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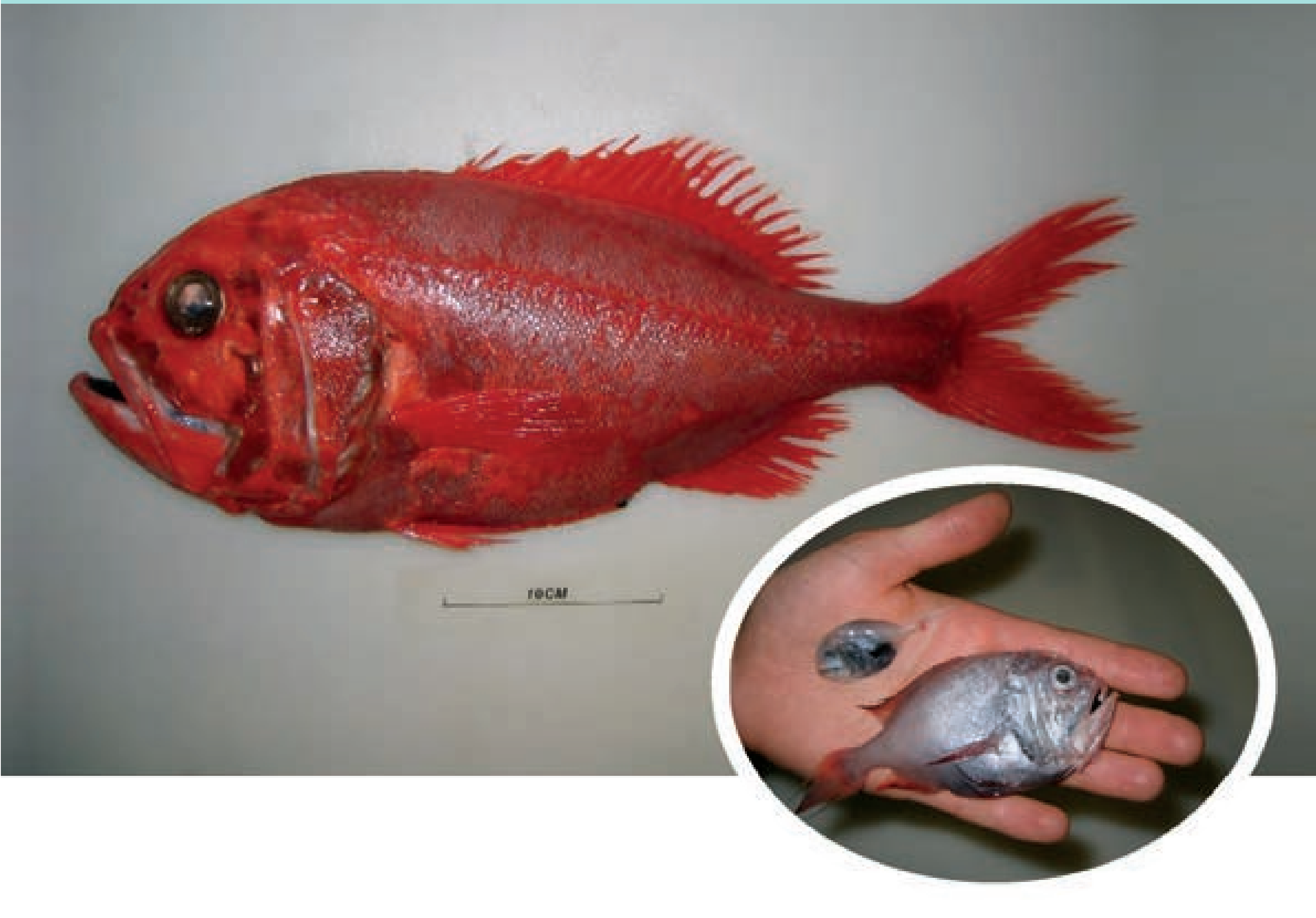
Global review of orange roughy (*Hoplostethus atlanticus*), their fisheries, biology and management



Cover photographs: Adult orange roughy (*Hoplostethus atlanticus*), and juvenile orange roughy (*H. atlanticus*) from central New Zealand, of ages about 1 year (left) and 5 years (right). *Photographs courtesy of M. Dunn*.

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Preparation of this document

This report has been developed by the authors, with guidance from workshop participants, and using material presented, at the FAO Workshop on the Science in Support of Management of the Fisheries for Orange Roughy (*Hoplostethus atlanticus*), held in Auckland, New Zealand, from 7 to 9 June 2016. The workshop was organized by FAO, with support from Dr Geoff Tingley, a consultant. In total, 13 participants attended the workshop in their individual capacities as regional experts on the subjects of the science of orange roughy and deep-sea fisheries, orange roughy stock assessment, and in the management of orange roughy fisheries. The workshop was organized as part of the FAO Deep-sea Fisheries Programme, which supports the implementation of the International Guidelines for the Management of Deep-sea Fisheries in the High Seas (adopted in 2008). These guidelines provide guidance to States and regional fisheries management organizations on arrangements to ensure the long-term conservation and sustainable use of deep-sea marine living resources in the high seas, including preventing significant adverse impacts on vulnerable marine ecosystems.

The workshop was financed with the support of two of the projects under the FAO Deep-sea Fisheries Programme: Support for the implementation of the International Guidelines on the Management of Deep-sea Fisheries in the High Seas, funded by Norway; and Fisheries Management and Marine Conservation within a Changing Ecosystem Context – deep-sea fisheries component, funded by Japan. The workshop addressed key issues identified in relation to the management of deep-sea fisheries in the high seas, and directly contributed to the goal and objectives of the Global Environment Facility-funded Areas Beyond National Jurisdiction (ABNJ) Deep Seas Project: Sustainable Fisheries Management and Biodiversity Conservation of Deep-sea Living Marine Resources and Ecosystems in the Areas Beyond National Jurisdiction.

Abstract

This publication is intended to provide a range of stakeholders and interested parties with an understanding of orange roughy fisheries around the world. The report covers historical aspects of the regional development of orange roughy fisheries, biology, stock assessment, ecosystem interactions, and key management issues. Recent developments in science and approaches to management are specifically highlighted with respect to future management of sustainable deepwater orange roughy fisheries.

The sustainability of orange roughy fisheries, or other fisheries for long-lived deepwater species, has been widely discussed. These reviews invariably draw on the common global experience of previous poor understanding about orange roughy productivity, rapid development of targeted industrial fisheries, the associated likelihood of overfishing and extended timescales for stock recovery, and an ensuing series of "boom and bust" orange roughy fisheries that frequently resulted in depleted stocks.

The more recent experience, with greater knowledge, improved technology, better approaches to modelling population dynamics in orange roughy, and a more considered and robust approach to setting up the management framework (harvest strategy, management strategy evaluation, appropriately estimated limit and target reference points or ranges, and effective harvest control rules), provides a different paradigm. Essentially, many of the assumptions about the unmanageability of these fisheries are not supported by the more recent evidence. Provided appropriate steps are taken to set and deliver a low and appropriate level of fishing mortality, orange roughy fisheries can be both well managed and sustainable. The improved understanding of the productivity and population response of orange roughy now provides a basis for better estimating yields and fishery value that are both more realistic and compatible with sustainable fisheries.

It is also of note that the regional fisheries management organizations that have the largest stocks and fisheries for orange roughy – the Southern Indian Ocean Fisheries Agreement and the South Pacific Regional Fisheries Management Organisation – have been ramping up their efforts to manage the fishing for the target species and at the same time address the benthic and vulnerable marine ecosystem impacts of bottom fishing through developing science-based, spatial management.

While there is still considerable discussion and opposed viewpoints on the sustainability of deepwater fisheries generally, aspects of the message have clearly changed: sustainable orange roughy fisheries are achievable. This review describes how, by making the right choices and employing the best science available, there are now some demonstrably sustainable orange roughy fisheries.

Even with this rather more positive perspective of the sustainability of these deepwater orange roughy fisheries, there remain some considerable challenges to address. These include improving understanding of deepwater benthic communities in general, their genetics and population distributions, their dispersal, and their ability to recover from fisheries (and other) impacts. With regard to the direct management of the fisheries, there are important opportunities and needs to improve ageing and acoustic biomass estimation, and to better understand the genetics and population structure of the stocks of orange roughy that are fished and managed.

FAO. 2018. *Global review of orange roughy (*Hoplostethus atlanticus*), their fisheries, biology and management*, by Geoffrey Tingley and Matthew Dunn. FAO Fisheries and Aquaculture Technical Paper No. 622. Rome, Italy.

Contents

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Finally, while all of the workshop participants contributed to this report in various ways, any errors or omissions remain the sole responsibility of the authors.

The author also acknowledges and thanks Enzo Luchetti for the layout of the publication.

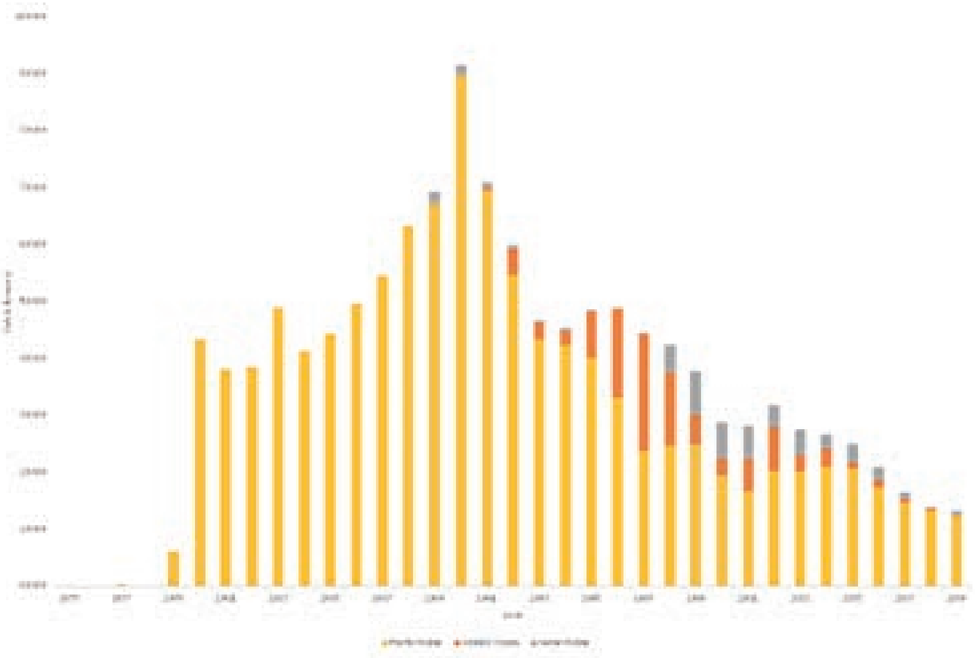
Abbreviations and acronyms

AFMA	Australian Fisheries Management Agency
AOS	acoustic optical system
B_0	initial, unfished biomass
B_{MSY}	biomass that produces the Maximum Sustainable Yield
BPA	benthic protection area
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CI	confidence interval
CMM	conservation and management measure
CSIRO	Commonwealth Scientific and Industrial Research Organisation
CPUE	catch per unit effort
CV	coefficient of variation
eNGO	environmental non-governmental organization
EEZ	exclusive economic zone
ETP	endangered, threatened and protected
F	fishing mortality
FIP	fishery improvement project
FMA	Fishery Management Area, New Zealand
GSI	Gonado-Somatic Index
h	steepness in the stock recruitment relationship
HCR	harvest control rule
ICES	International Council for the Exploration of the Sea
IUCN	International Union for Conservation of Nature
LRP	limit reference point
M	natural mortality
MCMC	Markov Chain Monte Carlo
MPA	marine protected area
MPI	Ministry for Primary Industries, New Zealand
MSC	Marine Stewardship Council
MSE	Management Strategy Evaluation
MSY	maximum sustainable yield
NAFO	Northwest Atlantic Fisheries Organization
NEAFC	Northeast Atlantic Fisheries Commission
RFMO	regional fisheries management organization
SEAFO	Southeast Atlantic Fisheries Organisation
SIODFA	Southern Indian Ocean Deepsea Fishers Association
SIOFA	Southern Indian Ocean Fisheries Agreement
SL	standard length (snout to end of caudal peduncle)
SPRFMO	South Pacific Regional Fisheries Management Organisation
TAC	total allowable catch
TACC	total allowable commercial catch
TL	total length (snout to end of tail)
TRP	target reference point
TS	target strength
UNGA	United Nations General Assembly
UTF	underwater topographical feature (e.g. hill, knoll, seamount, canyon)
VME	vulnerable marine ecosystem
YCS	year class strength

1. Introduction

Virtually every orange roughy fishery has followed the same basic developmental pattern, with early exploration leading to high initial catch rates and a rapid rise in both fishing effort and catch. This has then been quickly followed by a rapid drop in catch rates as stocks became overfished, and often then abandonment or closure of the fishery, a pattern often described as "boom and bust" (Koslow *et al*., 2000, Francis and Clark, 2005). This pattern was probably the result of a number of different factors, including opportunism by the catching sector, inadequate management responsiveness to changes in stock status and, in some places, an inability of managers to effectively curtail fishing effort, as well as inadequacies in the science used to advise managers, including the understanding of uncertainty in the science (Bax *et al*., 2005). These factors all combined with certain aspects of the biology of orange roughy that make this species vulnerable to overfishing – low productivity, and large, dense and predictable locations of aggregations). Together, the issues of biology, science, management and industry often combined to lead to substantial overfishing (Bax *et al*., 2005, Francis and Clark 2005), and also to make sustainable fisheries for orange roughy appear to be, at best, difficult (e.g. Clark 2001), and, at worst, unattainable (Sissenwine and Mace, 2007, Pitcher *et al*., 2010, Norse *et al*., 2012).

FIGURE 1
Global orange roughy landings by ocean from the start of the fisheries in the late 1970s to 2009



SOURCE: FAO (2018).

Orange roughy fisheries were first developed in New Zealand. Development then followed in the waters off Australia, Chile, Namibia, and in the northeast Atlantic, as well as high seas fisheries in the southwest Pacific and southern Indian Oceans, principally using fisheries expertise developed in New Zealand. This produced the now well-known rise and fall in global orange roughy catches (Figure 1). The early fisheries focused on fishing spawning aggregations of orange roughy, mostly on relatively flat areas of the continental slope but also often associated with underwater topographical features (UTFs) such as canyons, knolls, hills and seamounts (e.g. Clark and O'Driscoll, 2003). While the common understanding is that orange roughy fisheries operate on seamounts, in reality, true seamounts are only important in some areas, such as the Louisville Ridge in the southwest Pacific high seas fishery. Elsewhere, including in New Zealand waters, seamounts are rarely or never fished, with smaller UTFs (hills, knolls, ridges and canyons) being the principle focus of fishing. All UTFs can have associated large, epibenthic and biogenic fauna that are vulnerable to damage by mobile demersal fishing gear, such as the demersal trawls used to target orange roughy (Collie *et al*., 2017). Concerns about damage to the benthic environment (especially taxa that are used to describe vulnerable marine ecosystems [VMEs]) as a result of these deepwater fisheries led to activities to provide for benthic protection within exclusive economic zones (EEZs) and on the high seas (Koslow *et al*., 2000; Brodie and Clark, 2003; Penney *et al*., 2008; Penney, Parker and Brown, 2009; Parker, Penney and Clark, 2009; Helson *et al*., 2010).

With the relatively rapid collapse of most orange roughy fisheries after only a few years of operation, coupled with concerns about the possible scale of damage to vulnerable benthic fauna, orange roughy fisheries became one of the leading examples of overfishing by industry as a result of poor fisheries management. As a result, orange roughy fisheries became a focus for activities by campaign groups with an interest in marine environmental conservation, and orange roughy became a product labelled as a "fish to avoid".

In recent years, there have been considerable but largely unscientific advocacy efforts by environmental non-governmental organizations (eNGOs) to make demersal trawling politically and environmentally unacceptable (e.g. Weeber, Thomas and Dorey, 2010). These efforts have been especially pronounced for the deepwater trawl fisheries such as those for orange roughy. This is in addition to a more scientific and considered call for appropriate levels of benthic protection, especially for deepwater and high seas fisheries (Thompson *et al*., 2016), which can form the basis for an informed discussion on an appropriate balance between fishing and benthic conservation (Clark and Dunn, 2012).

The historically poor management performance of deepwater fisheries led to specific guidance from the United Nations General Assembly (UNGA) through Resolutions (e.g. UNGA 61/105 and 64/72), principally aimed at improving the management of high seas fishery areas. This has helped to promote the implementation of management measures to protect benthic habitats and VMEs in particular, especially at the regional level, implemented by regional fisheries management organizations (RFMOs). For a more specific and recent review of the occurrence and management of fisheries in relation to the protection of vulnerable marine ecosystems in the high seas, see FAO (2016) and Thompson *et al*. (2016).

At the time of writing, the only countries with commercial, targeted fisheries for orange roughy are Australia and New Zealand. The fishery off Namibia has been closed to fishing since 2008. However, Namibia conducted a vessel-based survey in 2016 to evaluate stock status to inform on management options for reopening the fishery (Esau, 2017). The fisheries off Chile and in the northeast Atlantic have both been closed to orange roughy fishing for some years. Two high seas fisheries also remain open, that in the southwest Pacific, under the management of the South Pacific Regional Fisheries Management Organisation (SPRFMO), and the southern Indian Ocean fishery, under the management of the Southern Indian Ocean Fisheries Agreement (SIOFA).

In both Australia and New Zealand, managers and fishers are, in general, now being considerably more cautious and conservative in their approach to these fisheries. Australia has a relatively high target reference point (stock at 48 percent of unfished biomass) and reopened its Eastern Zone orange roughy fishery on 1 May 2015 with a small total allowable catch (TAC) of 465 tonnes (including catch at Pedra Branca) following improvement in stock status.[1] The three largest New Zealand fisheries have, over a number of years, been improved in both science and management, in preparation for evaluation against the independent, third-party sustainability certification by the Marine Stewardship Council programme,[2] a process that has seen the implementation of very conservative catch limits with historically low fishing mortalities (MPI, 2016a). A fourth New Zealand orange roughy fishery is currently in a formal, publicly reporting, fishery improvement plan (FIP) in order to improve its sustainability.[3] Other, smaller New Zealand orange roughy fisheries are also being re-assessed as information needs are met, the most recent being the fishery at Puysegur and the west coast ORH 7B fishery, and there is a science programme to support this (MPI, 2016b).

There is a substantive literature covering orange roughy and their fisheries, including large quantities of grey and unpublished literature, much linked to the development of two RFMOs, and there have also been some substantive reviews, most notably that by Branch (2001), that summarize earlier publications. Thus, this work does not attempt an exhaustive review of earlier literature, but focuses instead on the most useful and relevant material to evaluate the current status of orange roughy fisheries, fisheries science and management, and consideration of future opportunities.

[1] See: www.AFMA.gov.au
[2] See: www.msc.org
[3] See: www.deepwatergroup.org, and https://fisheryprogress.org/

2. Orange roughy

2.1 TAXONOMY

The family Trachichthyidae (roughies) is a member of the order Beryciformes. The family is widespread globally, and consists of 8 genera and 49 species (Froese and Pauly, 2017). The roughies are also known as "slimeheads" or "sawbellies", making reference to their large heads with mucous cavities, and keel-like belly scutes, respectively. The name "orange roughy" is used to refer to a single species *Hoplostethus atlanticus,* which typically has a bright red to faded orange-grey colour, although this colouration is not unique within their family (*Hoplostethus melanopeza H. gigas*, and species of *Gephyroberyx* have similar colouration), nor within their order (e.g. the red-coloured alfonsinos of the genus *Beryx*). The orange colour of orange roughy is observed upon capture, but *in situ* fish can also be paler, or even appear white (Lorance, Uiblein and Latrouite, 2002). A species often referred to as "black roughy", *Diretmichthys parini*, is actually not a roughy (Trachichthyidae) but a "spiny fin" (Diretmidae) family. Orange roughy can be distinguished from other similarly coloured fishes by the large bony head, small irregular body scales, numerous but small belly scutes, an anus close to the anal fin, and 15–18 soft dorsal fin rays (McMillan *et al*., 2011). Orange roughy are also one of the largest species of their family, reaching lengths beyond 50 cm total length (TL).

2.2 DISTRIBUTION OF ORANGE ROUGHY

Orange roughy are widespread globally, but apparently absent from the northern Indian Ocean, and North Pacific (Roberts, Stewart and Struthers, 2015). Reports of orange roughy occurring north of the subtropical convergence (around 30°S) in the Pacific are probably misidentifications (Roberts, Stewart and Struthers, 2015). The largest fisheries for orange roughy have been found off Australia, Chile, Namibia, New Zealand, and in the northeast Atlantic (Branch, 2001). Smaller fisheries for orange roughy have been found throughout the southern Pacific and southern Indian Oceans (Clark *et al*., 2010a). In other areas, such as the northwest and southwest Atlantic, the reports of orange roughy have been limited to just a few fish (Kulka, Themelis and Halliday, 2003; Laptikhovsky, 2008; Wöhler and Scarlato, 2006). The distribution is likely to be somewhat underestimated given the reliance on commercial and research fishing operations to locate specimens.

Off New Zealand, juvenile orange roughy have been caught in demersal trawls at depths about 850–900 m, whereas the adults have been caught around 850–1 300 m (Dunn *et al*., 2009a). Isotope studies have suggested that juveniles in the northeast Atlantic may also start shallower than adults, but then go deeper than adults, eventually returning to occupy the adult depth range (Trueman, Rickaby and Shephard, 2013). Juvenile orange roughy have also been observed in large numbers from relatively deep water off southern Namibia (A. Smith, personal communication). While orange roughy are caught by demersal trawl gear, the fish can extend some distance vertically into the water column, especially during spawning, where plumes up to 200 m in height above the seabed have been reported (Branch, 2001), although the fish tend to dive towards the seabed in response to disturbance (Koslow, Kloser and Stanley, 1995; O'Driscoll *et al*., 2012).

Orange roughy are found in mesopelagic, bathypelagic and benthopelagic habitats, and have most often been caught near the seabed at depths of 700–1 300 m on the mid-continental slope (Branch, 2001). The depth distribution of orange roughy means they live at the limit of the range of sunlight penetration at mesopelagic depths and in complete darkness in bathypelagic depths, and in the colder (typically, < 10 °C) and more uniform water well below the thermocline. As a result, orange roughy are commonly referred to as being a "deep-sea" or deepwater fish.

Orange roughy can be ubiquitous in the deep sea, and caught on both flat (including the slope) and the complex seabed topographies of UTFs. On flat seabed, orange roughy are usually caught at relatively low densities, either targeted or as valuable bycatch in longer trawl tows with low catch per unit effort (CPUE). Large aggregations of orange roughy can also occur throughout the year, associated with spawning during the winter, and feeding outside of this time, and fished by highly directed trawls of short duration with high CPUE. The aggregations for feeding are typically associated with UTFs such as ridges, pinnacles, hills, knolls and seamounts. Spawning aggregations can be associated with UTFs, or over flat seabed, and there can be marked sex segregation within spawning aggregations (Pankhurst, 1988) and these aggregations can contribute a significant proportion of the overall catch (e.g. Clark and O'Driscoll, 2003; Clark, 2009; Anderson and Dunn, 2012). Spawning aggregations usually disperse within a few weeks of the end of spawning, and these outward, post-spawning migrations have been inferred to occur over large distances (Coburn and Doonan, 1997). The fish caught on UTFs are often larger than those caught on flat or sloping seabed (Dunn and Devine, 2010).

2.3 THE ECOLOGY OF ORANGE ROUGHY

Orange roughy eggs have been found in the mesopelagic habitat for a short time after spawning (Bulman and Koslow, 1995; Koslow *et al*., 1995a; Zeldis, Grimes and Ingerson, 1995; Zeldis *et al*., 1997; Branch, 2001), then early juveniles appear to have a benthic orientation, with benthopelagic movements suspected to increase as they become larger (Dunn and Forman, 2011). Orange roughy feed on benthopelagic and mesopelagic crustaceans, fish and squid (Branch, 2001). Their diet changes as they grow, with juveniles eating more small crustaceans, and adults eating more fish (Bulman and Koslow, 1992; Rosecchi, Tracey and Webber, 1988; Dunn and Forman, 2011; Forman, Horn and Stevens, 2016). Changes in diet composition have also been associated with changes in depth, area, year and water temperature (Bulman and Koslow, 1992; Rosecchi, Tracey and Webber, 1988; Dunn and Forman, 2011). Orange roughy feed on both flat and slope areas and in association with underwater features (Dunn and Forman, 2011). It has been assumed that the aggregations of orange roughy on features occur because of an improved access to mesopelagic food sources, combined with easier access to seabed refuges (Rowden *et al*., 2010a; Williams *et al*., 2010).

There are few known predators of orange roughy. Those that are known include sperm whales (Gaskin and Cawthorn, 1967), deep-sea sharks (Hallett and Daley, 2011, Pethybridge, Daley and Nichols, 2011), and very occasionally bony fish such as ling (*Genypterus blacodes*) and other orange roughy (Stevens, Hurst and Bagley, 2011).

Orange roughy show extreme longevity for a vertebrate, and have been validated to live to about 100 years (Andrews, Tracey and Dunn, 2009). Although age estimation has substantial uncertainty, several individuals have been aged at about 150 years (e.g. Payá *et al*., 2005; Doonan, Horn and Krusic-Golub, 2013a, 2013b; Doonan, Horn and Ó Maolagáin, 2014a, 2014b), and some have even been estimated to have approached 200 years old (e.g. Dunn, 2005). Orange roughy grow slowly, first reaching maturity at about 30 cm standard length (SL), and often 30 years or more in age, with females eventually growing larger than males (Branch, 2001). Orange roughy produce a relatively small number of eggs (Clark, Fincham and Tracey, 1994). The life history of orange roughy, in particular their high longevity, makes their stocks particularly unproductive.

2.4 ORANGE ROUGHY ON THE PLATE

Orange roughy produce a firm white fillet with no central red muscle band and with a mild flavour. Historically, the main international market has been the United States of America, but in recent years China has also become important, with a related change in demand from skinned fillets to whole fish. Orange roughy contain indigestible wax esters, which can cause diarrhoea and other acute gastro-intestinal symptoms when eaten (Branch, 2001; But, Ling and Cheng, 2008). The esters accumulate under the skin of the fish, but "deep skinning" filleting removes the ester-rich layer and substantially improves the palatability of the filleted product. When eaten whole, the fish must be correctly prepared (steamed).

Orange roughy are long-lived, and as a result, they naturally accumulate heavy metals, of which mercury is of particular concern for human health (Bosch *et al*., 2015). On average, the level of mercury in orange roughy flesh is about 0.5 $\mu g\ g^{-1}$ wet weight and is similar to the maximum level allowed for human consumption in Australia (van den Broek and Tracey, 1981), but lower than the maximum limit set by the European Union (1.0 $\mu g\ g^{-1}$ wet weight, Julshamn *et al*., 2011). The level of mercury in orange roughy is similar to that found in tunas, but substantially lower than found in many sharks, which can exceed 2 $\mu g\ g^{-1}$ wet weight (Bosch *et al*., 2015).

Orange roughy is favoured by processors and the retail trade, due, as noted above, to the mild flavour. Orange roughy is also able to be part-processed, frozen, defrosted, and reprocessed a number of times, without noticeable loss in quality. This attribute enhances the ability of processors and retailers to develop and market added value products for this high value fish.

3. Orange roughy fisheries, their exploration and development

3.1 INTRODUCTION TO FISHERIES EXPLORATION AND DEVELOPMENT

Orange roughy have a worldwide distribution as described by Branch (2001). Commercial fisheries of varying size have been established in a number of areas, notably around New Zealand, Australia, Namibia, Chile, and in the northeast Atlantic. The exploration and development of the fisheries in each of these areas is described in this section. Other areas have also seen some exploration for orange roughy, with reported catches in some areas but not in commercial quantities. For example, Kulka, Themelis and Halliday (2003) report orange roughy in the northwest Atlantic off Baffin Island and Newfoundland between 1982 and 2000, with catches reported as hundreds of individuals. All fisheries have targeted orange roughy using demersal trawls.

3.1.1 Australia

The origins and development of the Australian orange roughy fishery are well documented (Koslow *et al.*, 1997; Branch, 2001; Bax *et al.*, 2005). Orange roughy were first recorded in trawl surveys off New South Wales in 1972, with the first, small commercial catches made in the Southern and Eastern Scalefish and Shark Fishery in 1982. The first large aggregation was discovered off western Tasmania in 1986, and catches there rapidly increased. Between 1986 and 1988, several other non-spawning aggregations were discovered in Australian waters, and landings increased to 4 600–6 000 tonnes per year (Table 1).

TABLE 1

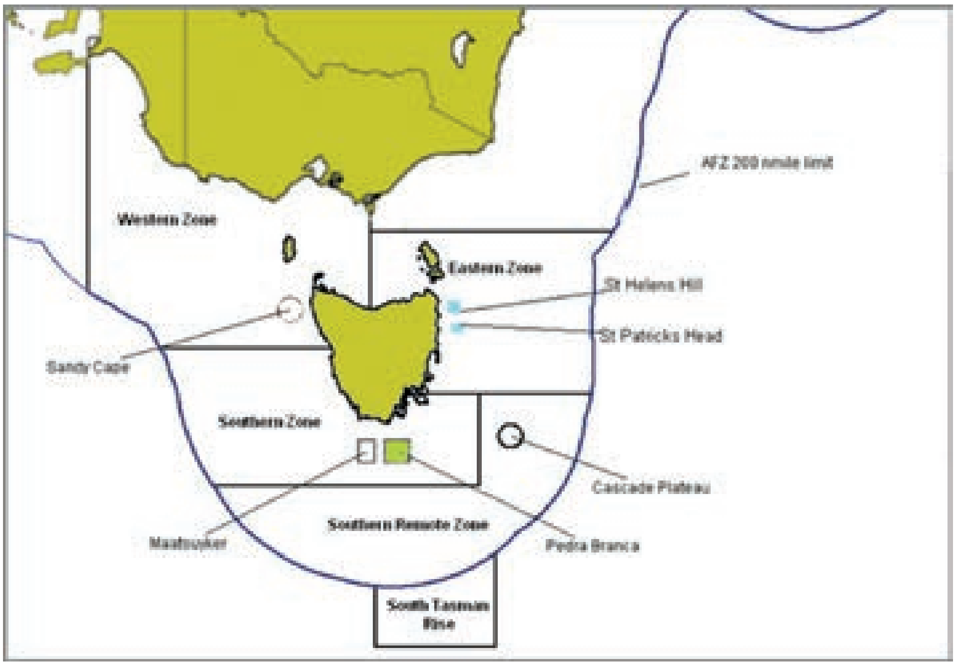
Reported catches, from logbooks (1985–1991) and landings (1992–2014), and agreed catch history of orange roughy for the East, South and West Management Zones and for Pedra Branca in the South, Australia

Year	East		East and Pedra Branca	Pedra Branca Only	South (including Pedra Branca)		West
	Reported	Agreed	Agreed	Agreed	Reported	Agreed	Reported
	(tonnes)						
1985	6	6	6	0	58	58	129
1986	33	33	60	27	631	631	3 970
1987	310	310	310	0	353	353	5 128
1988	1 949	1 949	1 949	0	469	469	4 765
1989	18 365	26 236	28 575	2 339	7 620	10 886	1 386
1990	16 240	23 200	34 502	11 302	24 801	35 430	802
1991	9 727	12 159	20 436	8 277	11 541	14 426	628
1992	7 484	15 119	24 265	9 146	7 947	16 054	1 141
1993	1 971	5 151	8 798	3 647	7 602	5 486	1 031
1994	1 682	1 869	4 140	2 271	4 345	4 828	927
1995	1 959	1 959	2 544	585	2 157	2 157	1 055
1996	1 998	1 998	2 231	233	802	802	1 320
1997	2 063	2 063	2 250	187	454	454	352

Year	East		East and Pedra Branca	Pedra Branca Only	South (including Pedra Branca)		West
	Reported	Agreed	Agreed	Agreed	Reported	Agreed	Reported
	(tonnes)						
1998	1 968	1 968	2 087	119	250	250	360
1999	1 952	1 952	2 052	100	174	174	244
2000	1 996	1 996	2 109	113	311	311	192
2001	1 823	1 823	2 027	204	357	357	248
2002	1 584	1 584	1 674	90	167	167	294
2003	772	772	877	105	210	210	243
2004	767	767	797	30	80	80	321
2005	754	754	772	18	99	99	281
2006	614	614	615	1	5	5	159
2007	113	113	129	16	22	22	31
2008	98	98	98	0	0	0	5
2009	193	193	193	0	10	10	16
2010	113	113	113	0	18	18	27
2011	160	160	162	2	17	17	37
2012	163	163	163	0	22	22	20
2013	150	150	150	0	8	8	45
2014	20	20	20	0	20	20	20

Notes: All seasons are included. The agreed catch history incorporates adjustments for fish lost due to lost gear and burst bags/panels, etc., as well as for misreporting. 2014 catches are estimates based on the landings as at October 2014.
Sources: CSIRO and TDPIF (1996); Wayte (2007); Upston *et al*. (2014).

FIGURE 2
Map of Australian orange roughy management zones and fishing areas



Note: The largest fisheries are located at St Helens Hill and St Patricks Head (blue) in the Eastern Zone, and Pedra Branca (green) in the Southern Zone.
Source: Upston *et al*. (2014).

Large spawning aggregations were found at St Helen's Hill, a seamount off eastern Tasmania, and non-spawning aggregations were found in the Pedra Branca and Maatsuyker areas off southern Tasmania (Figure 2), which resulted in further growth in the fishery in 1989. Catches were mainly from the Eastern and Southern Zones and increased from about 2 000 tonnes in 1988 to more than 58 000 tonnes in 1990, mirroring the rapid rise in catches seen in other orange roughy fisheries. Catches from the Eastern Zone and the Pedra Branca area of the Southern Zone for the four years 1989–1992 exceeded 100 000 tonnes. In this period, catches from the Cascade Plateau and the Great Australian Bight also peaked at 1 800 tonnes in 1990 and 3 400 tonnes in 1989. The landed value of the fishery was about USD 47 million per annum in 1989–1990 and was the most valuable fishery in Australia at that time. The introduction of management zones and enforceable catch quotas in 1992 prevented further increases in catch, and a managed reduction in catch limits began. Acoustic surveys to estimate orange roughy biomass were started in the Eastern Zone in 1990 (Kloser, Koslow and Williams, 1996).

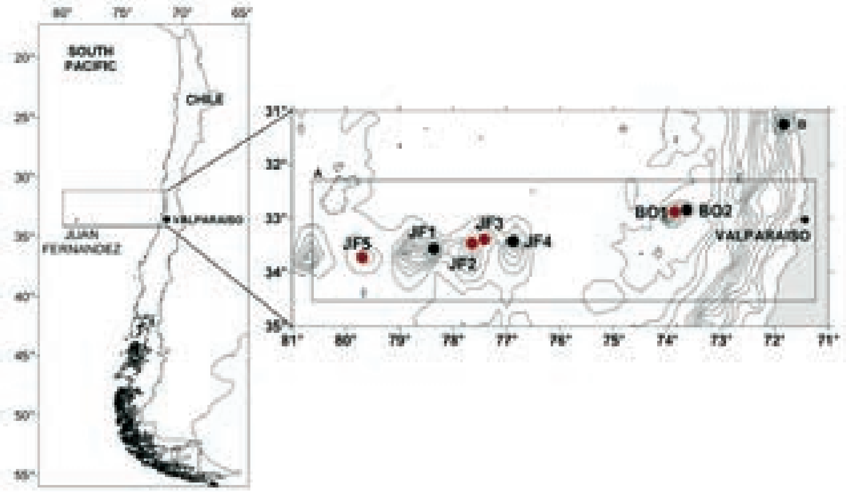
There followed a period of gradual reduction in Eastern Zone TACs from 2 000 tonnes in 1993 to 720 tonnes in 2005. The TACs in the other zones were also reduced over this period, with the exception of the Cascade Plateau, where the TAC was 1 600 tonnes annually between 1998 and 2005 and was subsequently reduced to 700 tonnes in 2006 (Patterson *et al*., 2015). The 2006 Eastern Zone stock assessment estimated spawning biomass had declined to 10 percent of the unfished level (Wayte, 2007). In 2006, with most stocks estimated to be below 20 percent of unfished biomass, orange roughy were listed as conservation dependent in Australian waters, and all target fishing (with the exception of that in the Cascade Plateau Zone) was prohibited (AFMA, 2006). A five-year conservation plan was adopted in 2007, and this was replaced by a rebuilding strategy in 2015 (AFMA, 2015). Long-term biological monitoring and acoustic biomass surveys indicated signs of recovery for orange roughy in Australian waters following these management actions (Kloser *et al*., 2015), and orange roughy in the Eastern Zone has recently been assessed to be above 20 percent of the unfished level (Upston *et al*., 2014). The Eastern Zone orange roughy fishery was re-opened in 2015 and an annual TAC of 465 tonnes was set for three years for the Eastern Zone and 35 tonnes for Pedra Branca within the Southern Zone (AFMA, 2015).

The most recent formal stock assessments of the Southern Zone (2000) and Western Zone (2002) orange roughy estimated the stocks at that time to be at 7 percent and 8 percent of unfished biomass, respectively, while Upston *et al*. (2014) reported the low point for the Eastern Zone stock to be about 12 percent of unfished biomass while its relative biomass level in 2015 was estimated to be 25–26 percent of unfished biomass and, therefore, was no longer considered overfished. The Cascade Plateau stock was estimated to have never been overfished, with the stock at about 60 percent of unfished biomass in 2011 (Morison *et al*., 2012). The present status of the Southern and Western Zone stocks is unknown.

3.1.2 Chile

The development of the Chilean orange roughy fishery is described by Payá *et al*. (2005) and Payá (2013). Aggregations of orange roughy that would support commercial fisheries were first found in Chilean waters in 1997–98. A series of exploratory scientific surveys were conducted in the following years and found orange roughy aggregations in only a small number of locations. The first two scientific surveys explored 23 seamounts in 1998 and found orange roughy on just five. These five locations were found in three areas of seamounts: the Juan Fernandez Archipelago, Bajo O'Higgins, and Punta Sierra. A third survey, in 1999, explored eight seamounts, with no orange roughy found, and a fourth survey in 2000 explored one seamount where orange roughy were found. The final exploratory survey, in 2003, visited two seamounts and found orange roughy in very low numbers.

FIGURE 3
Chilean orange roughy fishing grounds by area



Note: (JF = Juan Fernandez Archipelago, BO = Bajo O'Higgins and, B = Punta Sierra).
Source: Payá *et al.* (2005).

Following the exploratory surveys, eight seamounts were fished commercially for orange roughy: five at the Juan Fernandez Archipelago, two at Bajo O'Higgins, and one at Punta Sierra (Figure 3), with the development of overall orange roughy catches in Chilean waters given in Table 2. Between 1999 and 2003, most orange roughy (85 percent) were caught at the Juan Fernandez Archipelago, followed by Punta Sierra (10 percent) and Bajo O'Higgins (6 percent). The fishery was focused on and around the spawning season, where orange roughy aggregations associated with seamounts were targeted and catch rates were highest, with the fishing season lasting three or four months a year. Fish were typically caught towards the tops of the seamounts (Figure 4). All orange roughy caught were adults, and very few juveniles have ever been reported from Chilean waters (Lafon *et al*., 2010). Biomass estimates were made using the survey data (Roa and Nitlitschek, 2007).

Scientific observers collected logbook data describing commercial catch and effort for the fishery. Unstandardized catch rates were low in the first year as skippers were still learning how to fish for orange roughy. Catch rates doubled in the second year, and increased again in the third year. Only a few commercial vessels were involved in the fishery. Catch rates then decreased over the next two years of the fishery. In 2006, the commercial fishery was closed, with the closure accepted by the fishing industry as the catch rates and biological allowable catches were considered too low for the fishery to be economically viable. In 2012, a new Chilean Fishery Act made it illegal to trawl in vulnerable environments, such as may be encountered on seamounts, until it can

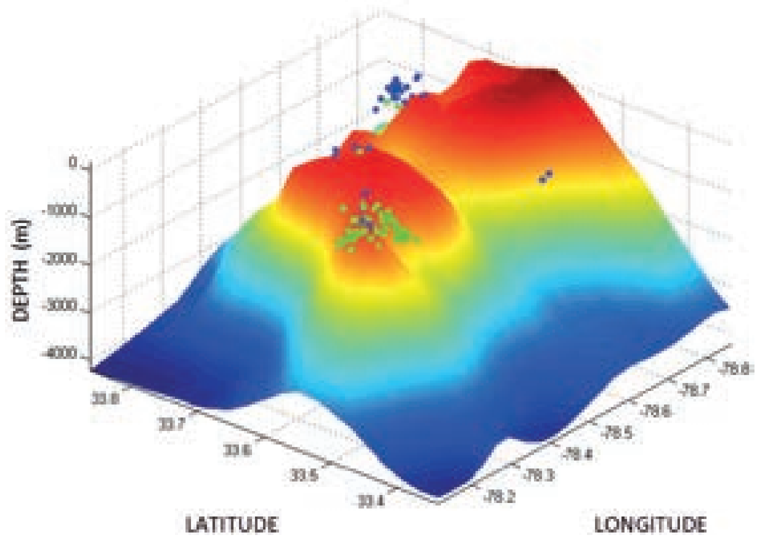
TABLE 2
Total allowable catch (TAC) and reported catch for the Chilean orange roughy fishery from 1999.

Year	TAC	Catch
	(tonnes)	
1999	1 500	726
2000	1 580	1 271
2001	2 140	2 109
2002	2 500	1 864
2003	2 500	1 300
2004	2 500	1 525
2005	2 000	830
2006	500	347
2007	300	0
2008	0	0
2009	0	0
2010	0	0
2011	0	0
2012	0	0

Note: The commercial fishery has been closed since 2006.
Source: Payá (2013).

be demonstrated that a trawl fishery will not cause adverse effects to the benthic ecosystem. In 2016, the ban on fishing for orange roughy (and some other deepwater species) was extended for a further five years, following the recommendations of a scientific committee, which considered the stock status, longevity and low productivity of these stocks (SUBPESCA, 2016).

FIGURE 4
Trawl locations on seamount JF1 (Juan Fernandez Archipelago) during May, 1999–2004



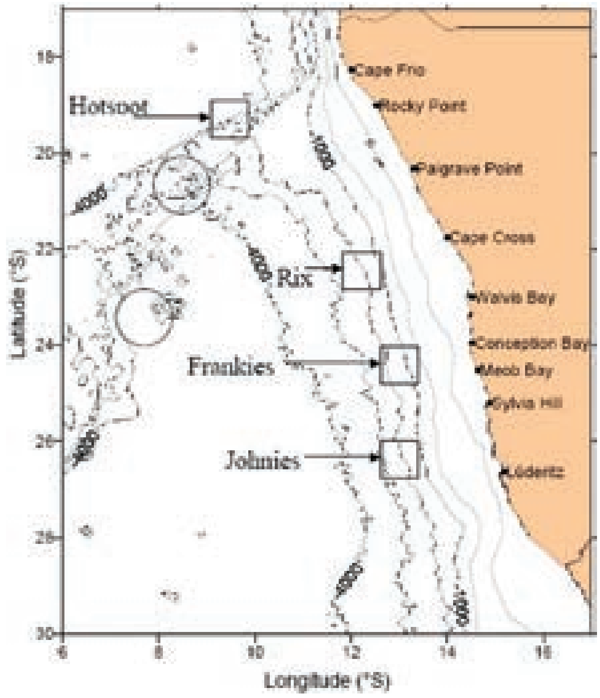
Note: Green dots – orange roughy; blue dots – alfonsino.
Source: Payá (2013).

During the time the fishery operated, there was little change in the lengths of orange roughy caught. The length frequency distributions were strongly unimodal, females were more commonly caught than males, and juvenile fish were generally absent. Chilean orange roughy in catches were relatively large, with the catch typically comprising fish in the range 32–49 cm SL (Payá *et al.*, 2005).

3.1.3 Namibia

Little information is available on the history or character of the Namibian fisheries, although stock assessments and management up until the early 2000s have been well described (Kashindi, 1999; Branch, 2001; Boyer *et al.*, 2001; McAllister and Kirchner, 2001, 2002; Brandão and Butterworth, 2005). Following initial exploratory fishing off South Africa in 1994, four distinct areas were defined off Namibia: Hotspot and Johnies in 1995, and Rix and Frankies in 1996 (Figure 5).

FIGURE 5
Map of the orange roughy fishing grounds off Namibia.



Source: McAllister and Kirchner (2002)..

TABLE 3
Catches and TACs for the orange roughy fishery in Namibia.

Year	TAC	Catch
	(tonnes)	
1994/95	–	1 872
1995/96	–	6 288
1996/97	–	17 381
1997/98	12 000	14 729
1998/99	12 000	10 040
1999/00	6 000	2 699
2000/01	1 875	1 344
2001/02	1 875	874
2002/03	2 400	1 985
2003/04	2 650	1 730
2003/05	2 600	1 106
2005/06	2 050	297
2006/07	1 100	429
2007/08	900	288

Note: By split year, as is normal for Southern Hemisphere fisheries.

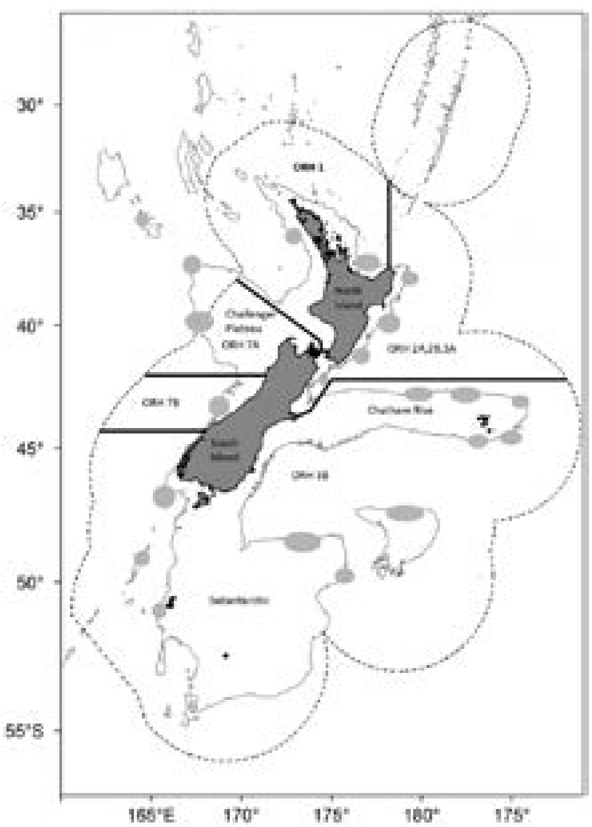
Catches increased from the start of the fishery in 1994/95, peaking at 17 381 tonnes in 1996/97, subsequently constrained by the application of a commercial TAC from 1997/98 (Table 3). The first major scientific research to support management was the development of swept-area biomass estimates in 1997, and both swept-area and acoustic estimates in 1998 (Kashindi, 1999).

The fishery was closed in 2008 and has remained closed since. A research survey conducted in 2016 has been reported (Esau, 2017) and there is therefore a possibility that commercial fishing will be restarted if there has been rebuilding of the stocks.

3.1.4 New Zealand

An extensive literature is available that describes the New Zealand orange roughy fisheries, with an annually updated overview published by the Ministry for Primary Industries (MPI) (see MPI, 2016a), and especially detailed descriptions of the fisheries in frequent ministry-published fisheries research reports (e.g. Anderson and Dunn, 2012), these two sources form the basis of this section. There have also been additional publications over the years that have summarized the history of, and broad issues concerning, the New Zealand orange roughy fisheries (e.g. Robertson, 1991; Clark and Tracey, 1994; Clark, 2001; Francis and Clark, 2005; Dunn, 2007), and a recent book providing perspectives and an overview of the development of the fisheries (Pankhurst, 2017).

FIGURE 6
The New Zealand region



Note: The map shows the Exclusive Economic Zone (broken line), management boundaries between main fishery regions (black solid lines), the 1 000 m isobath (grey solid lines), and indicative locations of the main orange roughy fisheries (grey shaded areas).

Orange roughy have been fished at many locations around New Zealand. The major fisheries are found around central New Zealand, on the Chatham Rise, Challenger Plateau, and off the southeast coast of the North Island (Figure 6 and Table 4). They have been well studied, with more than 130 dedicated research surveys completed, and numerous associated scientific studies.

New Zealand was the first country to develop orange roughy fisheries. This started in the late 1970s and early 1980s, and these were the only substantial orange roughy fisheries for several years. The fisheries started on the northern flank of Chatham Rise, and off the northeast coast of the South Island. The fisheries initially focused on targeting large aggregations of spawning orange roughy on flat areas of the continental shelf. As the stocks in these initial fisheries became depleted, and improvements to vessel positioning took place (in particular the advent of the Global Positioning System technology in the mid-1990s), the fisheries moved to also target aggregations of orange roughy on UTFs, such as canyons, ridges, hills, knolls, and seamounts, both during and outside of the spawning season.

Orange roughy has been a primary target species for New Zealand deep-sea, demersal trawl fishing. Other commercially valuable bycatch species are associated with orange roughy, including smooth oreo dory (*Pseudocyttus maculatus*), black oreo (*Allocyttus niger*), black cardinalfish (*Epigonus telescopus*), and less often, alfonsino (*Beryx splendens*) at the shallower end of the orange roughy depth distribution.

The main fishery areas are generally treated as separate management units, which are combinations, or subdivisions, of the standard New Zealand Fishery Management Areas (FMAs), and each may contain several geographically and/or temporally separate fisheries (Figure 6).

Since the introduction of the New Zealand Quota Management System in 1986, catches have been limited by the setting of TACs and total allowable commercial catches (TACCs) (Table 4). This system also accounts for any customary or recreational catches (which are considered to be zero for orange roughy). Overall reported catches peaked at about 50 000 tonnes a year between 1983 and 1990, and then declined to < 20 000 tonnes from 1994, and < 10 000 tonnes from 2009. Although the TACCs and agreed catch limits for individual FMAs or subareas of FMAs have restricted catches, this has not been the case for the overall New Zealand catch until recently (Table 4). Historically, true catches are known to have been greater than reported catches in a number of fishery areas. This was due to fish escaping from burst nets, through escape panels, or through errors in defining conversion factors (between, for example, fillets and whole fish). These additional catches have been between 5 percent and 50 percent of reported catches and have varied by stock and over time (Table 4), and they have been incorporated in the total catch estimates used in stock assessments (MPI, 2016a). Such catch over-runs are now considered to be minimal, with 5 percent usually being assumed.

TABLE 4: **New Zealand reported catches, TACCs, and estimated catches including over-runs, by fishery and fishing split year**

Fishing year	ORH1 Reported catch	ORH1 TACC	ORH1 With over-run	ORH 2A, 2B, 3A Reported catch	ORH 2A, 2B, 3A TACC	ORH 2A, 2B, 3A With over-run	ORH 3B Reported catch	ORH 3B TACC	ORH 3B With over-run	ORH 7A Reported catch	ORH 7A TACC	ORH 7A With over-run	ORH 7B Reported catch	ORH 7B TACC	ORH 7B With over-run	Total Reported catch	Total TACC	Total With over-run
	(tonnes)																	
1980	0	–	0	0	–	0	11 800	–	15 340	0	–	0	0	–	0	11 800	–	15 340
1981	0	–	0	0	–	0	31 100	–	40 430	33	–	43	0	–	0	31 133	–	40 473
1982	0	–	0	554	–	720	28 200	23 000	36 660	4 248	–	5 522	0	–	0	33 002	23 000	42 903
1983	<1	–	NA	3 763	–	4 892	32 605	23 000	42 387	11 839	–	15 391	0	–	0	48 207	23 000	62 669
1984	<1	–	NA	7 401	–	9 654	32 535	30 000	42 296	9 527	–	12 385	2	–	NA	49 465	30 000	64 336
1985	96	–	NA	8 438	–	10 322	29 340	30 000	38 142	5 117	–	6 652	282	–	NA	43 273	30 000	55 494
1986	2	–	NA	8 012	8 218	10 979	30 075	29 865	38 496	7 753	6 190	10 079	1 763	1 558	NA	47 605	39 641	61 319
1987	<1	10	NA	8 705	9 242	11 835	30 689	38 065	38 668	11 492	10 000	14 940	1 446	1 558	NA	52 332	58 875	66 889
1988	0	10	0	9 731	9 242	12 650	24 214	38 065	30 025	12 181	12 000	15 835	1 413	1 558	NA	47 539	60 875	59 924
1989	19	10	NA	9 520	10 266	11 900	32 785	38 300	39 998	10 241	12 000	12 801	1 750	1 708	NA	54 315	62 284	66 468
1990	86	190	NA	10 537	10 352	12 644	31 669	32 787	38 003	4 309	2 500	5 171	1 711	1 708	NA	48 312	47 537	57 615
1991	200	190	NA	10 001	10 352	11 501	21 521	23 787	32 282	1 357	1 900	20 355	1 683	1 708	NA	34 762	37 937	66 021
1992	112	190	NA	10 117	10 532	11 129	23 269	23 787	25 596	1 911	1 900	2 102	1 604	1 708	NA	37 013	38 117	40 543
1993	49	190	NA	9 052	10 632	9 957	20 048	21 300	22 053	2 087	1 900	2 296	1 129	1 708	NA	32 365	35 730	35 484
1994	189	190	NA	10 000	10 333	11 000	16 960	21 300	18 656	1 732	1 900	1 819	701	1 708	NA	29 582	35 431	32 365
1995	244	190	NA	8 642	9 660	9 074	11 891	14 000	12 486	1 636	1 900	1 718	290	1 708	NA	22 703	27 458	23 811
1996	965	1 190	NA	5 125	5 100	5 381	12 501	12 700	13 126	1 669	1 900	1 752	446	430	NA	20 706	21 320	21 671
1997	1 021	1 190	NA	4 613	5 100	4 844	9 278	12 700	9 742	1 308	1 900	1 373	425	430	NA	16 645	21 320	17 405
1998	511	1 190	NA	4 651	5 100	4 884	9 638	12 700	10 120	1 502	1 900	1 577	330	430	NA	16 632	21 320	17 422
1999	845	1 190	NA	4 174	4 600	4 383	9 372	12 700	9 841	1 249	1 425	1 311	405	430	NA	16 045	20 345	16 785
2000	771	1 190	NA	3 971	4 600	4 170	8 663	12 700	9 096	629	1 425	660	284	430	NA	14 318	20 345	14 981
2001	858	800	NA	2 054	1 700	2 157	9 274	12 700	9 738	<1	1	0	161	430	NA	12 347	15 631	12 914
2002	1 294	1 400	NA	1 667	1 700	1 750	11 325	12 700	11 891	<1	1	0	95	110	NA	14 381	15 911	15 031
2003	1 123	1 400	NA	1 122	1 000	1 178	12 333	12 700	12 950	4	1	4	90	110	NA	14 672	15 211	15 345
2004	986	1 400	NA	1 056	1 000	1 109	11 254	12 700	11 817	<1	1	0	119	110	NA	13 415	15 211	14 031
2005	1 151	1 400	NA	1 742	1 700	1 829	12 370	12 700	12 989	<1	1	0	106	110	NA	15 369	15 911	16 075
2006	1 207	1 400	NA	1 663	1 700	1 746	12 554	12 700	13 182	<1	1	0	77	110	NA	15 501	15 911	16 212
2007	1 036	1 400	NA	1 736	1 700	1 823	11 271	11 500	11 835	<1	1	0	125	110	NA	14 168	14 711	14 818
2008	1 104	1 400	NA	1 709	1 700	1 794	10 291	10 500	10 806	<1	1	0	6	1	NA	13 110	13 602	13 710
2009	905	1 400	NA	1 701	1 700	1 786	8 758	9 420	9 196	<1	1	0	1	1	NA	11 366	12 522	11 888
2010	825	1 400	NA	1 720	1 700	1 806	6 662	7 950	6 995	<1	1	0	<1	1	NA	9 207	11 052	9 626
2011	772	1 400	NA	1 690	1 700	1 775	3 486	4 610	3 660	476	500	500	<1	1	NA	6 424	8 211	6 707
2012	1 114	1 400	NA	1 145	1 430	1 202	2 765	3 600	2 903	511	500	537	<1	1	NA	5 535	6 931	5 756
2013	1 171	1 400	NA	1 124	1 130	1 180	2 515	3 600	2 641	513	500	538	<1	1	NA	5 322	6 631	5 531
2014	1 055	1 400	NA	1 171	1 430	1 230	4 492	4 500	4 717	497	500	522	<1	1	NA	7 216	7 831	7 524
2015	1 181	1 400	NA	693	725	728	4 747	5 000	4 984	1 594	1 600	1 674	2	1	NA	8 217	8 568	8 568
Total	20 892	–	–	159 000	–	185 011	539 350	–	637 972	95 383	–	137 516	16 447	–	–	874 006	–	1 053 652

Note: Catches are rounded to the nearest tonne. Fishing year is given as the end year, i.e. 2015 refers to the year 1 October 2014 to 30 September 2015 NA, not estimated.

Source: MPI (2016a).

It has been recognized in New Zealand for some time that there has been some level of historical fishing mortality that has not been reported from most orange roughy fisheries. These additional catches were not reported either because catch was lost at sea, or because of discrepancies in processed weights and conversion factors (e.g. converting headed and gutted weight to fresh weight). These unreported catches have been subsequently estimated as "catch over-runs" and have been added to the catch estimates used in the stock assessments to provide the best input data for assessments on which to base management advice (MPI, 2016a) (Table 4). In the early years of the fishery, large spawning aggregations were targeted, where it was relatively common for trawls to catch more than desired with catch rates exceeding 10 tonnes per minute, resulting in burst or damaged nets, or discarding of damaged fish. Greater skipper experience in the orange roughy fishery, as well as technological improvements (e.g. higher-resolution net monitors, increased GPS satellite coverage and catch sensors) contributed to substantially reduced catch over-runs by the early 1990s. When over-runs are included, the estimated total catch of orange roughy around New Zealand, from 1980 to 2015, is slightly more than one million tonnes (Table 4).

In the sections below, the main characteristics and changes in each of the New Zealand fisheries are described, summarizing the extensive information available in the literature and recent stock assessment reports, in particular MPI (2016a) and Anderson and Dunn (2012).

3.1.5 North of the North Island (ORH 1)

The targeted orange roughy fishery commenced in the early 1990s, with vessels targeting aggregations on several hills in the central Bay of Plenty (eastern ORH 1). Target fisheries were then developed on features in the northern Bay of Plenty, followed by those around the north of the North Island (from the mid-1990s to the late 1990s). The fisheries off the northwest coast of the North Island, both close to the mainland and at the edge of the EEZ, started in the early 2000s. New fishing areas have continued to be important to this fishery, although there is not a strong pattern of sequential depletion and most areas have continued to be fished, partly because of spatial restrictions limiting catches from specific features. Because the fishery has focused upon fishing features, most trawl tows are short, lasting 30 minutes or less. The catches peaked at 1 128 tonnes in 1998, then declined and stabilized at about 500–600 tonnes a year (Table 4).

In the early years of the fishery, catches and effort were highest in March, June and July, with the latter months coinciding with the occurrence of orange roughy spawning aggregations. In more recent years, fishing has been focused in October and November, and during the June–July spawning period (Dunn, 2017).

The number of vessels in the fishery peaked in 1996, although fishing effort reached a maximum in 1998 (Table 5).The CPUE has fluctuated considerably, influenced by declines in some established fisheries, good catch rates in new fisheries, and in some cases variable catch rates (Anderson and Dunn, 2012). The CPUE peaked in 2001 and 2006, and then trended slightly downwards to a level close to the long-term average for the fishery.

TABLE 5

Nominal fishing effort, measured as number of vessels that reported any catch and effort in the fishery, and the number of trawl tows reported

	ORH 1		ORH 2A, 2B, 3A		ORH 3B		ORH 7A		ORH 7B	
Fishing year	Tows	Vessels	Tows	Vessels	Tows	Vessels	Tows	Vessels	Tows	Vessels
1980	–	–	–	–	2 382	30	–	–	–	–
1981	–	–	–	–	4 790	38	7	2	1	1
1982	–	–	–	–	2 054	22	327	7	4	2
1983	–	–	256	11	3 228	32	411	18	6	2
1984	–	–	1 513	11	3 103	32	606	19	17	5
1985	–	–	2 280	18	3 124	31	764	11	90	10
1986	–	–	2 390	20	3 987	37	889	14	357	9
1987	–	–	1 952	19	4 209	35	1 270	21	405	11
1988	–	–	1 709	14	4 557	39	1 746	23	420	9
1989	–	–	779	20	5 492	33	1 402	23	368	10
1990	22	2	2 941	16	4 127	34	571	16	356	4
1991	2	2	3 304	17	3 231	32	294	8	632	9
1992	1	1	3 983	16	3 013	35	368	11	810	6
1993	23	4	4 303	27	3 312	28	540	17	784	12
1994	98	6	5 028	30	5 355	31	357	11	708	10
1995	149	7	5 389	34	3 794	30	386	12	361	11
1996	517	14	2 516	27	3 549	22	370	10	150	7
1997	759	11	2 390	29	3 416	29	529	12	182	9
1998	1 128	8	3 655	30	4 109	31	720	13	228	3
1999	944	7	3 730	30	3 942	36	905	12	566	1
2000	823	6	3 147	25	3 104	34	547	9	647	8
2001	231	6	1 397	24	3 542	30	1	1	431	10
2002	566	7	879	18	3 279	27	4	4	276	11
2003	717	8	773	18	3 891	30	15	3	231	4
2004	609	6	672	17	4 074	27	–	–	252	6
2005	585	8	930	14	3 621	26	55	3	393	6
2006	569	6	881	11	3 586	18	63	1	257	2
2007	591	5	736	11	3 069	15	2	2	167	4
2008	491	5	822	13	2 885	14	1	1	3	1
2009	490	4	851	10	2 699	8	76	1	1	1
2010	303	4	918	9	4 756	17	79	2	1	1
2011	375	4	1 045	8	3 033	17	113	4	1	1
2012	484	5	724	12	2 827	18	103	5	–	–
2013	479	4	637	9	2 128	19	157	7	1	2
2014	450	6	659	9	2 720	20	42	3	4	1
2015	416	6	434	7	2 554	19	434	5	9	3

Note: Fishing year is the year ending, i.e. 2015 refers to the year 1 October 2014 to 30 September 2015. Some fishing vessels are active in more than one FMA. The number of vessels for ORH 2A, 2B, and 3A, ORH 3B, and ORH 7B are minimum estimates (aggregate or complete statistics were not available). Data for 2010 and subsequent years are provisional.

Source: M. Dunn (unpublished MPI data).

3.1.6 East coast North Island (ORH 2A, 2B, 3A)

The east coast of the North Island (ORH 2A, 2B, 3A) was originally monitored and assessed as a single area, but later assumed to be separate stocks: the Mid-East Coast and East Cape. The separation between these two stocks has been made in ORH 2A, in order to separate a relatively discrete northern fishery (ORH 2A North, "East Cape") from the fisheries to the south (Figure 6). While there is no separation between the two stocks in terms of the TACC, industry manages a separation of catch limits for the two areas under an agreement with the ministry responsible (MPI, 2016a).

The targeted orange roughy fishery commenced in 1983, with vessels fishing off the southeast coast of the North Island until 1985, then farther south and in the central area (Ritchie Bank) in 1986, followed by the development of the hill areas to the south of Ritchie Bank in 1990–98 (Rockgarden), and then the development of the southern part of ORH 2A since the late 1990s. The fishery on and around Ritchie Bank focused on large orange roughy spawning aggregations occurring over relatively rough ground and hills. Nevertheless, catches have been made throughout the year, with most fishing during the spawning period between May and June, and very little during July–September. The East Cape fishery was developed as the Ritchie Bank fishery declined, and saw vessels move north to target newly found aggregations in that area. However, the large initial catches were relatively short-lived, declining from about 3 400 tonnes in 1994 to 300 tonnes in 2001, and remaining relatively small (< 250 tonnes) thereafter. Since 2005, about 12 percent of the catch from ORH 2A, 2B and 3A has been from East Cape. The catch over-runs have been estimated at about 50 percent for the fishery around Ritchie Bank in 1984–86, which is the highest catch over-run rate in New Zealand, but the over-runs declined to 10 percent or less for all parts of the fishery from 1992.

The vessels active in the fishery have generally been smaller than found in fisheries operating farther offshore, such as in the ORH 3B or ORH 7A fisheries. Both catches, effort and the number of vessels peaked during the mid-1990s, with effort and catches declining substantially in the early 2000s following a reduction in the TACC. Further reductions in the TACC, and catch, took place after 2011, following advice based on a stock assessment that indicated the stock was depleted (Anderson and Dunn, 2011). The number of vessels subsequently declined to a historical low in 2009.

The CPUE in the Mid-East Coast fishery declined rapidly and substantially in the first few years of the fishery, then was maintained until the mid-1990s, when it again declined and then remained at a relatively low level, with some suggestion of an increase since 2000. The CPUE for East Cape declined between the start of the fishery and 2003, with subsequent effort relatively low and catches sporadic.

3.1.7 Chatham Rise and Sub-Antarctic (ORH 3B)

The orange roughy fishery in ORH 3B has been the largest around New Zealand, with a complex fishing history and extensive history of research. The fishery started in 1979 on the northern flank of Chatham Rise, focused on an area of the northeast now known as the “Spawning Box”, where large spawning plumes occurred on flat grounds in July, but orange roughy in this area were scarce outside of the spawning season. In the 1980s, the fishery peaked in July, but also targeted pre- and post-spawning fish on flat ground (slope). Catches from both slope and feature areas of the western part of north and south Chatham Rise also grew rapidly in the early 1980s. Nominal fishing effort (number of tows) increased by about 50 percent between 1979–1981 and 1988–1990, with an increasing amount of the effort taking place around and on UTF complexes on south and east Chatham Rise. The importance of areas outside of the Spawning Box on Chatham Rise increased as the fishery in the Spawning Box declined in the early 1990s, with a temporary closure of the Spawning Box to fishing in 1993 and 1994. In the 1990s, the fishery was year-round, and became focused on UTFs. Through the 1980s and 1990s, there was an easterly movement of the fishery on features, as new fishing grounds were discovered, and a serial depletion of feature fisheries took place on southern Chatham Rise (but all from the same assumed stock). By the late 1990s, the feature fishery was centred outside of the spawning season on feature complexes on the southeast corner of Chatham Rise. As in all orange roughy trawl fisheries, tows on features have tended to be short, typically 30 minutes or less, with the gear hitting the bottom near the top of the feature and being towed down the flank. Since 2002, the catches from the northwest, south and east of Chatham Rise have declined, and the Spawning Box has again come to dominate the fishery.

Catch over-runs have been assumed for the Chatham Rise fishery and peaked during the 1980s, reducing to 10 percent or less from 1992. On Chatham Rise, the assumed number of discreet stocks has varied between one and five. It is currently thought that there are two, the Northwest Chatham Rise stock, and the East and South Chatham Rise stock, each having a separate, agreed catch limit under the overall ORH 3B TACC.

The ORH 3B area has been subdivided for fisheries management at 46°S, separating Chatham Rise from the rest. Non-Chatham Rise catches increased with the discovery of the Puysegur Bank grounds, off the southern tip of the South Island, in the early 1990s. The Puysegur Bank fishery was also focused on spawning aggregations and accounted for 30–40 percent of the entire ORH 3B catch for several years. While the fishery on Chatham Rise has been relatively stable spatially since the early 1990s, the fisheries in the Sub-Antarctic have shown serial depletion. In the Sub-Antarctic, the most persistent fisheries have been on Puysegur Bank (> 100 tonnes/year for 1990–97 and 2004–05), Auckland Islands (>100 tonnes/year for 1993–2000, 2002, and 2008–09), and north Pukaki Rise (> 100 tonnes for 2002–09) (data are not available after 2009). Some fisheries elsewhere have been intermittent or short-lived. Since 2002, the Sub-Antarctic fishery has been dominated by north Pukaki, peaking at 1 500 tonnes in 2006, which was 70 percent of the non-Chatham Rise catch, although this was only 12 percent of the ORH 3B catch.

For ORH 3B, concerns about sustainability resulted in agreed annual catch limits in 2010 to 2013 of 3 860 tonnes, 2 850 tonnes, and 2 850 tonnes, which were lower than the regulatory TACC. In these years, the excess TACC was "shelved" (i.e. the catch entitlement was passed to a third party, so it was not lost, but also could not be legitimately caught). In the Sub-Antarctic, the stock structure has not been formally evaluated, but Puysegur Bank and north Pukaki have been subject to agreed catch restrictions, and other discrete areas of fishing have been monitored separately.

The Spawning Box is the main site of reproduction, but it not the only one. Substantial spawning aggregations also occur on the Graveyard Hills complex within the Northwest Chatham Rise stock, with at least ten other spawning locations throughout ORH 3B (Dunn and Devine, 2010; MPI, 2016a). A presumed new spawning plume, named "Rekohu", was first detected in 2010 and first surveyed to estimate biomass in 2011. This plume occurs about 50 km to the west of the Spawning Box and within the East and South Chatham Rise stock boundary.

The fishery in the Spawning Box in the 1980s and early 1990s greatly reduced the stock biomass and also greatly reduced the spatial extent of the spawning aggregation (Clark *et al*., 2000; Dunn, Anderson, and Doonan, 2008). The Spawning Box fishery has maintained the highest CPUE, and it remained stable for most of the 1980s, before declining in the early 1990s. The CPUE on features typically declined rapidly in the first few years that each feature was fished (a five-fold decrease, or more, being typical), and then remained at relatively low levels. Some areas showed continued slow decline in CPUE (e.g. the Andes feature complex of the southeast Chatham Rise), whereas others remained stable (e.g. areas of the south Chatham Rise), and after a period of decline, some areas also saw some increase in CPUE (e.g. Graveyard Hills complex on the northwest Chatham Rise). Declines in CPUE for feature fisheries in the Sub-Antarctic have tended to be greater than those on Chatham Rise, with ten-fold or greater declines in CPUE occurring for most areas. The number of vessels in the fishery peaked in the late 1980s (Table 5), and then was fairly stable during the 1990s and until a consolidation of the fleet took place from 2005, followed by reductions in TACC from 2009. On Chatham Rise, about ten vessels were operating in the fishery after 2000, whereas in the Sub-Antarctic only four vessels remained after this time.

In the early years of the fishery (1980–81), the mean length of males on Chatham Rise was 31–35 cm SL, with females typically 32–37 cm SL (Liwoch and Linkowski, 1986). In the Spawning Box, the median length of combined sexes was much the same between 1984 and 1994, at 35 cm, with mean length only declining slightly from 34.7 cm in 1984 to 33.9 cm in 1994 (Clark *et al*., 2000; Dunn, 2006). The overall sex ratio also remained similar, and close to 1:1 (Clark *et al*., 2000); however, the spawning fish in the newly-discovered plume at Rekohu have been found to be smaller and younger than those in the historical main plume (Doonan, Horn and Maolagáin, 2014a, 2014b).

3.1.8 Challenger Plateau (ORH 7A)

The commercial fishery was developed in late 1981, and expanded rapidly, such that between 1982 and 1989 the Challenger Plateau was the second-largest orange roughy fishery in New Zealand. The fishery operated primarily during the spawning season, between June and August, targeting aggregations of spawning fish. The fishery was managed under an area-aggregated TACC from 1982, but received a specific ORH 7A TACC from 1986, when the TACC was raised to 10 000 tonnes. In 1989, only 8 200 tonnes of TACC was allocated because of concerns about overfishing, and in 1990 the TACC was reduced to 2 500 tonnes following a stock assessment indicating overfishing was occurring. A further stock assessment in 2000 indicated the stock was depleted (< 10 percent B_0 [Field and Francis, 2001]), and the commercial fishery was effectively closed in 2001.

The fishery targeted aggregations in three main areas, inside the EEZ on the main historical spawning grounds on a flat area of shelf ("Central Flat") and two hill complexes, one inside the EEZ referred to as "The Pinnacles", and one just outside of the EEZ on the "Westpac Bank". From the start of the fishery until the mid-1990s, the fishery was focused almost entirely on the Central Flat (> 80 percent of the catch), but then shifted to The Pinnacles, which produced 60–70 percent of the catch by 1989/90. Spawning was reported on the Central Flat as well as at The Pinnacles and Westpac Bank, with the proportion spawning in the latter area increasing during the 1980s (Clark and Tracey, 1994). The spatial extent of the fishery also expanded rapidly to other features and areas of the shelf during the five years before the closure in 2001. Catch over-runs were estimated for the Challenger fishery, and peaked in the 1980s, reducing to 10 percent or less from 1992. Research surveys recommenced in 2005 while the fishery was still officially closed, with a catch allowance intended to cover the surveys (a combined trawl and acoustic survey by single vessel). The surveys found substantial spawning aggregations in an area to the east of The Pinnacles, although no aggregations were found on the historical main spawning grounds of the Central Flat. The ORH 7A fishery was reopened in 2015 with a TACC of 1 600 tonnes following acceptance of a new stock assessment in 2014 that showed the recovery of the stock to a status above the management target (Cordue, 2014a).

In the 1980s and early 1990s, the period of the year during which large catches were made became progressively reduced to only the winter months (Clark and Tracey, 1994). The mean CPUE was higher in winter, and declined steadily between 1983 and 1989 to 15–20 percent of original levels. The CPUE increased in 1990 following reduction of the TACC, when there were fewer vessels and fewer trawl tows on the grounds.

The orange roughy on the Challenger Plateau, both inside the EEZ and outside the EEZ on the Westpac Bank, are regarded as a single but straddling stock. The fish are smaller and mature smaller and earlier than those from other locations around New Zealand (Clark and Tracey, 1994; Horn, Tracey and Clark, 1998). The length frequency distribution was similar in all years before the fishery closure, and strongly unimodal with a peak consistently at 32–33 cm SL, and the sex ratio remained constant at about 1:1. Age samples from surveys indicated that the stock in 2009 and later contained a much higher proportion of young fish, consistent with depletion of the original spawning stock and replacement through recruitment (Doonan, Horn and Maolagáin, 2014a; Cordue, 2014a).

3.1.9 Cook Canyon (ORH 7B)

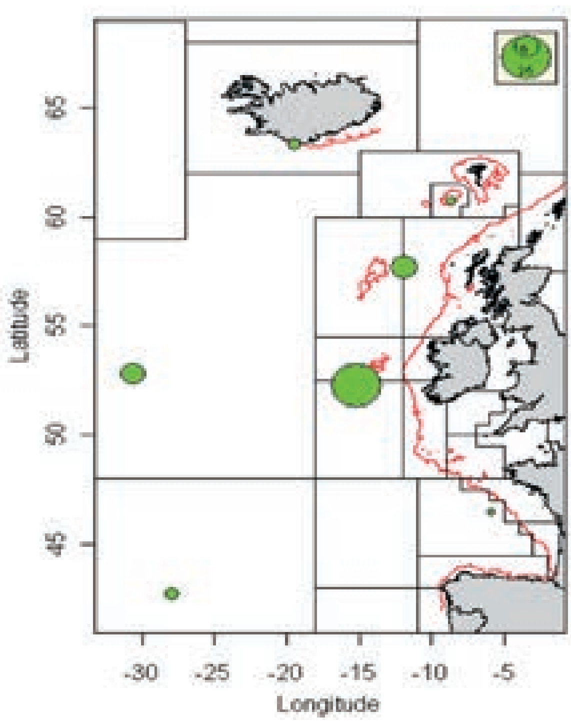
Orange roughy have been caught throughout ORH 7B, but the target fishery has been centred on an area near Cook Canyon, a trench running roughly east–west, off the west coast of the South Island, and an area just to the south, Moeraki Canyon. The fishery developed in 1985, with catches increasing and remaining relatively high for the following seven years, but then declining from about 1 100 tonnes in 1993 to 290 tonnes in 1995. The TACC was reduced in 1996, but catches declined further, with a further TACC reduction in 2002. Following a stock assessment in 2004, the fishery was effectively closed from 2008. Over time, the fishery became increasingly focused on spawning aggregations, with the focus of fishing and catches in June and early July. Catch over-runs have not been estimated or assumed for ORH 7B as there was no evidence that lost or otherwise unreported catches were an issue in this fishery.

While there has been no clear indication of sequential depletion in ORH 7B, the area fished increased throughout the 1990s, but did not increase after 1999. The mean tow distance increased abruptly in 2000 about three-fold, declining again in 2006, indicating some changes in fishing practice. The majority of the fishing effort reports for ORH 7B were on summary reporting forms, rather than detailed event-by-event forms, so the exact nature of the change is unknown. The CPUE was high at the start of the fishery, and began to decrease in the early 1990s, although relatively high CPUE was seen at Moeraki Canyon in 1993 and 1994. The CPUE was then relatively low but consistent, at less than 10 percent of initial levels, from 1997 until 2007, with catch rates very rarely greater than 5 tonnes/tow. Length frequency information for orange roughy from the commercial fishery in ORH 7B has not been reported, although size structure information has been collected during a number of commercial fishing trips.

3.1.10 Northeast Atlantic

The development of the northeast Atlantic orange roughy fishery is described by the International Council for the Exploration of the Sea (ICES) (ICES 2015a, 2015b; Large and Bergstad, 2003), and in more detail for selected countries and territories, including France, Ireland, Spain, the Russian Federation, and Faroe Islands in Hopper (1995). ICES reports landings and provides advice based on its statistical subareas and biological stocks and, as a result, some of the catch or landings reported by ICES subareas may have be taken on the high seas and not from the European Union (Member Organization) or other nation EEZ waters. Orange roughy were caught in the northeast Atlantic as early as the 1970s by Russian and German trawlers, but a targeted commercial fishery did not develop until 1991. Commercial

FIGURE 7
Northeast Atlantic orange roughy fisheries, by ICES area



Note: Circle size depicts relative historic accumulated catch for the years 1991–2013.
Source: ICES (2015a).

FIGURE 8
Map of ICES Subareas in the North Atlantic Ocean

Source: FAO.

fishing effort focused on targeting small orange roughy aggregations on seamounts and other features along the continental slope, largely in ICES Subareas VI and VII (Figures 7 and 8).

Demersal trawl tows targeting aggregations were as short as 20 minutes duration. In 2003, TAC) limits were introduced, and these have been set at zero since 2010. All orange roughy taken since 2010 have been as a bycatch in fisheries targeting other species.

The northeast Atlantic fishery started in ICES Subarea VI, to the northwest of Ireland. The fishery was a target fishery by French trawlers, and focused on orange roughy spawning aggregations around the Hebrides Terrace. French trawlers recorded the first major catches, taking about 5 000 tonnes in each of 1991 and 1992, catches dropped to just under 3 000 tonnes in 1993, and further to 1 800 tonnes by 1999 (Table 6). In the period 1990–99, French trawlers took about 75 percent of the total orange roughy catch from this area. Between 1996 and 2001, there was only one French deep-sea trawler fishing orange roughy, and in 2002 this vessel left the fishery. The CPUE declined fairly rapidly, by about 75 percent between 1991 and 1994, although it is thought that there was sequential fishing of aggregations.

After the collapse of the fishery in Subarea VI, fishing moved to Subarea VII. This target fishery peaked in 2002, and rapidly declined thereafter. Some

TABLE 6
Reported landings of orange roughy and by year for ICES Subareas VI and VII

Year	ICES Subarea VI TAC	ICES Subarea VI Landings	ICES Subarea VII TAC	ICES Subarea VII Landings
	(tonnes)			
1989	–	5		3
1990	–	15		2
1991	–	3 502		1 406
1992	–	1 422		3 101
1993	–	429		1 668
1994	–	179		1 722
1995	–	116		831
1996	–	116		879
1997	–	146		893
1998	–	102		969
1999	–	176		1 161
2000	–	138		1 020
2001	–	280		3 412
2002	–	323		5 465
2003	88	81	1 349	541
2004	88	56	1 349	467
2005	88	45	1 149	255
2006	88	33	1 149	489
2007	51	12	193	1646
2008	34	5	130	118
2009	17	2	65	15
2010	0	0	0	0
2011	0	0	0	0
2012	0	0	0	0
2013	0	0	0	0
2014	0	0	0	0
2015	0	–	0	-

Source: ICES WGDEEP (2016).

targeted fishing of orange roughy aggregations on and close to seamounts was carried out until 2008, while the rest of the orange roughy catch was as a bycatch in mixed species trawling on flat areas of the continental slope. This fishery was also dominated by French trawlers, except in 2001 and 2002, when Irish catches reached 2 367 tonnes in 2001, and 5 114 tonnes in 2002, representing 90 percent of the catch for the latter year. Subsequently, the Irish fleet landed a substantial portion of the (albeit small) catches. The Irish deep-sea trawl fisheries were short lived, and it has been suggested that they would not have been economically viable without subsidies from the European Union (Member Organization) (Foley, van Rensburg and Armstrong, 2011).

A small number of other orange roughy fisheries operate in the northeast Atlantic within EEZs and high seas, in Subareas Va, Vb, VIII, X, and XII (Table 7). Most catches in these other areas have not been made by vessels from the European Union (Member Organization). In Subarea Va, around Iceland, the fishery was dominated by Icelandic vessels, with catches starting in 1991 and peaking at 717 tonnes in 1993, then declining to < 100 tonnes per year since 1995, and < 10 tonnes between 2005 and 2011. In Subarea Vb, around Faroe Islands, the fishery was initially dominated by French trawlers, and then by Faroese vessels from 1993, with catches peaking at 420 tonnes in 1995 and generally being < 20 tonnes thereafter, effectively ceasing in 2006. In Subarea VIII, off the west coast of France, the fleet was composed largely of French trawlers, with a catch of 83 tonnes in the first year of the fishery (1992), and since then < 50 tonnes a year until 2006, thereafter < 15 tonnes a year, and with nothing since 2011. In Subarea IX, off the west coast of Spain, catches were primarily by Spanish and Portuguese trawlers, starting in 1997, with no reported catch between 2000 and 2010. In Subarea X, around and to the north of the Azores, catches have been variable, with most taken by Portuguese vessels in 2000 and 2001. Subarea XII, the mid-Atlantic Ridge, is known as a particularly difficult area for trawlers to operate in, and catches were almost entirely taken by Faroese trawlers, peaking in 1996 at 818 tonnes, of which 779 tonnes (95 percent) was taken by Faroese vessels.

TABLE 7
Reported landings of orange roughy and TAC by year for other ICES Subareas

	Subarea landings								Total
Year	IV	Va	Vb	VIII	IX	X	XII	TAC	Landings
	(tonnes)								
1990	0	0	22	0	0	0	0	–	22
1991	0	65	48	0	0	0	0	–	113
1992	0	382	13	83	0	0	8	–	486
1993	0	717	37	68	0	1	32	–	855
1994	0	158	170	31	0	0	93	–	452
1995	0	64	420	7	0	0	676	–	1 167
1996	0	40	79	22	0	471	818	–	1 430
1997	0	79	18	23	1	6	808	–	935
1998	0	28	3	14	1	177	629	–	852
1999	0	14	5	39	1	10	431	–	500
2000	0	68	155	52	0	188	259	–	722
2001	0	19	5	20	0	455	811	–	1 310
2002	0	10	1	20	0	30	6	–	67
2003	0	0	5	31	0	1	200	–	237
2004	0	28	7	43	0	403	307	–	788
2005	0	9	13	29	0	83	193	102	327

Year	Subarea landings								Total
Year	IV	Va	Vb	VIII	IX	X	XII	TAC	Landings
	(tonnes)								
2006	0	2	0	43	0	8	96	102	149
2007	14	0	1	1	0	0	20	44	36
2008	7	4	<1	8	0	37	71	30	127
2009	0	<1	2	3	0	26	34	15	66
2010	0	<1	<1	8	0	39	35	0	82
2011	0	4	0	0	<1	77	27	0	108
2012	0	16	0	0	28	45	94	0	183
2013	0	54	1	0	0	0	2	0	57
2014	0	0	–	0	0	47	11	0	58

Note: "Other" is defined as landings in the European Union (Member Organization) and TACs from/for European Community waters not under the sovereignty or jurisdiction of third countries.

The zero TAC set for orange roughy in all areas since 2010 could have led to discarding from the mixed species trawl fisheries, but available data suggest discarding was not substantial and that the cessation of targeting of orange roughy led to a strong reduction in risk to the stocks (Dransfield *et al*., 2013).

Orange roughy in the northeast Atlantic are typically larger than elsewhere, reaching 60 cm SL or more (ICES, 2015b). Orange roughy average length and weight in 2011–14 were unchanged from that observed in the fishery in 1992–98 (Thomson, 1998; ICES, 2015a). Trawl survey data from gentle slopes showed several modes in length smaller than the size at maturity, suggesting the presence of several juvenile cohorts (ICES, 2015b).

3.2 HIGH SEAS FISHERIES

The current high seas fisheries for orange roughy are also exclusively demersal trawl fisheries and are all found in the Southern Hemisphere, occurring in the southeast Atlantic, southern Indian Ocean and the western South Pacific Ocean. There was also a small fishery on the Mid-Atlantic Ridge that caught orange roughy. The fisheries have typically been winter fisheries that target spawning aggregations for a few weeks each year between June and August. Each of the high seas fisheries for orange roughy now falls under the control of a RFMO. The principal sources for the information on these fisheries as a whole are the RFMO websites for the Southeast Atlantic Fisheries Organisation (SEAFO, southeast Atlantic), SPRFMO (western South Pacific) and SIOFA (southern Indian Ocean), with additional information sources used being the government websites of the members of the RFMO and for the industry representative body, the Southern Indian Ocean Deepsea Fishers Association (SIODFA).

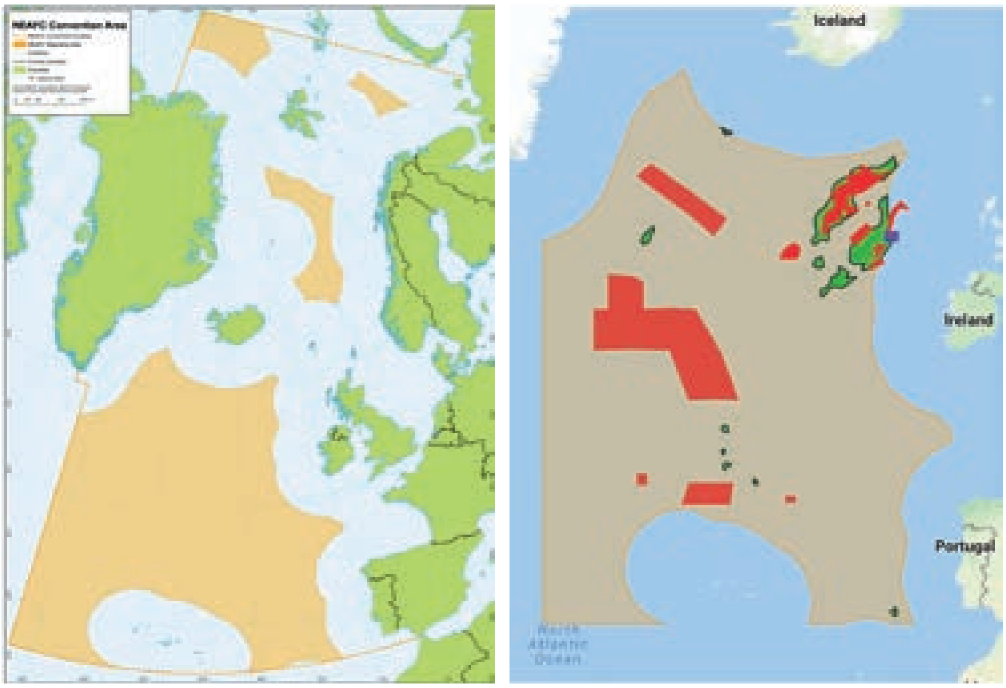
3.2.1 Mid-Atlantic Ridge, northeast Atlantic Ocean – Northeast Atlantic Fisheries Commission

The long-established Northeast Atlantic Fisheries Commission (NEAFC) includes both high seas and EEZ fisheries areas (Figure 9). The orange roughy fisheries within national jurisdictions are discussed in the northeast Atlantic section above and the high seas fishery in this section. Originally established in 1959, the NEAFC underwent considerable reorganization with a new convention in 1980 following changes in the then European Economic Community and the general extension of fishery limits to 200 nautical miles.[4] There are five contracting parties (members) to NEAFC and five cooperating non-contracting parties. The NEAFC manages a number of demersal and deepwater fisheries, including a small, high seas trawl

[4] See: www.neafc.org

FIGURE 9

Maps of the regulatory area (high seas) and convention area (high seas and EEZs) covered by the Northeast Atlantic Fisheries Organization



Note: Left map - regulatory areas of NEAFC in orange; Right map - areas closed to various forms of fishing in red, existing bottom fishing areas in green.
Source: www.neafc.org

fishery for orange roughy. The NEAFC seeks scientific advice from ICES, which in part accounts for the generally consistent approach to management of the deepwater fisheries in the northeast Atlantic region.

TABLE 8

ICES Working Group estimates of landings of orange roughy from ICES Subareas X and XII (covering the Mid-Atlantic Ridge

Year	Landings (tonnes)
1992	8
1993	33
1994	93
1995	676
1996	1 289
1997	814
1998	806
1999	441
2000	447
2001	1 266
2002	36
2003	201
2004	710
2005	323
2006	104
2007	20
2008	108
2009	60
2010	74
2011	104
2012	139
2013	2
2014	58
2015	84
2016	93

Source: ICES WGDEEP (2017).

Fishing on the Mid-Atlantic Ridge started in the 1970s, with a small demersal trawl fishery for orange roughy starting in the early 1990s, with catches peaking in the mid-1990s and early 2000s (Table 8). This fishery comprised a small number of European trawlers, principally French and Faroese vessels, that fished in ICES Subareas X and XII. Bensch *et al.* (2009) report a total of about 1 000 tonnes of orange roughy caught between 2003 and 2006, mostly taken by Faroese vessels. Landings of less than 100 tonnes per year from ICES Subareas X and XII have been reported for more recent years (Table 8) (Ofstad, 2017; WGDEEP, 2017).

Catches of orange roughy in the NEAFC regulatory area have effectively been reduced by management actions, including a zero TAC for vessels from the European Union (Member Organization), NEAFC requirements preventing targeting orange roughy in the NEAFC regulatory area by vessels of NEAFC contracting parties, and requirements for vessels to take measures to reduce bycatch of orange roughy (Recommendation 6: 2016).

3.2.2 Southeast Atlantic Ocean – South East Atlantic Fisheries Organisation

The convention of the South East Atlantic Fisheries Organisation (SEAFO) entered into force in 2003, with the first meeting of the commission in 2004. SEAFO is an intergovernmental fisheries management body with a convention area comprising the high seas part of the southeast Atlantic Ocean. SEAFO's primary purpose is to ensure the long-term conservation and sustainable use of all living marine resources in the high seas of the southeast Atlantic Ocean, and to safeguard the environment and marine ecosystems in which the resources occur. There are seven contracting parties (members) to SEAFO.

Seven Namibian-flagged vessels operated in the fishery in the SEAFO convention area using standard New Zealand orange roughy bottom trawl gear (see MFish [2008a] for a trawl gear description). Catches of orange roughy from the high seas of the southeast Atlantic Ocean SEAFO convention area have been small and sporadic, starting in 1995 and never exceeding 100 tonnes/year (Table 9).

There is minimal information to evaluate stock structure, and no information suggests that the fish taken from the SEAFO convention area are separate stocks from those taken in the adjacent EEZs. No stock assessment has been possible. Following a three-year fishing moratorium from 2008, the target fishery has remained closed, but SEAFO has retained an annual TAC for orange roughy of 50 tonnes to provide for bycatch in other fisheries (SEAFO, 2014). SEAFO has various management measures in place to provide protection for vulnerable benthic environments, marine turtles, seabirds and deepwater sharks.

TABLE 9
Landings of orange roughy reported to SEAFO for Management Area B1, 1994–2007

Year	Catch (tonnes)
1994	–
1995	40
1996	8
1997	5
1998	–
1999	<1
2000	75
2001	94
2002	9
2003	27
2004	15
2005	18
2006	–
2007	–

Notes: A further 27 tonnes was reportedly taken between 1993 and 1997. "-" indicates no fishing.

Source: FAO stock status report 2014.

3.2.3 Western South Pacific Ocean – South Pacific Regional Fisheries Management Organisation

The South Pacific Regional Fisheries Management Organisation (SPRFMO) had its first commission meeting in 2013, following a number of years of preparatory meetings and interim arrangements for managing the fisheries.[5]

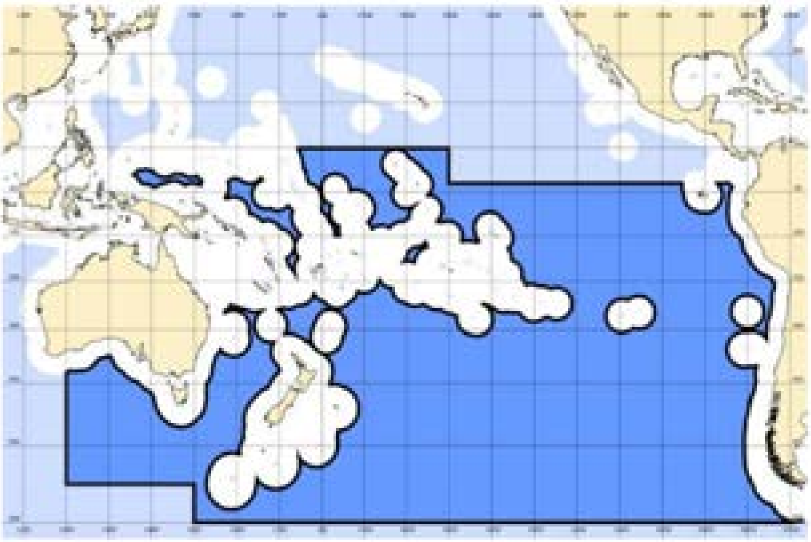
The SPRFMO is an intergovernmental organization committed to the long-term conservation and sustainable use of the fishery resources of the South Pacific Ocean and in so doing safeguarding the marine ecosystems in which the resources occur. Its convention area extends across the whole of the South Pacific (Figure 10). The main fisheries that are currently actively managed by the SPRFMO are the fishery for Chilean jack mackerel (*Trachurus murphyi*) in the eastern South Pacific and the demersal trawl fishery for orange roughy in the western South Pacific. The convention specifically requires that conservation and management measures (CMMs) include measures to protect habitats and marine ecosystems, while enabling sustainable fishing (Penney, Tingley and Loveridge, 2016).

As of early 2017, there are 15 members of the SPRFMO and 2 cooperating non-contracting parties.[6] Australia and New Zealand are the only SPRFMO members that have a recent track record of demersal fishing, and that are eligible under SPRFMO rules to fish for orange roughy by demersal trawling.

Fishing for orange roughy in the western South Pacific was begun by vessels of the Soviet Union in 1977, although the exact location of catches is unknown. It is thought

[5] See: www.sprfmo.int
[6] See: www.sprfmo.int

FIGURE 10
A map of the convention area (high seas) covered by the South Pacific Regional Fisheries Management Organisation (SPRFMO)



Source: SPRFMO website.

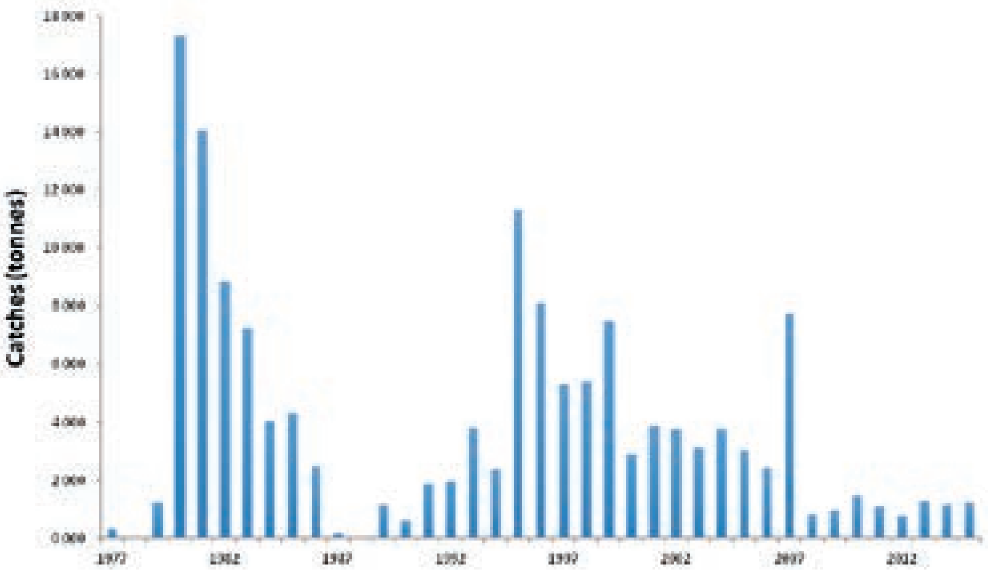
TABLE 10
Catches of orange roughy reported to the SPRFMO as taken within its convention area and the percentage caught by New Zealand flagged vessels

Year	Total catch (tonnes)	Percentage of catch taken by New Zealand flagged vessels
1977	319	0.0
1978	0	–
1979	1 251	0.0
1980	17 300	0.0
1981	14 076	0.0
1982	8 860	0.0
1983	7 229	0.0
1984	4 028	0.0
1985	4 306	0.0
1986	2 475	0.0
1987	151	0.0
1988	24	0.0
1989	1 153	0.0
1990	616	90.7
1991	1 868	7.5
1992	1 979	38.3
1993	3 787	67.8
1994	2 387	92.0
1995	11 306	99.0
1996	8 113	98.6
1997	5 320	72.6
1998	5 427	42.9
1999	7 469	66.2
2000	2 863	55.0
2001	3 864	64.7
2002	3 759	68.6
2003	3 117	63.3
2004	3 759	45.1
2005	3 020	52.9
2006	2 428	58.3
2007	7 742	11.2
2008	837	100.0
2009	928	100.0
2010	1 474	100.0
2011	1 081	99.8
2012	777	92.8
2013	1 292	96.2
2014	1 149	91.1
2015	1 223	98.4

Source: Catch data from SPRFMO website.

that virtually every UTF within fishable depths has been explored, with the majority experiencing some fishing, but fisheries have focused on major seamounts, ridges and plateaus (Penney, Tingley and Loveridge, 2016). There are five main fishing grounds in the region: the South Tasman Rise off Tasmania, the West Norfolk Ridge, Lord Howe Rise, the Northwest Challenger Plateau in the Tasman Sea west of New Zealand, and the Louisville Ridge to the east of New Zealand. There was some exploratory fishing by both Australian and New Zealand vessels on the Challenger Plateau and Lord Howe Rise from the mid-1980s, but it was in 1988 that the first major fishery in this region was developed on Lord Howe Rise, followed by the northwest Challenger Plateau two years later (Clark and Tilzey, 1996). Subsequently, commercial fisheries were developed on the Louisville Ridge (1993), the South Tasman Rise (1997), and the West Norfolk Ridge (2001) (Clark, 2008; MFish, 2008a; Williams *et al.*, 2011; Penney, 2013; MPI, 2015a).

FIGURE 11
Catches of orange roughy from western South Pacific as reported to the SPRFMO



Note: Later catches are all from the high seas, the location (high seas vs EEZs) of catches made by Soviet vessels between 1977 and 1991 are unknown.
Source: SPRFMO website.

Since 1992, catches have been dominated by New Zealand flagged vessels (Table 10, Figure 11). Before the advent of 200 nm management zones, it is not possible to be categorical as to whether catches came from what are now high seas areas, as opposed to from within what are now EEZs (Penny, Tingley and Loveridge, 2016).

Lord Howe Rise

Lord Howe Rise extends from the northwestern margin of the Challenger Plateau off the west coast of New Zealand out to Lord Howe Island in the western Tasman Sea and is mostly in international waters. The fishery developed in 1988, and although dominated by New Zealand and Australian flagged vessels, early effort also included vessels registered in Belize, Japan, Norway, Panama, the Republic of Korea, and the Russian Federation. Tows were relatively long, typically several hours, in the first three years when fishing effort concentrated on the flat ground of the broad Lord Howe Rise, but from 1991 there was a trend towards shorter tows as fishers shifted to areas of rougher ground. Fishing was originally focused on the winter months of June and July, but from 1993 spread out to most months of the year (Clark, 2006). Catches peaked in 1993 when a combined Australian and New Zealand catch of 1 900 tonnes was taken by 18 vessels. Since then, both catch and effort have varied considerably between years, with estimated annual catches of between 20 tonnes and 500 tonnes (e.g. MPI, 2015a).

Northwest Challenger Plateau

The northwestern corner of the Challenger Plateau features several clusters of hills, and orange roughy in this area were first fished in the winter of 1989. The first substantive catch was made in 1992, when the catch exceeded 200 tonnes, and rose in the following year to over 2 000 tonnes (Clark, 2008), as fishing became more successful using short tows on the hill features. Catches after 1994 have fluctuated between years, decreasing to several hundred tonnes per annum, before a brief resurgence between 1999 and 2003. The number of vessels involved in the fishery has varied considerably between years, with about 20 in some years between 1993 and 2003, but since then there have typically been fewer than 10. The corresponding number of tows carried out each year has also decreased, from levels of 1 000–2 000 to fewer than 200 tows per year since

2012. Fishing was originally mainly in the months of May–July, but tended to expand both in time and space, and by 2005 the fishery extended over much of the northern slope of the Challenger Plateau and occurred during most months of the year. Tows also became relatively long, averaging more than 10 nm. Combined Australian and New Zealand catches in the period 2010–15 averaged about 250 tonnes per annum.

Louisville Ridge

The Louisville Ridge is a chain of seamount and guyot features that rise to peaks of 200–1 000 m from the seafloor at 4 000 m depth. The seamounts that are fished are entirely in international waters. This has consistently been the largest of the orange roughy fisheries in the SPRFMO region. New Zealand vessels first fished this area in 1993, with catches rapidly increasing to more than 10 000 tonnes in 1995. From 1997 until 2006, catches varied between 1 000 and 3 000 tonnes, but then decreased. Following two years with no fishing effort in 2008 and 2009, the fishery restarted but at lower levels, with the average annual catch between 2010 and 2015 being about 500 tonnes.

At the start of the fishery, more than 30 vessels fished in one year, but from 1997 until 2004 numbers were between 10 and 20 vessels, and have since decreased to fewer than 10 in any single year. These have been mainly New Zealand vessels, with distance a probable limitation for the Australian fleet. In the early years, some vessels from Belize, China, Cook Islands, Japan, Norway, the Republic of Korea, the Russia Federation, and Ukraine also fished in this area, mainly as joint-venture partners with New Zealand companies. All New Zealand effort is now restricted to New Zealand flagged vessels.

Initially, effort in the fishery was spread over much of the year, but contracted from 1998 to be concentrated between June and August. The distribution of catches has varied considerably between years, with effort and this catch switching between seamounts (Clark, 2008).

West Norfolk Ridge

The West Norfolk Ridge is comprised of a chain of mixed types of underwater features that run from within the New Zealand EEZ northwest towards New Caledonia. The fishery started in 2001, and at most six vessels have worked the ridge in any year. It is a feature-based fishery, with little slope area at orange roughy depths. Catches increased from 200 tonnes to just less than 600 tonnes in the first two years, but then decreased until 2005 and 2006, when a new feature was located, and when catches peaked at more than 1 000 tonnes. From 2009, the fishery has decreased in size, with an annual catch total now less than 100 tonnes in the mid-2010s.

South Tasman Rise

The South Tasman Rise is a prominent ridge extending south from Tasmania into the Southern Ocean. It has a series of small peaks near its main summit at about 900 m just outside the Australian 200 mile EEZ. The fishery developed in 1997, with Australian vessels soon joined by New Zealand fishers. The catch that year was 1 800 tonnes, and increased to 3 500 tonnes for both 1998 and 1999, before decreasing with catches less than 100 tonnes after 2002. At its peak, the fishery involved about 20 vessels. The fishery was regulated from early 1998 by a conservation and management agreement between Australia and New Zealand, but this apparently did not prevent several vessels from other nations reportedly also fishing in the area during 1999. Various TACs were agreed for a March–February fishing year but catches were not maintained, and even declining TACs were not met, with the fishery closed in 2007. After 1998, fishery data show considerable variation in the distribution of effort between years, as fishing was affected by the management agreement, which limited catch in six-month blocks; hence

,much more fishing occurred in March and April than typical in most orange roughy fisheries, as fishers competed for the available quota (Clark, 2008).

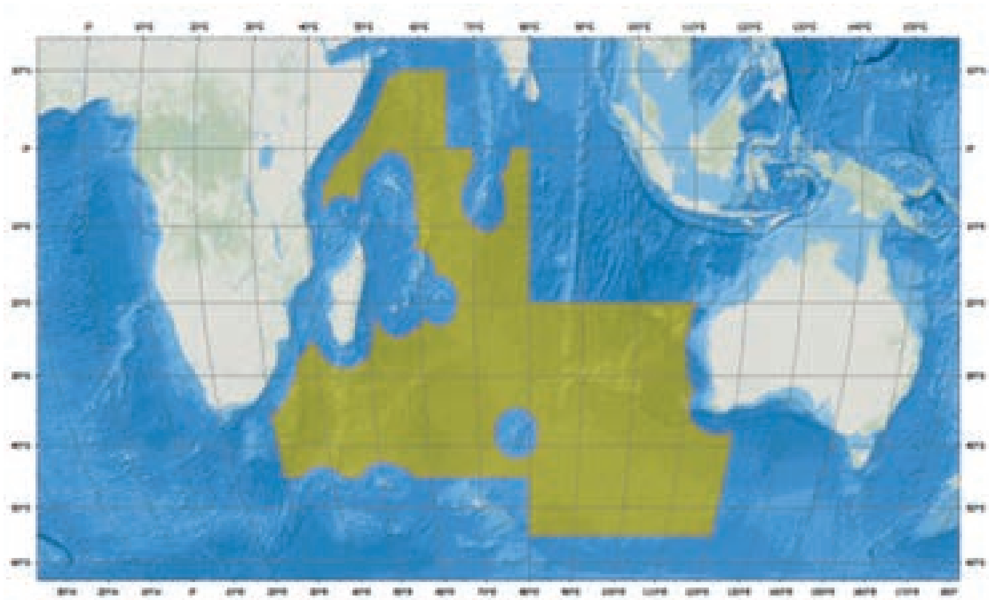
Regional management

The advent of the SPRFMO led to substantive restrictions on the orange roughy fishery from 2007. From 2007, only members and cooperating non-contracting parties that had submitted and had accepted a bottom fishing footprint for the period 2002–06 and an accepted bottom fishing impact assessment were permitted to fish with demersal trawls within the SPRFMO convention area, and then only within their accepted 2002–06 footprint. Catch and effort were also restricted to the average for the period 2002–06 for each flag state, restrictions that are still (as of 2017) in place. From 2007, only Australian and New Zealand flagged vessels participated in the fishery and annually reported catch, effort, bycatch, biological data and research to the SPRFMO (Hansen and Hobsbawn, 2015; MPI, 2015b) and it can be seen that the fishery has since seen relatively low and catches stable (Figure 11), with the fishery continuing to be dominated by New Zealand flagged vessels (Table 10).

3.2.4 Southern Indian Ocean – Southern Indian Ocean Fisheries Agreement

The Southern Indian Ocean Fisheries Agreement (SIOFA) is the newest of the non-tuna RFMOs, and covers much of the Indian Ocean (Figure 12).[7] The first SIOFA commission meeting took place in July 2016. The contracting parties to SIOFA are Australia, Cook Islands, European Union (Member Organization), France (on behalf of its Indian Ocean Territories), Japan, Mauritius, the Republic of Korea, and Seychelles.

FIGURE 12
Map of the area of application (high seas) covered by the Southern Indian Ocean Fisheries Agreement (SIOFA)



Source: SIOFA website

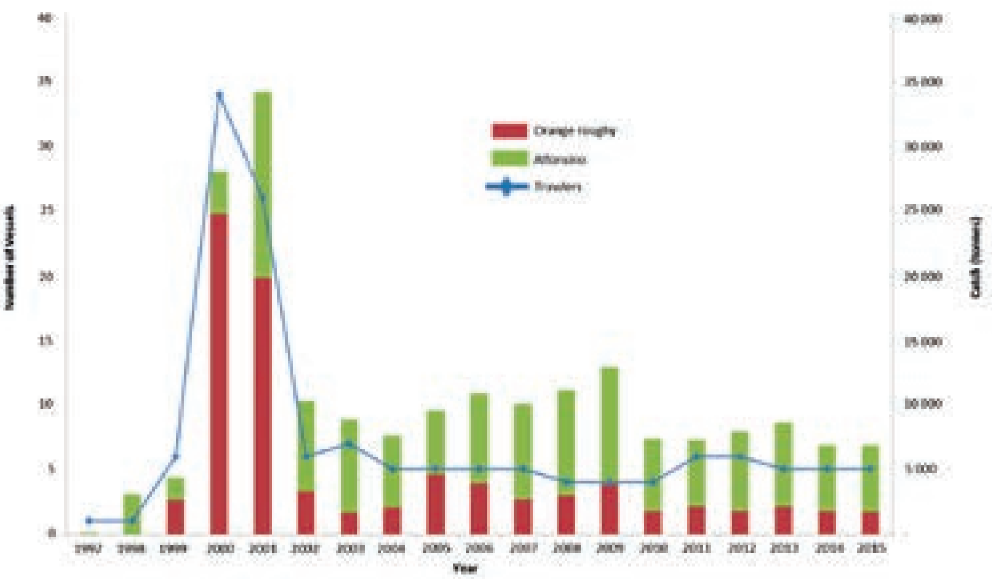
The stated objectives of SIOFA are to ensure the long-term conservation and sustainable use of the fishery resources in the area of competence through cooperation among the contracting parties, and to promote the sustainable development of fisheries, taking into account the needs of developing states bordering the competence area, and in particular the least-developed among them and small island developing states.

[7] See: www.siofa.org

Vessels of the Soviet Union started fishing for deepwater species in the southern Indian Ocean in the mid-1970s. They used a variety of gear types, including trawls, vertical mechanized lines, bottom longlines and traps. Following the breakup of the Soviet Union in 1991, some Ukrainian-flagged vessels continued to fish in the southern Indian Ocean. The Soviet fleet did not target orange roughy, but they were recorded in research vessel catches. The targeted fish species have all been deepwater and largely distributed in association with seamounts, ridges and other UTFs.

Commercial fishing of orange roughy in the southern Indian Ocean began in about 1998, and by the early 2000s there were up to 35 vessels in the fishery from 17 flag states. This rapid and unregulated growth in fishing effort led to the rapid depletion of the stocks of orange roughy in the southern Indian Ocean and a virtual collapse of the fishery, such that by 2004 the fleet had declined to about seven vessels, and catches of alfonsino were greater than those of orange roughy (Figure 13). The pattern of growth and decline in catches and landings of orange roughy after 1998 mirrors the pattern of fishing effort in the southern Indian Ocean.

FIGURE 13
Catch of the two principle target species and number of fishing vessels operating in the southern Indian Ocean, 1997–2011



Source: modified from SIOFA (2016).Source: SPRFMO website.

Trawl fishing in the high seas of the southern Indian Ocean now typically produces catches including alfonsino (*Beryx splendens*), ocean blue-eye trevalla (*Schedophilus labyrinthicus*), ruby fish (*Plagiogeneion* spp.) and southern boar fish (*Pentaceros richardsoni*), as well as orange roughy (see various national reports from SIOFA members at its website.

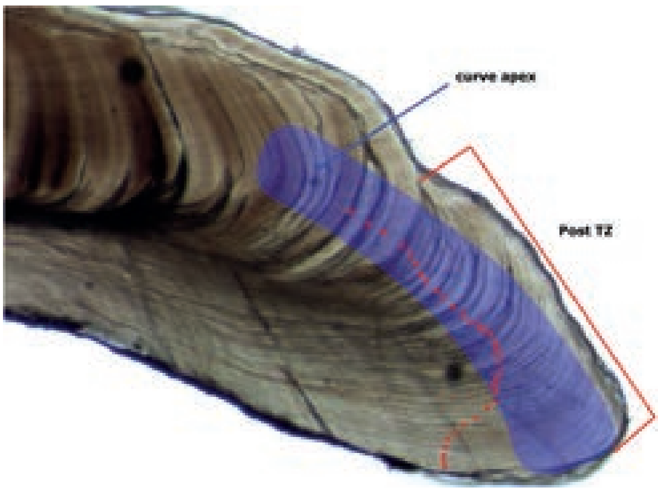
4. Age, growth and longevity

Orange roughy are a long-lived species, capable of living to ages of 100 years or more. The issue of substantial longevity in fishes used to be controversial, and especially for orange roughy, where great longevity for such a small fish was unexpected. One of the first reported ageing studies for orange roughy gave an estimated longevity of 24 years (Kotlyar, 1981). The first indication that the species might live a lot longer than this was reported a few years later, by Sullivan and Parkinson (1987), which was followed by partial validation of the estimated longevity by Mace *et al.* (1990) and through radiometric validation by Fenton, Short and Ritz (1991) and by Smith *et al.* (1995). However, such longevity remained controversial for some years, and Merrett and Haedrich (1997) noted that the "debate was not settled". The longevity debate continued, and by the late 1990s many orange roughy stocks had been substantially depleted. The stock assessments used had, by that time, recognized and assumed the greater longevity and concomitant low productivity of the species. Reinterpretation and support for the early radiometric studies was provided by Francis (1995), and a review by Tracey and Horn (1999) concluded orange roughy were most likely to be long-lived. Subsequently, there has been general consensus that orange roughy are long-lived, and concerns about longevity were replaced by concerns about precision and bias in orange roughy ageing (Francis 2006), which led to an international ageing study and a revised and tested ageing protocol (Horn *et al.*, 2016). Technical improvements also allowed a more complete radiometric age validation, finding that fish in the oldest age group were at least 93 years old, and confirming the centenarian longevity of orange roughy (Andrews, Tracey and Dunn, 2009).

4.1 DETERMINING THE AGE OF ORANGE ROUGHY

Several methods have been used to age orange roughy, including counting circuli on scales and growth zones on sagittal otoliths, both of which are assumed to show annual increments, and an independent method using radiometric dating of otoliths. The zone counts and age estimates from scales and whole otoliths suggested young ages (< 20 years), but zone counts on sectioned otoliths, and radiometric dating, suggested much greater longevity (Branch, 2001). Sectioned orange roughy otoliths (Figure 14) are, however, relatively hard to interpret (Horn *et al.*, 2016).

FIGURE 14
An orange roughy otolith section



Notes: The image shows the fine growth zones near the otolith margin after the transition zone and after the curve in otolith morphology. Discrete zones are marked with red dots. The initial 34 zones (approximately) have been marked along the curve apex, but in this example the most recently formed zones were clearest away from the curve apex.
Source: Image from Horn *et al*. (2016).

Mace *et al*. (1990) validated the growth of early juveniles using length-mode analysis, where separate size groups of fish were correlated with the number of growth zones visible, and demonstrated very slow growth, with a five-year-old fish being about 12.4 cm SL. In older and larger fish, the length modes start to overlap and the correlation between size and age cannot be clearly seen.

Andrews, Tracey and Dunn (2009) validated age estimates for adults using a lead–radium dating technique, which provided an independent estimate of age for groups of otolith cores, with the otoliths in each group judged, by zone counts, to be of similar ages. Improvements in technology allowed Andrews, Tracey and Dunn (2009) to sample otolith cores, rather than whole otoliths, alleviating many of the assumptions required in earlier studies (Fenton, Short and Ritz, 1991; Smith *et al*., 1995). The lead–radium ratios for each age group showed a high degree of correlation to an expected lead-to-radium curve, and while this did not confirm the age of individual fish, it did strongly indicate that the otolith interpretation method produced relatively accurate estimates of age.

Francis (2006) highlighted several problems with orange roughy age data, in particular that the age determination had poor precision, to the extent that important stock assessment information such as the occurrence of individual strong or weak cohorts could not be determined. Francis (2006) also raised concerns about "drift" in age estimates, suggesting otolith interpretation and age estimates were changing over time. The revised ageing protocol, developed in response to these concerns, specified where and how each section of the otolith should be read, recognizing that a different interpretation is required for the juvenile growth zone, adolescent zone, transition zone, and post-transition zone (Horn *et al*., 2016). This change in ageing protocol resulted in more consistent between-reader age estimates (Horn *et al*., 2016). Because of concerns about ageing, age data were removed from stock assessments in New Zealand after 2006, but were reinstated once samples had been aged using the revised protocol (Cordue, 2014a, 2014b). Re-ageing of Australian Eastern Zone fishery otoliths showed no between-year bias in ageing using the revised protocol and also found no evidence of a major bias in the early age estimates for Eastern Zone orange roughy (Upston *et al*., 2014).

4.2 GROWTH

The growth of orange roughy has been described using the von Bertalanffy growth curve (von Bertalanffy, 1938). Alternative growth models do not seem to have been fitted. Length-at-age is most often reported as SL, and occasionally as TL. The von Bertalanffy growth curve has three parameters: L_∞, the asymptotic length, or average maximum size, K, the Brody growth coefficient, which describes the rate of growth, and t_0, the hypothetical age of a fish at length zero. Because not all fish of a given age have the same length, variability around the length-at-age predicted from the von Bertalanffy growth curve is assumed, and is usually presented as a coefficient of variation (CV), the assumed or estimated CV is rarely reported outside of technical stock assessment reports. Finally, there is a conversion of length to weight, described using an exponential model with two parameters: a, and b. Parameter b is the exponent, and describes how "fat" a fish of a given length is, with a higher value indicating greater weight at a given length. Parameter a influences weight at length as well, but also varies depending on the units of length and weight in which the fish are measured.

Table 11 shows estimated or assumed values for growth parameters, from which it can be seen that females grow larger than males. Around New Zealand, the orange roughy are smaller on the Challenger Plateau, and those on the east coast are a similar size to those off Australia (note fish from the Cascade Plateau are large and are not included here). Orange roughy from Chile, and the northeast Atlantic, grow larger than those around Australia and New Zealand, whereas Namibian specimens are

TABLE 11
Growth parameters for orange roughy

Region	Source	Sex	Parameter	Estimate
Australia, Eastern	Upston *et al.* (2014)	Combined	L_∞	38.6
			K	0.06
			t_0	NA
			a (F, M)	0.0351 0.0383
			b (F, M)	2.97 2.94
			c.v.	0.07
Australia, Southern	Smith *et al.* (1995)	Combined	L_∞	41.4
			K	0.04
			t_0	–4.9
Chile	Payá *et al.* (2005)	Male	L_∞	43.5
			K	0.04
			t_0	–0.66
			a	0.06
			b	2.73
		Female	L_∞	48.7
			K	0.03
			t_0	–1.13
			a	0.06
			b	2.76
		Combined	L_∞	47.5
			K	0.03
			t_0	–0.60
			a	0.019
			b	3.04
			c.v.	0.04
Namibia, Johnies, Frankies and Rix	Brandão and Butterworth (2007)		L_∞	29.5
			K	0.069
			t_0	–2.00
			a	0.1354
			b	2.565
Namibia, Hotspot	Brandão and Butterworth (2007)		L_∞	37.2
			K	0.065
			t_0	0.5
			a	0.1354
			b	2.565
New Zealand, Chatham Rise	Cordue (2014a), MPI (2016a)	Combined	L_∞	37.8
			K	0.059
			t_0	–0.491
			a	0.08
			b	2.75
			c.v.	0.1–0.06
New Zealand, Ritchie Bank	Cordue (2014b), MPI (2016a)	Combined	L_∞	37.6
			K	0.065
			t_0	–0.5
			a	0.0921
			b	2.71
			c.v.	0.1–0.05
New Zealand, Challenger Plateau	MPI (2016a)	Male	L_∞	33.4
			K	0.07
			t_0	–0.4
			a	0.0921
			b	2.71
		Female	L_∞	35.0
			K	0.061
			t_0	–0.6
			a	0.0921
			b	2.71
Northeast Atlantic	Shephard and Rogan (2004) cited in ICES (2015b)	Combined	L_∞	47.6
			K	0.039
			t_0	2.61
	ICES (2015b)	Combined	a	0.169
			b	2.59

Notes: Units are: L_∞ in cm, K in per year, t_0 in years. Parameter a has been rescaled where necessary to give length (SL) in centimetres and weight in grams. The CV is the variability in length around the mean length-at-age.

among the smallest (Kashindi, 1999). Orange roughy at a given length have been estimated to be heaviest in the northeast Atlantic, and lightest in Chile, a 30 cm SL fish would weight about 650 g in Chile, 900 g in Australia and New Zealand, and 1 100 g in the northeast Atlantic. Analyses of weight-at-length, which is a measure of fish condition, have found variability by sex, maturity and season, and also variability or trends over years, with orange roughy in several areas showing a progressive reduction in weight-at-length over a period of 20 years or more (Dunn and Devine, 2010).

4.3 LONGEVITY AND NATURAL MORTALITY RATE

The natural mortality rate (*M*) is directly linked to longevity, with greater longevity resulting from lower *M*. The value of *M* also helps to describe the productivity of a species – to maintain a given population size, fish with greater longevity need to replace themselves less often, thereby requiring less reproductive output, meaning they are less productive.

Available estimates of maximum age from validation studies were 149 years (Fenton, Short and Ritz, 1991) and 125 years (Smith *et al*., 1995) from Australia, 93 years from New Zealand (Andrews, Tracey and Dunn, 2009), and 80 years from the northeast Atlantic (Allain and Lorance, 2000). Estimates of maximum age from otolith zone counts have been for: Australia, about 160 years (Upston *et al*., 2014); Chile, about 160 years (Gili *et al*., 2002); the northeast Atlantic, 169 years (Shephard and Rogan, 2006) and 187 years (Talman *et al*., 2002 cited in ICES, 2015b); and New Zealand 145 years for fish from the Challenger Plateau (Doonan, Horn and Krusic-Golub, 2013a), about 170 years for the Mid-East Coast and 155 years for Chatham Rise (Doonan, Horn and Ó Maolagáin, 2014a, 2014b).

The natural mortality rate (*M*) has been estimated to be 0.045 from a lightly fished stock on Chatham Rise, with 95 percent CIs of 0.030–0.060 (Doonan, 1994). A similar estimate was obtained for orange roughy off northern New Zealand, with $M = 0.037$ and 95 percent CIs 0.025–0.062 (Doonan and Tracey, 1997). Natural mortality has been estimated for fish from Chile to be between 0.03–0.058 (Gili *et al*., 2002). Stock assessment models for New Zealand have used an *M* of 0.045 (MPI, 2016a), those for the Australian Eastern Zone an *M* of 0.042 (Wayte, 2007) or 0.04 (Upston *et al*., 2014), and Cascade Plateau an *M* of 0.02 (Wayte and Bax, 2007). For Namibia, an *M* of 0.055 has been used in stock assessments (Brandão and Butterworth, 2005), for the northeast Atlantic, *M* has been assumed to be about 0.045, or 0.025 (WGDEEP, 2002, cited in ICES, 2015b), and for Chile *M* has been assumed to be 0.045 (Payá *et al*., 2005). While there are some differences, most researchers have estimated or assumed a similar value of *M*. Natural mortality estimated within assessment models for four orange roughy stocks in New Zealand in 2014 ranged from 0.032 to 0.041 (Cordue, 2014a).

5. Development and reproduction

5.1 MATURATION

Orange roughy are gonochoristic, and determination of sex outside of the spawning season (when gametes are expressed) requires examination of the gonads. The progression of maturation and spawning has been described using the condition of the gonads (Tables 12 and 13).

TABLE 12
Key for the assessment of orange roughy maturity from macroscopic and microscopic examination of female gonads

Stage	Macroscopic	Microscopic
1: Immature or resting	Immature or regressed, ovary clear.	Pre-vitellogenic oocytes only.
2: Early maturation	Ovary pink or clear, small clear oocytes visible against the light.	Endogenous vitellogenesis (yolk globule accumulation).
3: Mature	Orange oocytes present.	Exogenous vitellogenesis (yolk granule accumulation).
4: Ripe	Mature ovary, hyaline oocytes present.	Final oocytes maturation, nuclear migration and breakdown, coalescence of yolk material and oil droplet formulation.
5: Running ripe	Ovulated, eggs flowing freely when light pressure applied to abdomen.	Follicular separation and rupture.
6: Spent	Spent, ovary flaccid and “bloody”, residual eggs sometimes present in oviduct.	Post-ovulatory follicles, increased vascularization, follicular atresia.

Source: Pankhurst, McMillan and Tracey (1987).

TABLE 13
Key for the assessment of orange roughy maturity from macroscopic and microscopic examination of male gonads

Stage	Macroscopic	Microscopic
1: Immature or resting	Immature or regressed, testis threadlike.	Spermatogonia and primary spermatocytes predominate.
2: Early maturation	Testis increased in size but no milt expressible.	Secondary spermatocytes and spermatids present, spermatozoa in larger gonads.
3: Mature	Partially spermiated, viscous milt expressible.	Spermatozoa predominate.
4: Ripe	Fully spermiated, hydrated, freely flowing milt.	Spermatozoa predominate.
5: Running ripe	Spent testis “bloody” or grey, no milt expressible.	Residual spermatozoa, spermatogonia present towards testis margin.
6: Spent	Spent, ovary flaccid and “bloody”, residual eggs sometimes present in oviduct.	Post-ovulatory follicles, increased vascularization, follicular atresia.

Source: Pankhurst, McMillan and Tracey (1987).

The method proposed by Pankhurst, McMillan and Tracey (1987) remains the standard method for evaluating maturity status of orange roughy from the macroscopic (visual) examination of gonads (e.g. Minto and Nolan, 2006; Pitman, Haddy and Kloser, 2013). Further stages for fully atretic (female only) and partially spent gonads have also been used (Doonan *et al.*, 2000), as well as redefining stage 1 as immature only and adding a new stage (stage 9) for mature and resting adults (Anderson, 2011a). The use of the term “regressed” in stage 1 of Pankhurst, McMillan and Tracey (1987) indicates resting, adult fish, and although they provide no clear rationale for combining immature and regressed conditions into a single stage, the implication is that the two are difficult to tell apart. In practice, the use of "resting" for stage 1 orange roughy was discontinued on deepwater surveys in New Zealand from about 2000 onwards to enable immature stages to be potentially distinguished from early maturing or reverted (the stage after spent) gonads. Other samples have been collected and described using more generic macroscopic maturity stage keys, but these can be converted to specific orange roughy stages (O'Driscoll *et al*., 2011). Mature biomass is usually calculated assuming fish at stages 3 and above are mature.

Doonan *et al*. (2000) and Doonan, Tracey and Grimes (2004) compared microscopic and macroscopic assessments of maturity and found that macroscopically spent females could be clearly identified, meaning the appearance of spent females could be used to accurately determine the end of spawning. However, the proportion of post-ovulatory follicles was not clearly related to macroscopic stage, such that the macroscopic key (as applied in their samples) was not clearly tracking the progression of oocytes and release of eggs, and in particular raised doubt about the credibility of using a "partially spent" stage. It is normal in fish studies for the greatest confusion in maturity staging to arise in this way, between "maturing", "resting" and "spent" stages (Murua *et al*., 2003).

The other measure of reproductive condition that is often used is the Gonado-Somatic Index (GSI, the gonad weight divided by body weight). The GSI has been observed to peak at the time of spawning in females (e.g. Pitman, Haddy and Kloser, 2013).

5.2 AGE AND SIZE AT FIRST MATURITY

Three methods have been used to determine the age and length-at-maturity for orange roughy: (i) direct measurements of age and length with corresponding maturity measurements; (ii) age- and length-at-maturity inferred from changes in the appearance of otolith growth zones; and (iii) direct measurements of age and length in spawning plumes, with an assumption that the presence in a spawning plume implies maturity. Maturity ogives estimated from fish samples taken in and around spawning aggregations have been shown to be susceptible to bias, caused by the turnover of fish in the plumes (fish arriving to spawn coupled with fish departing after spawning) or by movement of fish (Francis and Clark, 1998).

Direct measurements of length-at-maturity have been made for most regions (Table 14), but direct measurements of age-at-maturity are rare, possibly because

TABLE 14

Estimated mean size (standard length) at first sexual maturity for orange roughy

Location	Male	Female	Combined sex	Source
	(cm)			
Australia, Eastern	–	28.0	–	Bell *et al*. (1992)
Australia, Tasmania	–	32.0	–	Bell *et al*. (1992)
Australia, Eastern	–	–	35.8	Wayte (2007), Upston *et al*. (2014)
Australia, Tasmania	34.0	34.0	34.0	Horn, Tracey and Clark (1998)
Chile	–	–	30.0	Young *et al*. (2000)
New Zealand, Bay of Plenty	31.9	33.7	32.5	Horn, Tracey and Clark (1998)
New Zealand, Ritchie Banks	30.1	30.8	30.4	Horn, Tracey and Clark (1998)
New Zealand, Chatham Rise	30.0	30.8	30.4	Horn, Tracey and Clark (1998)
New Zealand, Puysegur Bank	29.7	31.9	30.9	Horn, Tracey and Clark (1998)
New Zealand, Challenger Plateau	28.5	28.8	28.6	Horn, Tracey and Clark (1998)
Namibia	23.3	26.7	23.8	Horn, Tracey and Clark (1998)
Northeast Atlantic	–	–	41.0	Du Buit (1995)
Northeast Atlantic, Hatton Bank	41.8	40.1	40.8	Horn, Tracey and Clark (1998)
Northeast Atlantic, flats	–	34.0	–	Shephard and Rogan (2006)
Northeast Atlantic, aggregations	–	37.0	–	Shephard and Rogan (2006)
Northeast Atlantic, Porcupine Bank	–	37.0	–	Minto and Nolan (2006)

of the expense of ageing large samples of fish. Conversion of lengths to ages can be relatively imprecise for orange roughy, because slow growth after maturation and great longevity means that a 1 cm length class can include a large number of age classes.

The width of growth zones on otoliths is an indication of somatic growth, and the growth zones becomes narrower once the fish makes the transition from allocating resources to somatic growth, to the production of gametes, this transition is a cornerstone of life history theory (Mollet *et al.*, 2010). In orange roughy, the region on the otolith where it is believed that this change takes place is known as the "transition zone", and the age at the transition zone is an indicator of age at first maturity at about 25–30 years for most stocks (Table 15).

TABLE 15
Estimated mean age at first sexual maturity for orange roughy

Location, year	Method	Male	Female	Combined sex	Source
		(years)			
St. Helens, Australia (1992)	Transition zone	30.5	31.5	–	Kloser *et al*. (2015)
St. Patricks, Australia (1987)	Transition zone	30.0	31.0	–	Kloser *et al*. (2015)
St. Helens, Australia (2010)	Transition zone	28.0	29.0	–	Kloser *et al*. (2015)
St. Patricks, Australia (2010)	Transition zone	–	29.0	–	Kloser *et al*. (2015)
Chile	Lengths of mature fish	–	–	40.5	Payá *et al*. 2005
Bay of Plenty, New Zealand	Transition zone	26.1 (4)	27.5 (5)	26.5	Horn, Tracey and Clark (1998); MPI (2016a)
Namibia	Not reported	–	–	20–30	Boyer *et al*. (2001)
Namibia	Transition zone	21.2	23.3	21.8	Horn, Tracey and Clark (1998)
Ritchie Banks, New Zealand	Transition zone	26.0	25.5	25.7	Horn, Tracey and Clark (1998)
Ritchie Banks, New Zealand	Transition zone	–	–	31.5 (7.1)	MPI (2016a)
Ritchie Banks, New Zealand	Assessment model	–	–	35.0 (10)	MPI (2016a)
Chatham Rise, New Zealand	Transition zone	29.1	29.2	29.2 (3)	Horn, Tracey and Clark (1998); MPI (2016a)
East Chatham Rise, New Zealand	Transition zone	–	–	28.5 (4.6)	MPI (2016a)
East Chatham Rise, New Zealand	Assessment model	–	–	41.0 (12)	MPI (2016a)
Northwest Chatham Rise, New Zealand	Transition zone	–	–	28.5 (4.6)	MPI (2016a)
Northwest Chatham Rise, New Zealand	Assessment model	–	–	37.0 (13)	MPI (2016a)
Puysegur Bank, New Zealand	Transition zone	26.6	27.6	27.2	Horn, Tracey and Clark (1998)
Challenger Plateau, New Zealand	Transition zone	23.1	23.6	23.4 (3)	Horn, Tracey and Clark (1998); MPI (2016a)
Challenger Plateau, New Zealand	Assessment model	–	–	32.0 (10)	MPI (2016a)
Northeast Atlantic, Hatton Bank	Transition zone	36.8	34.2	25.4	Horn, Tracey and Clark (1998)
Northeast Atlantic	Not defined	–	–	30.0	Shephard and Rogan (2004) cited in ICES (2015b)
Northeast Atlantic, Porcupine Bank	Ages of mature fish	–	27.5	–	Minto and Nolan (2006)

Note: Values in parentheses refer to the number of years between the mean age at first maturity (A_{50}) and the age where 95 percent of fish are mature (A_{to95}).

In New Zealand, stock assessment models that assumed maturity estimated from the transition zone, with fishery selectivity estimated from length frequency samples taken from spawning aggregations or from UTF-based fisheries, found that the age-at-maturity was earlier than the age of selectivity (MPI, 2013). This resulted in a proportion of the mature stock that could not be caught by the fishery, a so-called "cryptic biomass". However, immature fish had been observed in commercial catch samples, suggesting fish are selected before becoming mature, and consequently the age of selectivity should be lower than the age of maturity (Francis, 2006). Dunn and Forman (2011) suggested cryptic biomass could be real, however, a consequence of spatially discrete fisheries interacting with ontogenetic structuring within the stocks. It has also been suggested that cryptic biomass might be partly a model artefact, caused by the model trying to fit trends in biomass indices (Dunn, 2009). Although the existence of, and rationale for, cryptic mature biomass has been unclear, the assumption of cryptic spawning biomass in relation to fisheries management has been considered unwise, and not precautionary, because it could result in assumptions that stocks were greater than estimated and, as a result, could not be overfished (Francis, 2006, Dunn, 2009). Recent stock assessments have made the simple and pragmatic assumption that age-at-maturity should be estimated from the age structure in the spawning aggregations (Cordue, 2014a, 2014b; MPI, 2016a), which resulted in estimated age-at-maturity of 4–13 years greater than that estimated from the transition zone (Table 15).

5.3 FECUNDITY

Orange roughy are determinate spawners that release eggs in at least two clutches (Doonan, Tracey and Grimes, 2004). Female fecundity increases almost linearly with body weight, or roughly exponentially with fish length (Pankhurst and Conroy, 1987; Bell, 1989; Clark, Fincham and Tracey, 1994; Minto and Nolan, 2006). Off New Zealand, an orange roughy of 1.2 kg has been estimated to produce about 30 000 eggs, with one of 2.8 kg producing about 80 000 eggs (Pankhurst and Conroy, 1987), although relative fecundity can vary between stocks (Clark, Fincham and Tracey, 1994). In the northeast Atlantic, relative fecundity has been found to be similar, with about 32 000 eggs at 1.2 kg and 87 000 at 2.8 kg, but fish in this location grow larger, leading to greater overall fecundity, with a 5 kg fish producing about 170 000 eggs (Minto and Nolan, 2006). However, Pitman, Haddy and Kloser (2013) reported a roughly linear relationship between fecundity and fish length, with fecundity increasing from about 45 000 eggs at 1.2 kg to 124 000 eggs at 2.8 kg. Koslow *et al*. (1995a) reported a decrease in fecundity after age 60 years, but the effect was weak, and Minto and Nolan (2006) found no decrease in fecundity with age.

Pitman, Haddy and Kloser (2013, 2014) concluded that fecundity had increased between the periods 1987–1992 and again by 2010, and that this could be a response to fishing and reduction in stock size. Some concerns have been raised about the veracity of this study, in particular that the fecundity estimates were based upon relatively few fish (Kennedy, 2014; Pitman, Haddy and Kloser, 2014). Koslow *et al*. (1995a) similarly reported an increase in fecundity off east Tasmania between 1987 and1992, a period when there was intense fishing and the stock declined substantially, while Clark, Fincham and Tracey (1994) found no change in fecundity of fish from the New Zealand Challenger Plateau over a four-year period when stocks were decreasing. Table 16 presents estimates of absolute and relative fecundity from different fisheries around the world.

TABLE 16
Estimates of orange roughy female fecundity.

Location	Date	Absolute fecundity	Average relative fecundity (per kg)	Reference
Australia, New South Wales	1988–89	42 787#	–	Koslow *et al*. (1995a)
Australia, East Tasmania	1987–92	31 085#	21 330	Koslow *et al*. (1995a); Pitman, Haddy and Kloser (2013)
Australia, South Australia	1988–89	35 339#	–	Koslow *et al*. (1995a)
Australia, East Tasmania	2010	23 927 – 121 360	36 890	Pitman, Haddy and Kloser (2013)
Chile	2000	33 699 – 379 216	16 056 – 115 944	Young *et al*. (2000)
New Zealand, Kaikoura Coast	1986	26 000 – 90 000	22 000	Pankhurst and Conroy (1987)
New Zealand, Cook Canyon	1988–89	15 700 – 103 570	27 180	Clark, Fincham and Tracey (1994)
New Zealand, Chatham Rise	1990 1994	- -	31 500 25 000	Clark *et al*. (2000)
New Zealand, Challenger Plateau	1987 1988 1989 1990	13 680 – 74 160 10 670 – 83 820 17 980 – 71 450 15 900 – 84 290	30 500 26 870 26 190 27 000	Clark, Fincham and Tracey (1994)
New Zealand, Puysegur Bank	1991	21 780 –184 820	49 530	Clark, Fincham and Tracey (1994)
New Zealand, Ritchie Banks	1990	25 550 – 65 530	28 550	Clark, Fincham and Tracey (1994)
Northeast Atlantic	–	70 000 – 380 000	–	Du Buit (1995)
Northeast Atlantic	–	–	48 530	Gordon (1999)
Northeast Atlantic, Porcupine Bank	2002	20 352 – 244 578	33 376	Minto and Nolan (2006)

Note: All fecundity estimates were based upon gravimetric methods. # - mean absolute fecundity for the sample.

Fecundity in fish varies greatly with reproductive strategy (Murua *et al*., 2003), and as a result it is not easy to interpret or compare fecundity estimates. Nevertheless, orange roughy are generally described as having “low fecundity” (e.g. Branch, 2001; Clark, 2001).

5.4 SPAWNING

Orange roughy are synchronous spawners, aggregating during winter for a spawning event that usually lasts 2–3 weeks (Pankhurst, 1988; Branch, 2001), and they tend to release all of their eggs at a single spawning event rather than in batches. In the Southern Hemisphere, spawning usually takes place in mid- to late June or in July, and in the north Atlantic between November and March (Branch, 2001; MPI, 2016a). Orange roughy spawning aggregations, often called “plumes”, can extend up to 200 m off the seabed into mid-water (Branch, 2001). Spawning plumes may occur on flat seabed, which are areas that are rarely occupied outside of the spawning season, or alternatively above features such as hills, knolls and seamounts. On the northeast Chatham Rise in New Zealand waters, spawning plumes form in relatively close proximity (within about 100 km), and at the same time, near a canyon (the Rekohu plume), on flat ground (the Old plume), and above a seamount (“Mount Muck”) (Doonan, Horn and Ó Maolagáin, 2014a). There can be marked sex segregation within aggregations during spawning, with trawl catches being dominated by either male or female fish, but generally not in any predictable way (Pankhurst, 1988). The locations of peak spawning are generally the same each year, but on flat grounds have been observed to move by up to 20 km from year to year (Pankhurst, 1988). This suggests spawning, on flat grounds at least, is not always linked to a specific submarine feature. For spawning fish associated with features, spawning can be clearly associated with a discrete feature (Pankhurst, 1988). The timing of spawning tends to be fairly consistent from year to year within a spawning area, but can differ between regions in both New Zealand and Australia (Pankhurst, 1988; Bell *et al*., 1992). In New Zealand, the day of peak spawning varied by no more than 7 days over 13 years on Chatham Rise (Clark *et al*., 2000), but was observed to shift forward by about 2 weeks from early July to late June over 16 years on the Challenger Plateau (Anderson, 2006).

Spawning plumes are usually located, observed and mapped using acoustic methods and within-plume observations made using both acoustic and optical methods (Kloser, Koslow and Williams, 1996; Ryan, Kloser and Macaulay, 2009), as well as through catch sampling. Fish within spawning plumes have been observed directly using underwater cameras (Ryan, Kloser and Macaulay, 2009; O'Driscoll *et al*., 2012), but their actual movements within and outside of the plumes remain unknown. Investigations of turnover of orange roughy in spawning plumes around New Zealand have yielded mixed results. In some areas, little turnover was suspected (Clark and Tracey, 1993; Doonan *et al*., 2000; Bull *et al*., 2001), whereas elsewhere turnover seemed likely (Zeldis *et al*., 1997). Orange roughy do not appear to immediately disperse after spawning, as spent fish can be found in spawning aggregations, but it is likely that dispersal is complete within 3–4 weeks after the end of spawning (Pankhurst, 1988).

In two stocks off New Zealand, the Challenger Plateau and the east and south Chatham Rise stocks, stock recovery following depletion by fishing has been associated with the appearance of new spawning plume locations, somewhat removed from the historical main spawning ground (MPI, 2016a). In the case of the Challenger Plateau, at the time of writing, the historical main spawning ground no longer contains a plume, whereas for the east and south Chatham Rise stock, the historical main plume remains, albeit greatly reduced in size, with a new plume having appeared about 50 km to the west (although quite when this took place is unknown). The relationship between historical plume locations and such "new" or relocated plumes, and the implications for stock productivity, remain unknown, although samples from Chatham Rise indicates that the new plume has a greater proportion of younger fish (Doonan, Horn and Ó Maolagáin, 2014a, 2014b).

Orange roughy may not spawn every year, potentially the result of living in a low-energy environment where resources for growth and reproduction are relatively limited (Bell *et al*., 1992). However, it is increasingly recognized that the tendency for such "skip spawning" is not just restricted to deep-sea species, but common in the females of many fish species (Rideout and Tomkiewicz, 2011). Estimates of the non-spawning ratio for orange roughy are 1.01–1.91, with an overall median of 1.46, this implies that roughly two-thirds of female orange roughy spawn each year (Table 17). Only a single study was found that considered the frequency of non-spawning male orange roughy (Table 17). The two cases where the non-spawning proportion was estimated in consecutive years suggested significant variability from year to year, between 1.56 and 1.91 for Ritchie Bank off the east coast of New Zealand, and 1.41–1.82 for northeast Tasmania (Table 17).

Fishing on spawning aggregations may influence the stock through mortality, disturbance of spawning activity, and potential impacts on spawning habitat (Overzee and Rijnsdorp, 2014). For orange roughy only the former has been considered, with potential disturbance and habitat impacts unknown.

TABLE 17
Published estimates of the ratio used to raise spawning to total mature biomass of orange roughy

Ratio	Area	Data	Comment	Source
1.82	Australia	Trawl survey	23% resting, 27% atretic.	Bell *et al*. (1992)
1.11–1.41	Australia	Trawl survey	Same as Bell *et al*. (1992) but for subsequent years. For 1991 and 1992, the ratio was 1.11 for males (CV 0.2) and 1.41 for females (CV 0.06).	Koslow *et al*. (1995b)
1.20–1.46	Namibia	–	–	Kirchener and McAllister (2002)
1.10	New Zealand, East Cape	Trawl survey	1995 survey, CV 0.02.	
	Unpublished data (Dunn, Anderson and Doonan, 2008)			
1.35	New Zealand, Chatham Rise	Trawl survey	CV 0.04.	Unpublished data (Dunn, Anderson and Doonan, 2008)
1.17–1.38	New Zealand, Chatham Rise	Trawl survey	For the hill survey was 1.17 (CV 0.04); for the Spawning Box, 1.38 (CV 0.1).	Doonan *et al*. (1999)
1.25	New Zealand, Chatham Rise	Acoustic biomass survey	Total spawning biomass 7 200 tonnes, total mature abundance of 9 035 tonnes.	Smith *et al*. (2008)
1.01	New Zealand, Chatham Rise	Acoustic biomass survey	Outside of the spawning plumes and hills, 11% of adult fish were classified as resting.	Doonan *et al*. (2006)
1.56–1.91	New Zealand, Ritchie Bank	Trawl survey	The 1992–94 surveys were, in order, 1.63 (CV 0.2), 1.56 (0.09) and 1.91 (0.07).	Unpublished data (Dunn, Anderson and Doonan, 2008)
1.01–1.02	Northeast Atlantic	Proportion classified as resting in commercial catch	Likely to be biased (low) if the fishing focused on spawning aggregations (unclear).	Berrehar, Du Buit and Lorance (1998)
1.09–1.26	Northeast Atlantic	Trawl survey	–	Shephard and Rogan (2006)

Notes: All but two of these estimates are based on comparing the proportion of stage 3 and greater fish (fish assumed to spawn that year) with those above the maturity L_{50} (fish assumed to be mature). The other two estimates are made directly from wide-area surveys. All estimates are for females, unless specifically stated.

5.5 EGG, LARVAL AND JUVENILE DEVELOPMENT

Orange roughy produce relatively large clear eggs of 2.0–2.5 mm diameter when hydrated (Pankhurst and Conroy, 1987; Bulman and Koslow, 1995; Zeldis, Grimes and Hart, 1998). Unfertilized and newly fertilized eggs have about 200 small, bright orange oil globules that coalesce into a single globule by the four-cell stage (Zeldis, Grimes and Hart, 1998), although the oil droplet may also fragment if the egg is disturbed (Bulman and Koslow, 1995). Initial sampling of eggs at 600–900 m, consistent with the depth of spawning, has been made, but the eggs are positively buoyant and older eggs were mostly taken in the upper 300 m of the water column (Bulman and Koslow, 1995). Eggs ascend in the water column from spawning depth to the surface mixed layers at about 250–350 m/day (Bulman and Koslow, 1995; Zeldis, Grimes and Ingereson, 1995). Bulman and Koslow (1995) also found the temperature where spawning was taking place was about 6 °C, the surface temperature 12.5 °C, and orange roughy eggs developed faster in warmer water. The mean development time to hatching has been estimated at 278 h at 8 °C, 235 h at 10 °C, and 146 h at 12 °C, and eggs at 6 °C did not hatch (Zeldis, Grimes and Hart, 1998). Time to hatching under natural conditions has been estimated at 175 h (7.3 days) (Bulman and Koslow, 1995). The stages of egg development are described in detail by Zeldis, Grimes and Hart (1998).

Orange roughy larvae are about 5 mm long at hatching (Zeldis, Grimes and Hart, 1998). The youngest juveniles, at age zero and lengths of about 5 cm SL, would pass through most trawl nets, and have rarely been caught (Branch, 2001). These 0-group fish have been reported from Chatham Rise and off the west coast of New Zealand (Gauldie, 1998; Mace *et al*., 1990; Dunn *et al*., 2009a), and also near the Frankies ground off Namibia (see Branch, 2001). Juveniles recently settled in the demersal environment appear to disperse slowly away from their initial settlement locations and may be constrained by water temperature, with these early juvenile fish preferring relatively warm water (Dunn *et al*., 2009a). Larger juveniles (> 10 cm SL) have been commonly reported around New Zealand, and at lengths > 20 cm SL their overall distribution in demersal survey trawls is similar to that of adult fish (Dunn *et al*., 2009a). Off New Zealand, juveniles have been caught in demersal trawls at depths shallower than adults, most often at 850–900 m, compared with 850–1 300 m for adults (Dunn *et al*., 2009a). The few juvenile fish that have been found off Chile were also caught in relatively shallow water, 7 fish at 520 m, and 3 fish at 770 m (Lafon *et al*., 2010). Depth distribution inferred from stable-isotope studies of otolith and muscle tissue has suggested that juveniles in the northeast Atlantic may similarly start life shallower than adults (< 800 m for years 1–3), but they then go deeper (about 1 200–1 700 m) until they approach the age at maturity (25–30 years), eventually returning to the somewhat shallower adult depth range, 1 000–1 500 m (Shephard *et al*., 2007; Trueman, Rickaby and Shephard, 2013). In the northeast Atlantic, trawl surveys have captured juveniles of 5–25 cm SL at depths of 1 000–1 500 m, but rarely at depths < 1 000 m (Trueman, Rickaby and Shephard, 2013). This contrasts with trawl survey captures around New Zealand, where adults have been caught down to about 1 400 m, but juveniles have rarely been caught below 1 200 m (Dunn *et al*., 2009a). Juveniles tend to be captured by trawls on flat areas of seabed away from large underwater features, with orange roughy more frequently caught around such features when adult (Dunn and Devine, 2010; Dunn and Forman, 2011). In New Zealand, 713 research tows using fine mesh cod-end midwater trawls at depths of 10–1 146 m over bottom depths of about 400–1 200 m failed to capture any juvenile orange roughy, despite catching adult orange roughy and other relatively large and active fish species, suggesting juvenile orange roughy may be demersal (Dunn *et al*., 2009a).

6. Population structure

6.1 STOCK DISCRIMINATION TECHNIQUES APPLIED TO ORANGE ROUGHY

A wide variety of data sources can be used to determine population and stock structures, but it is recognized that no one technique will provide a comprehensive result and different techniques often provide inconsistent results. Although many scientific studies on stock structure consider only a single technique, a holistic approach, where all relevant and available data are integrated and synthesized, has been recommended as the best approach to evaluating stock boundaries (Pawson and Jennings, 1996; Begg and Waldman, 1999).

Harden-Jones (1968) defined fish stocks as units that would have predictable responses to management measures, where they "respond largely independently to the effects of exploitation, because recruitment, growth and mortality within the stock are of more significance than emigration or immigration to the stock". Stocks can also be more simply defined as "units assumed homogeneous for particular management purposes" (Begg and Waldman, 1999). For orange roughy, many stock assumptions are most accurately described by the latter definition, recognizing the primacy of management issues. It seems reasonable to deduce that the expansive and homogenous deep-sea environment may lead to high genetic connectivity and expansive populations, an observation supported by genetic studies (see below) and expectations of species that, such as orange roughy, have a global distribution (Gaither *et al*., 2016). However, experience has shown that orange roughy on and around UTFs can be rapidly depleted. Stock definitions and associated management should, therefore, also take account of the potential for localized depletion, although it may not be an issue at the population scale. On Chatham Rise in New Zealand, for example, the current assumption of two stocks was based upon a holistic evaluation, but the region had previously been assumed to have between one and five stocks (Dunn and Devine, 2010). The five-stock assumption was recognized as biologically unrealistic, but at the time it was used to permit precautionary measures to help avoid localized depletion of resources on hill complexes particularly important to the fishery (Dunn and Devine, 2010). However, the genetic make-up of orange roughy throughout the entire New Zealand region (and further afield) indicates that there appears to be a single genetic population (Varela, Ritchie and Smith, 2012, 2013). Therefore, it is important to recognize that stock assumptions can integrate various biological, fisheries and management issues.

The information that has been used to evaluate stock structure in orange roughy is extensive, and has be categorized as: (i) fisheries data; (ii) fish appearance (morphology); (iii) life-history characteristics; (iv) fish movements and tagging data; (v) chemical and molecular techniques; and (vi) habitat boundaries.

6.2 FISHERIES DATA

The initial assumption for stock boundaries is often the default fishery management zones. This assumption has been used, for example, to define orange roughy stock boundaries in Australia (Fishery Zones), waters of the European Union (Member Organization) (ICES Subareas) and in New Zealand (FMAs). At a higher level of resolution, stock boundaries can be inferred from the spatial distribution of catches (commercial and/or research), with boundaries added where there are substantive gaps in the catch distribution. This has been used for orange roughy on Chatham Rise (Dunn and Devine, 2010), the Tasman Sea and western South Pacific (Clark *et* *al*., 2016a). However, the distribution of catches may reflect just patterns in fishing effort or fish abundance suitable for commercial fishing and not population or stock distribution. Commercial fishery CPUE trends have also been used to help define stocks (Dunn and Devine, 2010). If assumed to index biomass, then CPUE trends can be a direct measure of stock dynamics. However, when CPUE is localized, for example on specific UTFs, it may be indexing only a local resource or subcomponent of a stock, and may not be representative of the stock as a whole. This can result in biased stock assessments when only UTF CPUE is considered, because CPUE and catch data can indicate a stock size that proves to be inadequate to support observed future catches. This is because the UTF is not the entire stock, and future catches are being supported through immigration (Clark and Dunn, 2012).

6.3 FISH APPEARANCE (MORPHOLOGY)

Morphometrics is the measurement of organism (including fish) shape, and meristics is the enumeration of features such as fin rays or scales. Haddon and Willis (1995) observed differences in orange roughy head length, snout length, orbit diameter, maxilla width, premaxilla length, caudal peduncle, and gill raker count, from two sites in New Zealand waters. They concluded that phenotypic differences were greater between sites than within sites, suggesting this approach could be useful in distinguishing stocks. Elliott, Haskard and Koslow (1995) measured more numerous characteristics for orange roughy from five locations around southern Australia, and observed considerable variability, both within and between sites, with temporally and spatially discrete samples showing up as different. However, Elliott, Haskard and Koslow (1995) also found the greatest differences were between samples from the same site, taken in different years, a result that suggests morphology might have some severe limitations in discriminating between stocks. Meristics and otolith morphology have the advantage that they are less affected by handling and fish condition. Gauldie and Jones (2000) measured otolith shape and found standardized otolith widths indicated different stocks of orange roughy on the Chatham Rise compared to fish from Ritchie Bank and the Challenger areas. Smith *et al*. (2002) used a more sophisticated otolith shape analysis (Fourier analysis), which also indicated potential stock structure between the northern and southern parts of the Challenger Plateau.

6.4 LIFE-HISTORY CHARACTERISTICS

Differences in life-history characteristics are a potentially powerful tool for stock discrimination, as they directly describe aspects of stock dynamics. The mean age, length and otolith radius at the transition zone (which has been assumed to mark the onset of maturity) have all been found to differ among New Zealand, Namibian and northeast Atlantic stocks (Horn, Tracey and Clark, 1998). The mean age and length-at-maturity differed among four sites from the west of New Zealand, but did not allow a consistent picture of stock structure (Smith *et al*., 2002). Bell *et al*. (1992) noted that orange roughy off New South Wales (Australia) had a smaller length-at-first-maturity than other Australian stocks. However, the estimated age- or length-at-maturity can be biased by population structure, especially when the samples are taken from the spawning period when the population often becomes stratified by age and/or size (Francis and Clark, 1998).

There have been few studies comparing growth of orange roughy from different stocks, although at a global level there are clear differences (Table 11). Although Gauldie and Jones (2000) did compare growth rates among stocks, their ageing method is no longer considered valid. More frequently used are simple comparisons of length frequency distributions, or mean lengths, with a difference between sites taken as indications of stock structure (Ward and Elliott, 1993; Elliott, Haskard and Koslow, 1995; Smith *et al*., 2002; MPI, 2016a). Although length samples may show important pointers to stock differentiation, such as recruitment events, biased comparisons may also result from ontogenetic shifts between habitats (Dunn and Devine, 2010).

Trends in fish condition have also been examined as a tool to explore stock structure, with similar or divergent trends taken as indications of stock structure (Dunn and Devine, 2010). However, the condition of fish from the same stock may vary when sampled from different habitats (Dunn and Forman, 2011) as well as from different periods.

The location and timing of spawning is commonly used as an indicator of stock structure, with the assumptions that fish cannot be in two places at the same time and that there is site fidelity in spawning fish. Smith *et al*. (2002) observed simultaneous spawning in separate locations off the west of New Zealand, but also found considerable interannual variation in the timing of the start of spawning (3–4 weeks). Around New Zealand, although simultaneous spawning in different locations is the principal criterion used to separate some stocks (e.g. the East Cape and Mid-East Coast stocks), elsewhere multiple, simultaneous spawning locations are accepted to exist within a single stock (e.g. east and south Chatham Rise, where there are at least eight such locations) (Dunn and Devine, 2010; MPI, 2016a). Orange roughy are total spawners, releasing all their eggs in a short period, rather than batch spawners that release eggs in intervals over extended periods. This means that spawning females in one location are unlikely to spawn elsewhere in that year. In regions historically heavily fished off Australia, the lack of identified alternative winter spawning sites has been used as evidence to inform stock boundaries (Kloser *et al*., 2015). Bell *et al*. (1992) found orange roughy off New South Wales had a different time of spawning compared to other Australian stocks; the argument for stock separation in this case was augmented by New South Wales fish also having greater fecundity and smaller length-at-first-maturity (Bell *et al*., 1992). Differences in the timing of spawning in high seas fish on the Louisville Ridge in the western South Pacific have also lead to a re-evaluation of potential stock boundaries that were previously mostly based on the spatial separation of catches (Clark *et al*. 2016a).

The presence of nursery grounds has been used as a criterion for stock definition, under the assumption that a discrete stock should contain all components of the species life history (Dunn and Devine, 2010). Such patterns can be established from length-frequency analyses, but the rarity of juveniles in some areas effectively precludes this approach (Lafon *et al*., 2010), and the near-continuous distribution of juveniles despite other potential stock separation indicators reduces the usefulness elsewhere (Dunn *et al*., 2009a; Dunn and Devine, 2010).

Although mortality rates and recruitment patterns are also indicators of stock dynamics, and might be used as indicators of stock separation, these have not been used for stock definition in orange roughy to date. This is probably due to the difficulties of achieving precision in ageing and in the estimation of natural mortality.

The indirect (non-mortality) effect of selective fishing gear is becoming increasingly understood for shallower-water species, where selective fishing gear has brought about adaptive responses such as increased growth rate, earlier maturation, and greater reproductive investment in fishes (Kuparinen and Festa-Bianchet, 2017). Although increases in reproductive investment subsequent to fishing have been measured in orange roughy, such changes cannot be inherited because the fisheries have not yet existed for more than one generation of fish (Kloser *et al*., 2015).

6.5 FISH MOVEMENTS AND TAGGING DATA

Following the movements of individual fish provides a straightforward measurement of stock connectivity. Movements can be identified or inferred, and distance and direction estimated using natural or artificial tags. However, the interpretation of movement data is not always straightforward, as stocks may mix at certain times, such as during the spawning period, and be separate at other times. Such a complex temporal distribution has been offered as an explanation for conflicting results in genetic studies of orange roughy in the northeast Atlantic (Carlsson *et al*., 2011).

Orange roughy movements have been inferred from CPUE analyses. The migration of pre- and post-spawning orange roughy has been inferred by following the localized peak in commercial CPUE, which has suggested a migration rate of about 10 km/day (Taylor, 1993; Coburn and Doonan, 1994, 1997; Doonan and Coombs, 2004). Doonan, Tracey and Grimes, (2004) also combined trawl survey catch rates with measurements of macroscopic maturity stage to infer the post-spawning movement of orange roughy.

Parasites occur naturally in fish and can be used as biological tags. The unique occurrence of certain species or groups of species of parasites in fish sampled from different areas can indicate discrete fish populations, and therefore stock separation. Lester *et al*. (1988) collected parasites from orange roughy off Australia and New Zealand, and the occurrence of several species of larval nematodes and cestodes permitted discrimination between five Australian and three New Zealand stocks. No significant differences in parasite fauna were detected between samples of fish taken within the spawning season and those taken outside the spawning season in the same area. Despite genetic studies suggesting high connectivity (see below), such results suggest that for much of their life history orange roughy might be a relatively sedentary species, which would not be unexpected for fish living in a low productivity, deep-sea ecosystem. Gauldie and Jones (2000) did not determine the species of parasites, but found no differences in aggregate parasite load for orange roughy around New Zealand.

To date, there have been only exploratory attempts at tagging orange roughy with artificial tags (Latrouite *et al*., 1999; O'Driscoll *et al*., 2013), although equipment exists that might allow orange roughy to be tagged and released *in situ* in future (Sigurdsson, Thorsteinsson and Gústafsson, 2006).

6.6 CHEMICAL AND MOLECULAR TECHNIQUES

Edmonds, Caputi and Morita (1991) measured the elemental composition of whole otoliths from orange roughy from southern Australia, with multivariate analyses showing that patterns of elemental composition were specific to different areas, suggesting little movement of fish and therefore stock separation. Thresher and Proctor (2007) performed a more detailed analysis to examine ontogenetic variability in elemental composition of Australian, New Zealand, and northeast Atlantic orange roughy otoliths, and found spatial differences in the occurrence of strontium, again interpreted as an indicator of a sedentary lifestyle and complex population structure.

Allozymes are enzymes that are coded from the same location (locus) on DNA, but they are variable, and so can be used as molecular markers to gauge evolutionary histories and relationships. The analysis of allozymes uses gel electrophoresis to separate the proteins depending on their size, shape and charge, and it is a relatively straightforward and cheap technique to perform. However, this technique requires samples to be fresh, as enzymes can denature and be unusable within a short period. The ideal molecular marker for evaluating population structure is one that can be assumed to be neutral, meaning that the variability is not caused by natural selection, but by genetic drift. Allozymes may not be neutral (Lemaire *et al*., 2001; McPherson, O'Reilly and Taggart, 2004), but have nevertheless been successfully used for stock discrimination (Cuéllar-Pinzón *et al*. 2016). In orange roughy, Smith (1986) used allozymes and found little genetic differentiation among samples from the Tasman Sea, around New Zealand, and the northeast Atlantic, areas separated by as much as 21 000 km, and noted that this degree of homogeneity was unusual for a marine teleost.

Elliott and Ward (1992) also found no evidence of subdivision among allozyme samples from six sites across southern Australia, a distance of about 3 000 km, and one from New Zealand. They also estimated that as few as 200 migrants per generation would be sufficient to maintain the observed genetic homogeneity. Conversely, Smith and Benson (1997) found allozyme differences between five sites on the east coast of New Zealand, and those on Chatham Rise. They also found spatial and temporal heterogeneity within orange roughy from Chatham Rise, with temporal differences at individual sites suggesting a "more complex structure than simply geographically isolated stocks". Smith and Benson (1997) concluded there was evidence for isolation by distance for orange roughy to the east of New Zealand. Smith, Benson and McVeagh (1997) then reported significant differences between four sites from eastern and southern New Zealand, except for two from the east coast of New Zealand and the Chatham Rise.

Markers from mitochondrial DNA can be used to explore maternal relationships and often reveal greater genetic variation than allozymes, and with considerably less stringent tissue quality requirements. Mitochondrial DNA, like nuclear or genomic DNA (see below), may also not be entirely neutral (Ballard and Kreitman, 1995). Ovenden, Smolenski and White (1989) found evidence for separation between orange roughy off the east and west coasts of Tasmania. Smolenski, Ovenden and White (1993) found a similar pattern, with evidence of partial separation of fish from New South Wales from those around Tasmania and southern Australia, which could not be separated. At a larger geographical scale, Elliott, Smolenski and Ward (1994) found very limited, although significant, differences between Australian and northeast Atlantic samples, suggesting there was some gene flow between these distant areas, while Baker *et al*. (1995) found no significant differences between orange roughy from New Zealand, Tasmania and South Africa. Conversely, around New Zealand, Smith, McVeagh and Ede (1996), and Smith, Benson and McVeagh (1997) found evidence for separation between southern sites, fish from the Challenger Plateau, and sites off the east coast of New Zealand. Smith *et al*. (2002) found further differences between orange roughy from the Challenger Plateau and Lord Howe Rise, both off the west coast of New Zealand. While Oke, Crozier and Ward (2002) found no differentiation within Australian waters, and no evidence of differentiation between samples from Australia and New Zealand, and Australia and the northeast Atlantic, they did report evidence of a pattern of isolation by distance.

Varela, Ritchie and Smith (2012) used mitochondrial DNA sequences (subunits of the genes for cytochromes *b* and *c*) and found no significant differentiation among samples from Australia, Chile, Namibia and New Zealand, but low differentiation for two sites in the northeast Atlantic. Varela, Ritchie and Smith (2013) then used microsatellites (a microsatellite is a tract of repetitive DNA) and detected low but significant difference between the Southern Hemisphere regions (Australia, Chile, Namibia and New Zealand) and the northeast Atlantic, but with genetic homogeneity between New Zealand and Australia.

Nuclear DNA has more possibilities than mitochondrial DNA as a source of genetic markers because it is larger in size and is inherited from both parents. White *et al*. (2009) used nuclear microsatellite loci and could not reject panmixia among five samples from the northeast Atlantic, with some sampling sites separated by over 2 000 km. However, they found significant differentiation between the pooled northeast Atlantic samples and one sample from Namibia (White *et al*., 2009). In contrast, Carlsson *et al*. (2011) used a different set of microsatellites and reported low but significant differentiation within samples from the west of Ireland, mainly due to genetic differences between samples from a flat area and seamount sites. Carlsson *et al*. (2011) suggested that differences in seafloor structures (seamounts and flats) may provide discrete spawning structures with limited gene flow between them. Gonçalves da Silva, Appleyard and Upston (2015) analysed single-nucleotide polymorphisms in the genotype of orange roughy from five sites around Tasmania, and found very high levels of gene flow, with no evidence of local adaptation, indicative of a single genetic unit.

In general, studies focused on methods other than genetics have found differences between stocks, whereas genetics studies have not (Branch, 2001). There appears to be genetic connectivity over large areas, with some isolation by distance. Because egg and larval dispersal is moderate, connectivity seems likely to occur through active adolescent or adult dispersal (Varela, Ritchie and Smith, 2012, 2013; Gaither *et al*., 2016). However, this does not mean that populations are demographically connected, as genetic connectivity can occur with reproductive movement of relatively few fish per generation (200 or fewer) (Vucetich and Waite, 2000; Elliott and Ward, 1992), and the above demographic studies often inferred distinct stock differences.

6.7 HABITAT BOUNDARIES

A simple indicator of stock separation is a habitat boundary, which might be a land barrier, an oceanic trench, or extreme geographical distance. Smith *et al*. (2002) noted no major oceanographic features that might isolate Australian stocks. Dunn and Devine (2010) noted no clear breaks in geological habitat on Chatham Rise, but speculated that the presence of oceanic gyres close to spawning grounds, and an inferred preference of juveniles for relatively warm water (Dunn *et al*., 2009a), could contribute to stock separation.

6.7.1 Australia

There have been numerous studies related to orange roughy stock structure in Australian waters. The management authority asserts that "The population of orange roughy in Australian waters is known to be comprised of more than one stock, although the exact structure of these stocks is uncertain as is their relationship to one another" (AFMA, 2006). Stokes (2009) recommended a thorough review of all information relevant to determine stock structure in order to support assessment and management. It therefore appears that there has been no overall synthesis of stock structure for Australian orange roughy, and the structure used for assessment and management purposes is essentially a management zone approach, with nine discrete management areas.

The Eastern Zone and Pedra Branca from the Southern Zone produce the majority of catches and are assumed to constitute a single stock (Upston *et al*., 2014) and a separate stock assessment has been made for Cascade Plateau (Wayte and Bax, 2007). Some alternative stock structure hypotheses have been considered in stock assessments and run as sensitivities to base-case assessments (Wayte, 2007; Upston *et al*., 2014), which is generally considered good practice.

As noted above, the stock structure of orange roughy in Australian waters is uncertain (Koslow *et al*., 2000; Stokes 2009), despite the numerous studies, including morphology (Elliott, Haskard and Koslow, 1995), parasites (Lester *et al*. 1988), enzymes (Elliot and Ward, 1992), otolith biochemistry (Thresher and Proctor, 2007), otolith shape (Gauldie and Jones, 2000), and genetics (Ovenden, Smolenski and White, 1989; Elliott and Ward, 1992; Elliott, Smolenski and Ward, 1994; Smolenski, Ovenden and White, 1993; Oke, Crozier and Ward, 2002; Varela, Ritchie and Smith, 2012). Based on the available evidence, Hordyk (2009) reported that orange roughy from the eastern and western coasts of Tasmania appear to be distinct from each other, and from those on the Cascade Plateau and South Tasman Rise. A recent population genomics study found that orange roughy from sites off the Tasmanian coast, in the Southern, Eastern and Cascade Plateau Management Zones, formed several genetic stocks in "drift connectivity" and potentially a single panmictic population (Gonçalves Da Silva, Appleyard and Upston, 2015). Another genomics study reported low levels of genetic differentiation within Australian waters; however, the loci used were found to have signatures for natural selection, suggesting three separate areas: Albany/Esperance, Hamburger Hill (in the Great Australian Bight), and south-eastern Australia (Gonçalves da Silva, Appleyard and Upston, 2015). A key question for stock assessment and management of orange roughy is whether these defined areas are demographically isolated (Gonçalves Da Silva, Appleyard and Upston, 2015). High-resolution genetics, combined with demographic and migration information, could help to better describe the biological stock structure in the future.

For management and stock assessment purposes, Australian orange roughy are assumed to be structured according to nine discrete Management Zones (based on geographical areas and underwater features): Eastern, Cascade Plateau, Southern, Western, Southern Remote, Northeastern Remote, South Tasman Rise, East Coast Deepwater, and Great Australian Bight (AFMA, 2015). Exceptions are the Eastern and Southern Zones, which are thought to have separate resident stocks but a major, shared, winter spawning location in the Eastern Zone (St Helens Hill or nearby St Patricks Head) (Bax, 2000; Upston and Wayte, 2012a; Kloser *et al.*, 2015). The stock hypothesis for the Eastern Zone base-case stock assessment assumes that the Eastern Zone and Pedra Branca from the Southern Zone constitute a single stock (Upston *et al.*, 2014). This hypothesis reflects the prevailing theory that a proportion of Southern Zone orange roughy, primarily from the Pedra Branca area, migrate to the main spawning grounds in the Eastern Zone to spawn in winter. This perspective is based on otolith shape data, observations from fishers and processors that suggest separation of Pedra Branca and Maatsuyker in the Southern Zone (Bax, 2000), and only one known main spawning aggregation occurring in the East (Kloser *et al.*, 2015). However, given the uncertainty in stock structure, spawning biomass estimates for three alternative plausible stock structure hypotheses including the Eastern, Southern and Western Zones are also reported (Upston *et al.*, 2014; Stokes, 2009).

6.7.2 Chile

Chilean orange roughy are assessed as a single stock in the Juan Fernandez archipelago. Genetic studies (Galleguillos *et al.*, 2008), parasite analysis (George-Nascimento *et al.*, 2008), length-at-first-maturity and relative fecundity analyses (Roa, Niklitschek and Lafon, 2008) have not indicted significant differences in fish between fishing areas within this area.

6.7.3 Namibia

Although Namibian orange roughy have been analysed for stock structure in some global studies, there have been no specific and detailed regional studies. The four aggregations targeted by fisheries have all been treated as separate stocks, Hotspot, Rix, Frankies, and Johnies (Branch, 2001; McAllister and Kirchner, 2002), which was considered a precautionary approach given the absence of information.

6.7.4 New Zealand

Around the north of the North Island (ORH 1), spawning is known from several areas on both the east and west coasts. Simultaneous spawning is used to separate the fishery in the Bay of Plenty from that just over 200 km away at East Cape (ORH 2A) (MPI, 2016a). Stock structure for the remainder of ORH 1 remains undescribed, although a number of subareas were proposed on the basis of catch distribution, seafloor topography, spawning sites, reported differences in length-frequency distributions and in hydrographic patterns between various UTFs (Clark *et al.* 2002; Dunn, 2017).

Off the east coast of the North Island (ORH 2A, 2B, 3A), simultaneous spawning at East Cape and to the south at the Mid-East Coast, has been used to delimitate separate stocks for these two areas (MPI, 2016a). The Mid-East Coast stock has been assumed to be separate from the Chatham Rise stocks based on the allozyme studies of Smith and Benson (1997) (MPI, 2016a).

For the ORH 3B area (Chatham Rise and Sub-Antarctic), two main stocks were initially recognized, Chatham Rise and Puysegur, based upon the allozyme studies of Smith, Benson and McVeagh (1997) (MPI, 2016a). Chatham Rise has then been subdivided into two stocks, one on the northwest Chatham Rise, and one on the south and east Chatham Rise, a division based on a holistic analysis of catch distribution, CPUE trends, spawning and nursery grounds, inferred migrations, size, maturity and condition data, genetic studies, and habitat boundaries (Dunn and Devine, 2010; MPI, 2016a). Although other discrete fisheries have been described within the large ORH 3B management area, such as on the Arrow Plateau and in several regions throughout the Sub-Antarctic (outside of Puysegur), no stock structure for these regions has been evaluated (MPI, 2016a).

On the Challenger Plateau (ORH 7A), a single stock is assumed for the southwest Challenger Plateau and Westpac Bank on the basis of simultaneous spawning, size structure, parasite composition, flesh mercury levels, allozyme studies, and mitochondrial DNA studies (MPI, 2016a).

On the west coast of the South Island (ORH 7B), separation from Puysegur Bank and Challenger Plateau is assumed from genetic studies, size structure, parasite composition, and simultaneous spawning.

6.7.5 Northeast Atlantic

Genetic results for the northeast Atlantic have been equivocal. White *et al*. (2009) could not reject panmixia for orange roughy samples taken from a wide geographical range across the northeast Atlantic, but did indicate orange roughy from the northeast Atlantic were different to those from Namibia. Carlsson *et al*. (2011) concluded limited but significant population structure between mounds and flat areas to the west of Ireland and suggested their contrasting result was because they were sampled during the spawning season, whereas White *et al*. (2009) did not, and orange roughy populations mix over substantial distances outside of the spawning season.

The current ICES practice is to assume three assessment units: Subarea VI; Subarea VII; and orange roughy in all other areas (ICES, 2015a). ICES (2015b) notes that orange roughy is an aggregating species and the current scale of management would not prevent sequential depletion of local aggregations. ICES recommends that where small-scale distribution is known, this be used to define smaller and more biological meaningful management units.

6.7.6 High seas

Stock structure of orange roughy in the Tasman Sea and western South Pacific has been evaluated using a holistic approach, integrating largely existing information (Clark *et al*., 2016a). Techniques that were found to be consistent, and given greatest weight in stock evaluation, included size structure, location and timing of spawning, and fishery distribution. Information from genetics (allozyme, mitochondrial, and DNA), life-history parameters (age- and length-at-first-maturity), otolith microchemistry, morphometrics, and parasites were found to show inconsistent patterns (Clark *et al*., 2016a). Seven high seas management areas were proposed, based on possible stock boundaries, being Lord Howe Rise, Northwest Challenger Plateau, West Norfolk Ridge, South Tasman Rise, and three subareas of the Louisville Seamount Chain, with an eighth stock being the Southwest Challenger Plateau (a straddling stock of ORH 7A, with catches largely within the New Zealand EEZ). There remains a possibility of other straddling stocks existing, including for the South Tasman Rise and West Norfolk Ridge areas. Orange roughy stock structure elsewhere on the high seas has yet to be evaluated.

7. Stock assessment

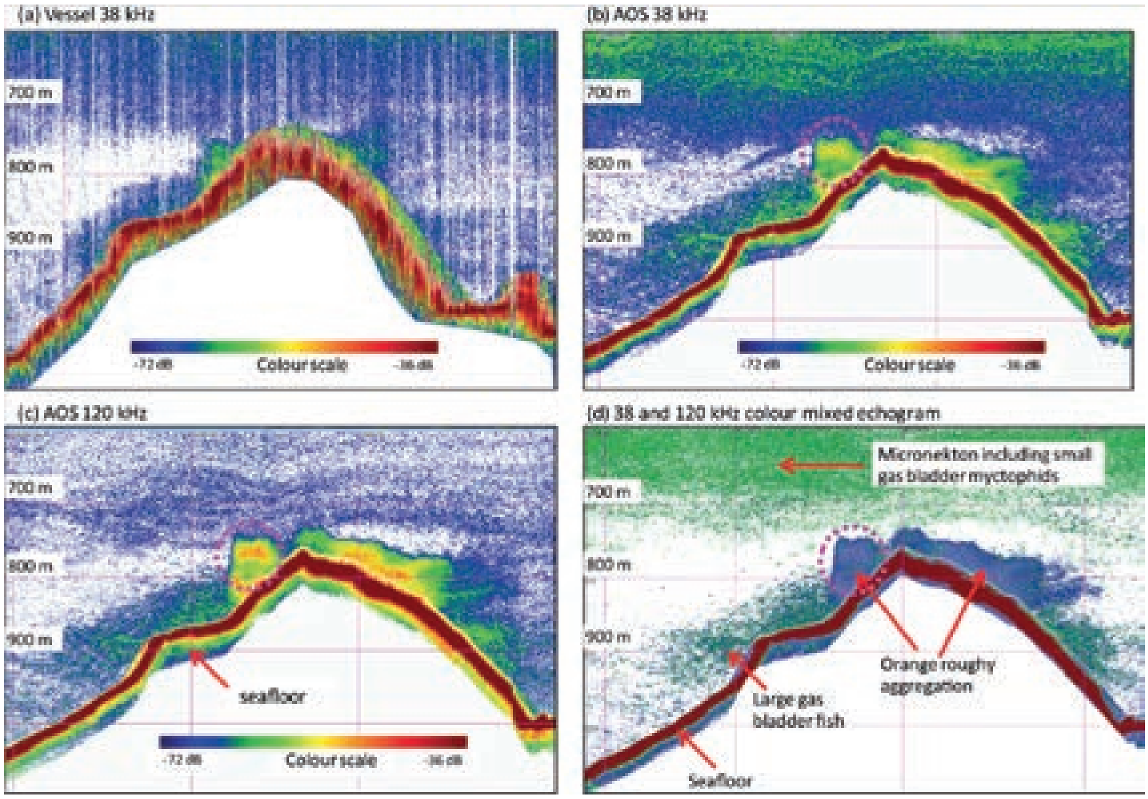
Recent developments in stock assessments in Australia and New Zealand have shown that many of the historical issues in the stock assessments for orange roughy were connected with inconsistencies or uncertainties in input data. These problems can be reduced through careful quality control and selection of those data, careful selection of plausible assumptions, and full documentation of all aspects of the assessment methods and assumptions. It appears that earlier assessments used data and/or assumptions of a nature and/or quality that compromised the ability of the models to provide credible answers on which to base advice. Historical published estimates of stock status, and perspectives of the fisheries based on such assessments (e.g. Hilborn, Annala and Holland, 2006a), may therefore be biased. This perspective is one of hindsight, while at the time there were plenty of sound reasons for the approaches used to collect the data and apply the assessment methodologies. In identifying this as an important issue, the intention is to advise future stock assessors to review their input data and assumptions in a careful, considered and critical way, and to exclude poor-quality data, or at the very least test the sensitivity of their models to the inclusion of such datasets and the underlying assumptions. Stock assessments will continue to develop over time as data, methodologies, and information on which assumptions are based are improved and updated. Full documentation of the data sources, methods and assumptions used in all stock assessments, including relevant historical information, should be part of the standard procedure to assess stocks.

Summary descriptions of approaches that have been used in stock assessments over time are given by Clark (1996), Branch (2001), Sissenwine and Mace (2007), and MPI (2016a). The main data and information constraints on assessments that are currently apparent include insufficient data to define stock structure in many areas (and understand within-stock structuring), a lack of information about recruitment variability and the stock-recruit model steepness parameter (h), only modest understanding of the natural mortality rate (M) and acoustic target strength (TS), poor precision in age determination, and too few, or unreliable, biomass indices in many areas.

While stock assessments over the years have correctly indicated reductions in stock biomass due to fishing, the more recent assessments for both Australia and New Zealand estimated the maximum levels of depletion to be lower than many of the earlier stock assessments (MPI, 2013; Cordue, 2014a; Upston *et al*., 2014).

A major advance in enabling the development of robust stock assessments has been through improvements in acoustic technology and associated analyses using acoustic echo integration methods, giving improved species identification and target strength estimation (Figure 15). This has delivered considerable improvement in the robustness of the estimates of spawning stock biomass using multifrequency echo sounders mounted on deep-towed platforms, including those attached to the headline of commercial fishing nets (Kloser *et al*., 2002; Kloser *et al*., 2011; Kloser *et al*., 2013; Macaulay, Kloser and Ryan, 2013; Ryan, Kloser and Macaulay, 2009; Ryan and Kloser, 2016). These improved spawning stock biomass estimates then underpin the statistical stock assessment models.

FIGURE 15
Echogram of volume backscatter (Sv) from transects over St Helens Hill, Australia



Notes: (a) Acoustic data at 38 kHz from vessel-mounted echo sounder (moderate weather conditions), echograms from a trawl mounted acoustic optical system (AOS) towed at a nominal depth of 450 m for (b) 38 kHz and (c) 120 kHz frequency transponders; (d) the corresponding composite colour-mixed 38 kHz and 120 kHz echogram with orange roughy schools and others sources of backscatter indicated.
Source: Kloser *et al.* (2013, Figure 2)..

7.1 AUSTRALIA

The most recent Australian orange roughy stock assessments are described in detail by Upston *et al.* (2014) for the Eastern Zone (see also Wayte, 2007), and for the Cascade Plateau by Wayte and Bax (2007) and Wayte (2009). Historical assessments, covering 1992 to the early 2000s, utilized a broad set of approaches depending on the available fishery information, ranging from simple CPUE analysis to more advanced stock reduction analysis and a full Bayesian assessment model (since 2000), and are described in CSIRO and TDPIF (1996), Bax (2000), Wayte and Bax (2002), and Wayte (2007), and reviewed by Deriso and Hilborn (1994), Francis and Hilborn (2002) and Stokes (2009). Deriso and Hilborn (1994) found the Australian stock assessment for the Eastern and Southern Zones, where most of the catch had been taken, to be much less ambiguous when compared with assessments of orange roughy in other areas. Deriso and Hilborn (1994) observed that three separate indices of abundance (acoustic surveys, total egg production, and CPUE) showed the same trend, which was a substantial decline approaching 30 percent of pre-fishery biomass by 1994, and the analysis of single or separate stocks hypotheses was also consistent (see Koslow *et al.*, 1995b; Bax, 2000). The Eastern Zone assessment, which has included Pedra Branca in the Southern Zone, was continually reviewed and refined over time. While uncertainty in the assessment remained substantial, the main premise, that the stock was substantially depleted, was accepted (Bax *et al.*, 2005). The 2002 full Bayesian stock assessment estimated that less than 15 percent of the pre-fishery spawning biomass remained (Wayte and Bax, 2002).

Until recently, few Australian orange roughy stock assessments had been completed since the early 2000s, partly due to the paucity of information from the commercial fishery, with declining fishing effort, and then subsequent closure of most areas of the Australian fishery in 2006. However, as part of the Orange Roughy Conservation Programme, biomass estimates and biological information continued to be collected from research surveys in the Eastern Zone and the Cascade Plateau (Ryan, 2006; Kloser, Sutton and Krusic-Golub, 2012). Stokes (2009) reviewed Australian stock assessments, focusing on the Cascade Plateau, Eastern, Western and Southern Zones, and highlighted some key areas for future work, including specifying Bayesian assessment model informed priors for the acoustic catchability parameter, improving documentation, a comprehensive review of all the data and assumptions, and completing Bayesian assessment model parameter estimation to the full Markov Chain Monte Carlo (MCMC) stage. A preliminary Eastern Zone orange roughy stock assessment was completed in 2011 (Upston and Wayte, 2012a) and further work was proposed, considering the reviews by Stokes (2009), Cordue (2011), and a CSIRO internal review (Upston and Wayte, 2012b). Subsequently, an Australian orange roughy workshop, held in May 2014 (AFMA, 2014a), discussed the fishery, reviewed available data, assumptions, and stock assessment approaches for the Eastern Zone stock assessment, and developed a base-case model definition.

The latest Australian orange roughy stock assessment for the Eastern Zone, including Pedra Branca within the Southern Zone (Upston *et al*., 2014), was a full Bayesian assessment and is outlined below.

7.1.1 Eastern Zone orange roughy stock assessment

The main modifications to the 2014 assessment for Eastern Zone orange roughy were the inclusion of revised relative indices of spawning biomass based on the average of acoustic biomass survey snapshots in each survey year for two survey areas, St Helens and St Patricks, combined. A series based on the maximum observed snapshot values was also calculated because these had been used in the previous stock assessments. Informative Bayesian prior distributions were developed for the catchability coefficient for the acoustic surveys, and the Francis (2011) data weighting method was applied to select the statistical weights for the age composition data, placing most weight (i.e. emphasis in the fitting procedure) on the acoustic survey indices (direct measures of abundance) when the model was fitted. Previously, the McAllister and Ianelli (1997) weighting method, which is implemented in the Stock Synthesis software package (Methot and Wetzell, 2013), had been used (Upston and Wayte, 2012a). Other new data inputs were a revised egg survey estimate of female spawning biomass, a catchability coefficient for that survey, and an ageing error matrix using data from a recent re-ageing experiment. The re-ageing experiment was designed to investigate between-year bias in age estimates, a potential issue that was raised in reviews. The experiment found no evidence of a substantive bias in the early age readings for Eastern Zone orange roughy.

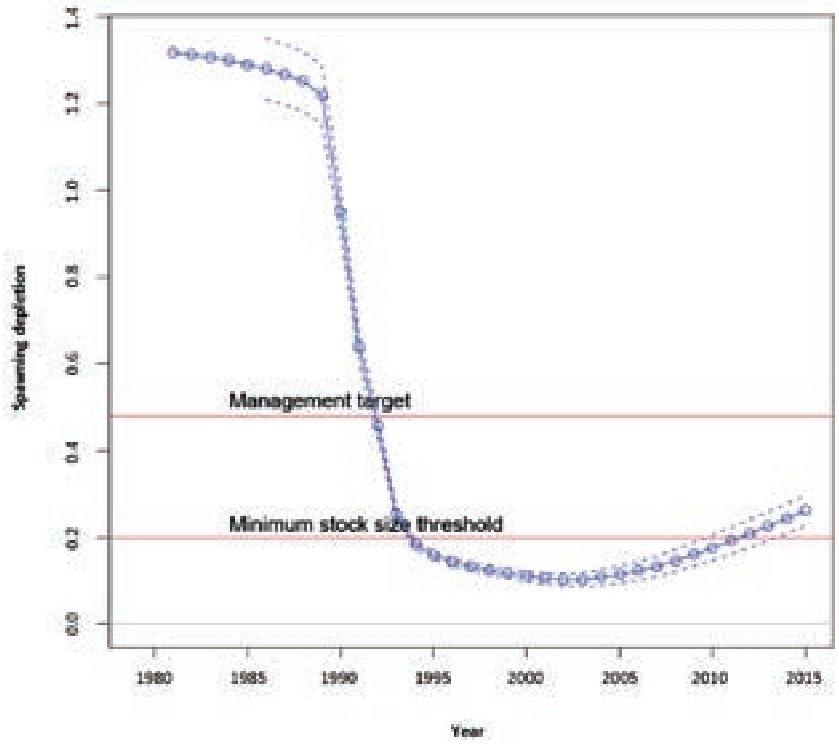
The assessment is based on a two-sex, age-structured population model that incorporated constant growth and natural mortality, but stochastic recruitment, to provide a series of annual female spawning stock biomass estimates given the catch history. The spawning population was modelled assuming the proportion of mature fish that were spawning in a given year was constant on average (as in Wayte, 2007). The base-case model assumed that the Eastern Zone and Pedra Branca from the Southern Zone constitute a single homogeneous stock. One trawl fishing fleet was modelled. Data inputs in the model included: catch (1985–2014); relative indices of abundance (spawning biomass) from acoustic towed body surveys (1991–1993, 1996, 1999, 2006, 2010 and 2012–2013) and vessel hull echosounder surveys (1990, 1991 and 1992); an absolute estimate of female spawning biomass from an egg survey (1992); and male and female age-compositions from spawning aggregations (1992, 1995, 1999, 2001, 2004 and 2010).

Biological parameters that were pre-specified in the model included the natural mortality rate (M), von Bertalanffy growth model parameters, length-weight parameters, and the variation in mean size-at-age. Recruitment was assumed to be distributed about a Beverton-Holt stock-recruitment relationship, with pre-specified values for recruitment steepness (h), variability (quantified by σ_R), and the extent of how bias-correction changes over time. Parameters estimated within the model included unexploited recruitment (R_0) and 76 recruitment deviates (year class strengths [YCS]), fishery selectivity (assumed to be a length-based logistic function, with parameters estimated for inflexion and width to 95 percent selection), and catchability (q) coefficients for the acoustic towed and hull surveys. Maturity was modelled as a logistic function of length. The model was fitted, and the parameters governing maturity as a function of length were set to match the estimated selectivity of the spawning aggregations. This approach of equating orange roughy being present on the spawning grounds with maturity (which will differ from functional maturity) is consistent with recent assessments of orange roughy in Australia and New Zealand that have been undertaken (Wayte, 2007; Cordue, 2014a). Fecundity-at-length was assumed to be proportional to weight-at-length.

Uncertainty in parameter estimates for the base-case model was calculated using asymptotic standard errors or MCMC methods. Sensitivity of the model to alternative assumptions, such as different stock structures, steepness (h), and alternative data weightings, were also investigated. The model fits to the expected values for abundance indices, and the age data were qualitatively assessed and were judged to be good (Upston *et al*., 2014).

The model estimated female spawning biomass in 2015 to be 26 percent of the unfished level (from the maximum posterior density estimate), which was close to the

FIGURE 16
Trajectory of spawning biomass as a proportion of initial biomass for the Australian Eastern Zone assessment



Notes: 95 percent asymptotic confidence intervals.
Source: Upston *et al*. (2014).

median Bayesian estimate of 25 percent (95 percent confidence interval [CI] of 23–28 percent). The outcome is consistent with that from the 2006 Eastern Zone orange roughy stock assessment, which forecasted that the biomass would reach the limit level of 20 percent of the unfished level in 2014 (Wayte, 2007).

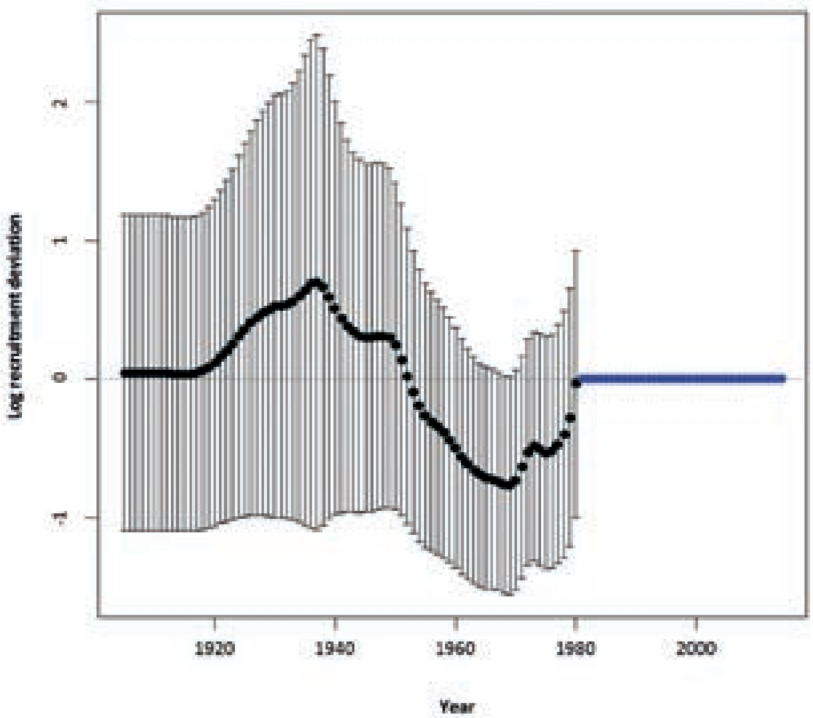
The trajectory of female spawning biomass relative to unfished levels implies a pattern of steep decline in the early 1990s, as the commercial fishery developed, followed by a period of further, gradual decline between 1995 and 2005, and a recent increase to levels above 20 percent (Figure 16).

The model estimated a pattern of recruitment that oscillates from high to low prior to the start of the fishery (Figure 17). As a result of this recruitment pattern, the unfished stock at the start of the fishery is estimated to be about 30 percent larger than it would be on average (Figure 16). The recruitment deviations were not estimated after 1980; instead, expected average recruitment from the spawner recruitment curve was assumed.

The catchability coefficients for the towed body and hull-mounted acoustic surveys were estimated by the base-case model to be 1.32 and 1.78, respectively. These were substantially higher than 1, which implies that the surveys saw substantially more fish than were actually present in the stock. While this is disconcerting, both estimates were within the bounds of the priors, and as such the result was statistically acceptable.

Assumptions regarding stock structure are a key uncertainty in the assessment, as the model outcomes differed depending on this assumption. The Australian assessments are unique in considering this aspect of uncertainty. The base-case model was also sensitive to the inclusion of recruitment deviations, higher earlier catches and, to a lesser extent, the data weighting method for the age compositions.

FIGURE 17
Trajectory of estimated recruitment deviations for the Australian Eastern Zone assessment



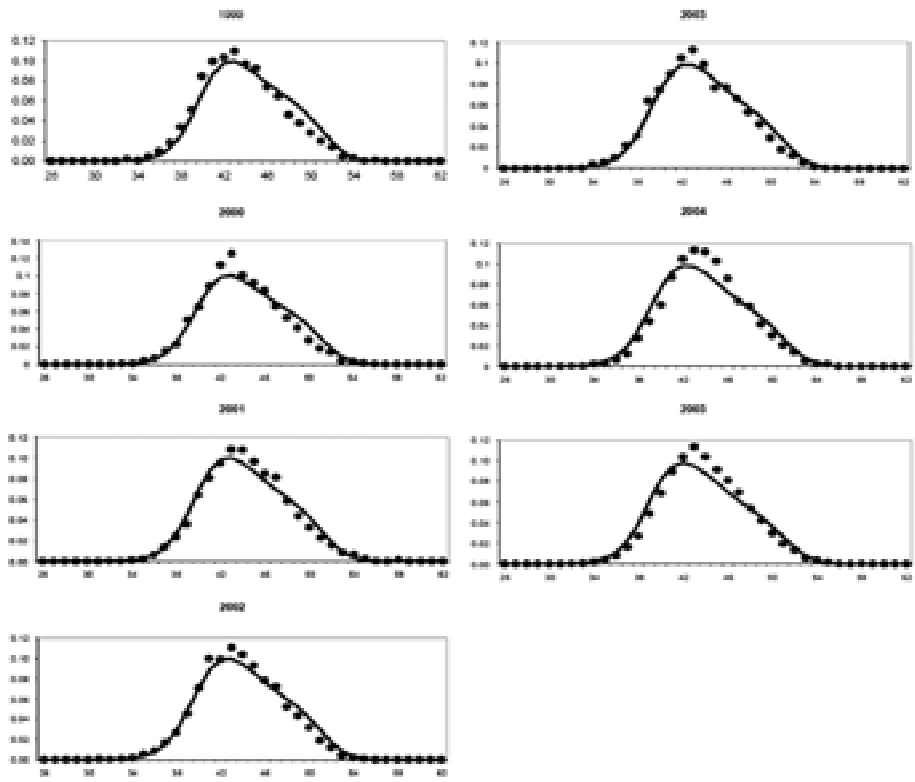
Notes: Year class strengths. Log scale. Recruitment deviations are not estimated after 1980; instead, expected recruitment (from the stock-recruitment curve) was assumed.
Source: Upston *et al*. (2014).

7.2 CHILE

The Chilean stock assessment for orange roughy is described in detail by Payá *et al.* (2005) and Payá (2013). The most recent population model was partitioned by age and sex. The observational data to which the model was fitted were a standardized commercial CPUE index for the years 2000–06, length frequency compositions from the commercial fishery (1999–2005), acoustic biomass survey estimates (2003–06), and length-frequency compositions from trawl catches made during the acoustic biomass surveys (2003–06). Both the CPUE and acoustic biomass series were treated as relative biomass indices. The selectivity of the commercial fishery and of the acoustic biomass surveys were modelled using logistic ogives. Proportions-at-length were assumed to have multinomial errors, and biomass indices to have lognormal errors. The pre-specified (fixed) parameters in the model were natural mortality rate (M), the steepness parameter of the Beverton-Holt stock-recruit relationship (h), the parameters of a logistic maturity-at-age ogive (A_{50}, A_{to95}), von Bertalanffy growth model parameters (L_∞, K, t_0), length-weight parameters (a, b), and the CV around mean length-at-age. The estimated model parameters included the selectivity parameters for both the commercial fishery and acoustic survey length frequency distributions, catchability coefficients for the CPUE and acoustic biomass indices, recruitment deviates, virgin biomass (B_0), and ageing errors (across ages 30–90+). Uncertainty in parameter estimates was calculated using asymptotic standard errors or MCMC methods.

The model fits to the observation data were good (Figures 18, 19 and 20). The spawning biomass was estimated to have declined between 1999 and 2005, and thereafter remained relatively stable following the closure of the commercial fishery. The model estimated maturity to take place before selectivity, although the difference between the ogives was small (about four years) and could have resulted from ageing

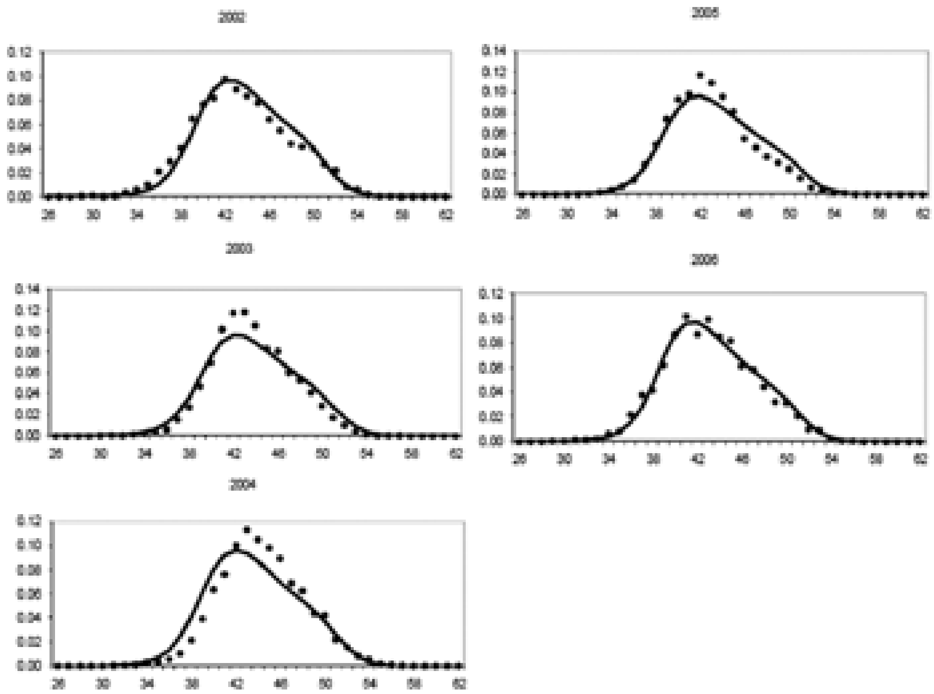
FIGURE 18
Chilean stock assessment model fits to the observed proportion-at-length in the commercial fishery



Notes: Lines = model fits; points = observed proportion-at-length in the commercial fishery.
Source: Payá (2013).

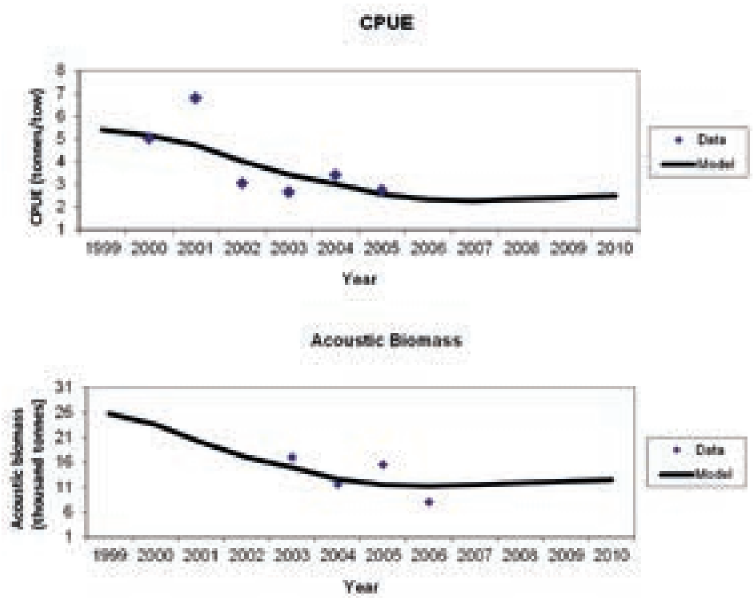
errors; however, this result would have implications for the estimation and selection of suitable biological reference points, because it would have resulted is a proportion of the mature biomass that was not available to fishing, i.e. a cryptic biomass.

FIGURE 19
Chilean stock assessment model fits to the observed proportion-at-length in the trawl catches associated with the acoustic biomass surveys



Notes: Lines = model fits; points = observed proportion-at-length in the trawl catches associated with the acoustic biomass surveys.
Source: Payá (2013).

FIGURE 20
Chilean stock assessment model fits to the observed relative biomass indices



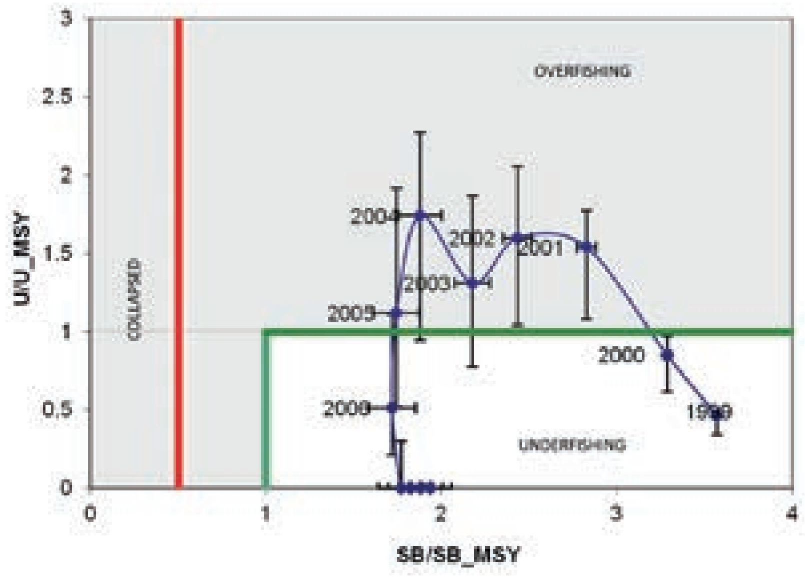
Notes: Lines = model fits; points = observed relative biomass indices. CPUE is in tonnes per tow, and the acoustic biomass index is in thousands of tonnes.
Source: modified from Payá, (2013).

The model was investigated for sensitivity to assumptions regarding the assumed natural mortality rate (*M*) and steepness (*h*). The derived quantities, including maximum sustainable yield (MSY) and long-term stock projections, were evaluated under different assumptions of mean age-at-maturity, fishery selectivity, steepness, and initial stock size (B_0). Stock assessment results were presented using phase plots (Figure 21). From the base-case model, MSY was estimated to be 2.5 percent B_0, similar to that estimated for New Zealand stocks (2.7 percent B_0) where similar model parameters were used (Francis, 1992). The biomass at MSY (B_{MSY}) was estimated to be 28 percent B_0, slightly greater than the B_{MSY} values of 21.8–24.5 estimated for four New Zealand stocks (Cordue, 2014a).

In terms of understanding stock status, biomass estimation for the first four years of the fishery was based on fitting to the CPUE index only; acoustic surveys to estimate abundance only started in the fifth year of the fishery. The model estimated a small initial vulnerable biomass to fit the rapid decrease in the CPUE index and an acoustic catchability coefficient close to 2.0 to scale the biomass of the later years (i.e. the acoustic survey was assumed to have overestimated the absolute biomass in the survey area).

The stock was assessed as underfished in the first two years of the fishery (1999 and 2000). It was then overfished from 2001 until the commercial fishery closed at the end of 2005, with the biomass having never been less than the estimated B_{MSY}. With the closure of the commercial fishery in 2005, overfishing stopped, catches in 2006 were research catches only (Figure 21). The commercial fishery ceased because the catch quota was insufficient to enable the fishery to be financially viable.

FIGURE 21
Phase (Kobe) plot showing spawning stock status and fishing rate for the period of catching orange roughy in Chile



Notes: SB/SB_{MSY} = spawning stock status; U = exploitation rate.
Source: modified from Payá, (2013).

The stock size that produces MSY (B_{MSY}) was estimated using the proxy $R_0 \times 0.4 \times SPR_0$, where R_0 is the virgin recruitment level and SPR_0 denotes unfished spawning biomass per recruit (i.e. $B_{MSY} = 0.4 \times B_0$). The biomass limit reference point, B_{LIM}, was assumed to be half B_{MSY}. From the base-case stock assessment model run, B_{MSY} was estimated to be 6 400 tonnes, with B_{LIM} 3 200 tonnes, and the associated F_{MSY} 0.07 (F_{LIM} = 0.15). Because of uncertainties about the stock-recruitment relationship, Payá *et al.* (2014) recommended a more precautionary approach to the management of orange roughy in Chile based on $0.5 \times B_0$.

7.3 HIGH SEAS

The lack of adequate data to define stock structure and to enable stock assessments for the high seas stocks remains a major issue, although progress has been made in some areas such as stock structure in the western South Pacific (Clark *et al*., 2016a).

There have been no stock assessments for the southeast Atlantic orange roughy stocks in the SEAFAO convention area, and the available data are unlikely to support an assessment in the foreseeable future (SEAFAO, 2014).

Some preliminary efforts to estimate stock status have been made for the southern Indian Ocean stocks in the SIOFA area (Niklitschek and Patchell, 2009). One fishing company has been using a net-mounted, multifrequency, acoustic optical system (AOS) aimed at verifying whether aggregations seen acoustically are orange roughy or not, and to collect acoustic data that could be used to estimate biomass following the methods of Ryan and Kloser (2016). While there are considerable useful data collected by industry relating to orange roughy and the fisheries in the southern Indian Ocean, to date there have been only limited analyses conducted and limited publication of results. Biological information of use in determining stock structure of orange roughy, including spawning sites, length frequencies, length-weight relationships, and sex ratios, were presented for the southern Indian Ocean to the orange roughy workshop that initiated this review (G. Patchell, unpublished data). It is expected that the establishment of SIOFA in 2012 will lead to more of this information being analysed and published in the future.

In the western South Pacific, all available information has been used to estimate an orange roughy stock structure (Clark *et al*., 2016a). This stock structure has been accepted for further research on stock assessments, at both the stock, and finer spatial structure levels of resolution (Clark *et al*., 2010a; Roux *et al*., 2017). The use of spatially analysed CPUE to provide a biomass index for use in a biomass dynamics model (or other approach) relies on the assumption that CPUE can index abundance orange roughy. Evidence from previous analyses for the ORH 7A orange roughy fishery suggested that CPUE time series (Field and Clark, 1996) declined more rapidly than could be easily explained by catches, and more rapidly than current estimates of stock biomass (Cordue, 2014a). This would lead naive CPUE-based assessments to underestimate stock biomass (Hicks, 2013). Traditional standardized CPUE analyses for the southwestern Pacific high seas fisheries failed to produce useful results (Clark, Dunn and Anderson, 2010). More refined spatially-disaggregated approaches following Walters (2003) are still in development, but have yet to be accepted for use in stock management in the SPRFMO convention area (Roux *et al*., 2017). This approach still suffers from the lack of an understood and reliable relationship between CPUE and abundance of these deepwater stocks, as well from CPUE data that are patchy in time and space.

7.4 NAMIBIA

The assessment of Namibian orange roughy has been described by Boyer *et al*. (2001) and McAllister and Kirchner (2001, 2002). In 1997, the virgin biomass was estimated using commercial CPUE as the biomass index, as these were the only data available (Branch, 1998). The assessment used a swept-area method to convert CPUE to an absolute biomass estimate. With only a single year of data, other methods requiring a time series could not be used. The scaling coefficient, q, to convert CPUE to biomass, included a number of potential biases, such as an assumed trawl catchability for orange roughy aggregations, and the amount of effort directed at aggregations. A Monte Carlo approach was then applied to sample from the q distribution, and develop a probability distribution for the average unfished biomass (B_0). To provide management advice, the assessment then used a population dynamics model to predict the impact of future catch scenarios, sampling from the B_0 probability distribution to allow a range of uncertainty to be expressed. The population dynamics models were based upon those assumed in New Zealand at the time, using parameters estimated for New Zealand stocks (excepting B_0). The limitations of this methodology were clearly recognized. In particular, not only had the spatial distribution of effort changed over time, and so the time series was not consistent, but the aggregations were spatially stable for 1994–98, but then also found to move. It was also concluded that the range of uncertainty, expressed through the q distribution, was too narrow.

In 1997, the first scientific acoustic biomass and research trawl surveys were conducted on the three southernmost fishing grounds. The biomass estimates that resulted were treated as absolute (q = 1), and those from the acoustic surveys were about half those from the commercial swept-area estimates. In 1998, the population dynamics model was run again, but this time using a probability distribution for B_0 estimated from the acoustic surveys. The assessment resulted in a lower MSY, higher risk of overfishing, and the conclusion that the swept-area method produced an assessment that was too optimistic (Table 18).

TABLE 18
Estimates of virgin stock size for orange roughy off Namibia from assessments made in 1997, 1998 and 2000

Year	B_0 estimate (tonnes)	Biomass estimation method
1997	305 000	Commercial swept-area
1998	230 000	Commercial swept-area
	150 000	Acoustic biomass estimates, after significant increases to the original survey estimates (see Huse *et al*., 1997)
2000	74 000	Commercial swept-area
	25 000	Acoustic biomass estimates

In 1999, the stock assessment was revised, and moved to an age-structured, Bayesian population model. The time series of available biomass observations was short, and the model assumed an informed prior on the q for the available acoustic biomass index for each ground (making the treatment of acoustic biomass data similar to that subsequently used in Australia and New Zealand). Fixed parameters in the model included those for growth and maturity. Priors were constructed for all parameters that could be estimated, including B_0, M, and recruitment deviates. Stock assessments were completed for each of the four grounds separately. Although the areas were close enough that they might reasonably be considered the same stock, treating them separately could be considered precautionary, as it may guard against localized depletion. The 1998 survey estimates used in the assessment indicated a severe decline in biomass since the previous surveys (a decline of 60 percent or more), but conflicts in the biomass signals from the acoustic and swept-area datasets resulted in an unclear assessment, and the TAC from the previous year was retained (although not caught).

A fourth assessment was conducted in 2000 (Boyer *et al*., 2001). The input data included revised acoustic and swept-area trawl biomass estimates (Boyer and Hampton, 2001; Kirchner and McAllister, 2002). A key feature of the assessment work in 2000 was that a dynamics model was developed that formally accounted for structural uncertainties. The large drop in biomass observed from acoustic and swept-area estimates could not be accounted for by catch removals. Therefore, four different structural assumptions were developed: (i) a catch removal model, where observed biomass declines were caused by fishing alone; (ii) a fishing disturbance model, where declines occurred because of disturbance of the aggregations by fishing, with fishing resulting in fish failing to aggregate on the grounds, and where, if fishing stopped, fish could re-aggregate; (iii) an intermittent aggregation model, where the decline occurred because of temporary factors unrelated to fishing, e.g. environmental factors, and where fish could re-aggregate but the timing would remain unpredictable; and (iv) a mass emigration or mortality model, where declines were caused by a mass (natural) mortality event or mass emigration, and the large initial biomass was unlikely to re-establish in the near future (McAllister and Kirchner, 2002). Each model was fitted to the data, and results combined across the four models, on the basis that there was no evidence to prefer any particular model. The results suggested far larger uncertainties than were obtained from single models. Over all four grounds, only the mass emigration or mortality model had moderate or high probability, and therefore seemed most likely, although considerable uncertainty remained. However, although the approach was insightful, the results were considered too preliminary to be used in management advice (Boyer *et al*., 2001).

Estimates of virgin biomass (B_0) have varied depending on stock assessment assumptions, with B_0 estimated to be in the range 21–111 kilotonnes for Johnies, 21–128 kilotonnes for Frankies, 19–44 kilotonnes for Rix, and 6– 57 kilotonnes for Hotspot (McAllister and Kirchner, 2002). The management goal was to fish the biomass of the stocks down to half of the unfished biomass (i.e. to 50 percent B_0). Despite restricting effort in the fishery to only five fishing vessels, the fishery was not sustainable and the stocks were reduced to low levels within six years, due in part to the over-optimistic initial assessments of abundance and associated fishing removals. Stock status at each of Hotspot, Johnies and Frankies was believed to be well below 30 percent B_0 by 2006, and the fisheries were closed in 2008.

7.5 NORTHEAST ATLANTIC

There has been no accepted stock assessment or model development for orange roughy, and advice has been based upon commercial CPUE trends (ICES, 2015b). For ICES Subarea VIa, annual French trawl CPUE showed an initial steep decline between 1991 and 1997, followed by low CPUE until 2005. Standardized CPUE for Irish deep-sea trawlers targeting orange roughy is available by month for the period from August 2001 to December 2003, and this index initially increased to a peak in June 2002, then declined steeply and rapidly to November 2002, followed by low CPUE until December 2003. ICES recommended that there should be no directed fisheries for orange roughy, and that bycatch in mixed fisheries should be as low as possible (ICES, 2015a). The European Union (Member Organization) has recently implemented a closure for vessels flagged to its member states for deep-sea fishing below 800 m (EU, 2016), it is therefore unlikely that any stock assessments will be developed for these stocks of orange roughy.

7.6 NEW ZEALAND

There is a relatively long history of orange roughy stock assessments in New Zealand, and this history until 2006–07 is summarized by Sissenwine and Mace (2007). The TACs for New Zealand orange roughy in the early 1980s were based on absolute biomass estimates from trawl surveys, requiring assumptions about the catchability of the fish to be made (Robertson, 1985, 1986). Stock assessments where population models were statistically fitted to observed data were not performed until the late 1980s, when trawl survey estimates were then used as relative rather than as absolute biomass estimates (Robertson, 1989). At around the same time, initial assumptions about productivity were revised, and an informed guess at natural mortality of 0.1 was replaced by a more realistic value of 0.05, later revised down to 0.045.

The focus of most stock assessment research was observational data on stock biomass. Research trawl surveys have been used to inform historical assessments for Puysegur (Annala *et al*., 2001), the Challenger Plateau (Field and Francis, 2001), and the Bay of Plenty (Clark, Anderson and Francis, 2001), and continue to be used for the east and south Chatham Rise (Cordue, 2014a), and Mid-East Coast (Cordue, 2014b). Nevertheless, there remain concerns about the veracity of some trawl surveys, because vessels and surveys were not always strictly comparable, the aggregating behaviour of fish often resulted in biomass estimates with high coefficients of variation, and the sex ratio of catches could also be highly variable indicating structure within the surveyed aggregations (Pankhurst, 1988; Francis *et al*., 1995).

While CPUE was used in historical assessments, it has been rejected from use in assessments conducted since 2014 (MPI, 2016a). Nevertheless, CPUE still forms the basis of the most recent assessments for ORH 7B (MPI, 2016a), East Cape (Anderson, 2003), and for stock evaluations of areas of ORH 1, and Sub-Antarctic areas of ORB 3B (MPI, 2016a). However, the use of CPUE from feature-based fisheries proved problematic, as CPUE typically started very high and declined more rapidly than could be accounted for by catch removals (Clark and Dunn, 2012). Moreover, CPUE data from fisheries on aggregations of fish are expected to be prone to hyper-stability (i.e. CPUE rates are maintained despite reduction in the biomass). Therefore, it appears that such observational data are not reliable indices of biomass as they are unlikely to be indexing the full stock biomass (MPI, 2013).

Acoustic biomass surveys for orange roughy were first trialled in the mid-1980s (Do and Coombs, 1989). Acoustic surveys using vessel-mounted equipment have been conducted regularly since the early 2000s, when results were initially used as absolute biomass estimates, later being used as relative biomass estimates with an informed prior on survey catchability from the early 2000s. The greatest acoustic problem was the combined issue of species identification and species mix, and thus how to obtain an accurate estimate of the target strength of orange roughy (the proportion of energy that an individual fish reflects). Problems with species identification/mix, where a small amount of fish having gas-filled swim-bladders can bias orange roughy biomass estimates when they are misidentified or in mixed species schools, were initially minimized by focusing surveys on spawning aggregations where the proportion of orange roughy (as evidenced by the estimated catch composition from trawls) was high and often very close to 100 percent. Further improvements in addressing species identification/mix issues have been delivered by surveying aggregations using deep-towed, multifrequency AOS (Kloser *et al*., 2000; Kloser *et al*., 2002). Development and use of such systems has enabled acoustic surveys to return visually verified target strengths, with species mixtures derived from post-survey analyses (Ryan, Kloser and Macaulay, 2009; Ryan and Kloser, 2016). This approach has yielded similar, positive results in both New Zealand and Australia (Kloser and Ryan, 2011; Macaulay, Kloser and Ryan, 2013; Kloser *et al*., 2013; Ryan and Kloser, 2016). The key outcome from these various developments has been an increased robustness in the estimation of spawning stock biomass from acoustic surveys.

Further observations and verification of species mix have also used moored underwater video recording (O'Driscoll *et al*., 2012). Target strength was not adequately resolved until 2012, with the development of a combined acoustic and optical system that could simultaneously acoustically "ping" and visually confirm the species identity and length of individual fish targets (Macaulay, Kloser and Ryan, 2013; Kloser *et al*., 2013; Ryan and Kloser, 2016).

Age composition samples were used in assessments of orange roughy until about 2006, when it became apparent that Australian and New Zealand age estimates were inconsistent (Francis, 2006). When age estimates had been used to inform recruitment variability, results were rejected because the models were likely to be statistically overfitted, given numerous cohorts but low precision in ageing, which would spread measurements of each cohort over many adjacent cohorts (MPI, 2006; Sissenwine and Mace, 2007). As a result, deterministic recruitment was assumed, but this resulted in stock biomass increases that were not supported by observational data (Sissenwine and Mace, 2007). The inability to estimate recruitment led to the abandonment of model-based stock assessments for the east and south Chatham Rise (Dunn, Anderson and Doonan, 2008). Therefore, the ageing of orange roughy was revised and a new protocol was developed and tested (Tracey *et al.*, 2009; Horn *et al.*, 2016). Age composition data were then used for an assessment of the Mid-East Coast stock in 2011 (Anderson and Dunn, 2011) and 2013 (Cordue, 2014b), and the Mid-East Coast, northwest Chatham Rise, east and south Chatham Rise, and Challenger Plateau in 2014 (Cordue, 2014a). The use of age data was crucial to the success of the 2013 and 2014 assessments, as it allowed the deterministic recruitment assumption to be rejected.

7.6.1 Current orange roughy stock assessments

The methods used in 2014 were different from those used in previous orange roughy assessments in a number of respects (Cordue, 2014a, 2014b; MPI, 2016a). The major differences were application of a high data-quality threshold, some aspects of model structure, the use of improved-quality age data, and the assumption of informative Bayesian priors. Bayesian estimation is used in almost all New Zealand statistical stock assessment models (MPI, 2016a), using the software CASAL (Bull *et al.*, 2012).

From 2013, the higher threshold imposed on data quality before it was used in an assessment resulted in the exclusion of a number of biomass indices and estimates that had previously been used. In particular, CPUE indices were not used in any of the assessments because they are very unlikely to be monitoring stock-wide abundance. Estimates of biomass from egg surveys were not used as it was found that all of the available estimates were from very problematic surveys (the assumptions of the survey design were not met and/or there were major difficulties in analysing the survey data). Finally, acoustic-survey estimates of biomass were only used when largely single-species aggregations were surveyed with appropriate equipment. Estimates of spawning orange roughy biomass were accepted for plumes on the flat (using hull-mounted transducer or towed systems) or plumes on underwater features (towed systems only, as otherwise the acoustic dead zone can be very large, meaning a substantial proportion of the fish habitat cannot be seen).

Despite the high quality threshold, sufficient data were nevertheless available to most assessments, including for the main stock areas of the Mid-East Coast, northwest Chatham Rise, east and south Chatham Rise and Challenger Plateau. Data available included a mix of research trawl survey biomass indices, acoustic spawning biomass estimates, length composition data from trawl surveys and from the commercial fisheries, age composition data from the trawl surveys and commercial fisheries, proportion spawning-at-age from the trawl surveys, and age composition data from the acoustic surveys. All models used the integrated stock assessment software CASAL (Bull *et al.*, 2012). Model structure was very similar across the four assessed stocks. In each case, the base model was a single-sex, single-area model with fish partitioned by age and maturity (mature or immature). Growth was assumed to be known and constant. Maturity was constant and estimated within the model from age compositions of spawning fish and, where available, female proportion spawning-at-age was also used. This is a major contrast to earlier assessments, where mature biomass was based on transition zone age-at-maturity, requiring acoustic and egg survey estimates of spawning biomass to be scaled up to transition-zone mature biomass before being used in an assessment. In the 2014 assessments, acoustic estimates of spawning biomass were used directly. Other estimated parameters included B_0, survey catchability (q), survey and fishery selectivity, recruitment deviates, and stock-recruitment steepness (h). In base model runs, M was fixed, but also estimated as a sensitivity run.

The major sources of recent abundance information in the assessments were acoustic surveys of spawning biomass (e.g. Ryan and Kloser, 2016). For each survey, the spawning biomass estimate was included as an estimate of relative spawning biomass with an informed prior on q. The two major sources of potential bias recognized in the prior were: (i) acoustic target strength; and (ii) availability of the stock to the survey (not all spawning fish within the survey area).

The target strength prior was derived from the estimates of Macaulay, Kloser and Ryan (2013) and Kloser *et al*. (2013) who both obtained TS estimates (from a 38 kHz echosounder) from visually verified orange roughy at New Zealand and Australian spawning sites, respectively. The two studies agreed on the mean TS; therefore, the prior was developed as –52.0 dB with a 99 percent CI that covered the uncertainty of estimates from both studies. The improvement in TS estimation was largely due to the development of equipment that could visually identify individual fish and estimate their target strength (Ryan, Kloser and Macaulay, 2009). This AOS, which included multiple frequency acoustic transducers, video and still-image capture, was specifically developed for addressing abundance estimation in these types of fish through a long-term cooperation between the Australian governmental institute Commonwealth Scientific and Industrial Research Organisation (CSIRO) and the New Zealand fishing industry. As noted elsewhere, development of biomass estimates using deep-towed, multifrequency systems has greatly improved the precision and confidence in those estimates (Kloser *et al*., 2015; Ryan and Kloser, 2016).

For one of the stocks, the ORH 3B east and south Chatham Rise stock, the mean of the acoustic priors was assumed to change from year to year according to an assumed proportion of the stock in each of two main spawning plumes (Cordue, 2014a). The priors reflected the assumption that a relatively new plume (Rekohu, first recorded in 2010) formed in 2002, and that this plume increased over time to eventually match the proportions of fish in each area as observed in 2011 and 2013. An assumption was also made that the surveyed spawning plumes contained "most" of the spawning stock biomass (applied in a prior having a mean of 0.8, i.e. it is assumed *a priori* that in those years, 80 percent of the spawning stock biomass was being surveyed). When the availability and target strength priors were combined (assuming they were independent), the result is a prior for the acoustic q. Sensitivities to the assumed priors were conducted during the assessment.

The estimation of YCS was also modified for the 2013 and 2014 assessments (Cordue, 2014a). The assumed priors for YCS were revised to be less restrictive on YCS estimates (using a Haist parameterization with "nearly uniform" prior [Bull *et al*., 2012; Cordue, 2014a]).

Numerous model sensitivity runs were conducted, of which the most useful were considered to be the runs that simultaneously increased/decreased M and decreased/increased the mean of the acoustic q priors by 20 percent (a lower stock status, reduced percent B_0, occurs when M is decreased and when the mean of the acoustic q priors is increased, similarly an increased stock status occurs for changes in the other direction).

The sizes of the four stocks, as estimated in 2014, varied considerably for both virgin and current biomass (Table 19). The east and south Chatham Rise stock was by far the largest, with a virgin biomass (B_0) estimated at more than 300 000 tonnes while the other stocks each had an estimate of less than 100 000 tonnes. In terms of current (2014) biomass, all of the stocks except for Mid-East Coast had median current biomass estimates close to or within 30–50 percent B_0, the management target range implemented in 2014 following management strategy evaluation (MSE) model simulations (Cordue,

TABLE 19

2014 base model, median MCMC estimates of virgin biomass (B_0), "current" biomass (B_{2014}) and "current" stock status (B_{2014}/B_0)

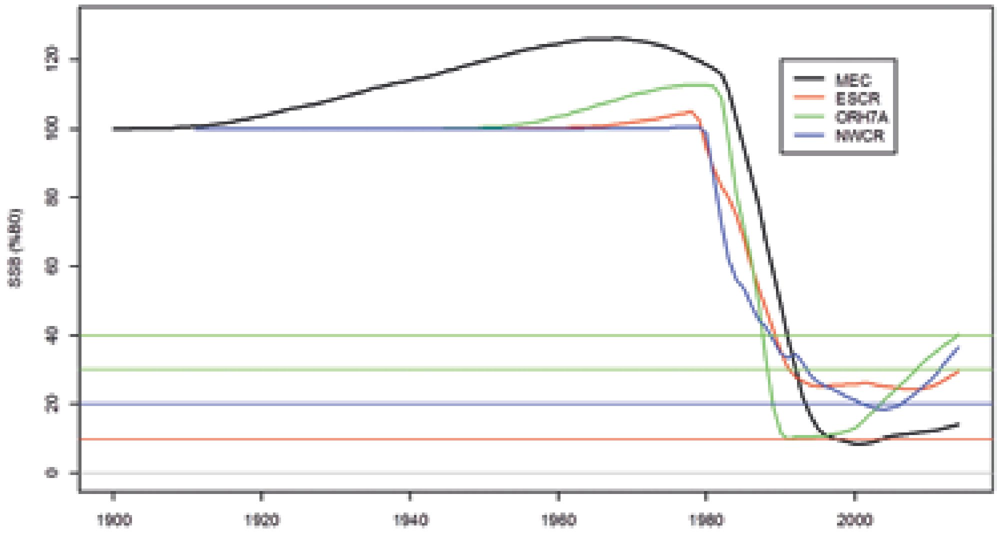
Stock	B_0	B_{2014}	B_{2014}
	(tonnes)		(% B_0)
Northwest Chatham Rise	66 000	24 000	37
East and South Chatham Rise	320 000	93 000	30
Mid-East Coast	95 000	14 000	14
ORH7A (including Westpac Bank)	87 000	35 000	40

Source: Cordue (2014c).

2014c; Table 19, Figure 22). The Mid-East Coast stock had a median estimate below the regulatory "soft limit" of 20 percent B_0, which under the New Zealand Harvest Strategy Standard (MFish, 2008b) should lead to the development and implementation of a time-constrained rebuilding plan.

FIGURE 22

MCMC base models: median estimates of stock status trajectory for the four stocks assessed in 2014, Mid-East Coast stock (MEC), east and south Chatham Rise stock (ESCR), northwest Chatham Rise stock (NWCR), and Challenger Plateau stock (ORH7A)



Notes: The horizontal lines show the management hard limit (10 percent B_0) in red, the soft limit and effective limit reference point (20 percent B_0) in blue, and in green, the lower (30 percent B_0) and upper (40 percent B_0) bounds of the management target range at the time of the assessments in early 2014. Following the results of a management strategy evaluation, the target reference point range was reset to 30–50 percent B_0 from August 2014.
Source: Cordue (2014a).

For each assessment, long-term deterministic projections and yield curves were estimated, from which to estimate deterministic reference points and yields (Table 20). Deterministic B_{MSY} was estimated to be similar for all four stocks, being in the range 21.5–24.5 percent B_0 (Table 20). In each case, very little yield would be lost when moving from deterministic B_{MSY} up to the 30 percent B_0 (the lower end of the biomass target range). The estimated long-term yields when fishing at U_{35} (the fishing intensity that forces the stock to deterministic equilibrium at 35 percent B_0) ranged from 1 300 to 2 100 tonnes for the smaller stocks, and was about 7 200 tonnes for the east and south Chatham Rise stock (Table 20). However, these yield estimates were considered unrealistic, because of the use of deterministic recruitment, and the exact application of a given level of fishing intensity.

TABLE 20

Base model, median MCMC estimates of deterministic B_{MSY}, MSY, deterministic long-term yield at U_{35}, and the exploitation rate corresponding to U_{35}

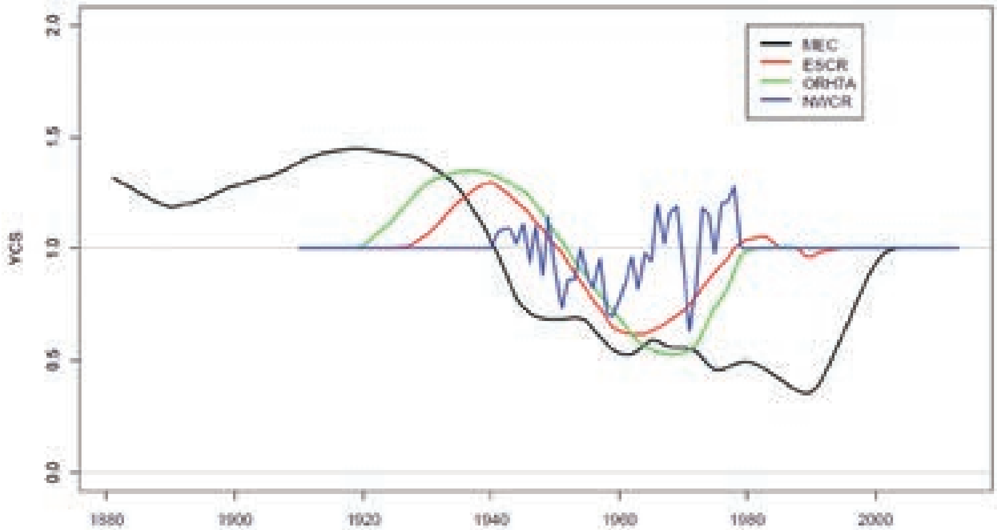
Stock	B_{MSY}	MSY	U_{35} yield	U_{35} fishing rate	U_{35} long-term yield
		(% B_0)		(%)	(tonnes)
Northwest Chatham Rise	23.7	2.1	2.0	5.3	1 320
East and south Chatham Rise	21.8	2.4	2.3	5.3	7 180
Mid-East Coast	22.5	2.3	2.2	5.1	2 080
ORH 7A (including Westpac Bank)	24.5	2.1	2.0	5.5	1 740

Source: Cordue (2014c).

Recruitment (YCS) was estimated for each of the four stocks (Figure 23), and three of the four estimated relatively high YCS in the 1920s–1940s, followed by a recruitment decline to about 50 percent or less of the long-term mean YCS by the 1960s, suggesting recruitment to the fisheries after the biomass fish-down was about half of what was expected from the initial size of the stocks. A similar pattern was estimated in Australian stocks (see Figure 17). The high recruitment estimated for the 1920s–1940s caused the estimated biomass to increase towards the start of fishery. When combined with the age at recruitment estimates (Table 15), the models indicate that recruitment to the spawning stocks moved to below average levels in the mid-1970s (Mid-East Coast), early 1980s (Challenger Plateau), and around 1990 (east and south Chatham Rise).

FIGURE 23

Estimated recruitment in four New Zealand orange roughy stocks, Mid-East Coast (MEC), east and south Chatham Rise (ESCR), Challenger Plateau (ORH 7A) and northwest Chatham Rise (NWCR)



Notes: MCMC base-case models: smoothed (where possible) median estimates of year class strength (YCS) for the four stocks assessed in 2014. A LOESS smoother (f=0.15) was applied to the MCMC median estimates for each cohort.
Source: MPI (2016a).

7.6.2 Development of a harvest control rule

An MSE was performed with a generic orange roughy stock to determine an appropriate limit reference point, target biomass range, and harvest control rule (HCR) for use in managing orange roughy stocks (Cordue, 2014c). The proposed management strategy was designed to be consistent with both New Zealand's Harvest Strategy Standard (MFish, 2008b) and the Certification Requirements of the Marine Stewardship Council (MSC) Sustainability Standard v1.3 (MSC, 2013). This MSE did not address social or economic interests in an explicit way but focused solely on the sustainability of the target stock in order to meet specific MSC requirements. In doing so, this also achieved some identified socio-economic goals through potential market access and price benefits that could be delivered by MSC certification as well and providing for relative long-term stability in catches.

The first step was to estimate stock recruitment steepness (the percentage of virgin recruitment, on average, produced when the stock is at 20 percent of virgin biomass, B_0) by performing extra stock assessment runs for the Mid-East Coast stock. Of the four stocks assessed in 2014, this was the only stock that had YCS estimated from age data on cohorts spawned at low stock size (and hence information on how average recruitment changes at low stock size). Assessment runs were done for

Beverton-Holt and Ricker stock-recruitment relationships. The results were similar for both, with median steepness (for the combined posterior) equal to 0.6 with a 95 percent CI of 0.31–0.95.

The large level of uncertainty in steepness, as well as the form of the stock-recruitment relationship, created a high degree of uncertainty in the estimates of B_{MSY}. For Beverton-Holt, the median estimate of B_{MSY} (and 95 percent CI) were 26 percent B_0 (12–39 percent), and for Ricker they were 42 percent B_0 (37–47 percent). As there was no basis for choosing between the Beverton-Holt and Ricker stock-recruitment relationships, it was assumed that the mid-point of the target range needed to be at about 40 percent B_0, a precautionary assumption.

The limit reference point was defined to be the greater of 20 percent B_0 or 50 percent B_{MSY}. Under this definition, the Bayesian estimate of the limit reference point was 20 percent B_0 with a very high level of certainty.

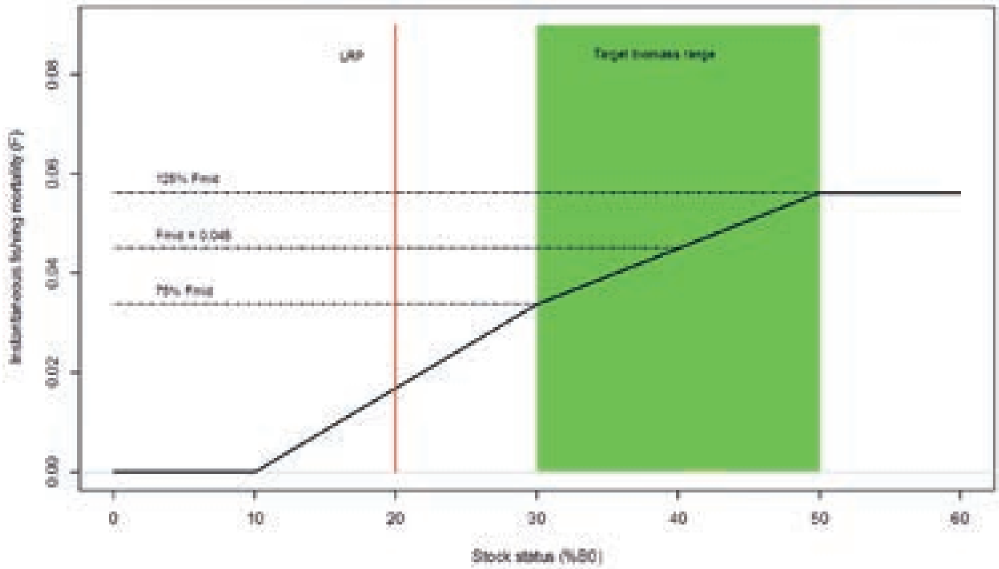
Experimentation with various HCRs showed that spawning biomass, even when managed with a (perfectly) constant fishing mortality, *F*, was prone to large, long-term fluctuations, because of the low natural mortality and pattern of recruitment. Therefore, a fairly wide target range was needed to accommodate these long-term fluctuations and a breadth of 20 percent of B_0 was proposed. Taken with the mid-point of 40 percent B_0, this gave a target biomass range of 30–50 percent B_0. With narrower biomass target ranges, the natural variability in the stock made it difficult to keep the stock within the range for a sufficiently high proportion of the time.

Based on the range and effective limit reference points (20 percent B_0), HCRs were tested in long-term simulations, sufficiently long to ensure stochastic equilibrium had been reached, and to check the HCRs performed adequately with regard to maintaining the biomass within the target range with little possibility of ever being below the limit reference point. An HCR was identified that was robust to the uncertainty in steepness and natural mortality, as well as one-off and multiple violations in major assumptions (Cordue, 2014c).

In a static HCR, there is a simple functional relationship between estimated stock status and *F*, where a low stock status results in management action to reduce *F* and vice versa. For example, a TAC could be set as a fixed proportion (such as the natural mortality rate, 4.5 percent) of the current spawning biomass estimate (Doonan, Fu and Dunn, 2015). In a dynamic HCR, there is an initial functional relationship and an additional rule by which that relationship can change over time that makes the HCR more responsive to changes in stock status (or whatever indicator is being used). The selected HCR was dynamic, and based on a "slope" HCR where *F* increased from 0 at 10 percent B_0 to 0.045 at 30 percent B_0, remaining constant thereafter (0.045 was the assumed value of *M* in the stock assessments, so the "slope" HCR was an "$F = M$" strategy). However, the dynamic HCR implemented in New Zealand was considerably more complex. In addition to the slope when the stock was below the target range, first, *F* further increased within the target biomass range of 30–50 percent B_0 as stock status increases (Figure 24), and second, there was an added dynamic component. The dynamic element operates by scaling the slope of the HCR when the stock is outside the target reference range, with a scaling down of *F* occurring every time that a stock assessment estimates stock status to be below the lower bound, and there is a specified limit to the scaling down that may not be exceeded (see Cordue, 2014c). There is also an equivalent scaling up if the estimated stock status is above 60 percent B_0. This is a complex HCR and is fully explained in Cordue (2014c). It is also an HCR that needs to be informed by appropriately frequent stock assessments (assumed every three years in the simulations).

FIGURE 24

The harvest control rule for orange roughy fisheries implemented in New Zealand in 2014



Notes: LRP = 20 percent B_0 (vertical red line), target biomass range = 30–50 percent B_0 (green band), initial F_{mid} = 0.045, slope within the target range: p = 25 percent, ramps down to zero at 10 percent B_0, rescaling limit points: lower = 30 percent B_0, upper = 60 percent B_0.

Source: Cordue (2014c).

The dynamic aspect of the HCR was found to be crucial in enabling long-term performance that was robust to uncertainty in parameter estimates (e.g. *h* and *M*, recruitment variability and correlation) and errors in assumptions (e.g. the form of the stock-recruitment relationship, bias in the estimators of stock status, and/or current biomass). The dynamic rescaling of the functional relationship essentially allows the HCR to be more responsive to changes in stock status, and the HCR becomes progressively more conservative as stock status falls (and vice versa).

The selected HCR was implemented by managers in the three orange stocks that were estimated to be within the target biomass range, and catch limits for the 2014–15 fishing year were set consistent with, or lower than, those indicated by the HCR (MPI, 2016a). The remaining stock, the Mid-East Coast, was treated separately as it was felt that stronger action was needed to ensure that it was rebuilt (noting that the HCR described above was not designed to rebuild stocks).

8. Ecosystem considerations

8.1 ASSOCIATED FISH SPECIES

8.1.1 Nature of the impact

The fisheries for orange roughy use demersal trawls capable of catching many fish species other than just the orange roughy target. This is true for all demersal trawls, whether used in shallow, productive waters to catch cod and haddock or in deepwater locations to catch species such as orange roughy, where a mix of bycatch species are taken, some of which are retained as commercial fish and others discarded. Generally, orange roughy fisheries have a relatively low fish bycatch when compared with other groundfish fisheries, especially when fishing on spawning aggregations. Much of the fish bycatch in orange roughy fisheries tends to be other valuable commercial species such as smooth oreo (*Pseudocyttus maculatus*) that are also retained, as is seen within both New Zealand waters and the SPRFMO jurisdiction convention area (Anderson, 2009, 2011b, 2013; MPI, 2015b). A detailed description of typical orange roughy trawl gear can be found in MFish (2008a).

8.1.2 Scale of the impact

All fisheries have impacts on bycatch and associated species, invariably with observed reduction in population size to a greater or lesser extent. This is an inevitable consequence of fishing (as it is for the target species), although often there are associated productivity increases. However, this does not in itself indicate a problem with, or lack of, sustainability. Acceptable levels of stock depletion have been defined for many fish species, including low productivity and low trophic-level species. It is also worth noting that the target stock is intended to be reduced in abundance to whatever the management goal is, either fluctuating around a target reference point (e.g. 48 percent B_0 for Australian orange roughy fisheries) or within a target management range (30–50 percent B_0 for New Zealand orange roughy fisheries).

The scale of the impact of an orange roughy fishery on the associated fish species will be driven by four factors: (i) the nature of the deepwater environment, which is cold and relatively unproductive; (ii) the size (catch quantity) of the fishery; (iii) the spatial expanse covered by the fishery (i.e. its extensiveness), which is generally small and often confined to specific areas; and (iv) spatial scale of tow impacts determined by the physical attributes of the gear (e.g. dimensions, design and weight) and aspects of the tow duration, especially the average tow duration.

The scale of impact that a fishery has on associated fish stocks can be viewed through the lens of the size and spatial extent of the fishery. A large-volume, extensive fishery is likely to have greater impacts in terms of numbers of stocks affected and in spatial extent compared with a relatively small-volume fishery operating in a restricted spatial area such as an orange roughy fishery. Thus, a number of metrics can be considered, including for example, the number of stocks impacted, the spatial extent of those impacts, as well as the level of depletion experienced by each stock.

The expectation for the amount of fish bycatch in orange roughy target fisheries is that there will be limited bycatch in both number of species and in volume, as a direct result of the relatively small areas fished and the short tow duration. What is seen in the orange roughy fisheries is exactly that, with a total of about 5 percent bycatch by weight across about 200 species, at a rate of between 25–62 g of non-commercial bycatch per kilogram of orange roughy in the New Zealand fisheries between 1990–91 and 2008–09 (Anderson, 2011b). Also, most of the bycatch are other retained, managed, commercial species such as smooth oreo (*Pseudocyttus maculatus*). Similar patterns are seen for other orange roughy fisheries and over other time periods (e.g. Ballara, 2015; Hansen and Hobsbawn, 2015; MPI, 2015b). The impact this bycatch has on bycatch species will depend on the distribution, abundance and productivity of the associated fish species. However, in general, there are few other associated species that aggregate to the same extent as orange roughy on the main fishing grounds (Tracey *et al*., 2004, 2012; Clark *et al*., 2010a) and hence fishing impacts would be expected to be less. For some high seas fisheries, bycatch levels have been estimated to be similarly low, or lower (MPI, 2015b).

Even the largest of the orange roughy fisheries are now small in volume, typically less than 5 000 tonnes per year from separate biological stocks, having previously been as high as 40 000 tonnes during the early fishing-down phase of the fisheries (Table 4) but never approaching the size of the large demersal whitefish or small pelagic fisheries around the world. Orange roughy fisheries also tend to be restricted to relatively small spatial footprints, often associated with specific underwater features (MPI, 2015c; Trumble, Punt and Stern-Pirlot, 2016). Given the small scale of the fisheries and the relatively low proportion of bycatch (Anderson, 2009, 2011b, 2013, Hansen and Hobsbawn, 2015; MPI, 2015b), the scale of the impacts of sustainable orange roughy target fisheries on associated fish species is likely to be relatively low, even for low-productivity species such as deepwater sharks, as assessed by Trumble, Punt and Stern-Pirlot (2016).

Bycatch species at orange roughy fishery depths are thought to typically have a higher proportion of low-productivity species, and consequently their resilience to fishing pressure tends to be limited, which increases risks of stock depletion. However, this does not mean that all or even any associated species in a fishery area are actually overfished, just that they are more susceptible to overfishing, especially if adequate management is not in place. However, if stock depletion does occur, it can take a long time to reverse.

Niklitschek *et al*. (2010) described significant reductions in abundance and changes in species composition of bycatch associated with Chilean orange roughy fisheries, but that the effects were highly localized to what was a very small fishery footprint. Clark *et al*. (2000) also reported local declines in New Zealand in some bycatch species for the Spawning Box on Chatham Rise. A subsequent examination of bycatch trends in the same area by Livingston, Clark and Baird (2003) found trends varied, with some species showing increases, and others decreases. However, Doonan and Dunn (2011) found little change in the majority of associated species, and where there was significant change (increase and decrease) there was no clear or consistent causal link for those changes.

The issue of potential impacts on the stocks of associated fish species has been specifically explored as part of the third-party evaluation of the three largest New Zealand orange roughy fisheries against the sustainability standard of the MSC and been found to meet the this standard (MSC, 2013) with regard to fish bycatch (Trumble, Punt and Stern-Pirlot, 2016). Specific concerns were raised about those associated fish species with higher-than-average vulnerability to fishing, which included a number of deepwater shark species. As noted above, it is important to distinguish between vulnerability to fishing and actual overfishing, as the identification that a species has higher-than-average vulnerability does not imply that it is, or will become, overfished, just that it is at greater risk and that management needs to address that level of risk. Risks to sharks from fishing have been specifically assessed in some areas, including for Australian fisheries (Wayte *et al*., 2007). Broadly similar approaches have been used for New Zealand sharks (Ford *et al*., 2015) and while this was not-fishery specific (i.e. neither separately for orange roughy fisheries or for the deepwater fisheries collectively), it does addresses risk for all fishing activity.

Owing to the paucity of relevant data from orange roughy fisheries in some areas of the world, in part due to the very small scale of some fisheries and because other fisheries only operated for a few years, there has been limited quantitative consideration of these issues. Good examples do exist, however, and this has been attempted for Baxter's dogfish (*Etmopterus granulosus*) in New Zealand, where this species is the deepwater shark that is most commonly caught in the orange roughy fisheries and was identified as at highest risk from fishing (Ford *et al*., 2015). While Ford *et al*. (2015) only considered recent impacts and not those from the historically larger orange roughy fisheries, as part of the MSC review of the three largest fisheries, biomass estimates from long time-series of swept-area trawl surveys were reviewed together with length-frequency information. The available survey-based biomass estimates showed no trend, and continuing presence of both small and large dogfish was indicative of on-going recruitment (persistent presence of small individuals) and relatively low fishing mortality (persistent presence of large individuals) (Trumble, Punt and Stern-Pirlot, 2016).

For the high seas fisheries, information on fish bycatch is somewhat limited (e.g. catch and length frequencies) but has been considered adequate for determining the need for management actions (Hansen and Hobsbawn, 2015; MPI 2015b). For the western South Pacific (SPRFMO convention area), catch and biological data are generally of high quality, partly because of the 100 percent observer coverage required for all permitted demersal trawlers. The proportion of bycatch is very small (MPI, 2015b; Hansen and Hobsbawn, 2015) and, given the scale of bycatch observed, while impacts remain uncertain, it is likely that the impacts of the orange roughy fishery on most non-target fish stocks are minimal. For example, New Zealand vessels represent the vast majority of the trawl fishing effort targeting orange roughy and report orange roughy as a percentage of total catch for 2002–2015 of 83.6 percent, with about 7 percent being non-commercial species that are discarded, and including about 0.6 percent of the total catch reported as sharks (MPI, 2015b). Australian vessels in this area report broadly similar information (Hansen and Hobsbawn, 2015). Recent developments within SPRFMO have seen ecosystem approach studies for sharks (Duffy, Geange and Bock, 2017) and preliminary ecological risk assessments for sharks (Georgeson *et al*., 2017).

For the southern Indian Ocean, industry has been proactive in studying the fishery and its impacts and there are a number of relevant documents on the SIODFA website.[8] However, the standard reporting of catch data is less well developed than that for the SPRFMO, as catch volumes are not typically published (Hansen and Hobsbawn, 2016), being regarded as confidential. There has been some focused research on non-target species vulnerable to demersal trawling, including sharks.[9]

In order for managers to receive the correct scientific advice in relation to associated fish species, it is important that appropriate information be collected and analysed and that data needs be periodically reviewed and monitoring strengthened where appropriate. Where such data are collected, analysed and reported, and while the target fisheries remain small and the quantities of bycatch remain small, as is the case for the SPRFMO fisheries, there appears little risk of overfishing for most species of fish bycatch. Little has so far been done to specifically assess risks to sharks from these demersal high seas fisheries. Substantial increases in the target catches or spatial extent of the fisheries, large biomass reductions, evidence of persistent poor recruitment, or reductions in the number of age classes or size classes for those species that are difficult to age, should trigger further investigation and appropriate management action.

[8] See: www.siodfa.org

[9] See: www.siodfa.org/programmes/nsf-sharks-tree-of-life-programme/

8.1.3 Impacts on benthic habitat and non-fish associated organisms

Nature of the impact

The impacts of demersal trawling during orange roughy fishing are largely restricted to the organisms on and attached to the seabed and to some level of physical damage to the substrate. Especially vulnerable are those sessile organisms growing up from the seabed.

The impacts of demersal trawl gear on the seabed ecosystem have been increasingly studied, and the impacts and the potential for sustainability and recovery are slowly becoming better understood (Thrush and Dayton, 2002; PFMC, 2005; Hiddink, Jennings and Kaiser, 2006; Hiddink *et al*., 2017; Kaiser *et al*., 2006; Pitcher *et al*., 2015, 2016). Understanding the specific benthic impacts from orange roughy fishing are more problematic, largely due to the difficulty of studying at such great depth and also due to the longevity of some of the organisms present, especially corals. However, the general nature of the benthic impacts will be essentially the same as in shallower environments, principally due to the physical contact of the trawl gear with the seabed and the associated epibenthic fauna (Clark *et al*. 2016b) although timescales for recovery are likely to be different. Reported tow duration when fishing for orange roughy around New Zealand is that 60 percent of tows have a duration less than 30 minutes, and almost one-third a duration of less than 15 minutes (MFish, 2008a). This reflects the fishing of aggregations and fishing on UTFs, where it is either unnecessary to fish for longer as large catches can be made from aggregations, or impossible to do so due to the roughness of the ground, or because the end of the tow would be too deep (on UTFs orange roughy tend to occur mostly around the top and flanks, and the trawls almost always proceed from the top down the flanks [never up]). Indirect impacts due to, for example, sediment plumes may also be important in some circumstances, especially when fishing on flat ground, but will be of less importance when fishing orange roughy aggregations on UTFs as there will tend to be less sediment on the slopes of UTFs and the bottom contact time for the trawl gear is, as noted above, relatively short. The amount of gear contact when fishing on UTFs is also expected to be less then when towing on flatter ground as the doors tend to be off the seabed, as are most of the sweeps and bridles (when used). With less gear in contact with the seabed when fishing on UTFs, both direct and indirect impacts will be proportionately reduced.

Where demersal gear does make contact with the seabed where there are larger, epibenthic fauna, there will be impacts on the individuals and colonies that lie in and close to the tow paths. This includes both physical damage from direct contact with the trawl gear that will result in mortality in many cases as well as smothering with re-suspended sediment in some locations. There have been a number of studies on the types of impacts in deepwater (Koslow *et al*., 2001; Clark and Rowden, 2009; Althaus *et al*., 2009) as well as studies seeking to address the likelihood of and timescales for recovery (Williams *et al*., 2010; Clark *et al*., 2010b).

Scale of the impact

The scale of the benthic impact is directly related to the scale and extent of the fisheries. This is important with respect to orange roughy fisheries, as many areas that were previously fished are currently closed (Chile, Namibia, the northeast Atlantic, and more than half of the historical fishing footprint in the western SPRFMO convention area) and the amount of directed effort in most other fisheries is now much reduced, as can be seen in the information on the fisheries detailed elsewhere in this report (see also MFish, 2008a; Trumble, Punt and Stern-Pirlot, 2016). Moreover, as discussed above, the average orange roughy tow duration is generally short, and short-tow duration limits the area impacted. Tow duration tends to be especially short for orange roughy fisheries on UTFs, which typically have limited towable areas with fish, or when fishing on spawning aggregations where catch rates of the target species can be very high and capacity catches can be made in a matter of minutes (MFish, 2008a). However, although short-tow durations on UTFs reduce the comparative footprint and impact, local effects on benthic fauna have the potential to be substantive. Some UTFs, such as hills, knolls and seamounts, can have much higher densities of sessile benthic fauna and biogenic taxa such as corals and sponges when compared with slope habitat (e.g. Rowden *et al*., 2010b; Tracey *et al*., 2011). The distribution of such high-density benthic fauna and biogenic taxa is not, however, uniform (Clark *et al*., 2015), and the scale of impacts depends on complex interactions between the patchy distributions of both habitat and fishery footprint, as well as how much fisheries explore new areas beyond the historical distribution of fishing.

The scale of the issue needs to be considered with respect to the area of unfished seabed that exists as a result of: (i) specific management actions (e.g. marine protected areas [MPAs], marine reserves and other closed areas); (ii) natural limitations of trawl fishing (areas too deep, too rough, or too steep to trawl and areas where orange roughy do not occur); and (iii) the interaction between the patchy distribution of both fishing effort and benthic fauna of interest. Each of these factors will provide areas that are unfished or where fishing does little or no damage to benthic species of conservation interest. For example, Clark, Consalvey and Rowden (2010) report dense stony corals occurring down a spur on a heavily-fished UTF, but commercial records and information provided by vessel skippers indicated that the spur in question had never actually been fished. Moreover, Clark *et al*. (2015) reported observing various corals and other vulnerable marine indicator taxa on heavily fished seamounts in the SPRFMO convention area.

Where estimates have been made, the size of the footprint of orange roughy fisheries within areas of fished habitat is extremely small. For example, for the Chilean fishery, Niklitschek *et al*. (2010) estimated the footprint at 2.7 percent of the habitat. In the SPRFMO, more than 99 percent of the total area is outside of any declared bottom fishing footprint and as such is closed to fishing by all parties (Penney, Tingley and Loveridge, 2016) and, for Australia and New Zealand, the combined trawl footprint for 2002–06 (which defines the maximum extent of permitted fishing) covered less than 10 percent of the available, fishable area (Penney, 2013). The same can be seen for the Australian footprint in the southern Indian Ocean (Hansen and Hobsbawn, 2016). However, care should be exercised when comparing different locations as it is likely that habitats have been described differently in the different locations. In addition, with respect to considering benthic impacts, "habitats" have usually been described in terms of very general properties such as "*area within fishable depth*", broad UNESCO-defined ecosystem bioregions, as well as subdivisions between slope and UTF fisheries (MFish, 2008a; Williams *et al*., 2011; Penney, 2013; Trumble, Punt and Stern-Pirlot, 2016).

In some areas, the scale of the potential impact as measured by overlap between the fishery and defined habitat can be greater on specific habitat types. In New Zealand waters, up to 80 percent of knolls and hills with summit depths between 600 m and 1 200 m are reported to have been fished (Clark and O'Driscoll, 2003). However, it is also known that, on many of these hills and knolls, fishing has been restricted to fishable areas and to areas that hold fish, leaving areas unimpacted or only very lightly fished during the exploratory phase of the fishery (see Trumble, Punt and Stern-Pirlot, 2016). The average percentage (in numbers) of such unfished or closed UTFs in New Zealand's EEZ has been estimated at about 68 percent (Black, O'Brien and Tilney, 2015).

For most orange roughy fisheries that are monitored, the scale of recent interactions between the fishery and the benthic environment has therefore been estimated to range from small to very small, and even when cumulative overlap between the fisheries and defined habitat across fisheries are considered, this was not sufficiently large to raise concerns about serious or irreversible impacts. For example, Trumble, Punt and Stern-Pirlot (2016) report that "in the 5 years to 2014, the maximum amount of structural damage to UTF habitats within the orange roughy distribution range that could be attributed to orange roughy fishing in the UoC areas is 13%, assuming 100% habitat destruction of habitat on the fished UTFs in the UoC areas."

The fishery footprint has been reported for the fisheries in New Zealand, the western South Pacific high seas fishery (SPRFMO convention area), and in Chile. Other fisheries have not been evaluated in a comparable way. Where assessed, the accumulated evidence suggested that, overall, the associated risk of an adverse impact on the structure or function of the benthic ecosystem would be relatively low for the largest of the New Zealand EEZ and straddling fisheries (Trumble, Punt and Stern-Pirlot, 2016), which have historically been and are also currently the largest orange roughy fisheries globally. Moreover, with the effective long-term closure of the orange roughy fisheries in Chile and the northeast Atlantic, further benthic impacts in these areas have been curtailed. However, it is important to consider two other aspects of this issue. First, most of these evaluations have considered current or recent fisheries, so that historical impacts (when the scale of the fisheries was larger) have not been adequately addressed; and second, given the patchy distribution of fishing effort and habitat components, it is likely that some significant local impacts may have occurred and may continue to do so in the most heavily fished areas.

A number of research studies have tried to estimate the time that it takes for species and habitats to recover from various types of fishing activity (gear types, frequency and intensity). While this approach has been relatively informative in the shallow coastal seas that are subject to the majority of fishing effort (Koslow *et al*., 2001; PFMC, 2005; Hiddink, Jennings and Kaiser, 2006; Clark and Rowden, 2009; Althaus *et al*., 2009; Pitcher *et al*., 2012, 2015, 2016; Pitcher, 2014), this has been less successful in understanding the ability or timescales for recovery in the deepwater fisheries such as those for orange roughy, with virtually no recovery of corals yet demonstrated (Williams *et al*., 2010; Clark *et al*., 2010b). This outcome should not really be unexpected as there are two key issues associated with such studies: (i) timescale, and (ii) positional accuracy. It is known that deepwater corals tend to grow extremely slowly and, thus, the expectation of being able to detect growth in individual specimens is likely to be on a timescale of (at least) decades (see review of age and growth in Clark *et al*., 2016b). While long-term observational studies have been initiated, they will need to run for many years before any regrowth is likely to be seen (Clark *et al*., 2010b). The issue of positional accuracy is that, in order to observe re-growth or recovery, it is a requirement that observations can be made repeatedly of the same organism or group of organisms, or on the same defined area of ground. If this cannot be reliably done, then it is likely that the timescale for researchers to be able to demonstrate recovery will be extended. Returning a camera, or other remote sampling device, to the exact same spot in a large and deep area of ocean is challengingly difficult, although the technology to do so is continually improving. Thus, it may not be so surprising that, at this time, studies of recovery in deepwater are sparse and largely inconclusive. Continued monitoring of these long-term study sites should be encouraged to improve understanding of the population process in the deep oceans. In the absence of such certainty, this issue should be dealt with through precautionary management that addresses the range of species and communities, assuming that recovery will not occur, or that it will be on such a long timeframe that recovery will **effectively** not occur.

Problems of benthic habitat classification, complexity, scale and estimation

In trying to interpret the likelihood of adverse impacts to benthic habitats from trawling in deepwater fisheries, a number of critical issues arise. Specifically, these are associated with the tools and information available to interpret the scale and intensity of interactions. For example, deepwater habitats are typically classified by the geology, geomorphology and the communities of fauna that live there, as well as by depth. In the deep-sea, these parameters are more difficult to study adequately than at shelf depths (Clark, Consalvey and Rowden, 2016c), and when marine habitat classifications are examined, it can be impossible to distinguish one class from another in any meaningful way to assist in management (Ford *et al*., 2016). There are other issues that are also difficult to work with, including the complexity of the seabed in terms of topography. Hard corals, for example, typically settle and develop on hard substrates. They can be found in small clumps over hard bottom, cover peaks and flanks of UTFs (Clark *et al*., 2010b), or can occur as continuous reefs extending over 1 km (Fosså, Mortensen and Furevik, 2002). However, it is not uncommon to find corals on very small areas of hard substrate, down to the scale of individual boulders in otherwise unsuitable habitat. As an example, habitat classification and appropriate protection is difficult for a poorly defined density of coral on a scattered boulder substrate. There is then the issue of scale, and at what scales should habitats be defined and, in particular, what are the appropriate scales from scientific, ecosystem and fishery management perspectives, and are they related in any way? There seem to be few answers to these questions, and all attempts to address them are confounded by the issues associated with estimating scale and distribution parameters from very limited data for deepwater habitats. More detailed data are required in order to understand the structure and distribution of deepwater benthic communities. For example, corals can also occur on firm muddy substrates, e.g. the bamboo corals (Isididae). Predictive modelling of the likely distribution of benthic taxa has advanced in recent years, and is increasingly being used to guide the development of spatial management. However, one group of studies has highlighted that the predicted pattern of suitable habitat for corals in the SPRFMO area differed dramatically depending on the scale of the data used (i.e. at the scales of the south Pacific Ocean (Anderson *et al*., 2016a), the New Zealand region (Anderson *et al*., 2016b), and the scale of an individual seamount on the Louisville Ridge (Rowden *et al*., 2017).

Issues of the nature and scale of interactions were addressed at a two-day workshop held in Wellington, New Zealand, in 2015 to review approaches to benthic science in support of fisheries management for New Zealand. This workshop determined that, as many of the issues described above were effectively insoluble in the near future, a risk-based approach was a more feasible way forward (Ford *et al*., 2016). The risk-based approach selected was pioneered in Australian inshore waters (Pitcher *et al*., 2012, 2015, 2016; Pitcher, 2014).

8.2 DEPENDENT SPECIES

The diet of orange roughy is known fairly well, and has been described in detail off Australia (Bulman and Koslow, 1992), in the northeast Atlantic (Gordon and Duncan, 1987), and for New Zealand on the Challenger Plateau and Chatham Rise (Rosecchi, Tracey and Webber, 1988; Forman, Horn and Stevens, 2016) and off the southeast of the North Island (Dunn and Forman, 2011). Further basic information on diet composition on Chatham Rise has also been published (Liwoch and Linkowski, 1986; Thomson, 1998; Clark *et al*., 2000; Jones, 2007) and extensive diet samples collected during research surveys around New Zealand have been summarized, although not analysed in any detail, by Stevens, Hurst and Bagley (2011).

More than 160 prey species or types have been identified for orange roughy, with the diet dominated by benthopelagic and mesopelagic crustaceans, fishes and squids. The prey often included crustaceans such as the mysids *Gnathophausia* spp. and *Boreomysis* spp., amphipods, natant decapods such as *Sergestes* spp. and *Pasiphaea* spp., mesopelagic and benthopelagic fishes such as myctophids Macrouridae, *Nansenia* spp. and *Bathylagus* spp., and cephalopods such as Cranchiidae and Onychoteuthidae squid. Diet composition has been found to change as orange roughy grow larger, with juveniles eating proportionately more small crustaceans, and adults eating more fish

(Bulman and Koslow, 1992; Rosecchi, Tracey and Webber, 1988; Dunn and Forman, 2011; Stevens, Hurst and Bagley, 2011; Forman, Horn and Stevens, 2016). Orange roughy diet composition has also been found to vary with depth, area, year, and water temperature (Bulman and Koslow, 1992; Rosecchi, Tracey and Webber, 1988; Dunn and Forman, 2011). Jones (2007) described a decline in squid in the diet of orange roughy off the west coast of New Zealand. Orange roughy during the spawning season appear to feed only rarely (Liwoch and Linkowski, 1986; Clark *et al*., 2000).

There are only a few records describing orange roughy predators. There are anecdotal reports of toothed whales (sperm whales [*Physeter macrocephalus*] in particular) being associated with orange roughy spawning aggregations, and Gaskin and Cawthorn (1967) found orange roughy were the commonest single species of fish found in the stomachs of sperm whales harvested from the Cook Strait, New Zealand, during 1963 and 1964. Sperm whale diet is usually dominated by cephalopods (Evans and Hindell, 2004), and where fish have been reported in sperm whale diets elsewhere, orange roughy was not identified (Roe, 1969). Wetherbee (2000) provided an anecdotal report of orange roughy in the stomachs of four species of deepwater shark from Chatham Rise: *Etmopterus granulosus* (= *E. baxteri*), *Centroscymnus owstoni*, *Centrophorus squamosus* and *Dalatias licha*. In a detailed study, orange roughy were found to be the most important prey of *E. baxteri* off Tasmania (Hallett and Daley, 2011). Pethybridge, Daley and Nichols (2011) found one orange roughy each in the diet of 98 *E. baxteri*, and 5 *D. licha* off Tasmania. Records of orange roughy being eaten by teleosts are extremely rare. Stevens, Hurst and Bagley (2011) reported just one orange roughy eaten in a sample of 18 000 ling, and four eaten in a sample of almost 106 000 orange roughy (cannibalized). No orange roughy were found in 11 254 stomachs from 25 species sampled at 200–800 m on Chatham Rise in the period 2004–07 (Dunn *et al*., 2009b).

Potential competitors of orange roughy have not specifically been described, although black oreo are known to have a similar diet to orange roughy (Clark, King and McMillan, 1989; Forman, Horn and Stevens, 2016). Potential competitors with juvenile orange roughy inferred from diet studies may include: alfonsino (*Beryx* spp.), lookdown dory (*Cyttus traversi*), macrourids such as Oliver's rattail (*Coelorinchus oliverianus*) and javelinfish (*Lepidorhynchus denticulatus*) (Dunn *et al*., 2009b). Potential competitors with adult orange roughy inferred from diet studies may include: squid, hakes (*Merluccius australis*) and hoki (*Macruronus novaezelandiae*), Ray's bream (*Brama* spp.), and a variety of sharks, including Owston's dogfish (*Centroscymnus owstoni*), longnose velvet dogfish (*Centroscymnus crepidater*), shovelnose dogfish (*Deania calcea*), Baxter's dogfish (*E. granulosus*), and Lucifer's dogfish (*E. lucifer*) (Dunn *et al*., 2009b, 2013). Johnson's cod (*Halargyreus johnsonii*) is also suspected to be a competitor, and the diet of smaller specimens is known to include the same natant decapods predated by orange roughy (Mauchline and Gordon, 1984). Off eastern New Zealand, Johnson's cod was one of the few species to show a significant increase in abundance following the depletion of an orange roughy stock (Doonan and Dunn, 2011).

8.3 ENDANGERED, THREATENED AND PROTECTED (ETP) SPECIES

8.3.1 Marine mammals and seabirds

The scale of interactions with marine mammals, seabirds and reptiles for orange roughy fisheries is generally comparatively low (see for example, MPI, 2015c). This is likely to be due to the depth at which the fisheries operate, and the geographical areas of the fisheries, which are often relatively far offshore and of limited spatial extent. For example, given the depth of fishing, the warps that connect the net to the vessel enter the water at a relatively steep angle close to the stern of the vessel, and this gives less opportunity for foraging seabirds to be accidently struck and killed by the warps when they fly around the stern of the vessel. Observational data, risk assessments, and reviews consistently demonstrate that these fisheries have low impact on seabirds (Abraham, Thompson and Oliver, 2010; Richard and Abraham, 2015; MPI, 2015c; Trumble, Punt and Stern-Pirlot, 2016). For marine air-breathers, the fishing depth is beyond the known range of all marine reptiles and all but the largest marine mammals, and the latter are then unlikely to be caught by the relatively small trawl nets, especially as tow durations are generally quite short (MFish, 2008a). For the period from 2002–03 to 2015–16, the only marine mammal with observed incidental captures in New Zealand's EEZ orange fisheries was the New Zealand fur seal, with a total of seven animals captured in this period. Estimated captures from 2002–03 to 2014–15, from a statistical model incorporating observed captures and total fishing effort, total 13 animals, with a peak of 5 animals in 2004–2005, averaging 1.0 per year over the period (Dragonfly Data Science, 2017).

The same is true for the high seas fisheries. Interactions between trawlers and seabirds, marine mammals and reptiles in the SPRFMO convention area are fully reported, as there is 100 percent observer coverage on trawlers, and interactions and mortalities are consistently minimal (Hansen and Hobsbawn, 2015; MPI, 2015b), with reports of only two seabird captures by bottom trawl vessels since 2007/08 (SPRFMO, 2017).

8.3.2 Fish

There are few ETP fish species that are likely to interact with orange roughy fisheries due to the fishing depth. Orange roughy fisheries are considered to pose virtually no risk to any ETP species of fish; most of these are large sharks that are not seen in the orange roughy fisheries, and the scale of the fisheries is also relatively small.

In New Zealand, there are seven protected shark species and two teleosts (both groupers). Captures of protected fish species in the orange roughy fisheries are not discussed in annual review documents, suggesting that there are none or very few protected fish captures in these fisheries (MPI, 2015c).

8.3.3 Coral

Damage to corals is a major concern of eNGOs and others with interests in marine conservation in relation to deepwater fisheries, and orange roughy fisheries in particular. Corals, especially hard branching corals, tend to be erect, can be large, and have a fragile branching structure. This makes them vulnerable to damage and destruction by demersal trawling in those areas where they occur (Koslow *et al*., 2001; Althaus *et al*., 2009; Clark and Rowden 2009; Freiwald *et al*., 2004; Hall-Spencer, Allain and Helge Fosså, 2002, Clark *et al*., 2016b).

There are no deepwater corals that co-occur in the areas of orange roughy fisheries that are identified as vulnerable, endangered or data deficient at a population level on the Red List of the International Union for Conservation of Nature (IUCN). Therefore, it may be deduced that the principal concern is of damage at a local scale and for defined vulnerable marine ecosystems, potential vulnerable marine ecosystems, or communities of taxa of biodiversity interest in areas of fisheries within EEZs and in the high seas.

Within the New Zealand EEZ, the majority of deepwater corals are protected under the New Zealand Wildlife Act (Anon., 1953). New Zealand appears to be the only jurisdiction with orange roughy fisheries that has protected corals in this way. However, no evidence has been presented that indicates these corals are in any way endangered regionally or at a species level, although as noted above it is possible that local population structure for some species increases the likelihood of local population-scale effects.

As evidenced from the determinants of the scale of the current and likely future Australian and New Zealand fisheries (i.e. the need to have a low level of fishing mortality), the future scale of these fisheries will be considerably smaller than in the past, with an associated proportionally reduced risk to all non-target species, including corals (see Trumble, Punt and Stern-Pirlot, 2016).

In the high seas, states have a duty to balance the development and operation of viable fisheries with protecting the wider environment. The UNGA, through its sustainable fisheries resolutions, supported by guidance from FAO, has called upon states, either individually or through RFMOs, to identify and adequately protect VMEs. In many cases, deepwater corals, sponges, and other erect sessile fauna are listed as VME indicator species by states and RFMOs, and, hence, damage or capture by fishing gear elicits a management response that leads towards their appropriate protection. Such measures are supported by the national legislation of states for their own jurisdictions or as RFMO members and cooperating non-members.

9. Management and sustainability

9.1 TARGET STOCKS

This section focuses on the management issues surrounding the current fisheries in Australia, New Zealand and on the high seas, but will also address Namibia given the possibility of a fishery returning in the southeast Atlantic. However, where relevant, examples will be drawn from other fisheries, including those that are closed.

As in many fisheries, in the historical orange roughy fisheries, managers did not always follow contemporary scientific advice. For example, in New Zealand on Chatham Rise as early as 1988 there were calls for substantial catch reductions, which were ignored (Robertson and Mace, 1988). It was not until the mid-1990s that catch limits on Chatham Rise, split into a number of smaller management subareas, were well aligned with stock assessment recommendations (e.g. see Annala, Sullivan and O'Brien, 2000).

The history of overfishing, changing political and socio-economic values, and the desire of the fish-product supply chain to see increased levels of sustainability have collectively led to a shift in the approach to management of many of the remaining orange roughy target fisheries, now restricted to Australia, New Zealand and the high seas in the southern Indian and western South Pacific Oceans. This is evident from the approach taken to reopening fisheries previously closed for sustainability concerns in both Australia and New Zealand, and the investment in science that underpins these management actions.

Australia and New Zealand have, for some time, had open and transparent science processes that are a key source of advice to fisheries managers. For example, Australia publishes calls for input to research planning and output science reports, as well as developing and publishing five-year strategic and annual research plans.[10] Similarly, New Zealand has developed and published minimum standards for "best available" science (MFish, 2008b), has open science review processes, as well as the publication of all relevant documents, including results from abundance surveys, stock assessments and supporting studies, and stock projections.[11] In addition, all key management documents are publicly available, including stakeholder consultation documents, policy papers and decision documents. This high degree of transparency in science and management is of fundamental importance in enabling those interested in the fisheries, be they fishers, eNGOs, other science and management organizations, processers or retailers, to be able to have confidence in the management of the fisheries.

In contrast, and in the context of the possibility that commercial fishing for orange roughy will be restarted in Namibia, Namibian fisheries science and management historically appears to have been less transparent than in other jurisdictions. This can be seen from the relative paucity of publicly available documents for stock assessments, stock status, management and policy decisions, about which public statements have been made by government and industry. Transparency in the processes of management tends to deliver better outcomes in fisheries sustainability.

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[10] See: www.afma.gov.au/research/

[11] Available at: http://fs.fish.govt.nz/Page.aspx?pk=91

9.2 LIMITING FISHING MORTALITY

Successful management of fisheries requires average fishing mortality to be limited to a level that the target fish stocks can sustain.

There are a number of tools available to assist managers to make appropriate sustainability decisions, including in the face of unreasonable demands from fishers, eNGOs or government officers, in order to achieve appropriate limits to fishing mortality. At their simplest, these can be described as decision points, such as limit reference points (LRPs) and decision tools, such as HCRs. While the development of target and limit reference points, and HCRs can be achieved fairly simply for some fisheries, or simply assumed following studies elsewhere, significant uncertainties remain in the understanding and monitoring of deep-sea fish and their fisheries, and a more coherent strategy developed using MSE generally gives better outcomes (Sainsbury, Punt and Smith, 2000; Dichmont *et al*., 2008; Mapstone *et al*., 2008; Butterworth *et al*., 2010; Punt *et al*., 2014).

The application of MSE techniques has been used in New Zealand since 2014 for the major orange roughy fisheries (Cordue, 2014c), and combined with the entry of these fisheries to the MSC certification process, has yielded marked improvement in the management of the fisheries. The process developed in New Zealand focused on long-term sustainability goals and so only considered biological performance without explicitly considering social or economic objectives. Essentially, an MSE is used to develop a harvest strategy acceptable to key stakeholders (in the New Zealand case, industry and the ministry), and then the different options of the components of the harvest strategy. This is best seen in the development of a range of candidate HCRs, comparison of their performance, and eventual selection of the one that best meets the defined management objectives and is most robust to the most important uncertainties (e.g. in a stock assessment model, the stock-recruitment steepness h, stock structure, natural mortality M, etc.). Those HCRs that are robust to those uncertainties will perform well even if some of the underlying assumptions are incorrect (Punt *et al*., 2014; Cordue, 2014c). An MSE can also be used to estimate LRPs, target reference points or ranges.

The eventual choice of an LRP, and target reference point (TRP) or TRP range, depends upon the strategic objectives of the chosen harvest strategy, coupled with the biological dynamics of the fish species (principally its productivity), including late maturity (indicative of higher reference points) and longevity with many age classes (indicative of lower reference points, see Cordue, 2014c). For orange roughy, the deterministic B_{MSY} has been estimated to be relatively low, because under constant recruitment, it makes sense to target the large number of accumulated cohorts that are undergoing little or no growth. Stochastic B_{MSY} is likely to be similar to the deterministic estimate because of the relatively large number of cohorts in the spawning biomass (P. Cordue, personal communication) This result, which suggests initial stock size should be reduced by perhaps as much as 75 percent to reach B_{MSY}, is at odds with accepted dogma about the relatively high vulnerability of long-lived deep-sea fishes (Francis and Clark, 2005). As a result, a more precautionary approach to limits **and** targets has generally been advocated (e.g. the defaults in the New Zealand Harvest Strategy Standard, and those used in Australia). Whatever the estimated value of B_{MSY}, best practice dictates that the adopted management target reference point should always be greater than B_{MSY} as a precaution to allow for possible errors in estimation or management of fishing mortality. This approach is applied in both Australian and New Zealand orange roughy fisheries (Upston *et al*., 2014; Cordue, 2014c).

9.3 FUTURE RECRUITMENT

Little is really known about patterns of recruitment in very long-lived species such as orange roughy. This becomes an issue in assessing the stocks, and in predicting trends in future abundance. This has been a longstanding concern around the sustainability of orange roughy fisheries (e.g. Clark, 2001). In addition, there is an expectation that the high longevity of orange roughy may have evolved to help the species withstand cyclical periods of good and poor recruitment (Leaman and Beamish, 1984; Longhurst, 2002). As a result, extended periods of poor recruitment might be expected; and over conventional fisheries assessment and management timescales (years to decades), the level of recruitment achieved by a stock might be much lower or higher than expected from the observed spawning stock biomass. Historically, orange roughy assessments assumed deterministic recruitment, which tended to generate overoptimistic model results and was also problematic when projecting future abundance (Francis and Clark, 2005). Improvements in assessing ageing (Tracey *et al.*, 2009; Horn *et al.*, 2016) have enabled age frequency estimations to better inform the assessment models of recruitment pattern, and provide a sampling base for assuming future recruitment in predictions (Cordue, 2014a). However, imprecision in ageing remains high enough that only general trends in recruitment can realistically be estimated (not individual YCS); stock assessment approaches must appropriately allow for this imprecision, but is arguably not done currently. It is of note that, for very long-lived fish, a lightly fished standing stock will be composed of a large number of accumulated age classes, and this will act to dampen the influence of recruitment variability on stock size compared with shorter-lived species.

The influence that the fisheries have had on reproductive dynamics and performance, through reduction of spawning stock biomass to low levels, and disturbance of spawning aggregations by fishing, remains unknown. The earliest substantial fisheries occurred in New Zealand in the mid-1980s, and with an estimated age of recruitment of 35 years or more, the effect of fishing on reproductive success and subsequent recruitment will not be known until at least 2020, although reduced future recruitment has been suspected (Dunn, Anderson and Doonan, 2008). The more recent biomass increases observed in some Australian and New Zealand fisheries appear to have been fuelled largely by new recruits, which will have been spawned well before the fisheries started. Therefore, fishery managers will need to be mindful of the potential for a period of reduced recruitment (in the near future, at the time of writing) resulting from the fisheries “boom” in the 1980s and 1990s.

9.4 REBUILDING STRATEGY

For stocks that are depleted, i.e. well below the target or below their LRPs (or proxies) or otherwise approaching the point of recruitment impairment, best practice requires the development and implementation of a time-bound rebuilding plan.

Australia has a clear, public, time-bound rebuilding strategy for orange roughy stocks (AFMA, 2015), which replaced an earlier conservation programme dating from 2006. In New Zealand, the stock status of the Mid-East Coast orange roughy stock was determined to be low in 2013 and 2014 (14 percent B_0), and it is now in a public FIP with a stated time-bound rebuilding plan intended to return the biomass to acceptable levels and enable this fishery to seek third-party certification (DWG, 2016).

The ultimate rebuilding strategy is to effectively prohibit catching any individuals from a depleted stock, as this will produce the maximum rate of rebuilding. Operationally, this approach would be seen through fishery closures, and also in very tight controls of any catch as bycatch in other fisheries. A fishery closure providing the maximum rate of rebuilding has been shown to be effective in a number of fisheries, including the ORH 7A (Challenger Plateau) fishery in New Zealand and in the St Helens fishery off Tasmania, and possibly also in Namibia. Approaches that provide for a more balanced outcome in terms of rebuild timeframe versus the scale of catch can also be implemented (e.g. the New Zealand Mid-East Coast FIP described above). Such balanced approaches, where more catch is permitted and a longer rebuild timeframe is accepted, can also enable more effective scientific monitoring of the stock through the collection of fishery data and having sufficient catch available for research surveys to be conducted.

As part of any rebuilding plan, careful consideration of monitoring needs is imperative. In order to detect change in status of a fishery under rebuilding, monitoring activities may need to be different from normal fisheries monitoring. This is certainly likely to be the case where rebuilding involves the closure of the commercial fishery, and a resultant lack of fisheries-dependent information. For orange roughy, such monitoring would probably require the application of acoustic methods to estimate abundance, coupled with estimation of age frequencies from the stock, stock assessments, as well as a small level of catch to enable checking the species identity and species mix of any acoustically sampled aggregations and collection of biological samples, including otoliths for ageing.

9.4.1 Management of impacts on associated species

While recognized current best practice is to manage all associated species within an ecosystem-based management framework, such frameworks are complex and have proved difficult to operationalize. Simpler approaches are usually applied with improved data collection and more analytical effort directed at those species that have a commercial value and/or are either a significant proportion of the bycatch or are known to be vulnerable to the effects of fishing for some reason (e.g. inherent biological attributes, or high spatial overlap between the stock distribution and the fishery). For these higher-priority species, one or more management targets against which to monitor stock heath are usually developed. These can be stock status for those species that have a stock assessment or can be simpler indicators of stock health, including such metrics as commercial or research CPUE compared with a reference period, and length and or/age frequency distributions over time.

If an associated species has a commercial value and is retained, then outside of incorporation within an ecosystem-based fisheries management framework, current best practice is for it to have its own management target (or targets) against which stock health can be monitored.

For unwanted bycatch, the quality of information is often inadequate to support full analytical stock assessments or even more rudimentary assessments such as changes in CPUE. Under such circumstances, fisheries should endeavour to reduce such catches where possible, using appropriate measures including gear modifications and avoiding temporal or spatial catching “hotspots”, and appropriate simple indicators should be defined and monitored.

The fundamental need with respect to sustainable fisheries management is to ensure, with an acceptably high degree of confidence, that fishery impacts remain below a level that is likely to lead to serious or irreversible harm (i.e. the scale of fishery impacts is sufficiently low to ensure sustainability).

For associated species, fishery managers are generally prepared to accept a higher risk with respect to stock biomass than for target species. This is often phrased by way of stocks not going below thresholds that might endanger recruitment, codified as staying above an LRP. Identifying such thresholds becomes important and, in the absence of good stock-specific information, these should be appropriately precautionary.

9.4.2 Management of impacts on benthic ecosystems

Recognized current best practice for the management of benthic impacts of demersal fishing is through the use of spatial management measures that result in areas where demersal fishing is permitted and areas that are closed to demersal fishing (and preferably other potentially damaging human activities) (Spear and Cannon, 2012; Ardron *et al*., 2014; Delavenne *et al*., 2012). Such closed areas are commonly recognized as MPAs, but this publication does not use this terminology because of the different interpretations that can be placed upon this term. There are three critical issues with the implementation of spatial management: (i) the identification of vulnerable areas needing to be protected through closures; (ii) the scale of spatial closures; and (iii) the distribution and representativeness of those closures. Representativeness has been variously addressed through design by managers (e.g. Helson *et al*., 2010) or though consultative processes involving a wider range of stakeholders (see CCAMLR, 2005; Cryer *et al*., 2017). In both approaches, various informative data should underpin the decision-making processes. These data have typically included both fisheries information used to define those areas of particular value to fisheries interests, and distributional information on taxa used to indicate areas of high biodiversity interest or of VMEs.

It has generally proved difficult to obtain agreement on both the scale and distribution of spatial management measures, partially because there is limited scientific analysis that can inform what is appropriate. The usual approach has been to optimize the scale and distribution of spatial closures so as to limit the impacts on fishing opportunity, while at the same time maximizing the protection of areas of high conservation interest. This requires some implicit or explicit trade-offs that can prove difficult for some to accept, such as the loss of possible future fishing opportunities, or that benthic damage will occur within fished areas. The value of spatial management is that benthic damage is limited to those areas open to fishing and that sufficient areas are completely protected from fishing impacts. Embedded in this process are a variety of other complex issues that can generate dissatisfaction and disagreement from some or all stakeholders. For example, there can be considerable disagreement about what constitutes an area closed to fishing (Trumble, Punt and Stern-Pirlot, 2016; Gianni and Bos, 2012). In addition, there may be a need for a "move-on rule" and the definition of its operational details, both of which are often contentious and difficult to agree (Auster *et al*., 2011; Hansen, Ward and Penney, 2013; FAO, 2016; Cryer and Nichol, 2017).

The calls by both fisheries and conservation interests, and their associated government departments, with respect to the scale of spatial closures have typically been increasing over time, as have the cumulative areas of seabed closed to fishing in EEZs (Trumble, Punt and Stern-Pirlot, 2016) and in the high seas (Thompson *et al*., 2016).

While there may be limited science to support general approaches for defining appropriate scale and distribution of protection for a benthic environment, pragmatic solutions have been proposed (Spear and Cannon, 2012), and implemented (Helson *et al*., 2010), also with recommendations for best practice discussed with respect to the MSC programme (Grieve, Brady and Polet, 2014, 2015). In specific areas, benthic information derived from both fishing operations and research surveys, coupled with habitat modelling, has been used to inform the discussion and selection of all spatial management parameters (Rowden *et al*., 2015).

It is important to note that recovery is not a requirement for sustainability. This is true provided that the balance between any spatial management (open and closed) areas protects an adequate proportion of the habitat and that the distribution of the protection is also appropriate (i.e. representative of habitat distribution rather than being confined to one area). This applies whether considering EEZ areas under national jurisdiction, or high sea areas managed by RFMOs and the subject of various benthic-focused UNGA resolutions. The adequate proportion of benthic habitat requiring protection remains uncertain for most deepwater ecosystems, although ranges of 20–30 percent and 30–50 percent are cited (e.g. Botsford, Hastings and Gaines, 2001; Airame *et al*., 2003), with proponents of best practice supporting proportions within these ranges (Greive, Brady and Polet, 2014; Spear and Cannon, 2012).

Both Australia and New Zealand have taken a wider approach to benthic protection across all of their fisheries, and have included areas specifically to protect the types of habitat that also occur where orange roughy fishing occurs. Other jurisdictions have generally less-developed approaches. Australia and New Zealand have extensive ranges of MPAs intended to principally protect different wildlife and habitat. About 30 percent of the area of the New Zealand EEZ is closed to demersal fishing using mobile gear through a network of large benthic protection areas (BPAs) (Helson *et al*., 2010) that are relevant to a range of benthic species including a large number of coral species also found in the depth range of the orange roughy fisheries. Pelagic fishing, where the gear remains at least 100 m above the seabed, and non-mobile demersal gear (longlines and pots) are permitted within the BPAs, but these methods are not used to catch orange roughy. With specific relevance to protecting habitat that is associated with orange roughy fishing, there are also a series of "seamount closures" (mostly hills and knolls) within the EEZ (Brodie and Clark, 2003), which are completely closed to fishing and which together with the BPAs protect about 30 percent of hills, knolls and seamounts in the EEZ from mobile bottom gear (Helson *et al*., 2010; Trumble, Punt and Stern-Pirlot, 2016). Australia has an extensive network of MPAs for various purposes, including protecting the benthic environment in the areas and depth of orange roughy fisheries, including for example, fishing closures at the Tasmanian Seamounts and the East Coast Deepwater Zone trawl exclusion zone (Figure 25; AFMA, 2003, 2014b).

FIGURE 25

Trawl closures in the Southern and Eastern Scalefish and Shark Fishery, Australia, 2014

Source: AFMA (2014b).

Chile has conducted some specific impact work on the area of its former orange roughy fisheries and the commercial fishery has been closed since 2006 (Niklitschek *et al*., 2010), with trawling in vulnerable environments (such as seamounts) prohibited under the Chilean Fisheries Act 2012 (Payá, personal communication).

Namibia has been developing a considered approach to defining and implementing MPAs, but this has not yet been applied to deepwater environments (Currie, Grobler and Kempe, 2008).

With regard to orange roughy fisheries on the high seas, two areas stand out as having given specific consideration to protecting benthic habitat, and both have used spatial management approaches. The more advanced approach is for the South Pacific Ocean within the SPRFMO convention area, which has had interim measures in place managed by Australia and New Zealand from 2007, which were subsequently developed into formal RFMO conservation and management measures (see CMMs 2.03, 4.03 and CMM 03-2017).[12] The effectiveness of the New Zealand approach to protecting benthic habitat on this area has been reviewed by Penney and Guinote (2013). The other area is the southern Indian Ocean, where industry developed a series of spatial closures that have subsequently been considered for formal protection by the newly formed RFMO and where member-implemented encounter protocols are also in place (Shotton, 2006; SIODFA, 2013; Sanders and Thompson, 2016). The approach taken by the SPRFMO is more thorough and developed, but has taken longer to implement that that applied in the Indian Ocean.

For the high seas, given the relative scales of orange roughy fishing effort, there has been a disproportionately large amount of discussion and interest in the impacts of demersal fishing compared with within-EEZ areas. This has been driven in part by the historical lack of specific measures implemented for the demersal high seas fisheries, and concern about unregulated fisheries in those areas. This has contributed to the setting up of RFMOs that included management of benthic impacts of fishing as part of their remit. In the last decade, largely in response to a number of UNGA resolutions concerning sustainability of high seas, deepwater fisheries and protection of VMEs, these RFMOs have increasingly supervised a range of research to define the sensitivity of their benthic environments, and to define, monitor and manage the scale of interactions and impacts. The approaches have varied a little between the different RFMOs, although the same basic principles apply. Most have identified specific areas with known or likely VMEs that may be at risk of significant adverse impacts from bottom fisheries and to close those areas to bottom fishing (e.g. the Commission for the Conservation of Antarctic Marine Living Resources [CCAMLR], Northwest Atlantic Fisheries Organization [NAFO], NEAFC and SEAFO). In addition, others, such as the SPRFMO, have opted to allow continued fishing in some areas previously fished during a particular time reference period and close all unfished areas to demersal fishing unless specific impact assessments and exploratory fishing plans are prepared and formally reviewed by the scientific committee and the commission. The current position is that, for those high seas areas under RFMO control where orange roughy are fished, most areas, whether the whole convention area or just the area at fishable depths, are protected from open-access fishing to a substantial extent (see Thompson *et al*., 2016).

In terms of ensuring that closed areas are respected, RFMO members and cooperating parties are typically required to monitor such areas using vessel monitoring systems and observer programmes. Moreover, there are reporting actions in place to enable the impacts in open areas to be monitored, such as the approach used in the SPRFMO, where there must be 100 percent observer coverage and observers record benthic interactions and monitor the requirements for triggering move-on rules. In this case,

[12] See: www.SPRFMO.int

ministry observers record all benthic catches, and these data are reported to flag states and to the SPRFMO (Hansen and Hobsbawn, 2015; MPI, 2015b).

On the high seas, the SPRFMO has a network of spatial management areas implemented independently by flag states that can fish for orange roughy using demersal trawls (currently, only Australia and New Zealand). These arrangements are still of an interim nature, but are fully regulated through a bottom-fishing CMM. The management of fishing impacts on benthic habitats and potential VMEs (no VMEs have yet been designated) is conducted though a spatial approach whereby some areas are open to fishing and others are closed, built up from 20' latitude by 20' longitude blocks. The approaches implemented separately by Australia and New Zealand operate in rather different ways. Australia implements areas open and closed to fishing, with its 2002–06 footprint open to fishing, except that on the Tasman Rise, which has not been fished by New Zealand vessels since 2001 and has been closed to fishing since 2007 (Williams *et al*., 2011). The rest of the SPRFMO convention area is closed to demersal fishing (Williams *et al*., 2011). In the open area, vessels are subject to a bycatch move-on rule with relatively high trigger thresholds, but which exceeded lead to the temporary closure of the area within five nautical miles around the path of the tow to all Australian vessels for the rest of the year (Williams *et al*., 2011). New Zealand's approach to spatial management has been to consider the New Zealand 2002–06 fishing footprint and divide the total area into three approximately equal parts: lightly fished, moderately fished, and heavily fished. The lightly fished areas have been closed to demersal trawling, the moderately fished areas are open to demersal trawling but subject to a move-on rule with low trigger thresholds also incorporating a biodiversity trigger, and the already heavily fished areas are open to trawling with no move-on rule (MFish, 2008a). Some 20' × 20' blocks in the moderately and heavily fished areas have also been closed to demersal trawling due to the known presence of VME indicators. Australia and New Zealand implement 100 percent observer coverage for demersal trawlers in the SPRFMO convention area. The observers record and report details of benthic encounters, and ensure that the move-on rules are followed. In order to ensure compliance with move-on rules, complete observer coverage in the trawl fisheries is a requirement.

Hansen, Ward and Penney (2013) reviewed the utility and effectiveness of move-on rules as a mechanism to protect benthic habitat, biodiversity and VMEs to inform the SPRFMO. This review was unequivocal in describing the use of move-on rules as suitable for interim arrangements only, confirming other work that has suggested that move-on rules tend to redistribute effort, including into areas that may have a higher conservation value (e.g. Auster *et al*., 2011; Clark and Dunn, 2012).

There is a growing body of considered, scientifically-based, best practice relating to implementing benthic protection in the marine environment, to assist managers and their advisors in developing appropriate programmes (Hilborn, Micheli and De Leo, 2006b; Spear and Cannon, 2012; Clark and Dunn, 2012; Greive, Brady and Polet, 2014). In essence, the advice is to develop and implement a balanced spatial approach to the management of fisheries impacts on the benthic environment. This approach is usually, and needs to be, focused on the wider management of fisheries within a specific jurisdiction, rather than for deepwater fisheries alone.

Elements of this collective of proposed best practice include implementing a network of relatively large-scale closed areas to protect benthic organisms, communities, and habitat structure and function. These networks should be broadly representative of the main habitat types present in the management area, and ideally selected though an open, inclusive, science-based process. The networks should seek a balance between maximizing the benthic environment conservation goals (i.e. protection of individual species, communities and habitats) and minimizing the social and economic impacts on the fishing industry at the time of implementation and into the future.

The process of selecting areas to close has often proved controversial and requires workable approaches to define areas of conservation interest, including identification of VMEs in high seas areas (e.g. Ardron *et al*., 2014), and especially where such areas may be at risk of significant adverse impacts from bottom fishing. A similar, practical approach needs to be taken to define areas of fisheries interest, with a practical, rational and justifiable approach to addressing the representativeness of the pattern of closures (e.g. Helson *et al*., 2010). However, there are a number of tools to assist in some of these difficult tasks, including various software packages to enable visualization of the range of possible outcomes to assist the discussions around the conservation–fishing trade-offs, including the Zonation conservation planning software (Moilanen *et al*., 2005; Moilanen, Kujala and Leathwick, 2009). Implementation of spatial planning with respect to deepwater fisheries and orange roughy in particular has been recently developed and applied (Rowden *et al*., 2015). The RFMOs have seldom considered the issues of creating a network of, or representativeness in, benthic coverage in spatial management, something that has been more common within EEZs.

The current best practice guidance for managing fishing impacts on benthic habitat proposes very large-scale closures to mobile demersal fishing gear (trawls, dredges, etc.) but may also extend to cover static demersal gear (bottom-set gillnets, bottom-set longlines, pots, traps, etc.). The scale of the suggested closures is typically about one-third of the management area. While many jurisdictions have so far achieved much lower levels of benthic protection, a number, including those with the largest orange roughy fisheries (Australia and New Zealand), have either achieved these levels or are moving towards this scale of closure. This specifically includes a number of RFMO-managed high seas areas, and the SPRFMO, which manages the largest high seas orange roughy fishery, has closures of a much greater extent, amounting to more than 95 percent of the total convention area, and about half of the fishable depth within the convention area (MFish, 2008a; Williams *et al*., 2011; Penney, 2013).

For the southern Indian Ocean, SIOFA members have embarked on collecting information, and have been developing a process of addressing the management of benthic impacts through the development of fishing footprints (Hansen and Hobsbawn, 2016) and bottom-fishery impact assessments (e.g. Williams *et al*., 2011). This has been coupled with substantial and detailed acoustic imaging of considerable areas of the southern Indian Ocean (Rogers *et al*., 2009; SIODFA, 2016). The industry in this area was also proactive in closing areas to demersal fishing that held extensive and or high densities of biota of conservation and/or biodiversity interest, especially areas of corals and sponges, which have now been formalized by the newly created RFMO (Shotton, 2006; SIODFA, 2013; Sanders and Thompson, 2016).

The fishing industry targeting orange roughy and alfonsino in the southern Indian Ocean has been proactive in understanding the distribution of benthic habitat of conservation interest, including areas that may in future be considered as VMEs. It has also demonstrated a highly responsible approach to self-regulation to protect some important areas of habitat in advance of the existence of a fully functioning RFMO (Rieser, Watling and Guinotte, 2013; SIODFA, 2016).

There are two important issues that are often misunderstood or misreported. First, in order to protect the benthic environment, it is not a requirement to have no-take MPAs. Pelagic fishing with no seabed contact can be conducted with no risk to benthic habitats. Such fishing could include mid-water trawling for pelagic fish, such as the krill fishery in the CCAMLR and pelagic longline fishing for tunas, billfish or sharks. Second, such closed areas are de facto MPAs, but the defined purpose should specifically be to protect benthic habitat and not the target or bycatch fish species. The sustainability of the stocks of fish species should be addressed though specific population targets, monitoring, and management of fishing mortality (i.e. through properly managing fishing effort and catches). Additional spatial measures may be used if there is, for example, a clear functional association with a particular habitat (e.g. juveniles in a biogenic habitat for shelter); something yet to be shown for orange roughy.

The benthic protection measures that have been developed and implemented are primarily for the protection of the benthic component of biodiversity, while enabling viable commercial fishery operations. While questions remain about how to evaluate the effectiveness of the various measures and approaches, especially as key factors such as connectivity are poorly understood, it is the scale and representational coverage of spatial closures that are driving acceptance of adequacy of measures. Benthic habitat and biodiversity protection are still often political or managerial, with science needing to advance faster and provide information that moves from qualitative (describing impacts on species or species groups) to the quantitative, population and biodiversity estimates that managers require.

9.4.3 Management of impacts on endangered, threatened and protected species

Risk to ETP species from orange roughy fisheries has been determined to be relatively low (e.g. Wayte *et al*., 2007; Abraham, Thompson and Oliver, 2010; Richard and Abraham, 2015; MPI, 2015c; Trumble, Punt and Stern-Pirlot, 2016), and active management may not be specifically required for interactions with ETP species. However, in order to ensure that this remains the situation, it is important that there be a programme of effective monitoring of interactions and a strategy for intervention should active management be needed. This approach can then be supported by periodic risk assessments and population studies.

A key function of government, with respect to the impacts of fisheries on particular species or taxonomic groups, is to have transparent and robust processes to protecting species at risk, with appropriate legislation. For those areas where orange roughy have been fished, this is well developed in Australia, Chile and New Zealand, as well as with the European Union (Member Organization).

Jurisdictions that are leading best practice in managing fisheries impacts on ETP species tend to be implementing a fishery-wide approach, with monitoring focused on species at particular risk or fisheries with higher levels of interactions or impacts. The RFMOs generally follow approaches used by their members but often implemented to a different degree or in a different way, and either more or less effectively. Elements of successful strategies typically include: scientific observers on vessels, with a coverage of the fishery related to defined aspects of interactions (i.e. the percentage coverage of the fishing activities by observers is related to the risks, status of key species, uncommonness of interactions, etc.); assessment of risk to specific species or species groups; apportioning risk to different fishery types; population monitoring; regulations to retain or discard captures; and requirements to report all captures of ETP species by vessels.

Generally, higher observer coverage, including 100 percent coverage, is perceived as beneficial as there is proportionately less uncertainty in the collected information, and 100 percent coverage also fits well in areas where there may be a benthic move-on rule.

9.5 MARINE MAMMALS

All species of marine mammals are protected in Australia, New Zealand and within the SPRFMO convention area. In Namibia, there is a managed harvest of South African fur seals (*Arctocephalus pusillus pusillus*), while all cetaceans are protected. The distance offshore for most orange roughy fisheries, coupled with the great depth of the fishing operations, means that there are few observed interactions between vessels fishing for orange roughy and marine mammals, and very little risk to the mammals. The three largest orange roughy fisheries in New Zealand recorded no incidental captures of any marine mammals between 2002 and 2012 (Thompson and Berkenbusch, 2013). As such, bycatch of marine mammals is not a sustainability concern for current orange roughy fisheries in New Zealand, and by extension elsewhere.

9.6 SEABIRDS

With a few specific exceptions, seabirds are fully protected in Australia, Namibia, New Zealand and the European Union (Member Organization), and there is some protection afforded to seabirds, especially petrels and albatrosses, through international agreements for the RFMO managed areas (Anon., 1953; Bianchi *et al*., 1999; EU, 2009; Ramirez *et al*., 2017). New Zealand and Australia have well-defined requirements relating to observing and reporting incidental captures of marine mammals, marine reptiles, and seabirds both within their EEZs and on the high seas, and both states also have well-developed approaches to the use of mitigation measures, especially for seabirds (MPI, 2015c; Hansen and Hobsbawn, 2015). For seabirds, mitigation measures for trawl vessels typically take the form of devices intended to keep seabirds away from the area immediate astern of the vessel. This area is where the paired trawl warps pass from the blocks at the back of the deck into the water, and in this area seabirds are at risk of colliding with warps or being dragged under the water by the warps due to the movement of the vessel through the water. Devices used include Brady bafflers and tori lines (Lokkeborg and Thiele, 2004). Due to encounter rates being below a management threshold, Australia does not require its trawlers to use seabird mitigation in the SIOFA area (Hansen and Hobsbawn, 2016). Apart from using appropriate mitigation devices, best practice management includes implementing vessel operating procedures to minimize activities that would increase risk to marine mammals, reptiles and seabirds, such as controls on the release of offal. These can be regulatory or voluntarily implemented by industry (DWG, 2017). As much of the current orange roughy catch is not processed at sea, there tends to be less offal than in equivalent fisheries for other species, which makes vessels fishing orange roughy less attractive to foraging seabirds. Generally, incidental capture of seabirds is not a sustainability concern for current orange roughy fisheries. However, impacts on those seabird species of greatest conservation concern should continue to be actively monitored.

9.7 MARINE REPTILES

Marine reptiles, especially turtles, are afforded protected status in all areas where orange roughy fishing occurs, with the exception of the southern Indian Ocean. It is expected that when SIOFA is fully established, marine reptiles will receive similar protection to that afforded by other RFMOs. As noted above, the depth of fishing operations makes interactions between reptiles and vessels very rare, and risks from orange roughy fishing to turtles in particular are very low. As such, bycatch of marine reptiles is not a sustainability concern for current orange roughy fisheries.

9.8 ECOSYSTEM SUSTAINABILITY

The requirements of an ecosystem-based approach to fisheries management necessitates that sustainable fisheries avoid significant adverse impacts, especially unnecessary impacts, to all key ecosystem components, including to bycatch populations, key predators and habitats, and to the wider ecosystem. Significant adverse impacts are often defined in terms of a level of reproduction that is impaired to the point that population recovery is inhibited. As noted, most jurisdictions have regulatory requirements relating the management of interactions between fishing and animals such as marine mammals, marine reptiles, and seabirds. These regulatory requirements take various forms but usually include reporting, avoidance of and minimizing impacts, mitigation, and retention or discarding.

In order to understand and be able to monitor the interactions of fishing vessels with all elements of the ecosystem, comprehensive observer coverage is highly desirable. Complete observer coverage of fishing fleets allows high-quality data to be collected on both target and bycatch species, and also helps ensure compliance with fishery management regulations. It and is therefore recommended for any fishery. Vessels fishing orange roughy are generally large and so capable of carrying observers. Adequate observer coverage is a key component to ensure demonstrably sustainable fisheries. In future, some elements of the observer function may be enhanced and/or replaced by electronic monitoring, using, for example, at-sea camera systems with subsequent data review ashore. While electric monitoring systems will be useful in data gathering and in fulfilling a regulatory function, they are not a panacea and will not provide solutions in all situations. For example, electric monitoring equipment will not be able to identify the gender of most fish, including orange roughy, the stage of sexual maturity or collect otoliths for ageing, and it is unlikely to be able to identify to species some groups of fish (e.g. macrourids). Hence, there will continue to be a need for either an at-sea observer programme and/or a port or landings sampling programme.

It must be acknowledged that understanding ecosystem function and, hence, long-term sustainability is a **major** challenge. Ecosystem-scale information requires an extensive scientific research programme that is unlikely to be a realistic option for many single fisheries. Hence, monitoring as many components of the system from as early a stage as possible is important. For new fisheries, there should be an increased emphasis compared with the past on establishing baseline conditions, and describing the structure of the pre-fished system. Although deep-sea ecosystem functions have been described in general (e.g. Armstrong *et al*., 2012; Thurber *et al*., 2014), specifics and their quantification remain poorly known. In particular, there are difficulties in describing the level of impact that is likely to constitute a significant adverse impact at the ecosystem level. Such research, using, for example, ecosystem models, is in its infancy. To date, the research has essentially focused on finding simple models that work; they have not yet been developed to account for, for example, spatial heterogeneity in habitat, or basic biological characteristics such as ontogenetic (size) specific shifts in species' ecological role (Heymans *et al*., 2011; Morato *et al*., 2016).

In the ecosystem evaluations done for almost 500 fisheries against the ecosystem component of the MSC standard for sustainable fisheries, no fisheries were judged to have failed and only four fisheries received a conditional pass (MSC, personal communication). While this could suggest that the standard is weak at this point, there is a strong consensus among MSC assessment teams that the fisheries reviewed are operating in a way that does not risk ecosystem function, and that this is consistent with the lack of major ecosystem sustainability problems that can be attributed to fisheries.

It is also notable that very few commercial-scale fisheries have ever been clearly recognized as having caused a major ecosystem shift as a result of the fishery alone (Frank *et al*., 2005, 2011; Daskalov *et al*., 2007; McCain *et al*., 2016).

While evidence from other fished (mainly shelf) ecosystems suggests that they are inherently robust to the historical and current levels of fishing, the low productivity of deep-sea fisheries and their supporting ecosystems means that any major ecosystem shifts (e.g. similar to the Northwest Atlantic cod) would be expected to take considerably longer to establish and also to be reversed. This means that the full impacts of the existing orange roughy fisheries may not yet have been fully felt. There has yet to be one orange roughy generation time since the start of the fisheries. Other similarly long-lived species have also had little time to respond to the changes in orange roughy abundance. It may be many decades before a competitor to orange roughy could increase in abundance to fill the gap in the ecosystem created by fishing orange roughy. Any population or ecosystem recovery (beyond that fuelled by pre-fishery recruitment) may therefore take place on a multidecadal time-scale.

Of all of the areas of fishery-related science, ecosystem sustainability is one of the most in need of development. Appropriate scientific and especially modelling innovation is beginning to deliver a suite of tools that will be of use in the future. However, their application to deep-sea fisheries questions will continue to lag behind the shelf-seas, principally due to the paucity of data about the deep oceans, and the relatively high cost of filling that data gap.

9.9 NEW SUSTAINABILITY TOOLS

Starting in the late 1990s and with increasing effectiveness over time, a new approach to driving fisheries sustainability has developed. This has increasingly brought new tools and new players to promote and deliver this development. Collectively, these can be seen be seen as market- or supply-chain-led drivers, with the best-known example being the (MSC).[13] The MSC, dating from 1997, is a not-for-profit global NGO that enables the application of rigorous, evidence-based evaluation of sustainability criteria, specifically linked to market access and or/pricing. The MSC programme has typically attracted fisheries that were already well on the way towards meeting sustainability criteria through having appropriate management in place. The advent of FIPs, first advocated by the Sustainable Fisheries Partnership,[14] in the mid-2000s brought a different perspective, in that any fishery had marketable worth provided it had an improvement plan and was publicly and transparently making sustainability improvements.[15] Both of these approaches can help, and have helped, promote improved sustainability substantially in many hundreds of fisheries, including some for orange roughy and other deepwater fish.

With respect to orange roughy fisheries, the use of these new sustainability tools has been led by the New Zealand industry and applied to the Mid-East Coast orange roughy fishery, which is in a public FIP intended to improve the fishery to meet the MSC standard at some point in the future, once the key issue of low stock status has been addressed.[16] However, it is seen most strongly in the New Zealand fishing industry having achieved MSC certification for the three largest orange roughy fisheries (Trumble, Punt and Stern-Pirlot, 2016) – a journey from overfishing to sustainability using scientific innovation and the application of best practice in developing and applying fisheries management. For fisheries vilified for years by various eNGOs, this is a remarkable turnaround and is a demonstration that, with sufficient focus and investment, virtually any fishery, including the deepwater fisheries, can be managed sustainably. This is a very different route to that taken for the European deepwater fisheries, which have all been effectively closed (EU, 2016).

9.10 CHOICES IN FUTURE APPROACHES

The workshop that provided much of the material for this report was not tasked with trying to define what best practice should look like for orange roughy fisheries. In fact, best practice may need to differ somewhat between fisheries due to inherent factors of location, fish-stock carrying capacity or information quality. Instead, this report has drawn together what has been done in the different orange roughy fisheries around the world, and discussed what has worked or not worked well in terms of providing for long-term sustainable fisheries. From this, it is possible to identify some key areas where making the correct choice of approach is critical to delivering sustainable fisheries. Specifically these include the following.

Management of target fishery

- Ensure that the mechanism designed to limit fishing mortality, whether it be a TAC or an effort limit, is effective and that there is no or limited scope for unrecorded or unrecognized fishing mortality to occur. This includes the elimination of all forms of IUU fishing.
- Continue to develop and apply appropriate mathematical model frameworks to assess the stock (stocks) and provide uncertainty information.

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13 See: www.msc.org
14 See: www.sustainablefish.org
15 See: www.fisheryprogress.org
16 See: http://deepwatergroup.org

- Apply MSE techniques to assist in the selection of precautionary LRPs and TRPs (or TRP ranges), together with an HCR, that are robust to the principal uncertainties (e.g. stock structure, M, and stock-recruitment relationship).
- Ensure that all assessment procedures and management decisions are made in an open, transparent and publicly available manner to encourage critical evaluation and improvement over time.
- Determine stock structure on an appropriate spatial scale. Populations may be extensive, but a precautionary approach would advocate avoiding local depletion. If stock structure remains poorly known, assessments and management should consider stock definition at a scale sufficient to avoid unsustainable local depletion.
- Implement an efficient and effective biological data collection programme to provide input data for stock assessments (scientific observer programme, port/landings sampling programme). This should also cover collecting appropriate data from bycatch species (ages and other biological information), ETP species and benthic species.

Estimating target stock status

- Avoid using deterministic recruitment in stock assessments and projections unless the assumption can be fully justified. Instead, use age representative samples of the population (usually by the collection and reading of otoliths) with sufficient precision to estimate informative age-frequencies. In addition, age at maturity should be estimated from the age composition of spawning aggregations, not from transition zones on otoliths.
- To obtain indices of abundance for use in stock assessments, preference should be given to well-designed and well-executed fishery-independent surveys, and acoustic biomass surveys in particular.
- Acoustic biomass surveys should incorporate equipment calibration, multifrequency acoustic methods, and deep-towed acoustic equipment if UTFs or steep slopes are to be surveyed. Spatial and temporal coverage of these surveys should be carefully considered.
- Commercial or research CPUE should only be used as an index of abundance if potential biases can be accounted for, and if no suitable fishery-independent survey index is available, and should be given an appropriately high degree of uncertainty.

Management of interactions with the rest of the ecosystem

- Estimate bycatch by species. Develop effective monitoring programmes and indicators, and continue to monitor associated and dependent species for any changes (abundance, distribution, etc.).
- Appropriately monitor those species that the fishery may have a disproportionate impact on, such as rare ETP species or vulnerable fish such as elasmobranchs. This may, for example, be achieved through direct population monitoring or the application of population risk-assessment processes.
- Progressively move those bycatch species with the largest catches and/or the highest vulnerabilities to fishing into quantitative assessments.
- Implement a representative network of permanently closed areas for benthic/biodiversity protection as part of a benthic management strategy. This network should balance the needs of fishing and benthic conservation.
- Continue to explore methods to reduce bycatch through innovation by, for example, gear switching, gear modifications, spatial and temporal fishing restrictions, as well as implementing the use of *in situ* observation platforms, such as underwater, gear-mounted cameras, where practical.

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Appendix 1: List of workshop participants

Workshop on the Science in Support of Management of the Fisheries for Orange Roughy (*Hoplostethus atlanticus*), Auckland, New Zealand, 7–9 June 2016

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Appendix 2: Workshop agenda

AGENDA (Draft)

Workshop on the Science in Support of Management of the Fisheries for Orange Roughy (*Hoplostethus atlanticus*).

Rydges Hotel, Auckland, New Zealand
7th – 9th June 2016

Tuesday 7th June			
1	09:15 – 09:30	**Opening of the meeting** (i) Introductions (ii) Adoption of agenda	FAO / Chair (Tingley)
2	09:30 – 10:00	**Workshop objectives** (i) Links with ABNJ Deep Sea Project and FAO programme (ii) Discussion about objectives, work processes and outputs	FAO / Tingley
3	10:00 – 10:30	Coffee	
4	10:30 – 11:30	**Orange roughy fisheries** (i) Description of main high-seas and EEZ fisheries by region and country (e.g. historical catches, effort, fleet information, management and markets)	Participants – ~10–15 mins per fishery
5	11:30 – 12:30	**Biology of the species relevant to management (Part1)** (i) Age at recruitment to the fisheries (ii) Age at maturity (iii) Growth parameters (iv) Fecundity (v) Natural mortality	Participants – ~10–15 mins per fishery
6	12:30 – 13:30	Lunch	
7	13:30 – 14:30	**Biology of the species relevant to management (Part 2)** (vi) Population structure (vii) Predator-prey relationships (viii) Relationship with benthic environment (ix) Other factors	Participants – ~10–15 mins per fishery
8	14:30 – 15:30	**Data (requirements, availability, and reporting)** (i) Global level data (ii) Regional level data (iii) Fishery (fleet size and make-up, gear-types, seasons, footprint, catch and/or effort limits, etc.) (iv) Catches - quantities, species, size, age composition (v) Catch-effort relations (vi) Stock structure (vii) Estimation of recruitment (viii) Estimation of biomass/stock abundance	Participants – General discussion
9	15:30 – 16:00	Coffee	
10	16:00 – 16:30	**Discussion** (i) General discussion (ii) Housekeeping announcements (iii) Close of workshop for the day	

Wednesday 8th June			
1	09:00 – 10:00	**Stock structure (methods)** (i) DNA/genetics/clades (ii) Meristic analyses, including length (iii) Age (iv) Spawning period (v) Natural barriers (vi) Parasites (vii) Other methods (e.g. chemical composition)	Participants – ~10–15 mins per fishery
2	10:00 – 10:30	Coffee	
3	10:30 – 11:00	**Estimation of abundance** (i) Acoustic survey methods (ii) Trawl survey methods (iii) Combined acoustic-trawl survey methods (iv) Use of CPUE (v) Egg surveys (vi) Yield models (surplus production models, Y/R models, etc.) (vii) Summing removals (viii) Other methods	Cordue / Kloser
4	11:00 - 12:30	**Stock assessment approaches** (i) Review of what does and does not work (ii) Current approach in New Zealand	Cordue / Upston
5	12:30 – 13:30	Lunch	
6	13:30 – 15:00	**Ecosystem interactions of orange roughy fisheries** (i) Associated species (fish bycatch) (ii) Dependent species (predators and prey) (iii) Seabirds, marine mammals, and reptiles (iv) Benthic habitat, benthos and VMEs (v) Others	Participants – ~10–15 mins per fishery
7	15:00 – 15:30	Coffee	
8	15:30 – 17:15	**Limitations, challenges and opportunities in the management of orange roughy fisheries** (i) Data quality, availability and reporting (ii) Uncertainty (iii) Survey practicality (iv) Catch, effort and CPUE data (v) Stock assessments (vi) Management options – catch (vii) Management options – impacts (viii) Regional management issues (ix) Global management issues	General discussion
9	17:15	Close of workshop for the day	

Thursday 9th June			
1	09:00 – 10:00	**Social and economic issues in the orange roughy fisheries** (i) Socio-economic data availability and quality (ii) Social issues (iii) Economic issues (iv) What makes for a viable industry?	General discussion
2	10:00 – 10:30	Coffee	
3	10:30 – 11:00	**Market issues for orange roughy** (i) Current markets (ii) Future markets	Smith/Patchell
4	11:00 – 12:00	**Fit-for-purpose science** (i) Developing appropriate national science programmes to support future management orange roughy fisheries (ii) Developing appropriate international science programmes to support future management orange roughy fisheries	General discussion
5	12:00 – 13:00	Lunch	
6	13:00 – 15:00	**Summary and synthesis of orange roughy fisheries** (i) Open discussion – all subjects, anything overlooked (ii) Identification of overall conclusions (iii) Drafting of recommendations	General discussion
7	15:00 – 15:30	Coffee	
8	15:30 – 16:00	**Summing up** (i) Main conclusions (ii) Principal recommendations (iii) Next steps	Tingley
9	16:00 – 16:15	Workshop close	FAO / Tingley

This publication is intended to provide a range stakeholders and interested parties with an understanding of orange roughy fisheries around the world. The report covers historic aspects of the regional development of the fisheries, biology, stock assessment and key management issues. Recent developments in science and approaches to management are specifically highlighted with respect to the future for the sustainable management of deepwater, orange roughy fisheries.

There are a number of considered, published documents that discuss whether it is possible to have sustainable orange roughy fisheries (or other deepwater fisheries for long-lived species). These reviews draw on the common global experience of previous poor understanding about orange roughy productivity and the associated likelihood of overfishing, and the potential timescale for stock recovery, which led to 'boom and bust' orange roughy fisheries that frequently resulted in depleted stocks.

The more recent experience, with improved technology, better approaches to modelling population dynamics in orange roughy, and a more considered and robust approach to setting up the management framework (harvest strategy, management strategy evaluation, appropriately estimated limit and target reference points or ranges, and effective harvest control rules) provides a different paradigm. Essentially, assumptions about the unmanageability of these fisheries are flawed and that provided appropriate steps are taken to set and deliver a low and appropriate level of fishing mortality, orange roughy fisheries can be both managed and sustainable. The improved understanding of the productivity of orange roughy now provides a basis for better estimating yields and fishery value that are both more realistic and compatible with sustainably managed fisheries.

This review, contrasts these two perspectives and, whilst there is still considerable discussion and opposed view points, the message has clearly changed: sustainable orange roughy fisheries should be achievable. This review describes how, by making he right choices, this can be achieved.

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