Deep-water fisheries along the British Isles continental slopes: status, ecosystem effects and future perspectives

RUI P. VIEIRA[1,2,3] | CLIVE N. TRUEMAN[3] | LISA READDY[1] | ANDREW KENNY[1] | JOHN K. PINNEGAR[1,2]

[1]Centre for Environment, Fisheries & Aquaculture Science, Lowestoft Laboratory, Pakefield Road, Lowestoft, Suffolk, UK.

[2]School of Environmental Sciences, University of East Anglia, Norwich, UK.

[3]Ocean and Earth Science, University of Southampton, Southampton, UK.

Correspondence

Rui P. Vieira, Centre for Environment, Fisheries and Aquaculture Science, Pakefield Road, Lowestoft, NR33 0HT, UK

Email: rui.vieira@cefas.co.uk

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ABSTRACT

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In this paper, we revisit the state of deep-water fisheries to the west of the British Isles and aim to provide an overview on the key drivers behind community changes along continental margins. The deep-water fisheries to the west of the British Isles that extend from the shelf-slope break down to the lower slope and along banks and seamounts of the Rockall Basin, mainly target blue ling *Molva dypterygia*, roundnose grenadier *Coryphaenoides rupestris*, orange roughy *Hoplostethus atlanticus*, with by-catches of black scabbardfish *Aphanopus carbo* and tusk *Brosme brosme*. These fishing grounds experienced a long period of exhaustive exploitation until the early 2000s, but subsequently the implementation of management strategies has helped to relieve excessive fishing pressure. It is widely accepted that a better understanding of the long-term implications of disturbance is needed to understand patterns in deep-water communities and what sustainable use and exploitation of resources might look like in this context.

KEYWORDS

ecosystem disturbance, ecosystem management, food webs, modelling, north-east Atlantic Ocean

1 | IMPORTANCE OF CONTINENTAL SLOPES FOR DEEP-WATER FISHERIES

Continental slopes cover around 5.4% of the global ocean floor (Harris *et al*., 2014) and play an important role for deep-water fisheries (Koslow, *et al*., 2000). This is a highly productive environment, favourable to a broad diversity of habitats and associated fish species (Haedrich & Merrett, 1988; Gordon & Bergstad, 1992; Lorance, 1998; Priede *et al*., 2010; Campbell *et al*., 2011).

Extensive deep-water fisheries (deeper than 400 m, near the lower limits of the upper continental slope) have developed following a decline of the continental-shelf fisheries (Hopper 1995; Koslow *et al*., 2000, Thurstan *et al*., 2010). Total global catch of important deep-water fisheries reached a peak of about 3.7 Mt in 2003 (FAO, 2011; Priede, 2017), but long-term effects of exploitation caused a significant reduction of many fish stocks (Devine *et al*., 2008; Bailey *et al*., 2009; Priede *et al*., 2011; Victonero *et al*., 2018), with several species still being fished outside safe biological limits (Norse *et al*., 2012; Pauly ||& Zeller, 2016).

Depletion of deep-sea fish populations is commonly associated with long-term effects on the benthic fauna (Benn *et al*., 2010), with concerns regarding the ecosystem services they provide, such as benthic nutrient cycling (Ruhl *et al*., 2011). Fishing can have a direct effect on the continental slopes and fishing-induced changes in food web or biomass structure may also disrupt nutrient transfer *via* the active biological pump (Trueman *et al*., 2014). It may also cause damage to sensitive coral or sponge aggregations (Clark *et al*., 2016) that have been suggested to provide essential habitats for some commercial important fish (Costello *et al*., 2005; Buhl- Mortensen *et al*., 2010; Söffker *et al*., 2011; Baillon *et al*., 2012; Milligan *et al*., 2016).

In the north-east Atlantic Ocean, multispecies fisheries target *inter alia* blue ling *Molva dypterygia* (Pennant 1784), roundnose grenadier *Coryphaenoides rupestris* Gunnerus 1765 and orange roughy *Hoplostethus atlanticus* Collett 1889 (Gordon, 2001; Lorance & Dupuoy, 2001; Lorance *et al*., 2001; Gordon *et al*., 2003) and crustaceans such as Norway lobster *Nephrops norvegicus* and the shrimps *Parapenaeus longirostris* and *Aristeus antennatus*. Such fisheries are typically characterised by significant amounts of by-catch and discards of numerous unwanted species, including by-catch of epibenthic fauna and vulnerable fishes, such as elasmobranchs (Monteiro *et al*., 2001; Clarke *et al*., 2015; Bueno-Pardo *et al*., 2017).

This paper aims to provide a brief overview of the current status of deep-water fisheries to the west of the British Isles (Figure 1) and the key drivers shaping deep-water communities along the continental margin. Finally, we discuss the role of ecosystem-based approaches in deep-water fisheries management as a tool towards the achievement of the United Nations Sustainable Development Goals (SDG).

2 | State of deep-water fisheries in ICES subareas 6 and 7

Following the decline of more traditional continental-shelf fisheries, deep-water fishing effort in the north-east Atlantic Ocean has expanded dramatically since the 1960s, with Russian fishing vessels targeting *C*. *rupestris* in international waters, beyond the western limits of the UK continental margin. German trawlers targeted *M*. *dypterygia* in the Rockall Trough in the 1970s and a major French trawl fishery to the west of the British Isles for the same species developed during the same decade (Charuau *et al*. 1995; Gordon, 2001). Fisheries targeting black scabbardfish *Aphanopus carbo* Lowe 1839, *H*. *atlanticus* and *C*. *rupestris* began in the late 1980s, following a French incentive to land species with low commercial value but high by-catch rates (Charuau *et al*. 1995; Gordon, 2001). In the mid-1990s, deep-water fisheries became attractive to other fleets, including Scottish, Irish and Spanish fleets, but the French fleet has continued to dominate along the NE Atlantic continental slopes.

The development of deep-water fisheries was mainly driven through the increase in fleet capacity and fishing power, particularly in the 1990s (Villasante, 2010) and a lack of fishing opportunities elsewhere. Other important fisheries exist for species such as blue whiting *Micromesistius poutassou* (Risso 1827)), boarfish *Capros aper* (L. 1758) and monkfishes *Lophius* spp., but these primarily occur on the continental shelf, in depths < 400 m and, therefore, are not considered here. The average landings of the main deep-water fisheries to the west of the British Isles (ICES subareas 6 and 7; http://www.ices.dk/marine-data/maps/Pages/default.aspx) between 1950 and 2016 are given in Tables 1, 2.

The expansion of global fishing towards deeper areas means fishing for, generally, vulnerable species, that are long-lived with relatively low fecundity and growth rates, late maturation, commonly (but not exclusively) seen in deep-water fishes (Morato *et al*., 2006). This low biological productivity implies that these fish stocks can only sustain a low to moderate fishing mortality (Priede, 2017); *i.e*., the level of fishing mortality that slope stocks can sustain is lower than for shelf stocks.

Qualitative assessments on deep-water fish stocks in the North Atlantic Ocean have been provided since 1994 by the ICES expert working group (EWG) on the biology and assessment of deep-sea fisheries resources (WGDEEP; formerly SGDEEP, study group on the biology and assessment of deep-sea fisheries resources). Recent developments such a new accepted assessment framework and methods and available data have enabled the EWG to quantitatively assess some of these assessment units. WGDEEP provides scientific advice on 29 assessment units, including those for *C. rupestris*, *M. dypterygia*, *A. carbo*, *H. atlanticus* and *B. brosme*. Stocks are managed within units that do not always correspond to a biological population. Biological stock units generally represent self-perpetuating units where there is limited genetic exchange with other stocks (Pawson & Jennings, 1996; Pawson & Ellis, 2005). For example, ICES consider there to be five stocks of *B. brosme* in the north-east Atlantic Ocean based on genetic studies (ICES, 2018a), yet there were five management units, which cover different spatial areas (EU, 2018). This is because biological assessment units are updated as new information is made available (Knutsen *et al*., 2009), but there is a delay in amend management units.

The introduction of regulatory measures under the EU Common Fisheries Policy (CFP) in 1983 has had little or no effect on deep-water fish stocks (Villasante *et al*., 2012). Following ICES advice, in 1992, the European Commission (EC) introduced new policies regulating the exploitation of deep-water stocks, which remained ineffective as EU fleets continued to target deep-water fish beyond safe biological limits during the 1990s and 2000s (Gordon, 2003; Villasante *et al*., 2012). Since the implementation of further restrictive measures, *i.e*., total allowable catch (TAC), by the EC in 2002, the overall fishing pressure on deep-water fish species has declined and the overall biomass of commercial fish stocks in ICES subareas 6 and 7, those adjacent to the British Isles, has increased markedly (ICES, 2016a). In addition, the EC has recently adopted a regulation that limits the use of certain gears below depths of 800 m (EU, 2016), including the prohibition of bottom fishing activities in areas of occurrence of vulnerable marine ecosystems (VME).

The main fleets targeting *M. dypterygia* are French and, in recent years, Scottish trawlers. Estimated landings in ICES subareas 6 and 7 reached a peak of about 18,000 t in 1973 (ICES, 2018a), but from 2000s total reported landings have declined (Figure 2). *Molva dypterygia* fishing mortality has decreased since 2002 and stock biomass has increased since 2004, but stocks have been exploited at a level considered above sustainable levels since 2010. Total landings in ICES subarea 7 are low and restricted to by-catch (ICES, 2018b).

Like other stocks around the world (Clark *et al*., 2000), *H. atlanticus* catch per unit effort (CPUE) has declined since 1990, which has been linked to depletion caused by fishing activities in ICES areas 6 and 7 (Bailey *et al*., 2009). Owing to the decline of the fishery, total landings of *H. atlanticus* have been low since 2002 (Figure 2) and there has been no directed fishery following the introduction of a zero TAC for this stock unit in the NE Atlantic Ocean since 2010 (ICES, 2018a). Owing to the restrictive measures for this fishery, limited data on catch and its composition are available, which, coupled with current monitoring programmes, are insufficient to evaluate the status of *H. atlanticus* populations in ICES subareas 6 and 7 (ICES, 2018b).

Most landings of *C. rupestris* from ICES subareas 6 and 7 are from French and Spanish bottom trawlers. This fishery has declined since 2004, particularly in the last few years with the implementation of the regulation prohibiting bottom trawling below depths of 800 m (EU, 2016; ICES, 2018a). Landings of *C. rupestris* have declined sharply in subarea 6 from 14,800 t in 2001 to *c*. 630 t in 2017 and no landings were reported in subarea 7 in the same year (Figure2; ICES, 2018b). Discard rates of *C. rupestris* in other fisheries have also reportedly declined over the past few years and this has been attributed to the decline of the deep-water fishery overall (ICES, 2018a). The current status of *C. rupestris* stocks is unknown (ICES, 2018b) mainly due to insufficient data.

Historical landing data for *A. carbo* in ICES subareas 6 and 7 indicate an increasing trend from 1999 reaching a maximum of over of 5800 t in 2002 (Figure 2), just before the implementation of EU TAC management policies for deep-water fish stocks. The stock assessment for this species indicates a moderate increase in abundance since 2010, with stable landings up to 2016 (ICES, 2018b).

Brosme brosme is a by-catch species in the deep-water fisheries in ICES subareas 6 and 7 and it has been mainly landed by the Norwegian longline fleet. Historically, reported landings varied substantially in ICES subarea 6, peaking at about 5600 t in 1983 and at 4400 t in 2000, but showing a slight decline since 2001. Landings from ICES subarea 7 are historically low; they peaked at 785 t in 1969, but have remained very low since 2003 (ICES, 2018a). Discards are negligible in the longline fishery, but most of *B. brosme* by-catch in trawl fisheries is discarded. Owing to limited information of *B. brosme* life-history parameters, no stock-management reference points are defined for this stock in ICES division

6b and a precautionary approach was advised in 2018 for stocks in ICES subareas 6 and 7 (ICES, 2018a).

Current fishing opportunities for *M. dypterygia* and *C. rupestris* are based on an ICES category 1 assessment approach with full analytical assessment, forecast and maximum sustainable yield (MSY) reference points, a precautionary approach is used for the *H. atlanticus* (ICES category 6) and *A. carbo* (ICES category 3) in ICES subareas 6 and 7 (ICES, 2018a). Although *A. carbo* is assessed as a category 3 stock, it is qualitatively evaluated in relation to management reference points, like that of the *H. atlanticus*. There is evidence of improvements in the state of some of the stocks in the slopes to the west of the British Isles. However, there remains some uncertainty in this assertion given that historical landings and discards data are not always accurately recorded, or not provided by all EU member states (Pawlowski & Lorance, 2009). Additionally, assessments could be improved with fisheries-independent data, for example estimates on recruitment (Large *et al*., 2010; Carlsson *et al*., 2011; Neat, 2017), but spatial and temporal coverage does not always provide robust estimations (Large *et al*., 2003; Thorson & Barnett, 2017). Despite recent efforts, there is still a scarcity of information for most of the deep-water fish species, particularly on the spatial distribution and habitat uses, biological and genetics data. Previous studies have attempted to estimate the age for certain species around the British Isles, such as Baird's slickhead *Alepocephalus bairdii* Goode & Bean 1879, *C. rupestris*, blackbelly rosefish *Helicolenus dactylopterus* (Delaroche 1809)) and *H. atlanticus* (Gordon & Swan, 1986; Smith *et al*., 1995; Kelly *et al*., 1997; Kelly *et al*., 1999; Allain & Lorance, 2000), but although crucial for estimation of growth rate, mortality rate and productivity, data on the population structure and studies on age validation of the long life of several deep-water species are still limited (Treble *et al*., 2008; Andrews *et al*., 2009).

3 | Population Connectivity and Stock Identity

Little is understood about connectivity of deep-sea fish populations. Recent genetic analysis has revealed evidence of *C. rupestris* population substructure along the central and eastern North Atlantic Ocean (White *et al*., 2010), suggesting limited connectivity across ocean basins. This is particularly evident in the Rockall Trough area, where limited migrations of *C. rupestris* occur once a fish settles out and adopts a benthopelagic behaviour pattern (Knutsen *et al*, 2012). Geochemical analysis of *C. rupestris* otoliths showed a clear separation of population units along the NE Atlantic Ocean (Longmore *et al*., 2010, 2011). A complex depth-dependent genetic structure in *C. rupestris* was revealed recently (Gaither *et al*., 2018) identifying genetically based ontogenetic depth segregation whereby juvenile populations inhabiting shallow waters exhibit mixing, but adults segregate by depth forming genetically distinct populations. In contrast, the roughhead grenadier *Macrourus berglax* Lacépède 1801 shows significant gene flow across North Atlantic locations (Coscia *et al*., 2018). A general lack of understanding of potentially complex population demographic processes across broad oceanic areas and behaviours limits the ability for an effective management. A combination of tools (*e.g.*,, genetic and geochemical approaches) have potential to inform on the mechanisms for population connectivity and reveal insights in terms of the duration of different life stages (Trueman *et al*., 2016).

Deep-water fish stocks have been exploited outside their safe biological limits over decades in the north-east Atlantic Ocean with unknown long-term effects on the ecosystems. Moreover, while > 100 deep-water demersal fish species are known for the British Isles (Neat & Campbell, 2011; Froese & Pauly, 2018), only few species and stocks are assessed for advice by ICES. However, stock assessments are hampered owing to incomplete or lack of available data on deep-water fisheries catches in the NE Atlantic Ocean (ICES, 2016b, 2018a,c), significantly challenging a sustainable management of deep-water fisheries (Norse *et al*., 2012; Large *et al*., 2013). For this reason, there is a clear need to undertake further research to develop and improve stock assessment methodologies and to collect systematic data to identify important habitats and location of spawning areas.

4 | Trophodynamics in continental slopes

Energy availability plays a key role in determining community and biomass trends, as well as patterns of diversity and distribution of assemblages along continental slopes (McClain *et al*., 2012; Woolley *et al*., 2016). Abundant and diverse deep-sea communities are supported on slopes by enhanced food availability (McClain & Schlacher, 2015), which appears to be controlled by lateral and vertical (sinking) fluxes (Ichino *et al*., 2015). At greater depths, where absence of light, high pressure and near freezing temperatures are major constraints, detrital flux of particulate organic matter (POM), secondary production and benthic recycling have an important role in fuelling benthic and benthopelagic communities (Figure 3; Jones *et al*., 2014; Stasko *et al*., 2016).

The active transport of carbon and nutrients by diel vertical migration of zooplankton and small pelagic fishes is a common phenomenon, which has been extensively documented (Sutton, 2013), and it is crucial in fuelling bentho-pelagic production at mid-slope depths (Mauchline & Gordon, 1991; Trueman *et al*., 2014; Drazen & Sutton, 2017). However, the diversity and composition of higher trophic-levels communities can be modified by changes in primary production (Blanchard *et al*., 2012). For instance, a shift in the dominant phytoplanktonic species can induce phenological change in zooplankton abundance, which may ultimately affect fish recruitment and fisheries; at present, however, these cascading effects have not been comprehensively documented (de Madron *et al*., 2011). Sea surface temperatures are predicted to continue rising over the coming decades (Domingues *et al*., 2008; Medhaug *et al*., 2017) and theoretical and empirical evidence shows that mean phytoplankton body size reduces with increasing temperature (Morán *et al*., 2010; Thomas *et al*., 2012). This reduction may be critical for deep-sea communities, as smaller phytoplankton may be less readily exported to deep waters, potentially reducing the total nutrient supply to deep-water fish communities (Yool *et al*., 2017).

It is now acknowledged that deep-water fish communities may provide a significant carbon sequestration ecosystem service by capturing carbon from the deep-scattering-layer communities and transferring carbon to the greater depths (Trueman *et al*., 2014). In this sense, it is conceivable that the contribution of deep-water benthic fauna in regulating atmospheric carbon might be higher than previously appreciated (Thistle *et al*., 2006; Kahn *et al*., 2015). Long-term changes in organic matter fluxes and, ultimately, in deep-water fish communities are therefore likely to influence energy pathways and the capacity of carbon storage, but the sensitivity of deep communities to change is poorly understood (Sweetman *et al*., 2017).

5 | Ecosystem effects and implications for the seabed

Despite limitations, historical data do enable an examination of temporal trends in catches and evaluation of contribution from individual species to ecosystem level. Studies of human activities in the deep sea have received increasing attention over the past 20 years, mainly due to appreciation of the ecological and economic importance of these poorly-studied marine ecosystems (Basson *et al*., 2001; Gage, 2001; Rogers *et al*., 2015).

The geographical extent of deep-sea fisheries effects on the deep seabed of the North Atlantic Ocean has been highlighted as a long-running concern, especially since benthic trawling has by far the greatest effect on the seafloor, much more so than the hydrocarbon industry, seafloor cables and other activities combined (Benn *et al*., 2010). An increasing body of evidence suggests severe perturbations caused by bottom trawling on the integrity of the seafloor and its biological assemblages along continental slopes (Daly *et al*., 2017; Eigaard *et al*., 2017). Previous studies identified trawling-induced sediment displacement, suggesting that industrial bottom trawling had long-term effects on seabed morphology (Puig *et al*., 2012; Martín *et al*., 2014). Bottom trawling increases sediment resuspension, affecting sedimentation rates at and around the affected areas and altering geochemical properties of the sediments, resulting in biological and ecological effects such as nutrient resuspension and sediment homogenisation (Amaro *et al*., 2016; O'Neil & Ivanović, 2016). Physical disturbance by bottom trawling is now widely spread in coastal and shelf zones (Brown *et al*., 2005; Bueno-Pardo *et al*., 2017), but, in these environments, sediment transport and erosion may buffer the capacity of trawling to modify seafloor morphology (Nittrouer & Wright, 1994).

Direct consequences of bottom trawling also include biomass removal or damage of sensitive benthic communities (*e.g*.,, sponges and deep-sea corals), resulting in high mortality rates (Ramsay *et al*., 2001; Clark *et al*., 2016). Bottom-trawl fishing has been shown to damage or destroy long-lived benthic communities, but also to harm the complexity of the seabed, reducing species richness and biomass (Gage *et al*., 2005; Huvenne *et al*., 2016). Numerous studies have demonstrated that deep-sea fisheries in the North Atlantic Ocean occur at the same depths as known VMEs (Muñoz *et al*., 2009; Fonseca *et al*., 2014). The effects of bottom trawling on benthic communities, particularly on VMEs, are often direct and immediate (*e.g*.,, damaging sponges and corals, removing non-target species through by-catch), persisting for decades or longer (Hall-Spencer *et al*., 2002; Clark *et al*., 2016). Muñoz *et al*. (2012) provided insights on the presumed effects of towed gears, suggesting that apparent differences in cold-water corals and sponge aggregations in different habitats along the Hatton Bank might be a consequence of bottom trawling activity. Likewise, it has been proposed that extensive fisheries induced considerable changes in biomass of deep-water fish populations in the Porcupine Seabight (Bailey *et al*., 2009) and along the slopes of the Rockall Trough (Mindel *et al*., 2018).

The effects of persistent trawling are said to result in reduced diversity of epifauna and endofauna (Roberts *et al*., 2000; Jennings *et al*., 2001; Tillin *et al*., 2006; Pusceddu *et al*., 2014; Almeida *et al*., 2017), with a shift in communities towards a dominance of scavenger species (Jennings & Kaiser, 1998; Blanchard *et al*. 2004). However, the degree to which deep benthic fauna are resilient to long-lasting bottom trawling and the implications on marine food webs is still poorly understood (Arroyo *et al*., 2017; Cunha *et al*., 2017; Hiddink *et al*., 2017; Vieira, 2017). Ecological indicators, such as trophic and size structure, community biomass and diversity, reflect overall changes in ecosystems (Blanchard *et al*., 2017; Mindel *et al*., 2018), but substantial knowledge gaps and long-term data deficiency contributes to deficient assessment of the extent of human-induced effects, particularly through fisheries, on deep-water ecosystems (Rogers *et al*., 2015). Nevertheless, greater perturbations of the benthic communities could lead to the disruption of the benthic energy pathways, through the changes in nutrient transfer and the removal of food resources exploited by the fish communities. This could result in potential changes on the ecosystem services associated with benthic nutrient cycling and ultimately affecting secondary production of deep-water fish communities.

Some preliminary studies of long-term changes in deep-sea fish communities to the west of the British Isles have been carried out, most notably by Basson *et al*. (2001), Neat *et al*. (2008) and Bailey *et al*. (2009). Various scientific institutes collected information in the 1970s and 1980s, prior to the development of major deep-water fisheries (Basson *et al*.,

2001) and in recent years Marine Scotland (Aberdeen, UK) and the Marine Institute (Oranmore, Ireland) have conducted detailed surveys of shelf edge and seamount communities as part of their annual monitoring programme (O'Hea *et al*., 2009; Neat *et al*., 2010). The species richness was observed to reach a maximum value at 1000 m and decline at greater depths. Basson *et al*. (2001) calculated various diversity indices based on deep-water survey data from the pre-and post-exploitation era. Data available post-exploitation, from the SAMS OTSB(S) survey in 1999, suggested that diversity was lower than the pre-exploitation state. Godbold *et al*. (2013) reported a decline in total demersal fish biomass in the Porcupine Seabight and adjacent abyssal plain of 36% within the depth range of the commercial fishery (< 1500 m). Whilst there were significant declines in target (*e.g*., *C. rupestris* decreased by 57 %) and non-target [*e.g*.,, *Coryphaenoides guentheri* (Vaillant 1888) and *Antimora rostrate* (Günther 1878)] species, not all species declined significantly.

Identifying temporal changes and geographical gradients in community dynamics can be informative to improve ecological theory (May & McLean, 2007) and ecosystem modelling (Thorpe *et al*., 2015; Spence *et al*., 2018), both of which can be used in support of adequate fisheries management (Hyder *et al*., 2015). However, species-based food web models such as Ecopath with Ecosim (Christensen & Walters, 2004) are less effective in size-structured ecosystems and recent studies have highlight the important role that body size plays in energy flux and trophic ecology within deep demersal fish communities (Polunin *et al*. 2001; Trueman *et al*., 2014; Mindel *et al*., 2016; Vieira, 2017). Food web models combining taxonomic–functional trait and body-size approaches are likely to be more effective (Hartvig *et al*., 2011; Blanchard *et al*., 2014). Therefore, understanding the mechanisms supporting deep-water fish community structure would be useful to parameterise ecosystem models [*e.g*.,, Porcupine Seabight (Howell *et al*., 2009), Rockall Trough

(Heymans *et al*., 2011)] in order to simulate, on a decadal timescale, the effects of benthic community changes on fish abundance and production (Blanchard *et al*., 2009, 2011).

Accumulated proof of anthropogenic effects on deep sea communities and habitats confirms the significant threat to biodiversity (Ramirez-Llodra *et al*., 2011). Information on fisheries activities and related pressures (Weaver *et al*., 2011; McCauley *et al*., 2016; Amoroso *et al*., 2018), as well as the distribution and type of litter often lost or discarded (*e.g*., Pham *et al*., 2014), are powerful tools to increase societal and scientific awareness regarding the degradation of even the most remote marine ecosystems (Ruhl *et al*., 2011). Such information is key to identify anthropogenic pressures operating in these ecosystems and is therefore essential for effective management and achievement of ‘good environmental status’ (EU, 2008) and to support further actions to target commitments under the United Nations Sustainable Development Goal 14: conserve and sustainably use the oceans, seas and marine resources (www.sustainabledevelopment.un.org/sdg14).

6 | Scope for future work

Generally, bottom trawls are relatively unselective fishing gears (Cashion *et al*., 2018), for this reason the wider cost-benefit of deep-water fisheries should be considered when reviewing fisheries regulations (Mangi *et al*., 2016). Collecting further data on the biodiversity and ecosystem functioning of deep-sea communities is important to consolidate knowledge and assess state and trends and to provide scientific-based information towards conservation and sustainable exploitation of marine resources (Ehler & Douvere, 2009; Kaiser *et al*., 2016).

It is crucial to better understand resilience, recoverability and the role of disturbance on deep-water communities. Additionally, it is relevant to investigate the role of pelagic communities; *e.g.*, the vertically-migrating component of ecosystems, their structure and function (Woodall *et al*., 2018). This knowledge gap in a community functionally responsible for active benthic–pelagic coupling in continental slope ecosystems severely hampers our ability to predict ecosystem response to climate or human-induced change (Anderson *et al*., 2018).

Understanding historical trends of fisheries, climate and anthropogenic effects on ecosystems is important for the development of predictive models that can be used to identify the gaps in knowledge and better inform policy-makers in the support for future deep-sea management and conservation decisions. As evidenced by the recent introduction of depth ban on trawl fishing bellow 800 m, there is an intention to reduce of bottom-trawl fisheries to allow the sustainability of deep-water fisheries and to protect benthic ecosystems (EU, 2016). An integrated ecosystem-based approach coupling the management of deep-water fish stocks and the state of benthic ecosystems (Kenny *et al*., 2018) would allow a better understanding of direct and indirect effects of bottom trawling on benthos and how fisheries effects reflect changes in food webs and ultimately in fish stocks. Implementing such strategies can be a step towards the achievement of the SDGs and contribute to the conservation and sustainable use of marine resources.

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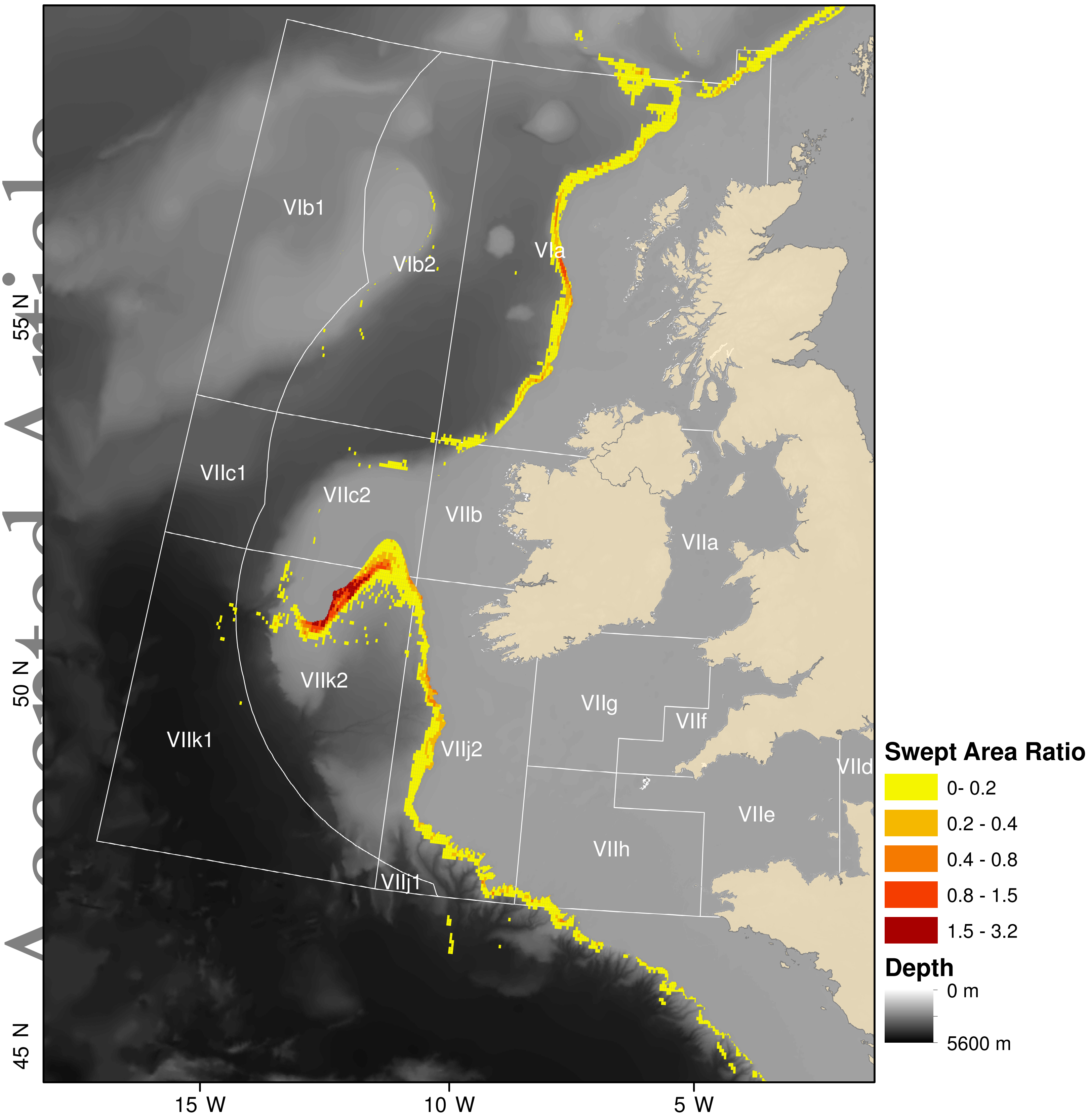
Figure captions

FIGURE 1 Map showing ICES Subareas and Divisions to the west of the British Isles, and fishing intensity deeper than 400 m in 2015 (www.odims.ospar.org)

FIGURE 2 (a) Trends in official nominal catches from ICES subarea 6 and (b) ICES subarea 7 of *Aphanopus carbo* (■), *Molva dypterygia* (■), *Hoplostethus atlanticus* (■), *Coryphaenoides rupestris* (■) and *Brosme brosme* (■). *n.b.*, In 1999 data were not provided by all EU member states (www.ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx),

FIGURE 3 Schematic illustration of important physical and biological processes responsible for fueling benthic and pelagic communities on the continental slope and in the water column (after Vieira, 2017).

Figure 1


VIb1
VIb2
VIa
VIIc1
VIIc2
VIIb
VIIa
VIIk2
VIIk1
VIIj2
VIIg
VIIf
VIId
VIIe
VIIh
VIIj1
55 N
50 N
45 N
15 W
10 W
5 W
Swept Area Ratio
0- 0.2
0.2 - 0.4
0.4 - 0.8
0.8 - 1.5
1.5 - 3.2
Depth
0 m
5600 m

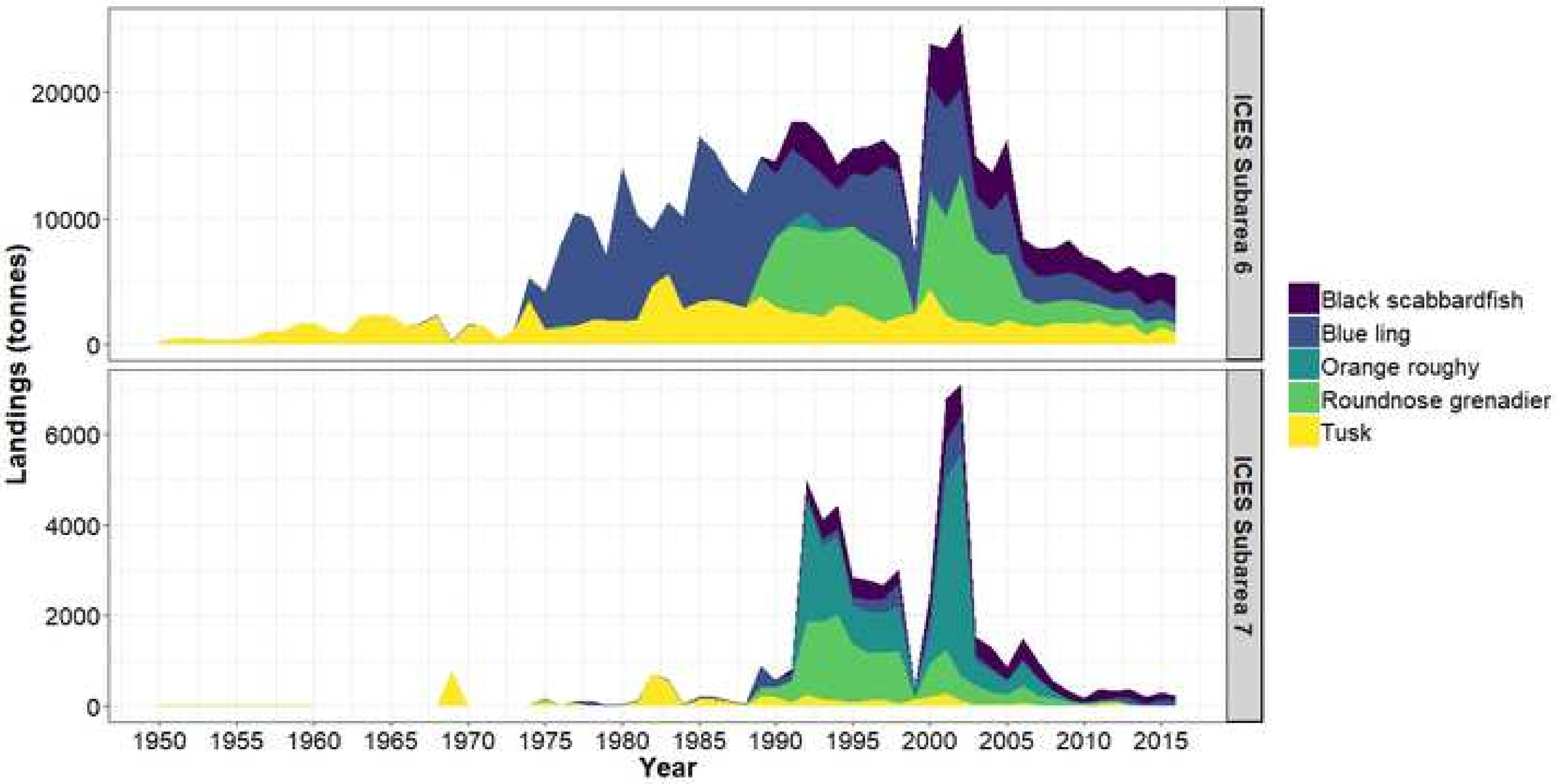

Landings (tonnes)
20000
10000
0
6000
4000
2000
0
1950
1955
1960
1965
1970
1975
1980
1985
1990
1995
2000
2005
2010
2015
Year
ICES Subarea 6
ICES Subarea 7
Black scabbardfish
Blue ling
Orange roughy
Roundnose grenadier
Tusk

Figure 3

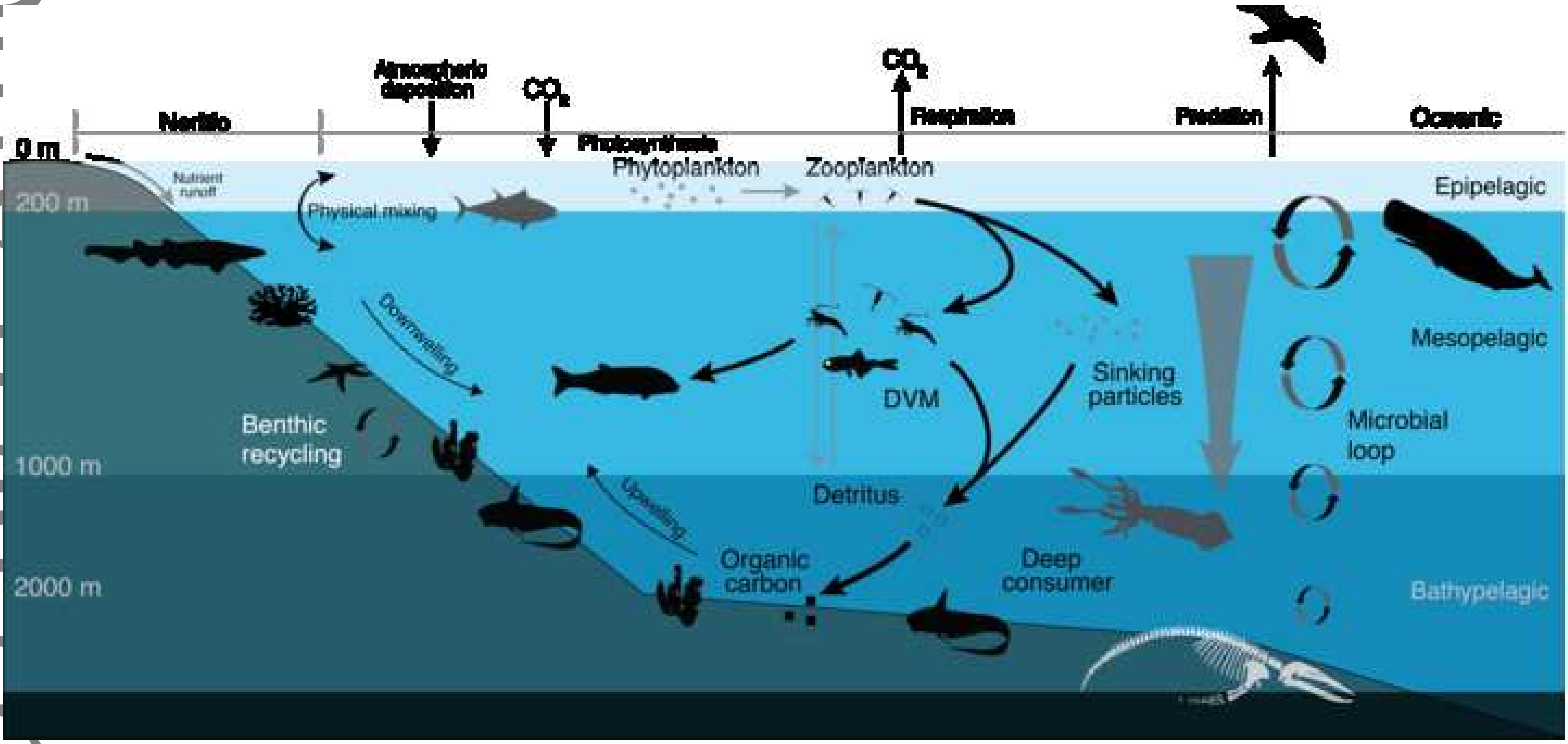

Neritic
Atmospheric deposition
CO_2
Photosynthesis
CO_2
Respiration
Predation
Oceanic
0 m
Nutrient runoff
200 m
Physical mixing
Phytoplankton
Zooplankton
Epipelagic
Downwelling
DVM
Sinking particles
Mesopelagic
Benthic recycling
Microbial loop
1000 m
Detritus
Upwelling
Organic carbon
Deep consumer
2000 m
Bathypelagic

TABLE 1 Average annual historical nominal catches of selected deep-water fish species from ICES Subarea 6 1950–2016 (ICES: www.ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx)

		Average landings (t)						
		1950	**1960**	**1970**	**1980**	**1990**	**2000**	**2010**
Bony fishes								
Beryciformes								
Alfonsino	*Beryx* spp.	0	0	0	1	177	5	1
Orange roughy	*Hoplostethus atlanticus*	0	0	0	1	276	118	0.1
Gadiformes								
Tusk	*Brosme brosme*	690	1623	1601	3397	2523	2013	1381
Roundnose grenadier	*Coryphaenoides rupestris*	0	0	38	231	5470	5192	1139
Roughhead grenadier	*Macrourus berglax*	0	0	0	0	100	59	162
Blue ling	*Molva dypterygia*	0	34	3368	9062	4895	4429	1470
Ling	*Molva molva*	4348	8719	14,500	14,459	9191	5956	5227
Greater forkbeard	*Phycis blennoides*	0	0	2	377	640	1080	717
Roughsnout grenadier	*Trachyrincus scabrus*	0	0	0	0	0	127	0
Osmeriformes								
Greater argentine	*Argentina silus*	0	0	0	0	1	1697	2665
Perciformes								
Blackspot (red) seabream	*Pagellus bogaraveo*	0	0	336	125	5	3	0
Black scabbardfish	*Aphanopus carbo*	0	0	0	15	1893	3226	2048
Elasmobranchs								
Squaliformes								
Leafscale gulper shark	*Centrophorus squamosus*	0	0	0	0	0.4	169	26

Portuguese dogfish	*Centroscymnus coelolepis*	0	0	0	0	18	766	6
Kitefin shark	*Dalatias licha*	0	0	0	0	3	148	0
Birdbeak dogfish	*Deania calcea*	0	0	0	0	0	3	0
Carcharhiniformes								
Blackmouth catshark	*Galeus melastomus*	0	0	0	0	0	0.1	0

TABLE 2 Average annual historical nominal catches of selected deep-water fish species from ICES Subarea 7 1950–2016 (ICES: www.ices.dk/marine-data/dataset-collections/Pages/Fish-catch-and-stock-assessment.aspx)

		Average landings (tonnes)						
		1950	**1960**	**1970**	**1980**	**1990**	**2000**	**2010**
Bony fishes								
Beryciformes								
Alfonsino	*Beryx spp.*	0	0	2	14	115	57	12
Orange roughy	*Hoplostethus atlanticus*	0	0	0	0.3	975	1176	2
Gadiformes								
Tusk	*Brosme brosme*	24	84	23	208	137	77.5	16
Roundnose grenadier	*Coryphaenoides rupestris*	0	1	10	21	1048	406	36
Roughhead grenadier	*Macrourus berglax*	0	0	0	0	0	18.5	3
Blue ling	*Molva dypterygia*	0	0	14	67	220	238	77
Ling	*Molva molva*	1852	3492	6092	11,015	9395	5459	2876
Greater forkbeard	*Phycis blennoides*	0	0	0.4	218	1565	1739	805
Osmeriformes								
Greater argentine	*Argentina silus*	0	0	0	0	0	4	2
Perciformes								
Blackspot (red) seabream	*Pagellus bogaraveo*	0	0	967	366	43	83	60
Black scabbardfish	*Aphanopus carbo*	0	0	0	0	282	446	157
Elasmobranchs								
Squaliformes								
Leafscale gulper shark	*Centrophorus squamosus*	0	0	0	0	0	258	4
Portuguese dogfish	*Centroscymnus coelolepis*	0	0	0	0	13	488	1

Kitefin shark	*Dalatias licha*	0	0	0	0	0	66	1
Birdbeak dogfish	*Deania calcea*	0	0	0	0	0	32	0.2
Carcharhiniformes								
Blackmouth catshark	*Galeus melastomus*	0	0	0	0	0	2	1