BOTTOM TRAWLING: DIRECT AND INDIRECT IMPACTS ON CETACEANS, WITH A FOCUS ON THE MEDITERRANEAN SEA

Issue: Bottom trawling

Background

The ACCOBAMS Secretariat received the "Bottom trawling: direct and indirect impacts on cetaceans, with a focus on the Mediterranean Sea" from OceanCare.

BOTTOM TRAWLING: DIRECT AND INDIRECT IMPACTS ON CETACEANS, WITH A FOCUS ON THE MEDITERRANEAN SEA

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Preamble

At its 15th meeting, the Scientific Committee of ACCOBAMS concluded that "The nexus of activities that contribute to climate change and have an impact on cetaceans should also be further explored" (Conclusion 34, p. 19; ACCOBAMS 2023). In that regard, we point out and summarize the effects of bottom trawl fisheries – largely based on information contained in a report[1] prepared for OceanCare (Bearzi et al. 2024a). Bottom trawling has impacts on cetaceans and their habitat, and it also contributes to climate change. Most of these impacts are either likely to occur or have been documented to occur in Mediterranean Sea areas where bottom trawling is permitted.

What is bottom trawling?

Bottom trawling is a type of fishing characterized by the active towing of nets along the seabed by a moving boat. Bottom trawls 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 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; He et al. 2021).

Midwater trawls are a separate category of towed nets that normally do not touch the seabed. However, the term “midwater” can be misleading, considering that some midwater trawls do not fish only in midwater. For instance, the pollock fishery in Alaska uses midwater gear that is in contact with the seabed roughly half the time (Hilborn et al. 2023; Stratton and Wilson 2023). In the Adriatic Sea, benthic taxa were caught in 12% of 496 midwater trawl hauls, indicating net proximity with the seabed (Casale et al. 2004).

Trawl vessels vary greatly in size, ranging from small boats fishing in shallow inland channels and coastal waters to large factory ships up to 150 m long, with fleets that can operate in deep offshore waters and stay at sea for weeks or months (Bakkala et al. 1979; Clark et al. 2016; FAO 2021). While the smallest trawlers may only catch enough fish to feed a family or a small community, large factory freezer trawlers can process up to 350 tonnes of fish a day and store up to 7,000 tonnes of catch (Kose 2010).

Bottom trawling has become a cornerstone of the global food supply, accounting for more than a 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 (Steadman et al. 2021). Bottom trawling, however, has long been known to be detrimental to marine life. It has been 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 livelihoods (Roberts 2007; Pitcher and Lam 2015; De Nicolò 2016, 2018). The introduction of steam and diesel engines

[1] The report can be downloaded from: http://www.oceancare.org/trawlsupremacy

(in the 1830s and 1930s, respectively) led to 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.

The harmful nature of bottom trawling

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. Countless scientific studies, encompassing decades of fishery research, have documented the harmful nature of bottom trawling. Table 1 provides a summary of the main direct and indirect impacts of trawling and lists key references. The largely unselective removal of organisms reduces the biomass, diversity and complexity of benthic communities. In addition to the target species, most types of trawl gear take unwanted species, such as threatened elasmobranchs, turtles, birds and mammals. The action of trawl gear on the seabed also causes dramatic mechanical and chemical alterations, compromising the seabed’s functionality and productivity.

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 in intensively trawled areas such as the Adriatic Sea (Barausse et al. 2011; Pitcher et al. 2022; Hilborn et al. 2023; Fortibuoni et al. 2010, 2017; Lotze et al. 2011; Sguotti et al. 2022) where, for instance, elasmobranchs declined by >94% across 60 years, and 11 of 33 studied species could no longer be detected (Ferretti et al. 2013).

Table 1. Some of the main direct and indirect impacts of trawling (from Bearzi et al. 2024a, with additions).

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, 2024a, 2024b; 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; 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; Scheinin et al. 2014; Wakefield et al. 2014; 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; Izquierdo-Serrano et al. 2022; 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; Robertson and Chilvers 2011; 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. 2016; 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

Apart from the biological impacts listed above, recent studies indicate that bottom trawling has a considerable carbon footprint, with high direct and indirect greenhouse gas emissions contributing to climate disruption. The carbon footprint of bottom trawling is summarized in a section below.

Because of all these impacts, bottom trawling has often been characterized as a “destructive” practice—though use of the term destructive has been inconsistent and controversial (Willer et al. 2022). For instance, Watling and Norse (1998) likened bottom trawling to forest clearcutting. 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, Halpern et al. (2019, Table S3, p. 10) classified bottom trawling as being destructive and having high bycatch. 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”. In October 2024 the United Nations compiled 400 inputs on marine conservation received from stakeholders based in 90 countries, which characterized overfishing and bottom trawling as destructive practices contributing to biodiversity loss and ecosystem degradation (UN 2024).

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 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. In recent years, several campaigns were conducted to ban the practice within protected areas and inshore zones[2]. 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.

Impacts on cetacean habitat

<u>Physical and chemical alteration of the seabed</u>

The scraping and ploughing of the seabed changes 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 can 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. 2021a). A 12-m long bottom otter trawler operating on a muddy Baltic seabed displaced an estimated 1,000 m^3 (500 tonnes) of sediment and suspended 9.5 tonnes of sediment per kilometre of track (Bradshaw et al. 2021a). 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. 2021a). Dissolved methane concentrations were elevated in the water for at least 20 hours, and two hours after trawling there was a pulse of dissolved nitrogen, phosphorus, and manganese 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. 2021a).

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. 2016). By combining the worldwide distribution of soft sediments on continental shelves with estimates of bottom trawling intensity, Oberle et al. (2016) 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.

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

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. 2014a; 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).

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.

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).

Abandoned, lost and discarded trawl 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; Kuczenski et al. 2022). Fishing gear that is abandoned, lost or discarded at sea has environmental, social and economic impacts that go beyond those of plastic pollution, as such 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).

Richardson et al. (2022) estimated that, globally, nearly 220 km^2 of trawl netting was 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 (by all vessels combined) is small when compared to estimates of more than 75,000 km^2 of purse seine netting, nearly 3,000 km^2 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. More than 50 metric tons of abandoned, lost, or otherwise discarded fishing gear accumulates annually in the Northwestern Hawaiian Islands (Dameron et al. 2007), and entanglement in such gear is a major threat to the endangered Hawaiian monk seal *Neomonachus schauinslandi*, one the rarest pinnipeds in the world (Baker et al. 2024). 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).

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 (Kebe and Ndiaye 1993; Menon et al. 2016; Scholtens 2016; Kularatne 2020). 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).

Impact on seabed organisms

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). 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.

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. 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 highly vulnerable to trawling (Collie et al. 2017).

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).

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).

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 preceded the baseline years of the 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 influence management decisions.

Impacts on target and nontarget 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, 2024a, 2024b; Foley et al. 2011; Hiddink et al. 2011; Dransfeld et al. 2013; Barausse et al. 2014; Johnson et al. 2015; Novaglio et al. 2020). Some authors contend that well-managed bottom trawl fisheries don't necessarily lead to the decline of target species (Fernandes and Cook 2013; Zimmermann and Werner 2019; Hilborn et al. 2021, 2023). Bottom trawling, however, causes impacts that go well beyond those on the target species and populations (Bearzi et al. 2024a).

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 can 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 in bottom trawls they 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; Edelist et al. 2013; FAO 2022). 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).

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). Non-edible 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).

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 midwater trawls (Zeeberg et al. 2006; Bonanomi et al. 2018). 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 2022). Similarly, almost 93% of the elasmobranchs caught in bottom otter trawls in the Aegean Sea (eastern Mediterranean) are discarded (Damalas and Vassilopoulou 2011). 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).

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*, which is 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).

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). In this region, 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 about 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).

Direct impacts on cetaceans

<u>Behavioural effects</u>

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 odontocete cetaceans (Bonizzoni et al. 2022), pinnipeds (Tixier et al. 2021), elasmobranchs (Ferretti et al. 2013; Mitchell et al. 2018) and seabirds (Oro and Ruiz 1997; Karris et al. 2018).

At least 19 odontocete cetacean species are known to scavenge and forage 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). In the Mediterranean Sea, common bottlenose dolphins *Tursiops truncatus* are the species most frequently involved in these interactions (Table 2).

Table 2. Some reports of odontocete species interacting with bottom trawlers in the Mediterranean Sea.

Species	Trawl type	Vessel size (m)	Location	References
Delphinus delphis	Bottom otter	17–22	Spain: Alborán Sea	Giménez et al. 2021
	Bottom otter	Unreported	Greece	Miliou et al. 2018
	Bottom otter	13–24	Israel	Kerem et al. 2012, Brand et al. 2019
Sousa plumbea	Bottom otter	7.5	Turkey: Mersin Bay	Ozbilgin et al. 2018
Tursiops truncatus	Bottom otter	17–22	Spain: Alborán Sea	Giménez et al. 2021
	Bottom otter	16–23	Spain: Balearic Islands	Gonzalvo et al. 2008
	Bottom otter	Unreported	Algeria: off Ghazaouet	Laurent 1991
	"Trawlers"	Unreported	Italy: Ligurian Sea	Bellingeri et al. 2011, Salvioli et al. 2016
	Bottom otter	Unreported	Italy: Sardinia	Lauriano et al. 2004
	Bottom otter	<16	Italy: Sardinia	Rako-Gospić et al. 2020
	Bottom otter	Unreported	Italy: Sicily Channel	Boldrocchi et al. 2013
	Bottom otter	20	Italy: Lampedusa	Pace et al. 2012
	Bottom otter	24	Italy: Adriatic Sea	Di Nardo et al. 2023
	Beam, Bottom otter	9–29	Italy: Adriatic Sea	Bonizzoni et al. 2021, 2023
	Bottom otter	9–15	Slovenia and Italy: Adriatic Sea	Genov et al. 2008, 2019
	Bottom otter	<12	Croatia: Adriatic Sea	Bearzi et al. 1999, Rako-Gospić et al. 2017
	Bottom otter	Unreported	Croatia: Adriatic Sea	Pleslić et al. 2020
	Bottom otter	12–16, <24	Montenegro: Adriatic Sea	Miočić-Stošić et al. 2020
	Trawlers "including beam"	>12	Montenegro: Adriatic Sea	Rudd et al. 2022
	Bottom otter	Unreported	Greece	Miliou et al. 2018
	Bottom otter	Unreported	Turkey: Antalya Bay	Miliou et al. 2018
	Bottom otter	13–24	Israel	Scheinin et al. 2014, Brand et al. 2019, Galili et al. 2023, Zuriel et al. 2023

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 and scavenge 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, as well as by the negative impacts of exposure to noise and depletion of their natural prey. Therefore, foraging behind trawlers can either help the predators involved survive in an overexploited and degraded marine environment, or ultimately prove maladaptive by decreasing fitness, 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 trawl gear), 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 scavenging and foraging behind trawlers may, at least in some contexts, reflect a lack of options: the animals may simply be exploiting the resources that remain available to them (Bearzi et al. 2019).

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).

Information on the underwater noise produced by trawlers is scant and methodologically inconsistent (e.g. De Robertis and Wilson 2006; Rako et al. 2013; Daly and White 2021). 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).

<u>Bycatch</u>

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). Cetaceans can 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). 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 sublethal effects of cetacean bycatch in fishing gear are almost always overlooked (Wilson et al. 2014).

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 (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 *Delphinus delphis*), 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 (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 bottom trawlers (Rayment and Webster 2009) and are bycaught at least occasionally (Starr and Langley 2000). 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: in one study, 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, in this area, as many as 9,040 (95% CI 6,640–13,300) common dolphins die 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 of 2021/2022.

Reports of cetacean bycatch in the Mediterranean have been relatively few, if one considers the magnitude of trawling there. That may be due either to low levels of bycatch, or to low levels of monitoring and reporting. In this region, however, it is probably also, at least in part, a consequence of the decline of species such as common dolphins (Bearzi et al. 2003; Bearzi and Genov 2022), which are known to be especially susceptible to bycatch in trawl gear.

In the Alborán Sea, observers onboard bottom otter trawlers reported no bycatch of cetaceans across 70 hauls between August 2000 and October 2001 (Giménez et al. 2021). Interviews with bottom trawl fishers operating in the northwestern Mediterranean (Valencia region, 148 trawlers) reported monthly bycatch rates of 0.01 dolphins per vessel. In that area, extrapolation of interview-based information to the entire trawl fishery resulted in bycatch estimates of 23 delphinids per year (95% CI 7–39). The reported species included common bottlenose dolphins and striped dolphins *Stenella coeruleoalba* (Izquierdo-Serrano et al. 2022). In the central and northern Adriatic Sea, dolphin bycatch by the Italian fleet of midwater pair trawlers was assessed in the context of an extensive monitoring programme (Bonanomi et al. 2022), but there is no information on bycatch rates in bottom otter and beam trawls. Between 2006 and 2019, 19 common bottlenose dolphins were bycaught in midwater pair trawls, ranging between zero and three animals per year, resulting in an estimated bycatch rate of 0.00075 individuals per haul (Bonanomi et al. 2022). Off Israel, reports of common bottlenose dolphin mortality in bottom otter trawls included 7 individuals "in the last two years" before 1995 (Goffman et al. 1995), 8 individuals between 1996 and 2008 (Scheinin et al. 2014) and 26 individuals between 2000 and 2010 (Kerem et al. 2014).

Mitigation of bycatch in trawl gear

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 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 a readily available 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 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).

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 be 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).

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).

Impacts on climate

Climate disruption caused by human activities has global consequences that include loss of biodiversity and catastrophic ecological changes (Cheng et al. 2019; Halpern et al. 2019; IPCC 2019; Bradshaw et al. 2021b; Georgian et al. 2022). Warming of both surface and deep waters has long been observed in the Mediterranean Sea, with biological responses including disease outbreaks, shifts in faunal distribution and the spread of invasive species (Lejeusne et al. 2010; Lionello and Scarascia 2018). These effects are expected to increase in the coming decades, as water temperatures continue to rise and species are either displaced, replaced by more tolerant and tropical ones, or extirpated. Rising sea temperatures and acidification linked to climate change affect marine biodiversity in ways that are poorly understood, and difficult to predict for cetaceans (Whitehead et al. 2008; Lacoue-Labarthe et al. 2016). Climate change may have relatively less impact on certain species (particularly those that are more resilient and mobile), while other species suffer from effects that are either direct or are mediated via changes to the ecosystem, such as reduced prey availability (Learmonth et al. 2006; Simmonds and Elliot 2009; Sousa et al. 2019; van Weelden et al. 2021). Changes in social organization were observed in some cetacean communities as a response to shifts in water temperature (Lusseau et al. 2004). Increasing sea temperatures may cause changes in species distribution and range (MacLeod 2009).

The Scientific Committee of ACCOBAMS recently noted that "the composition of most of the present marine and coastal ecosystems is expected to change under continued warming and there will be a greater risk of species extinctions – especially those with a restricted climatic distribution, those that require highly specific habitats, and/or small populations which are naturally more vulnerable to modifications in their habitats" (ACCOBAMS 2023). It was also noted that the future ability of some cetacean species to adapt to Mediterranean Sea warming and other shifts related to the global climate breakdown is of concern. For example, the only known prey species of the fin whale *Balaenoptera physalus* in the Mediterranean, i.e. the Norwegian krill *Meganyctiphanes norvegica*, is living there at the thermal limit of its distribution: rising sea temperatures may lead to a decline in krill abundance and reduced food availability and quality for Mediterranean fin whales (ACCOBAMS 2023).

Carbon footprint of bottom trawling

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).

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.

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), equivalent to 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). While there have been sporadic attempts to replace diesel-powered systems in Mediterranean trawling vessels with more "sustainable" alternatives (e.g. batteries, liquid natural gas, methanol, ammonia, soybean/biodiesel/diesel blend, hydrogen; Koričan et al. 2022), these attempts have been largely inconsequential.

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, the vessels used in trap fisheries targeting crustaceans such as lobster are an exception as they can require substantial amounts of fuel (Parker and Tyedmers 2015; Hilborn et al. 2018).

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). Those authors argued, however, that a few well-managed bottom trawl fisheries had carbon footprints below those of chicken and pork production.

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 thousands of years (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. 2016; 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 those inconsistencies, 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".

While the carbon footprint of trawling in the Mediterranean Sea has not been assessed, a recent study compared different size categories of bottom trawlers and purse seiners operating in the Spanish Mediterranean (Muñoz et al.

2023). Bottom trawlers over 12 m had higher CO_2 emissions per kg of landed fish (but significantly lower profit) than all purse seine categories. Those authors also found that CO_2 released from the seabed by bottom trawling was between 3 and 10 times greater than the CO_2 buried in the seafloor by the biological pump, "potentially turning the continental shelf from a CO_2 sink to a CO_2 source" (Muñoz et al. 2023).

The Mediterranean scenario

Fishery 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 they have been declining over the past three decades, with annual landings down to 750,000 tonnes in 2015, and to an annual average of 665,053 tonnes in 2020–2021 (FAO 2023). 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).

Discard ratios varied widely depending on fishing method and geographic area (FAO 2022). 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 (2022), 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%).

Geographic intensity of trawling

FAO (2022) 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. Figures encompassing both the Mediterranean and the Black Sea indicate that "trawlers and beam trawlers" generated 16.5% of total fishery employment and 37% of total revenue[3], whereas "purse seiners and pelagic trawlers" generated 16.5% of total employment and 27% of total revenue; small-scale fisheries generated 61% of total employment and 26% of total revenue (FAO 2023).

A study of bottom trawling intensity (Amoroso et al. 2018) indicated that, of 24 worldwide regions included in the analysis, several European regions had the highest intensity of trawling. Intensively trawled regions included the Adriatic Sea (which ranked 1st), the Tyrrhenian Sea (4th), and the Aegean Sea (9th). 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, and these included the Adriatic Sea, Tyrrhenian Sea and Aegean Sea. The Adriatic Sea had by far the poorest seabed status (Pitcher et al. 2022; Hilborn et al. 2023).

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). The spatial distribution of trawlers indicated intensive trawling off the Mediterranean coasts of Spain, Italy and Greece, in the Gulf of Lions, in

[3] 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).

the Sicily Channel, and particularly in the Adriatic Sea (Merino et al. 2019). 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. Patterns were largely consistent with those obtained by Ferrà et al. (2018, 2020).

Overfishing

The Mediterranean and Black Sea (FAO area 37) is one of the world regions with 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/FMSY = 2.25; FAO 2022). According to FAO, exploitation rates have been decreasing in the past decade, and that was interpreted as indicative of improved fishery management (FAO 2023). However, Colloca et al. (2017) 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, the 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” effect (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.

Fishery subsidies

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 2021; Shen and Chen 2022; Vaughan et al. 2023). 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 these subsidies have historically contributed to 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).

In the Mediterranean Sea, the reform of harmful economic subsidies is an objective of the Mediterranean Strategy for Sustainable Development 2016-2025 (MSSD 2016), which advocated a transition from harmful to "beneficial" economic subsidies (Sumaila et al. 2010a) to help correct market distortions, promote social equity, and stimulate a more sustainable economy (Plan Bleu and UNEP/MAP 2024). In the case of subsidies provided under the EU's European Maritime and Fisheries Fund (EMFF) and earlier funding schemes, however, 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) – at times even supporting trawl fisheries that have severe environmental impacts. For instance, a part of the fishery subsidies the EMFF allocated in 2016 and 2017 was 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).

Capacity-enhancing subsidies are also used by "developed" nations to enhance their own fishing fleets in foreign waters. Skerritt et al. (2023) documented that Asian, European and North American nations provide harmful subsidies to promote their distant-water fishing fleets. For example, fishing in African waters was supported by substantial harmful subsidies originating in 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. 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). 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). In other words, 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).

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. 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 by, inter alia, 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 arduous (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 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. (2024b), “adopted resolutions often clash with the broad economic interests involved in the human uses of the marine environment, and these interests have invariably prevailed”.

Regulatory and enforcement effort varies greatly across areas, and environmental protection measures are often ineffective—to the point that the intensity of bottom trawling can be greater inside than outside some European Marine Protected Areas (Dureuil et al. 2018). 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 2023). 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 MPAs by 2030, starting with Marine National Parks by 2026, represents an important commitment[4], which may encourage other EU Member States to take similar action.

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). In November 2024, the GFCM established a FRA in the Otranto Channel, between Italy and Albania, which includes a ban on bottom fishing in an area of approximately 2,000 km^2 (mostly in waters deeper than 700 m; GFCM 2024).

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. Off Israel, Zuriel et al. (2023) reported a trawl ban in waters less than 30 or 40 m deep, depending on subarea. 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

[4] The announcement was made at the 9th “Our Ocean” Conference (Athens, Greece, 16-17 April 2024; see https://www.ourocean2024.gov.gr/commitments).

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.

Conclusions

This report summarises evidence that bottom trawling is a fundamentally destructive practice that damages marine habitats and depletes marine life. The impacts of bottom trawling on Mediterranean cetaceans are both direct (noise, bycatch, behavioural disruption) and indirect (habitat degradation, prey depletion, climate change). Habitat degradation stands out as being the principal threat posed by bottom trawling.

Bottom trawling brings not only disruption associated with the removal of target and non-target species at "unsustainable" levels, but also broad-spectrum collateral damage caused to marine food webs by the destructive and unselective fishing gear, the disposal and loss of such gear, and the high carbon footprint which contributes to climate change. While bycatch and entanglement in fishing gear is the most significant immediate threat to several cetacean species elsewhere (e.g. Brownell et al. 2019), bottom trawling in the Mediterranean is pervasive enough to seriously affect the cetaceans living in this region – not because of high rates of mortality in trawl gear, but because the quality and health of their habitat is being (and has been for a very long time) severely worsened. Habitat degradation caused by trawling can exacerbate the effects of threats originating from activities other than trawling, given that the animals' overall resilience is undermined from living in degraded, contaminated and noisy environments with less and lower-quality prey. Only the most resilient and opportunistic cetacean species (primarily common bottlenose dolphins) appear to survive within intensively trawled areas, often scavenging and foraging behind trawlers as a coping strategy (Bearzi et al. 2019, 2024b; Bonizzoni et al. 2022, 2023).

Within the Mediterranean, a striking and well-documented example is 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). Here, damage caused primarily by trawling resulted in a major regime shift (Fortibuoni et al. 2017; Lotze et al. 2011; Sguotti et al. 2022). Some vulnerable species were lost (e.g. elasmobranchs; Fortibuoni et al. 2010; Ferretti et al. 2013), and rich three-dimensional habitats (including oyster reefs; Thurstan et al. 2024a, 2024b) were turned into flattened plains that trawlers continue to exploit. The extreme habitat degradation in the Adriatic Sea doubtless contributed to the almost complete local extirpation of common dolphins and to the low diversity and abundance of cetaceans (Bearzi et al. 2004, 2024b).

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 and other non-wild organisms (Suuronen et al. 2020; Naylor et al. 2021). Whether one leans towards the pro- or anti-trawl side, it is obvious and unquestionable that trawling provides substantial food and employment. 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 does not justify the damage that bottom trawling causes to sea life generally. 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, would be a major step towards regaining healthy, resilient marine ecosystems that sustain, among many other things, diverse and abundant cetacean populations as well as thriving local fishing communities.

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For citations of most of the references in this document see:

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