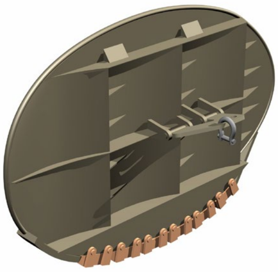
Flat door of wood/steel

V-door

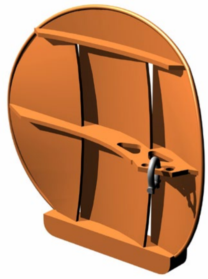


Oval cambered door

Cambered V-door



Foiled door



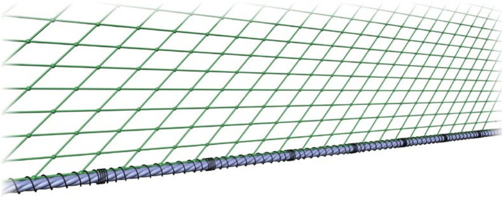
Round, foiled door

Flat cambered door

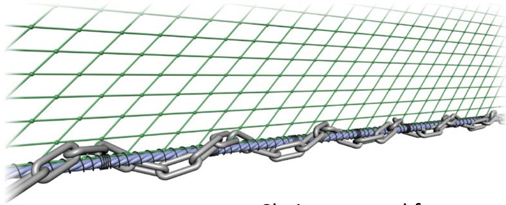
Multifoiled door

Semipelagic door

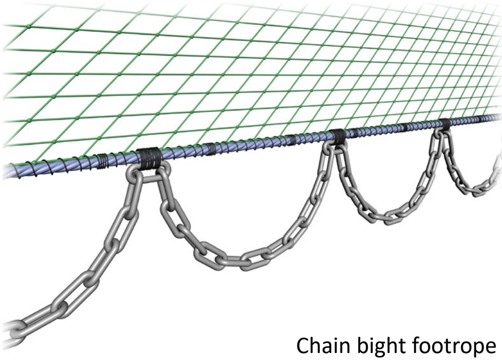
Examples of trawl doors / otter boards
(from Seafish seafish.dash.app)



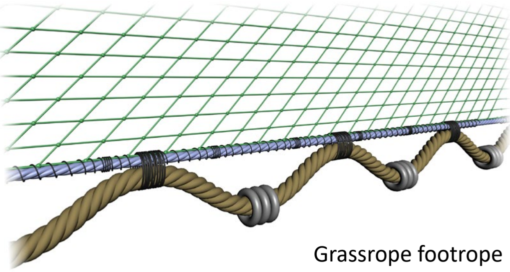
Bare footrope



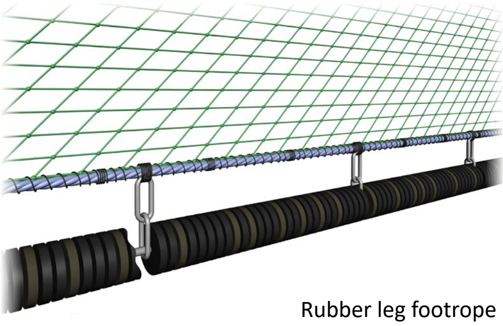
Chain-wrapped footrope



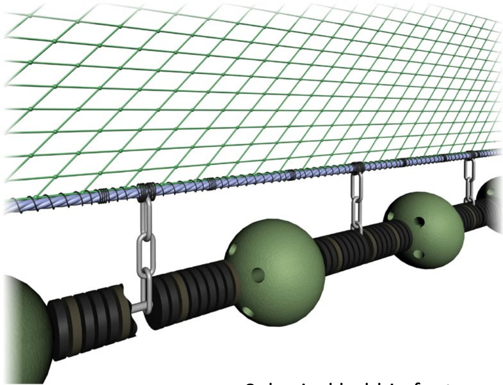
Chain bight footrope



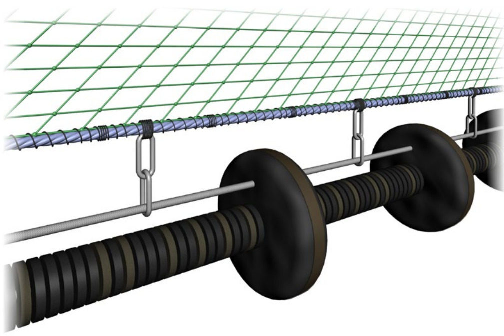
Grassrope footrope



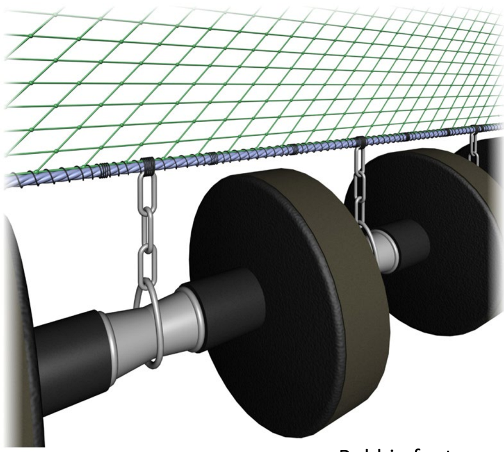
Rubber leg footrope



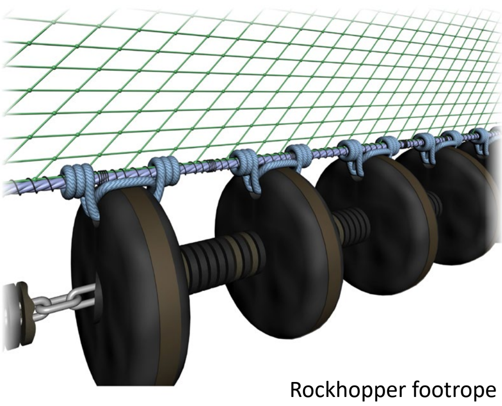
Spherical bobbin footrope



Large disc footrope



Bobbin footrope



Rockhopper footrope

Examples of trawl groundgear
(from Seafish seafish.dash.app)

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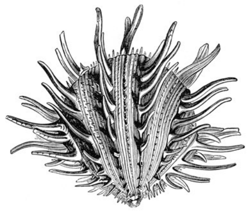

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