Fisheries New Zealand

Tini a Tangaroa

Aquatic Environment and Biodiversity Annual Review 2021

A summary of environmental interactions between the seafood sector and the aquatic environment

Te Kāwanatanga o Aotearoa
New Zealand Government

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Cover Image – Mary Livingston. Sentinels of the ocean. Kelp and other seaweeds are important indicators of the health of coastal waters and have the potential to offset our carbon footprint if managed correctly.

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PREFACE

Aquatic Environment and Biodiversity Annual Reviews (AEBAR) have represented a significant annual output of Fisheries New Zealand and its predecessors since 2011. The AEBAR is now more than 700 pages long and provides detailed summaries of the fisheries and scientific information available to Fisheries New Zealand. It primarily addresses interactions between the seafood sector, biodiversity, and the aquatic environment. The AEBAR is supported by more detailed reports and science papers published on-line each year. The information contained within the AEBAR informs decision-making in Fisheries New Zealand.

The AEBAR provides an environment-based analogue of Fisheries New Zealand's annual Fisheries Assessment Plenary reports. It includes the most recent data and analyses on particular aquatic environment issues and, where appropriate, the current status against any specified targets or limits. Whereas the Fisheries Assessment Plenary reports are organised by fish stock, the AEBAR is organised by topic (e.g., protected species, bycatch, benthic impacts, etc.), and almost all topics involve more than one fish stock or fishery.

Fisheries New Zealand Science Working Groups contribute substantially to the AEBAR. These are primarily the Aquatic Environment Working Group (AEWG), the Antarctic Working Group (ANTWG), and the Biodiversity Research Advisory Group (BRAG). Over time, constructive relationships with industry, recreational, and environmental interests have been developed through the working groups and it has become expedient to finalise AEBAR chapters through the Science Working Group processes.

As more data are collected, more analyses are conducted, and more sophisticated models are developed, the Fisheries New Zealand peer review processes have become increasingly more rigorous. Research and science standards and the terms of reference for each working group have been developed to ensure a high quality and consistent peer review process. A wide variety of relevant, peer reviewed research that has not been contracted through Fisheries New Zealand is summarised in the AEBAR. This work is included only when it meets the Fisheries New Zealand Research and Science Information Standard.

Improvement of the AEBAR is ongoing and each chapter is reviewed annually and updated when new research results or data become available. An appendix summarising aquatic environment, Antarctic, and marine biodiversity research projects commissioned since 1998 is regularly updated for reference. The status of each chapter is clearly stated in the first row of the overview table for each chapter. Technical summary sheets are located at the start of each chapter.

The AEBAR 2021 has been led by the Science Group within the Directorate of Fisheries Science and Information in Fisheries New Zealand. It has also relied on the input of members from the AEWG, ANTWG, and BRAG working groups, as well as the Department of Conservation's Conservation Services Programme Technical Working Group (CSP-TWG) and other individuals who were commissioned to assist. I would like to recognise and thank all members of the Aquatic Environment Science Team (Mary Livingston, Ian Tuck, Marco Milardi, Ben Sharp, Karen Lisa Tunley, William Gibson, Josh van Lier, Suze Baird, Campbell Murray, Fabrice Stephenson and Jean Davis) for completing a major milestone for our team. I also thank the Aquaculture and the Fisheries Management Directorates at Fisheries New Zealand, and the large number of scientists from research organisations (in particular Di Tracey, Phil Sutton and Matt Pinkerton from NIWA for their contributions to the Climate Chapter), academia, the seafood industry, environmental NGOs, Māori customary teams at the Department of Conservation, and MPI, along with all other participants in numerous AEWG, ANTWG, BRAG, and CSP-TWG meetings for their active participation and contributions to this review.

I am pleased to endorse this document as representing the best scientific information on the environmental effects of fishing, marine biodiversity, and other environmental information available to Fisheries New Zealand as at May 2022.

Philip Heath

Manager Aquatic Environment Team, Fisheries Science and Information Directorate, Fisheries New Zealand

May 2022

CONTENTS

1 INTRODUCTION

Status of Chapter	This chapter has been updated for AEBAR 2021.

1.1 CONTEXT AND PURPOSE

Improving environmental performance and modernising fisheries management is at the core of extensive reforms that are currently underway in Fisheries New Zealand. These include changes to the Fisheries Act, moving towards electronic monitoring of commercial catch, and management measures to further protect New Zealand's protected species from the effects of fishing. Underpinning such reforms is the science required to inform management decisions about environmental matters, to identify risks and to optimise the use of new data streams that will be generated in the coming years. The Fisheries Science and Information Directorate is responsible for identifying and ensuring that the science needs for fisheries management both domestically and internationally are met.

When the Quota Management System (QMS) was introduced in 1986, most of the fisheries management related research focused on collecting the data required for stock assessment of key species in the QMS. In addition to the collection of catch records, scientific surveys to estimate fish abundance were conducted. Biological research to investigate the age structure of fish populations, their reproductive capacity, and natural mortality was also undertaken. Concerns about the effects of fishing on the environment, biodiversity, and climate change as risks to sustainability were not part of the dialogue at that time. Understanding the interactions between the aquatic environment and fishing activity has become increasingly important as New Zealand moves towards a more ecosystem based approach to fisheries management.

The Aquatic Environment and Biodiversity Annual Review (AEBAR) is a key document that is updated each year to provide transparency about research that has been commissioned by Fisheries New Zealand. The AEBAR complements Fisheries New Zealand's annual reports from Fisheries Assessment Plenaries. The Plenary documents report on the assessments of individual fish stocks, and the AEBAR reports on aquatic environment fisheries-related issues and biodiversity responsibilities. These often apply across many fish stocks, fisheries, or activities, and sometimes include the responsibilities of multiple agencies.

The AEBAR has been developed by the Aquatic Environment Fisheries Science Team. It is updated and drafted each year with assistance from working group members (primarily the Aquatic Environment Working Group, AEWG and the Biodiversity Research Advisory Group, BRAG) and selected research providers. As with the reports from Fisheries Assessment Plenaries, it has already grown substantially since its first publication in 2011 and is expected to grow and evolve further as new information becomes available and more issues are considered. Fisheries New Zealand aims to update as many chapters as possible each year. The need for an update is prioritised when new information becomes available through Fisheries New Zealand Science processes and for emerging topics. This year, significant updates have been made to sections within the Biodiversity, Seabirds, Antarctic, and Climate & Oceans chapters. Other chapters have been updated to a lesser extent. The status is provided at the start of each chapter.

The AEBAR provides a national overview on each environmental issue considered. For instance, the benthic (seabed) effects of mobile bottom-fishing methods are dealt with at the level of all such fisheries combined rather than at the level of a target fishery. The details of environmental issues for individual fisheries are documented in selected chapters in the May or November Report from the Fisheries Assessment Plenary (Fisheries New Zealand 2021a, Fisheries New Zealand 2021b).

The first part of this document describes the legislative and policy context for aquatic environment and biodiversity research commissioned by Fisheries New Zealand, and the science processes used to generate and review that research. The second, and main part of the document, contains chapters on key aquatic environment issues for fisheries management. Those chapters are under six broad themes: protected species, non-QMS bycatch (mostly fish), benthic effects, ecosystem issues (including New Zealand's oceanic setting), marine biodiversity, and aquaculture. The final part of the review includes appendices for reference.

1.2 LEGISLATION

The primary legislation for the management of fisheries, including the effects of fishing on the aquatic environment, is the Fisheries Act 1996 (Table 1.1). The main guidance to avoid, remedy, or mitigate any adverse effect of fishing on the aquatic environment is given in sections 8, 9, and 15, although sections 10, 11, and 13 are also relevant to decision-making under this Act (Table 1.2). Fisheries New Zealand also administers a range of other acts on behalf of the Ministry for Primary Industries (MPI) and there are some acts administered by other agencies (Table 1.1) that lead to a requirement for Fisheries New Zealand to work with other government departments (especially Department of Conservation (DOC), the Ministry for the Environment (MfE), the Natural Resources Sector[1]), and with various territorial authorities (especially regional councils) to a much greater extent than is required for most fisheries stock assessments.

Various layers of Regulations and Orders in Council (see http://www.legislation.govt.nz/) exist under the primary legislation. It is beyond the scope of this document to summarise these.

In addition to its domestic legislation, the New Zealand government is a signatory to a wide variety of International Instruments and Agreements that bring with them various International Obligations (Table 1.3). Section 5 of the Fisheries Act requires that the Act be interpreted in a manner that is consistent with international obligations and with the Treaty of Waitangi (Fisheries Claims) Settlement Act 1992.

1.3 POLICY SETTING

1.3.1 STRATEGIC INTENTIONS AND OUR STRATEGY

Fisheries New Zealand is the principal adviser to the Government on fisheries and aquaculture. This is a business unit located in the Ministry for Primary Industries that also includes other business units which together have responsibilities for fisheries, agriculture, horticulture, forestry, food safety, animal welfare, and the protection of New Zealand's primary industries from biological risk, i.e., biosecurity. MPI's Strategic Intentions (formerly called Statement of Intent, SOI) document is an important guiding document for the short to medium term. This document is available on the MPI website at: https://www.mpi.govt.nz/dmsdocument/31056/direct.

Table 1.1: New Zealand Acts of environmental relevance to Fisheries New Zealand. * Denotes the Act of primary importance for the management of fisheries.

Acts that Fisheries New Zealand administers	Acts requiring Fisheries New Zealand to work with others
Fisheries Act 1996*	Wildlife Act 1953
Fisheries Act 1983 (residual parts)	Marine Mammals Protection Act 1978
Treaty of Waitangi (Fisheries Claims) Settlement Act 1992	Marine Reserves Act 1971
Fisheries (Quota Operations Validation) Act 1997	Conservation Act 1987
Maori Fisheries Act 2004	Hauraki Gulf Marine Park Act 2000
Maori Commercial Aquaculture Claims Settlement Act 2004	Resource Management Act 1991
Aquaculture Reform (Repeals and Transitional Provisions) Act 2004	Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012
Driftnet Prohibition Act 1991	Environmental Reporting Act 2015
Antarctic Marine Living Resources Act 1981	

Table 1.2: Sections of the Fisheries Act 1996 relevant to the management of the effects of fishing on the aquatic environment and biodiversity.

Fisheries Act 1996

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[1] The Natural Resources Sector is a network of government agencies established to enhance collaboration. Its main purpose is to ensure a strategic, integrated, and aligned approach is taken to natural resources development and management across government agencies. The network is chaired by the Chief Executive of the Ministry for the Environment.

s8 Purpose –
(1) The purpose of this Act is to provide for the utilisation of fisheries resources while ensuring sustainability, where
(2) "Ensuring sustainability" means –
(a) Maintaining the potential of fisheries resources to meet the reasonably foreseeable needs of future generations, and
(b) Avoiding, remedying, or mitigating any adverse effects of fishing on the aquatic environment.
"Utilisation" means conserving, using, enhancing, and developing fisheries resources to enable people to provide for their social, economic, and cultural well-being.

s9 Environmental Principles
Associated or dependent species should be maintained above a level that ensures their long-term viability,
biological diversity of the aquatic environment should be maintained,
habitat of particular significance for fisheries management should be protected.

s10 Information Principles
All persons exercising or performing functions, duties, or powers under this Act, in relation to the utilisation of fisheries resources or ensuring sustainability, shall take into account the following information principles:

a. decisions should be based on the best available information,

b. decision makers should consider any uncertainty in the information available in any case,

c. decision makers should be cautious when information is uncertain, unreliable, or inadequate,

d. in the absence of, or any uncertainty in, -information should not be used as a reason for postponing or failing to take any measure to achieve the purpose of this Act.

s11 Sustainability Measures. The Minister may take into account, in setting any sustainability measure, (a) any effects of fishing on any stock and the aquatic environment.

S13, 2b Total Allowable Catch. The Minister may set a TAC that enables the level of any stock whose current level is below that which can produce the maximum sustainable yield to be altered within a period appropriate to the stock, having regard to the biological characteristics of the stock and any environmental conditions affecting the stock;

S13, 2A b Total Allowable Catch. For the purposes of setting a total allowable catch under this section, if the Minister considers that the current level of the stock or the level of the stock that can produce the maximum sustainable yield is not able to be estimated reliably using the best available information, the Minister must have regard to the interdependence of stocks, the biological characteristics of the stock, and any environmental conditions affecting the stock;

s15 Fishing-related mortality of marine mammals or other wildlife. A range of management considerations are set out in the Fisheries Act 1996, which empower the Minister to take measures to avoid, remedy, or mitigate any adverse effects of fishing on associated or dependent species and any effect of fishing-related mortality on any protected species. These measures include the setting of catch limits or the prohibition of fishing methods or all fishing in an area, to ensure that such catch limits are not exceeded.

Fisheries New Zealand's broad approach was updated in 2017 with a refresh of *Our Strategy 2030*. The new strategy was called *Our Strategy* (Figure 1.1) and is available on the Ministry's website at: http://www.mpi.govt.nz/about-mpi/our-strategy/. The Ministry's purpose is unchanged in *Our Strategy* as "growing and protecting New Zealand" but a new ambition is defined as "*New Zealand is the most trusted source of high value natural products in the world*". Four key outcomes are also outlined:

- **Growth**: New Zealand's food and primary sector grows the value of its exports;
- **Sustainability**: New Zealand's natural resources are sustainable, in the primary sector;
- **Protection**: New Zealand is protected from biological risk and our products are safe for all consumers; and
- **Participation**: New Zealanders participate in the success of the primary industries.

To provide relevant information to fulfil these roles in terms of interaction with the environment, Fisheries New Zealand commissions the following types of research:

- **aquatic environment research** to assess the effects of fishing and aquaculture on marine habitats, protected species, non-target species of fish and other species caught by fishing, and to understand habitats of special significance for fisheries;

- **marine biodiversity** research to increase our understanding of the systems that support resilient ecosystems, productive fisheries, and aquaculture, including their trophic linkages and the effects of climate change. There is increasing focus on ecosystem-scale studies that develop the information needed for Ecosystem Based Fisheries Management.

Table 1.3: International agreements and regional agreements to which New Zealand is a signatory, that are relevant to the management of the effects of fishing on the aquatic environment.

International Instruments	Regional Fisheries Agreements
• **Convention on the Conservation of Migratory Species of Wild Animals (CMS)** Aims to conserve terrestrial, marine, and avian migratory species throughout their range. • **Agreement on the Conservation of Albatrosses and Petrels (ACAP)** Aims to introduce a number of conservation measures to reduce the threat of extinction to the Albatross and Petrel species. • **Convention on Biological Diversity (CBD)** Provides for conservation of biological diversity and sustainable use of components. States accorded the right to exploit resources pursuant to environmental policies. • **United Nations Convention on the Law of the Sea (UNCLOS)** Acknowledges the right to explore and exploit, conserve, and manage natural resources in the State's EEZ…with regard to the protection and preservation of the marine environment including associated and dependent species, pursuant to the State's environmental policies. • **Convention on the International Trade in Endangered Species of Wild Fauna and Flora (CITES)** Aims to ensure that international trade in wild animals and plants does not threaten their survival. • **United Nations Fishstocks Agreements** Aims to lay down a comprehensive regime for the conservation and management of straddling and highly migratory fish stocks. • **International Whaling Commission (IWC)** Aims to provide for the proper conservation of whale stocks and thus make possible the orderly development of the whaling industry. • **Wellington Convention** Aims to prohibit drift net fishing activity in the convention area. • **Food and Agriculture Organisation – International Plan of Action for Seabirds (FAO-IPOA Seabirds)** Voluntary framework for reducing the incidental catch of seabirds in longline fisheries. • **Food and Agriculture Organisation – International Plan of Action for Sharks (FAO –IPOA Sharks)** Voluntary framework for the conservation and management of sharks. • **Noumea Convention** Promotes protection and management of natural resources. Parties to regulate or prohibit activity likely to have adverse effects on species, ecosystems, and biological processes. • **Food and Agriculture Organisation - Code of Conduct for Responsible Fisheries** Provides principles and standards applicable to the conservation, management, and development of all fisheries, to be interpreted and applied to conform to the rights, jurisdiction, and duties of States contained in UNCLOS.	• **Convention for the Conservation of Southern Bluefin Tuna (CCSBT)** Aims to ensure, through appropriate management, the conservation and optimum utilisation of the global Southern Bluefin Tuna fishery. The Convention specifically provides for the exchange of data on ecologically related species to aid in the conservation of these species when fishing for southern bluefin tuna. • **Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR)** Aims to conserve, including rational use of Antarctic marine living resources. This includes supporting research to understand the effects of CCAMLR fishing on associated and dependent species, and monitoring levels of incidental take of these species on New Zealand vessels fishing in CCAMLR waters. • **Convention on the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPFC)** The objective is to ensure, through effective management, the long-term conservation and sustainable use of highly migratory fish stocks in accordance with UNCLOS. • **South Tasman Rise Orange Roughy Arrangement** The arrangement puts in place the requirement for New Zealand and Australian fishers to have approval from the appropriate authorities to trawl or carry out other demersal fishing for any species in the STR area • **Convention on the Conservation and Management of High Seas Fishery Resources in the South Pacific Ocean** (a Regional Fisheries Management Organisation, colloquially **SPRFMO**) has been negotiated to facilitate management of non-highly migratory species in the South Pacific. • **Te Mana o te Taiao – Aotearoa New Zealand Biodiversity Strategy 2020 and the Aichi Agreements** In August 2020 New Zealand launched a refreshed biodiversity strategy for the protection, restoration, and sustainable use of biodiversity, which has goals and targets relevant to fisheries management.

1.3.2 FISHERIES PLANS

Fisheries planning processes for deepwater and middle-depth species, highly migratory species, inshore finfish, inshore shellfish, and freshwater fisheries use objective-based management to drive the delivery of services. The planning processes are guided by a series of National Fisheries Plans, which recognise the distinctive characteristics of these fisheries. The first National Plans for deepwater and middle-depth fisheries and highly migratory species fisheries were approved by the Minister in September 2010 and a suite of three draft plans for inshore species was released in 2020. Fisheries New Zealand is currently reviewing the plans and is, or will be, consulting on such reviews. Fisheries plans establish management objectives for each fishery, including those related to the environmental effects of fishing. All are available on the Fisheries New Zealand's website together with a wide variety of other information on the management of these fisheries.

Deepwater and middle-depth fisheries:
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/deepwater-fisheries/

Highly migratory species (HMS) fisheries:
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/highly-migratory-species/

Inshore fisheries (comprising finfish, shellfish, and freshwater fisheries):
http://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/inshore-fisheries/

Antarctic and other international (high seas) fisheries are not covered by fisheries plans but, rather, by the plans and strategies of the respective international organisations (CCAMLR, SPRFMO, WCPFC, CCSBT, etc.).

Figure 1.1. The four outcomes of the Ministry for Primary Industries Strategy, released in 2019.

1.3.3 OTHER STRATEGIC DOCUMENTS

A number of strategies or reviews have been published that interface with aquaculture and fisheries values and research requirements. These include: the Aquaculture Strategy; Biosecurity Strategy (2003, followed by its science strategy 2007 and more recently Biosecurity 2025); the Marine Protected Area Policy and Implementation Plan (2005); MfE's discussion paper on Management of Activities in the EEZ (2007, now translated to the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act 2012); Fisheries 2030 (2009); MfE's Roadmap for Environment Science (2016); the Revised Coastal Policy Statement (2010); the National Plan of Action to Reduce the Incidental Catch of Seabirds in New Zealand Fisheries (2004, revised and updated in 2013 and 2020); the New Zealand Sea Lion Threat Management Plan (2017 https://www.doc.govt.nz/globalassets/documents/conservation/native-animals/marine-mammals/nz-sea-lion-tmp/nz-sea-lion-threat-management-plan.pdf); the

Hector's and Māui Dolphin Threat Management Plan Review (https://www.mpi.govt.nz/dmsdocument/34971); and the New Zealand National Plan of Action for the Conservation and Management of Sharks (2013); MfE and Stats New Zealand Environmental Reporting Act 2015, Our Marine Environment 2019 (https://www.mfe.govt.nz/publications/environmental-reporting/our-marine-environment-2019); New Zealand's Biodiversity Action Plan 2016; and Te Mana o te Taiao - Aotearoa New Zealand Biodiversity Strategy 2020.

In 2012, the Natural Resources Sector https://www.mfe.govt.nz/about-us/who-we-work/overview-who-we-work formed a Marine Directors' Group to improve data sharing and information exchange across key agencies with marine environmental responsibilities, particularly Fisheries New Zealand, DOC, MfE, Environment Protection Agency, Land Information New Zealand, Ministry of Business, Innovation, and Employment, and Stats NZ. Another initiative between business and government to address environmental issues in New Zealand is the Aotearoa Circle https://www.theaotearoacircle.nz/.

1.4 SCIENCE PROCESSES

1.4.1 RESEARCH PLANNING

Fisheries New Zealand has adopted an approach of specifying management objectives for fisheries in Fisheries Plans and using these to develop implementation strategies and required services, including research. Services specific to a fisheries plan are identified in Annual Operational Plans that are updated each year (available via the links in section 1.3.2). Alongside this process, and in close consultation with fisheries managers and the Department of Conservation, Fisheries New Zealand also develops a portfolio of research on aquatic environment issues related to fisheries. This portfolio is designed to meet information needs that span multiple fisheries (e.g., incidental captures of seabirds across multiple fisheries in multiple areas) as well as the specified needs of individual fisheries plans. Also included in the Aquatic Environment portfolio is work aimed at addressing emerging issues (e.g., effects of climate change on fisheries, ecosystem based fisheries management, aquaculture) and emergencies (e.g., Kaikōura earthquake). Marine biodiversity research has a much broader and more strategic focus, and planning of such research is conducted through the Biodiversity Research Advisory Group (BRAG) chaired by Fisheries New Zealand in consultation with MBIE research programmes, science challenges, local government initiatives, and DOC. Fisheries New Zealand will produce a Medium Term Research plan for Aquatic Environment research in 2022. The plan will set the research direction for the next 5 years with a focus on science that helps to deliver Ecosystem Based Fisheries Management.

1.4.2 RESEARCH REVIEW AND CONTRIBUTING WORKING GROUPS

Any research that is intended or likely to inform fisheries management decision-making must be reviewed against the requirements of the Research and Science Information Standard for New Zealand Fisheries (RSIS, 2011) (https://www.mpi.govt.nz/dmsdocument/3692-research-and-science-information-standard-for-new-zealand-fisheries).

The main contributing working groups for this document are Fisheries New Zealand's Aquatic Environment Working Group (AEWG), the Antarctic Working Group (ANTWG), and the Biodiversity Research Advisory Group (BRAG). The Department of Conservation's Conservation Services Programme Technical Working Group (CSP-TWG, see http://www.doc.govt.nz/our-work/conservation-services-programme/meetings-and-project-updates/) also considers a wide range of DOC-funded projects related to protected species, sometimes in joint meetings with the AEWG. Fisheries New Zealand Fishery Assessment Working Groups occasionally consider research relevant to this review where there is particular relevance to a fishery.

Terms of reference for Fisheries New Zealand working groups are periodically revised and updated (see Appendices 19.1–19.7 for those working groups relevant to this document).

The AEWG is convened for Fisheries New Zealand peer review purposes with an overall purpose of assessing, based on scientific information, the effects of fishing, aquaculture, and enhancement on the aquatic environment for all New Zealand fisheries. The purview of the AEWG includes: bycatch and unobserved mortality of protected species, fish, and other marine life; effects of bottom fisheries on benthic biodiversity, species, and habitat; effects of fishing on biodiversity, including genetic diversity; changes to ecosystem structure and function as a result of fishing, including trophic effects; and effects of aquaculture and fishery enhancement on the environment and on fishing. Where possible, the AEWG may explore the implications of any effects, including with respect to any standards, reference points, and relevant indicators. The AEWG is a technical forum to assess the effects of fishing or environmental status and make projections. It has no mandate to make management recommendations or decisions. Membership of the AEWG is open (and current participants are listed in Appendix 19.2).

The ANTWG is convened with an overall purpose assess the stock status and the effects of fishing for Antarctic fisheries. The purview of the ANTWG includes: stock status of target species, bycatch and unobserved mortality of protected species, fish, and other marine life; effects on biodiversity and benthic biodiversity, species, and habitat; and changes to ecosystem structure and function as a result of fishing, and including trophic effects. The ANTWG also provides peer review of documents and papers submitted to the scientific working groups of CCAMLR to aid and inform its management. The ANTWG is a technical forum to assess the stock status, effects of fishing or environmental status, and make projections. It has no mandate to make management recommendations or decisions. Membership of ANTWG is open (and current participants are listed in Appendix 19.2).

The two main responsibilities of the BRAG are: to review, discuss, and convey views on the results of marine biodiversity research projects contracted by the Fisheries New Zealand; and to discuss, evaluate, and make recommendations on annual fisheries research plans and individual projects. Both tasks have evolved from the strategic goals in both the New Zealand Biodiversity Strategy (2000) and the Strategy for New Zealand Science in Antarctica and the Southern Ocean (2010). More recently, the programme has become aligned to research on emerging issues such as climate change and ocean acidification. The BRAG has provided advice and oversight of some large cross-government survey projects such as NORFANZ, BIOROSS, Fisheries and Biodiversity Ocean Survey 20/20; and International Polar Year (IPY) Census of Antarctic Marine Life (IPY-CAML). Membership of the BRAG is open and current participants are listed in Appendix 19.2.

Following consideration at one or more meetings of appropriate working groups, final reports from individual projects are also technically reviewed by Fisheries New Zealand before they are finalised for use in management and/or for public release. Fisheries Assessment Reports (FARs) and Aquatic Environment and Biodiversity Reports (AEBRs) are also subject to editorial review whereas Final Research Reports (FRRs) and Research Progress Reports (RPRs) are not. Finalised FARs, AEBRs, historical FARDs (discontinued Fisheries Assessment Research Documents), and MBBRs (discontinued Marine Biodiversity and Biosecurity Reports), and some FRRs are in the Document library at: http://fs.fish.govt.nz/Page.aspx?pk=61&tk=297. More recent reports are available from the MPI website at: http://www.mpi.govt.nz/news-and-resources/publications/.

1.4.3 REVIEW OF RESEARCH NOT FUNDED BY FISHERIES NEW ZEALAND

Almost all research of direct relevance to management of fish stocks is commissioned by Fisheries New Zealand and reviewed through Fisheries New Zealand fishery assessment working groups. This is a structured approach to meet the requirements of the RSIS. However, research on various aspects of the environmental effects of fishing is also commissioned by a range of external organisations and is commonly published in science journals. It is not always clear that the requirements of the RSIS have been met in these cases. Fisheries New Zealand working groups, including the AEWG and BRAG, can provide an excellent and well-informed forum to discuss such research, and researchers are encouraged to bring their work on the environmental effects of fishing to this forum for review and assessment against the requirements of the RSIS. This is particularly important if researchers wish their work to be used to inform Fisheries Management processes. Whether or not a working group has considered them, reports or journal papers that are intended or likely to inform fisheries management decision-making are technically reviewed by the Fisheries New Zealand's fisheries science team before they can be used.

1.5 REFERENCES

Fisheries New Zealand (2021a) Fisheries Assessment Plenary, May 2021: stock assessments and stock status. Compiled by the Fisheries Science Team, Fisheries New Zealand, Wellington, New Zealand. 1782 p.

Fisheries New Zealand (2021b) Fisheries Assessment Plenary, November 2021: stock assessments and stock status. Compiled by the Fisheries Science and Information Group, Fisheries New Zealand, Wellington, New Zealand. 663 p.

Ministry of Fisheries (2011) Research and Science Information Standard for New Zealand Fisheries. Ministry of Fisheries, Wellington, New Zealand. 31 p.

2 RESEARCH THEMES COVERED IN THIS DOCUMENT

Status of chapter	This chapter has not been updated for AEBAR 2021.

Fisheries New Zealand commisions five broad themes of research that reflect our legislative responsibilities and obligations outlined in Chapter 1:

Theme 1. Protected species

Theme 2. Non-target fish and invertebrate catch

Theme 3. Benthic impacts

Theme 4. Ecosystem effects

Theme 5. Marine Biodiversity

Each theme has several chapters that provides the latest science available to Fisheries New Zealand Working Groups on topics relating to the effects of fishing on the aquatic environment, ecosystem scale effects and marine biodiversity. Where topics overlap, crosslinks between chapters are provided.

1. PROTECTED SPECIES

- **CH. 3. SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT:** Describes the methodology developed for advanced spatially explicit risk assessment.
- **CH.4. NEW ZEALAND SEA LION (PHOCARCTOS HOOKERI)**: Describes the scientific information that underpins the New Zealand sea lion Threat Management Plan 2017.
- **CH.5. NEW ZEALAND FUR SEAL (ARCTOCEPHLAUS FOSTERI):** Describes the current state of knowledge about fur seals in New Zealand waters.
- **CH.6. HECTOR'S DOLPHIN (CEPHALORHYNCHUS HECTORI HECTORI) AND MĀUI DOLPHIN (C. H. MAUI):** Describes the scientific information that underpins the Hector's and Maui Dolphins Threat Management Plan 2020.
- **CH. 7. NEW ZEALAND COMMON DOLPHIN (DELPHINUS DELPHIS DELPHIS):** Describes the current state of knowledge about common dolphins in New Zealand waters
- **CH.8. NEW ZEALAND SEABIRDS:** Provides the most recent risk assessment results and demographic studies for seabirds affected by fishing in New Zealand.

2. NON-TARGET FISH AND INVERTEBRATE CATCH

- **CH. 9. NON-TARGET FISH AND INVERTEBRATE CATCH**: Provides estimates of incidental capture of fish and invertebrates (bycatch) and discard rates.
- **CH.10. CHONDRICHTHYANS (SHARKS, RAYS AND CHIMAERAS):** Describes the target and non-target catch of cartilaginous fish. Including protected shark species.

3. BENTHIC IMPACTS

- **CH. 11. BENTHIC (SEABED) IMPACTS:** Describes the annual trawl footprint of fishing, the impacts of fishing on the seabed, seabed habitat classification and seabed recovery projects.

4. ECOSYSTEM EFFECTS

- **CH. 12. NEW ZEALAND'S CLIMATE AND OCEANIC SETTING:** Describes oceanic circulation and productivity of the New Zealand region; regime shifts and environmental trends.
- **CH.13. TROPHIC AND ECOSYSTEM-LEVEL EFFECTS:** The potential effects of fishing on marine foodwebs are described.
- **CH. 14. HABITATS OF PARTICULAR SIGNIFICANCE FOR FISHERIES MANAGEMENT**: Highlights habitats that could be considered of

particular significance to fisheries management (e.g., spawning grounds; nursery grounds).

- **CH. 15. LAND-BASED EFFECTS ON FISHERIES**: Describes the main known threats of land based activities on marine ecosystems (eg., nutrient levels, sedimentation).

- **CH.16. ECOLOGICAL EFFECTS OF MARINE AQUACULTURE:** Describes known and potential ecological effects of current marine aquaculture operations in New Zealand.

- **CH.17. ANTARCTIC SCIENCE RESEARCH:** Describes research that underpins New Zealand engagement in the management of the Ross Sea region and the Southern Ocean.

5. MARINE BIODIVERSITY

- **CH. 18. MARINE BIODIVERSITY**: This chapter reports on research that has been driven largely by the New Zealand Biodiversity Strategy (2000). In recent years there has been a focus on the effects of climate change on the seafood sector.

Research progress is not uniform across these themes; for example, our knowledge of the quantum and consequences of fishing-related mortality on protected species is much better developed than our knowledge of the consequences of catching non-target fish, bottom trawl impacts, or land management choices for ecosystem processes or fisheries productivity.

Ultimately, the goal of research described in AEBAR complements information on fishstocks to ensure that the Ministry has the information required to underpin progress towards an ecosystem approach to fisheries management. Stock assessment results have been published for many years in Fisheries Assessment Reports, Final Research Reports, and the Annual Report from the Fishery Assessment Plenary ('the plenary'). Collectively, these provide a rich and well-understood resource for fisheries managers and stakeholders. In 2005, an environmental section was first included in the hoki plenary report as part of the characterisation of that fishery and to highlight any particular environmental issues. Similar, fishery-specific sections have since been developed for several other fisheries and included in the plenary, but work on environmental issues has otherwise been more difficult to access for fisheries managers and stakeholders. Fisheries New Zealand continues to explore better ways to document, review, publicise, and integrate information from environmental assessments with traditional fishery assessments, including annual publication of this document. The AEBAR relies heavily on studies that are published in Aquatic Environment and Biodiversity Reports and Final Research Reports but, given the overlapping mandates and broader scope of work in this area, results published by other organisations and in the scientific literature are also important. The integration of all this work into a single source document analogous to the Report from the Fishery Assessment Plenary has advanced considerably since the first edition in 2011 but it will take time for all issues to be included.

THEME 1: PROTECTED SPECIES

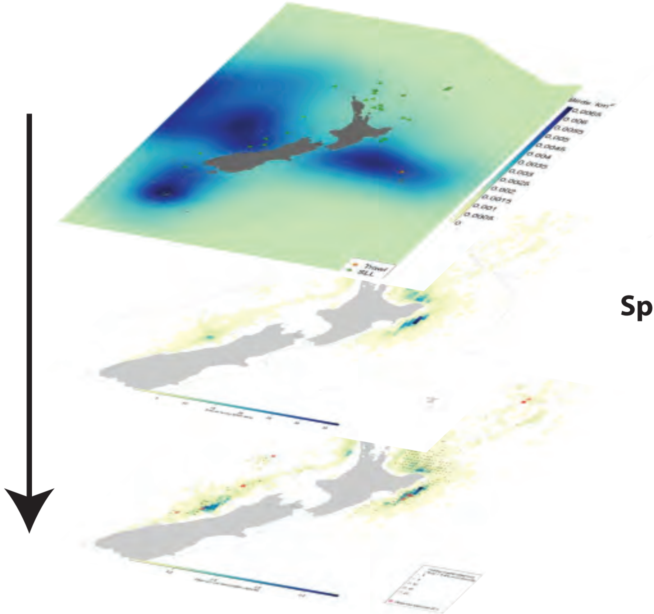
Chapter 3: *Snapshot* of Chapter 3 - Spatially Explicit Fisheries Risk Assessment (SEFRA)

1. THE METHOD IN BRIEF

- SEFRA is a method to estimate the risk to protected marine species posed by fishing (or other threats)
- Risk is represented as a ratio between the estimated mortality of a species across all fisheries and an estimate of mortality that the species population can withstand while achieving a desired population outcome (generally a stable or increasing population)
- This method is particularly useful in instances where data on observed fisheries mortality is very low (e.g., rare species or low observer coverage), and thus cannot be used directly in risk assessment
- Detailed inputs can allow development of robust and tailored management options
- Uncertainty in model outputs can be evaluated and tracked (see 5.)
- The method can be used to estimate risks from multiple threats simultaneously (e.g., fishing, diseases, etc.)

2. ESTIMATING INTERACTIONS

- The SEFRA method uses the spatial distribution and abundance of a species, combined with the distribution and intensity of fishing or other threats, as a way to estimate their overlap
- Where there is no overlap, there is no interaction. In overlap areas, the method can model the interactions (e.g., captures or warp strikes) between species and fishing gear (see 3.)



Spatial distribution and abundance of the species

+

Spatial distribution and intensity of fishing/other threat

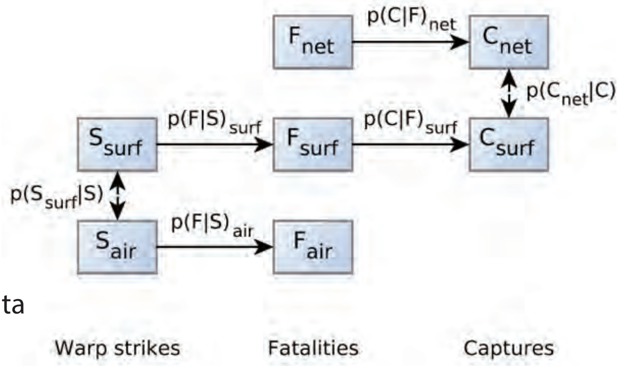
=

Overlap (areas where interactions can occur)

3. ESTIMATING CAPTURES

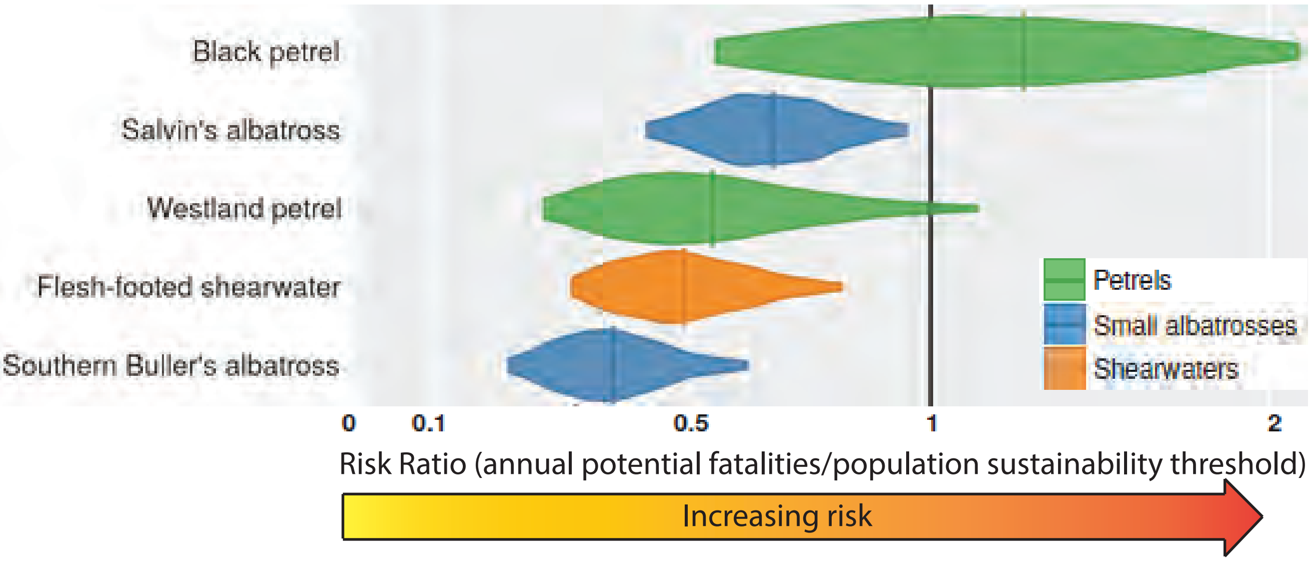
- The probability that an interaction will result in a capture or death, termed 'vulnerability', is estimated using fisheries observer data. Vulnerability is estimated separately for different types of fishing gear, and may be adjusted by mitigation measures (e.g., tori lines)

- Fatal interactions that cannot be observed, i.e., 'cryptic mortality', are estimated by species/gear, from available data or expert judgment

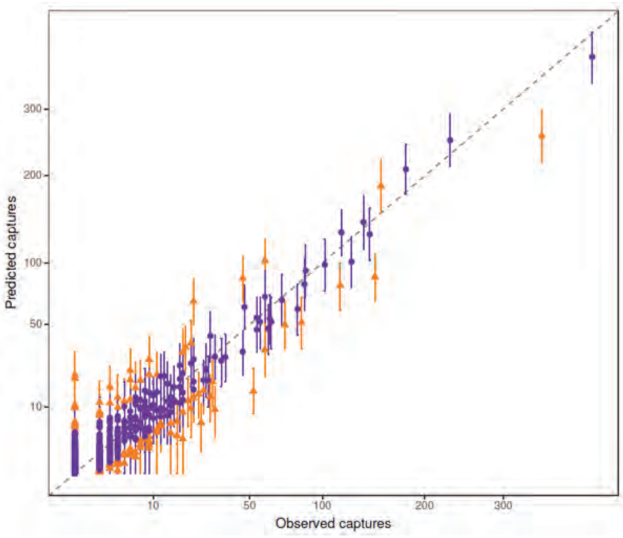


4. ESTIMATING POPULATION RISK

- SEFRA compares the fatalities with a 'Population Sustainability Threshold' (PST), to see whether the total number of deaths pose a conservation risk to the population or not
- Population size, demography, growth rate, and natural mortality are used to define how much mortality the population can withstand while still achieving a defined population outcome
- Population outcomes reflect policy targets and may vary between species
- If the mean estimated mortality from an activity/threat is higher than the population ability to reproduce and increase, the SEFRA method assigns a very high risk to that interaction (risk ratio > 1)
- Risk values above 1 indicate that the defined population outcome will not be achieved



5. UNCERTAINTY AND EVALUATION



- Uncertainty derives from poorly-understood biological inputs, low observer coverage, or poor fits between observed and predicted capture rates

- Uncertainty can be traced through the modeling process

- Model outputs are carefully assessed using observed values, before being accepted or used to inform management options

Figure shows the comparison of the number of seabird captures recorded by observers and predicted by the model used to estimate the number of annual potential fatalities (2006–17)

6. APPLICATIONS

- The SEFRA methodology has been applied to e.g., NZ sea lion (see Chapter 4), Hector's and Māui dolphin (see Chapter 6), and seabirds (see Chapter 8), risk assessments
- Similar methods could be applied to assess low-information fish stocks or the risk to sharks

3 SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT (SEFRA)

Status of chapter	This chapter has not been updated for AEBAR 2021.
Scope of chapter	This chapter describes New Zealand's Spatially Explicit Fisheries Risk Assessment method, which has been designed to estimate fisheries impact and risk for non-target species, and to inform risk management responses within a quantitative and statistically rigorous framework. The chapter includes: i) a description of the conceptual and specific mathematical application of this method to New Zealand seabird and marine mammal species; ii) a description of required data inputs and potential pitfalls in the application of this specific method; and iii) a more general discussion of other planned or in progress applications of the SEFRA framework, e.g., applied to non-target fish or benthic invertebrates, for which the conceptual approach is the same but modified methods will be developed in the implementation stage.
Area	The SEFRA method can be applied at any spatial scale at which spatial data representing species distributions and fishing effort distributions are available. The most fully developed implementations, for New Zealand seabirds and marine mammals, have been applied at the scale of the New Zealand EEZ.
Focal localities	Outputs from each implementation of the SEFRA method will identify different key locations at which fisheries risk occurs, based on the spatial overlap between species distributions and the fishing effort to which that species is most vulnerable.
Key issues	To assess and manage fisheries risks across large numbers of potentially affected non-target populations, fisheries managers are forced to make difficult decisions in the context of poor and/or sparse information. Innovative methods are required to enable maximum use of available data in a transparent and statistically rigorous framework. Application of the SEFRA method to the New Zealand Seabird Risk Assessment (NZSRA) has been iteratively improved since the initial design of the method in 2009. The updated NZSRA will constitute a 'full' implementation of the method as designed, providing a useful methodological template for other risk assessments.
Emerging issues	The first application of the SEFRA method to the New Zealand Marine Mammal Risk Assessment (NZMMRA) is now complete (Abraham et al. 2017), closely following the method template provided by the NZSRA. Modified applications of the method are in development (for individual protected species, global seabirds, non-target fish, and pelagic protected fish) or planned as future work.
MPI research (current)	The current New Zealand Seabird Risk Assessment (NZSRA) is delivered under contract PRO-2014-06. The current New Zealand Marine Mammal Risk Assessment (NZMMRA) is delivered under contract PRO-2012-02. A customised user-driven query and simulation tool to inform risk management is in development under contract PRO-2016-06. Cetacean spatial distribution modelling to inform an improved MMRA is delivered under contract PRO-2014-01. A global seabird risk assessment is in progress under contract PRO-2013-13. SEFRA implementations for particular mammal or bird species are in progress under SEA2016-30, PRO2017-12, and PRO2017-10, and PRO2017-19.
NZ government research (current)	Risk assessment outputs are routinely used to inform the prioritisation of biological and population monitoring research under the DOC Conservation Services Programme (CSP) and MPI protected species programme research, to focus research efforts on populations or variables for which uncertain parameter inputs have significant effects on risk estimation for species of interest.
Related chapters/issues	Results of the NZMMRA are summarised in species-specific marine mammal chapters for NZ sea lions, New Zealand fur seals, Hector's and Māui dolphins, and common dolphins (i.e., Chapters 4–7) Results of the NZSRA are included in New Zealand seabirds, Chapter 8. Future implementations of the SEFRA framework may inform updates of these chapters and/or Chapters 9–11.

3.1 CONTEXT

3.1.1 SCOPE

The scope of the Spatially Explicit Fisheries Risk Assessment framework (hereafter SEFRA) is *to assess the population-level risk to non-target species arising from direct incidental mortality in commercial fisheries.* The SEFRA framework combines an impact assessment to estimate the level of incidental fisheries mortality with a biological assessment of the associated effect on the population, as a function of population size and demographic parameters influencing population productivity. The SEFRA framework does not address potential indirect fisheries effects, e.g., trophic effects.

This paper outlines the conceptual and mathematical basis for the application of the SEFRA framework to estimate fisheries risk to seabirds and marine mammals, for which the method is nearly identical. Other applications of the framework, e.g., applied to non-target fish or benthic invertebrates, are in progress but will require modifications to the mathematical framework described below. These will be described separately.

3.1.2 BACKGROUND

The SEFRA framework was developed initially with specific reference to commercial fisheries impacts on New Zealand seabirds. The scope and nature of the SEFRA framework was designed to address the specific information needs of fisheries managers charged with managing seabird impacts by New Zealand fisheries, and with reference to the level and quality of available data in New Zealand to inform the risk assessment process. Risk assessments that carefully consider management needs and data limitations in the design stage are likely to be more effective than generic templates applied universally for different kinds of threats and for a wide range of management applications (such as the templates described by Hobday et al. 2007).

The specific New Zealand seabird context is as follows:

- At a global scale New Zealand has a disproportionately high number of resident or breeding seabird populations. For many of these species, reliable demographic or population data are unavailable, and are not feasible to obtain, for example due to remote colony locations.

- New Zealand seabirds are exposed to risk from a wide variety of fishing methods. The quality and availability of fisheries observer data useful to estimate incidental capture rates varies greatly, from relatively well-observed deepwater fisheries (30–50% of fishing events observed) to very poorly observed primarily inshore fisheries (often less than 1% of fishing events observed).

- Fisheries observer coverage is variable, and what data is available is almost always spatially unrepresentative of the whole, due to spatially non-random distribution of observers and highly variable vessel interaction rates with seabirds in different locations. Direct estimation of seabird impacts from observed capture rates without reference to spatial overlap patterns therefore has the potential to be dangerously biased.

- Some seabirds have very low population sizes, or are impossible for non-expert fisheries observers to identify reliably at sea, so that observed capture rates on a species-specific basis are not a reliable means of estimating population-level risk.

Data availability and the needs of fisheries managers drove the following decisions in the design and application of the SEFRA framework to New Zealand seabirds:

- The fundamental unit at which risk is assessed is *per seabird species or distinct population*. Biological risk assessment only makes sense with reference to units that are biologically meaningful. Only subsequently does it make sense to disaggregate and assign the risk to particular fisheries or areas. Assessment frameworks that assign risk on the basis of administrative categories but do not relate these to total risk at the species or population level (e.g., Campbell & Gallagher 2007) are inadequate for this purpose.

- The SEFRA method *can be applied to every species of seabird for which spatial distributions have been estimated.*

- The risk assessment stage *does not rely on species-specific population models or monitoring studies;* these are unavailable for most species.

- The impact assessment *does not rely on the existence of universal or representative fisheries observer data* to estimate seabird mortality. Fisheries observer coverage is generally too low and/or too spatially unrepresentative to allow direct estimation of seabird bycatch at a species level. *The SEFRA method can be applied for any fishery for which some observer data exists, and modifications of this method (see Section 3.2 below) are useful even where no observer data are available to estimate capture rates.*

- The SEFRA framework assigns risk to each species in an *absolute* sense, i.e., species are not merely ranked relative to one another (e.g., as in the PSA approach; Hobday et al. 2007, Waugh et al. 2008). An absolute as opposed to a relative risk score is required to set clear performance standards to meet conservation goals, and to track changes in performance over time arising from mitigation or management.

- Risk is estimated as a function of population-level impact and of biological parameters that are generally available from published sources, reducing reliance on new or location-specific population data which are often unavailable. Risk can be estimated even for species for which no estimate of population size is available.

- Both impact and risk are *quantitative* and *objectively scalable* between fisheries or areas, so that risk at a species level can be disaggregated and assigned to different fisheries or areas based on their proportional contribution to total impact. This allows managers to identify risk hotspots to target management interventions effectively, to track location- or fishery-specific change over time, and to equitably assign responsibility for necessary risk management responses. It also provides tangible incentive for the adoption of mitigation to reduce impact on a location- or fishery-specific basis.

- *The estimation of risk for each species is quantitative and repeatable* without reference to subjective interpretation or expert knowledge, enabling managers to utilise a consistent decision framework for necessary management action to meet performance standards, and to track changing risk over time.

- *The SEFRA framework allows explicit (Bayesian) treatment of uncertainty, and does not conflate uncertainty with risk* (see Kaplan 1997). Because risk is calculated from numerical inputs for which confidence intervals are explicit, it is possible to track the propagation of uncertainty from uncertain parameter inputs and/or noisy data through to output estimates of risk. The outputs distinguish between situations where information is sufficient to ascertain that impacts are unacceptably high (i.e., high impact, low uncertainty, requiring management intervention) and those where information is insufficient to estimate impacts reliably (i.e., unknown impact, high uncertainty, suggesting the need for additional data collection). It is also possible to identify the origins of the uncertainty (i.e., which input parameters are most responsible for uncertainty of the output estimates) to target new research most effectively.

- The SEFRA framework is designed to *readily incorporate new information.* Assumptions in the impact assessment stage are transparent and testable; as new data become available or assumptions change, the consequences for the subsequent impact and risk calculations arise logically *without the need to revisit other assumptions or repeat the entire risk assessment process,* which would otherwise constitute a major and cost-prohibitive institutional burden to managers.

3.1.3 ITERATIVE DEVELOPMENT OF THE NEW ZEALAND SEABIRD RISK ASSESSMENT

The SEFRA method was initially developed arising from a New Zealand Ministry of Fisheries workshop hosted 18–19 February 2009 (described in Sharp et al. 2011) to support the revision of New Zealand's National Plan of Action – Seabirds. Subsequent to the workshop, application of the SEFRA method has been updated and substantially improved in multiple iterations of the New Zealand seabird risk assessment (hereafter NZSRA), arising from productive collaboration between MPI scientists and contracted research providers, with input from the MPI Aquatic Environment Working Group and the Seabird Stakeholder Advisory Group. Sequential iterations of the seabird risk assessment from 2009–15 are described in Waugh et al. (2009), Richard et al. (2011), Richard & Abraham (2013b), Richard & Abraham (2015), and Richard et al. (2017).

Cognisant of structural or methodological improvements that had not yet been actioned in SEFRA implementations to date, the full method framework was described here for the first time (in 2017) to guide future work. Subsequently, the first SEFRA implementation fully consistent with the method described herein was a single-species assessment for Hector's- Maui dolphins (Roberts et al. 2019). A multi-species seabird implementation consistent the framework is in development (D Webber, in prep).

In the National Plan of Action – seabirds (NPOA-Seabirds; Ministry for Primary Industries 2013), the SEFRA method was adopted as the means by which species-level risk to seabirds is assessed, and to provide a performance metric by which risk-reduction goals are defined and evaluated.

3.1.4 APPLICATIONS OF THE SEFRA FRAMEWORK TO OTHER RISK ASSESSMENTS

It is planned that variations on the SEFRA method will be used in New Zealand to deliver risk assessments across a wide range of direct fisheries impacts. In addition to the New Zealand seabird risk assessment, the method has been or is being applied also as follows:

- Waugh et al. (2012) applied a variation of the SEFRA method to characterise risk to multiple seabird species on a global scale associated with tuna fishing effort under the Commission for the Conservation of Southern Bluefin Tuna (CCSBT).
- Currey et al. (2013) used a simplified precursor to the SEFRA method to estimate commercial trawl and set-net fishery risk to Māui dolphins, as part of an expert workshop to characterise risk to this species from both fisheries and non-fisheries threats. Outputs of this workshop were subsequently used to evaluate the relative efficacy of alternate risk-reduction strategies and inform management.
- The first iteration of a New Zealand Marine Mammal Risk Assessment (hereafter NZMMRA) was completed in 2017 (Abraham et al. 2017).
- A species-specific implementation of the SEFRA method focused on Māui and Hector's dolphins is in progress (MPI project SEA2016-30) to estimate fisheries risk and inform the evaluation of hypothetical risk management scenarios.
- The SEFRA framework will be adapted to also address non-fishery threats in a multi-threat risk assessment (PRO2017-12) to inform the update of the Māui and Hector's dolphin Threat Management Plan in 2018.
- Species-specific implementations of the SEFRA are planned for New Zealand sea lions and fur seals once available satellite telemetry has been analysed to estimate spatial foraging distributions (PRO2017-10).
- A Southern Hemisphere seabird risk assessment is currently in progress to assess risk to globally distributed New Zealand seabird species from all commercial High Seas and EEZ fishing effort.

- Adaptations of the SEFRA method are being considered to evaluate harvest rates for non-target and/or low information fish species.

Adaptations of this method are also being considered to evaluate fisheries risk to other protected species and harvest rates for non-target fish in other areas. The SEFRA method is also fully compatible with a spatially explicit bottom fishing impact assessment method described in Sharp et al. (2009) and further developed (with simulations including recovery from impacts and management strategy evaluation) in Mormede & Dunn (2012). The existence of comprehensive spatially explicit risk assessments evaluating all fisheries impacts simultaneously, and with the ability to evaluate alternate management scenarios via management strategy evaluation (MSE), will provide a powerful tool to inform fisheries management.

3.1.5 CHAPTER OVERVIEW

This chapter describes the SEFRA framework at the conceptual and methodological level, without reference to one particular implementation of the method. Section 3.2 outlines the mathematical formulation a multi-species implementation of the method, which applies a fully integrated Bayesian model to estimate capture rates and risk across multiple species and different fisheries simultaneously, as in the current NZSRA and NZMMRA. Section 3.3 describes in detail the structural assumptions and necessary input parameters to inform the model formulation outlined in Section 3.2. Section 0 briefly describes potential alternative applications of the method to address different types of problems, or to accommodate situations where the data are not available to inform all of the standard inputs in the fully integrated Bayesian modelling method.

Where appropriate, the method description is illustrated with examples from one or more of the existing SEFRA implementations listed above, or where necessary from unpublished implementations still in development. Because the SEFRA method was first designed in the context of the NZSRA, many of these examples are extracted or reproduced from Richard & Abraham (2015) or from the unpublished subsequent iteration of the NZSRA described in Chapter 8, but where alternative methodological choices are best illustrated by other existing risk assessments, these are cited in turn. Results of the most recent NZSRA are included separately in the seabird chapter of this AEBAR, Chapter 8. Results of the NZMMRA are published separately in Abraham et al. (2017).

3.2 METHODS

3.2.1 INTEGRATED BAYESIAN MULTISPECIES IMPACT ESTIMATION: MATHEMATICAL OVERVIEW

Mathematical parameters and their support are summarised in

Table 3.1.

3.2.1.1 OVERLAP

The SEFRA method estimates the encounter rate between non-target species and fishing effort as a function of the *overlap* (in space and time) between mapped species distributions and mapped fishing effort distributions. Every fishing event *i* is assumed to be within the 2-dimensional space $\mathbb{X}$ (i.e., $i \in \mathbb{X}$) and to occur at some time (i.e., $i \in \mathbb{T}$).

For each species s, at the location and time of every fishing event *i*, O_{si} is the *overlap* parameter, estimated as the product of the fishing intensity a_i and species probability density p_{si} at the location of fishing event *i*, i.e.:

$$O_{si} = a_i \ * p_{si} \tag{1}$$

where a_i is a metric of *fishing effort intensity* (e.g., number of hooks, kilometres of net) assigned to every fishing event *i*; and p_{si} is the *species probability density* at that location and time, i.e., the probability that an individual of species s selected at random from the population occupies that spatial cell at the time of the fishing event; the sum of all cells in the spatial domain must equal one.

Table 3.1: Mathematical variables and their support as utilised in equations (1) – (30).

Variable	Support	Description
Indices		
i		Fishing event index
s		Species index
z		Species group index
g		Fishery group index (all fishing events *i* are assigned to a fishery group denoted *g*)
Covariates		
a_i	$a_i > 0$	Fishing intensity per event (e.g., number of tows, number of hooks, length of nets)
p_{si}	$p_{si} \geq 0$	Species (individual) probability density
k_{zg}	$k_{zg} \geq 1$	Cryptic mortality multiplier
r_{zg}	$0 \leq r_{zg} \leq 1$	Live release rate
L_{zg}	$0 \leq L_{zg} \leq 1$	Live release survival rate
Derived quantities		
O_{si}	$O_{si} \geq 0$	Species (individual) overlap
∂_{zi}	$\partial_{zi} \geq 0$	Species group density in space
Θ_{zg}	$\Theta_{zg} \geq 0$	Species group density overlap
q_{sg}	$q_{sg} \geq 0$	Catchability
$\varkappa_{zg}$	$\varkappa_{zg} \geq 0$	Total fisheries related deaths multiplier
I_{zg}	$I_{zg} \geq 0$	Fishery interactions
D_{sgi}	$D_{sgi} \geq 0$	Fisheries related deaths
U_s	$\mathrm{U}_s \geq 0$	Species impact ratio
R_s	$\mathrm{R}_s \geq 0$	Species risk ratio
PST_s	$\mathrm{PST}_s \geq 0$	Population Sustainability Threshold
Data		
C_{zgi}	$C_{zgi} \geq 0$	Observable captures
C'_{zgi}	$C'_{zgi} \geq 0$	Observed captures
Parameters		
v_g	$v_g \geq 0$	Fishery group vulnerability
v_z	$v_z \geq 0$	Species group vulnerability
$\mathbb{N}_{si}$	$\mathbb{N}_{si} \geq 0$	Available population size
N_{si}	$\mathrm{N}_{si} \geq 0$	Biological population size
φ	$\varphi \geq 0$	PST adjustment factor
r_{max}	$r_{max_s} \geq 0$	Maximum population growth rate

3.2.1.2 FISHERY GROUPS

All fishing events *i* are assigned to *fishery groups g* within which the gear configuration and vessel behaviour is assumed to be similar, such that species catchability and vulnerability estimates for each species group can be applied uniformly to all effort in the fishery group. The overlap of a species with all fishing effort in the fishery group is obtained by summing across all fishing events in the group.

$$O_{sg} = \sum_{\mathrm{i}} O_{sgi} \tag{2}$$

3.2.1.3 TOTAL OBSERVABLE CAPTURES

A *capture* is an event whereby an individual of the non-target species in question is entangled or restrained by fishing gear (alive or dead) and is unable to free itself under its own power. *Captures* include animals that are killed and their bodies recovered on board the vessel, plus animals released alive, but exclude *cryptic deaths* (see below). *Observable captures* include all captures that occur and would be recorded if 100% of fishing events were observed. *Observed captures* refer to only that subset of observable captures that are actually recorded by fisheries observers.

Total observable captures *C* of each species per fishing event in fishery group *g* is a product of the probability of encounter per individual (proportional to overlap *O*), times the probability of capture per encounter (*q*), times the available population size at time *t* of fishing event *i*:

$$C_{sgi} \;=\; q_{sg} O_{sgi} * \mathbb{N}_{si} = q_{sg}\theta_{sgi} \tag{3}$$

where $C_{sgi} \geq 0$ is implied.

> $q_{sg} \geq 0$ is the catchability for species s in fishery group g; (analogous to catchability in a fisheries context, hence abbreviated *q*); and
>
> $\mathbb{N}_{si} \geq 0$ is the *available population size* of species *s* at time *t*, i.e., the biological population size *N* adjusted to reflect the proportion of that population that is within the spatial domain of the assessment at the time of fishing event *i*.
>
> $\theta_{sgi} \geq 0$ is the *density overlap* of species *s* with fishing event *i* (see below).

Total observable captures in fishery group *g* is obtained by summing across captures at all events:

$$C_{sg} = \sum_{\mathrm{i}} C_{sgi} \tag{4}$$

3.2.1.4 DENSITY OVERLAP

The overlap term *O* represents the probability or frequency that a particular individual animal selected at random from the population will encounter a fishing event of a particular fishery group. In contrast, the *density overlap* θ represents the number or frequency of encounters for all individuals of that species. Overlap is converted to a *density overlap per event* by multiplying by species *available population size*:

$$\theta_{sgi} = O_{sgi} * \mathbb{N}_{si} \tag{5}$$

where:

> $\mathbb{N}_{si}$ is the *available population size*, i.e., the number of animals of species s that are present within the spatial domain of the risk assessment at the time t corresponding to fishing event i;

Note that where available population size is seasonally variable (i.e., $\mathbb{N}_{si}$ is not the same for all events i throughout the year), density overlap θ_s must first be calculated at the level of fishing events as in equation (5) and only subsequently summed across events in a fishery group. One consequence is that relative values of O between species reflect relative exposure to fishing effort per individual animal, which scales directly with risk, whereas θ values reflect absolute encounter rates per species, which scales with expected captures but not risk because θ is confounded with population size. For this reason, *O* rather than θ is used until such time as actual densities are estimated across all species in a species group (equations (8) – (9)).

3.2.1.5 IMPROVED CATCHABILITY ESTIMATION USING SPECIES GROUPS

In its most rigorous application, the SEFRA method allows fully quantitative estimation of species-level catchability, applying Bayesian inference to estimate capture rates per encounter for each combination of species x fishery group (q_{sg}), as a function of observable captures C_{sg} and overlap O_{sg}, as in equation (3). However risk assessment methods are designed for application to data-poor problems; if sufficient data existed to estimate catchability for every species x fishery group combination individually, it is unlikely that a risk assessment approach would be required at all; instead captures could simply be estimated directly. In New Zealand, direct estimation is used to estimate captures of the most commonly caught seabirds by the most well observed fisheries (Abraham & Richard 2017; see http://data.dragonfly.co.nz/psc), but this approach is not feasible for the majority of species and fishery groups. In early iterations of the NZSRA (e.g., Richard et al. 2011) application of the approach in equation (3) to species x fishery group combinations for which there were few or no observed captures yielded unacceptably unconstrained answers: estimates of q_{sg} and C_{sg} sometimes varied by more than two orders of magnitude, and extended into biologically implausible bounds.

To better estimate q the dimensionality of the model can be reduced by aggregating individual species s into species groups z on the basis of common physical and behavioural characteristics thought to affect capture rates, such that all species in the group are assumed to have the same catchability q_{zg}.

$$C_{zi} = \sum_{s} q_{zg} O_{si} * \mathbb{N}_{si} \tag{6}$$

3.2.1.6 COMBINED DENSITY OVERLAP

To combine species within a species group, probability density values for each species in the location of every fishing event *i* (p_{si}) at time *t* are converted to actual animal densities and summed across all species in the species group *z* per fishing event, as follows:

$$\partial_{zi} = \sum_{s} (p_{si} * \mathbb{N}_{si}) \tag{7}$$

where:

∂_{zi} is the actual density of all individuals of species group *z* at the time and location of fishing event *i*;

$\mathbb{N}_{si}$ is the *available population size* (see below) of species *s* at the time t of the fishing event *i*.

The use of *available population size* $\mathbb{N}_{st}$ in equations (5) and (7) recognises that the number of individuals actually present in the spatial domain of the risk assessment at the moment of fishing event *i* may be different than the size of the biological population *N* against which impacts are evaluated.

Subsequently, the *density overlap* between species group z and fishery group g (Θ_{zg}) can be estimated simultaneously across all fishing events *i*, by combining equations (1) and (7), as follows:

$$\Theta_{zg} = \sum_{i} (a_{gi} * \partial_{zi}) \tag{8}$$

Note that *density overlap* Θ is different from the previously used overlap *O* in that it refers to the combined actual density of all individuals rather than a probability distribution per individual; this is necessary in order to accurately reflect variable abundances across species when summing distributions across multiple species in a species group. Total observable captures per species group across all fishing events is then:

$$C_{zg} = q_{zg}\Theta_{zg} \tag{9}$$

3.2.1.7 CRYPTIC MORTALITY AND TOTAL FISHERIES RELATED DEATHS

Especially for protected species such as seabirds and marine mammals, not all observable captures result in death, and conversely not all deaths arising from fishery interactions result in an observable capture. Estimation of fishery related deaths D_{sg} from captures data is as follows:

$$D_{sg} = (C_{sg} * k_{sg}) - (C_{sg} * r_{sg} * L_{sg})$$

$$= C_{sg}\left(k_{sg} - \left(r_{sg} * L_{sg}\right)\right) \tag{10}$$

where:

k_{sg} is the *cryptic mortality multiplier,* i.e., a multiplier of the observed captures to account for the additional individuals that die as a direct result of their interaction with the fishing effort but are not recovered on board the vessel and recorded as captures; and

r_{sg} is the *live release rate*, i.e., the proportion of captured individuals that are released alive; and

L_{sg} is the *live release survival rate*, i.e., proportion of live releases expected to survive.

To aid subsequent algebraic manipulation, it is useful to combine these parameters (with uncertainty) into a *total fisheries related deaths multiplier denoted by* $\varkappa$ (kappa), to facilitate conversion between total observable captures *C* and total fishery related deaths *D*, as follows:

$$D_{sg} = C_{sg} * \varkappa_{sg} \tag{11}$$

where $\varkappa_{sg} = \left(k_{sg} - \left(r_{sg} * L_{sg}\right)\right)$

3.2.1.8 SPECIES VULNERABILITY TO INTERACTION

Non-target species capture rates are modelled separately within each of several broadly defined *fishing methods*. The NZSRA defines four such fishing methods: trawls, bottom longlines, surface longlines and set nets. The NZMMRA includes also purse seines as a fifth method. *Fishery groups* are nested subsets of *fishing methods*.

Within each such method-specific model, interaction rates between species groups and fishery groups are estimated at the level of *interaction incidents* rather than deaths or captures in isolation. Interactions I_{zg} are defined as captures (alive or dead) plus cryptic deaths, i.e.,

$$I_{zg} = C_{zg} * k_{zg} = q_{zg}\Theta_{zg}k_{zg} \tag{12}$$

Species *vulnerability v* is defined as the probability of interaction per encounter with fishing effort (i.e., vulnerability *v* includes captures plus cryptic deaths, as opposed to catchability *q*, which is the probability of capture excluding cryptic mortality).

$$v_{zg} = q_{zg}k_{zg} \tag{13}$$

$$I_{zg} = v_{zg}\Theta_{zg} \tag{14}$$

A major innovation first utilised in the third iteration of the NZSRA (Richard & Abraham 2013b) was to split the vulnerability parameter v_{zg} into two parameters representing species group vulnerability v_z and fishery-group vulnerability v_g separately, as follows:

$$I_{zg} = v_z v_g \Theta_{zg} \tag{15}$$

The species group vulnerability term v_z reflects that some species groups are more attracted to fishing vessels, or otherwise more susceptible to capture or cryptic death than other species groups. The structural assumption imposed by splitting the vulnerability parameter in this way is that the relative difference in species group vulnerability will apply across all fishery groups within a broadly defined fishing method (e.g., a bird species that aggressively interacts with trawl fisheries will be more vulnerable to capture in all trawls than is a less aggressive bird species, reflected by a higher v_z, and this relationship will be constant across trawl fishery groups).

Similarly, the fishery group vulnerability term v_g reflects that within each fishing method, some fishery groups will be expected to capture or kill non-target species more often than do other fishery groups, e.g., reflecting mitigation uptake or offal discard practices, and this fishery group effect will apply across all species groups in common.

By separating the vulnerability term v_{zg} into these separate components, this model structure effectively allows capture rates in data-limited species x fishery group combinations to be informed or constrained by data from species x fishery group combinations for which more data are available (i.e., because of higher populations, or higher capture rates, or higher levels of observer coverage). In the example of the NZSRA, replacing the single-parameter approach in equation (3) with the split-parameter approach in equation (15) yielded substantially improved model power.

Estimation is applied to *interactions* rather than *captures* (i.e., *vulnerability* not *catchability*) on the assumption that the inherent species group and fishery group properties represented by the v_z and v_g terms affect the rate at which the species will physically interact with fishing gear, but that subsequent retention of corpses affecting the cryptic mortality multiplier k_{zg} (hence capture rate C_{zg}) may operate independently per combination of fishery x species group. This formulation has significant implications for the way that cryptic mortality multipliers are applied, especially in poorly estimated fishery group x species group combinations. Most or all of the factors affecting cryptic mortality multipliers are by necessity estimated outside the integrated model, using input priors to represent uncertainty (see below).

Re-expressing capture rates (for which fisheries observer data are useful) in terms of vulnerability rather than catchability yields:

$$C_{zg} = \frac{v_z v_g \Theta_{zg}}{k_{zg}}$$

(16)

3.2.1.9 BAYESIAN ESTIMATION OF CATCHABILITY

To estimate total observable captures and catchabilities from available fisheries observer data, the most rigorous application of the SEFRA method applies a Bayesian model for each of the broadly defined fishing methods (e.g., trawl, surface longline, bottom longline and set net), using data from observed fishing events to estimate capture rates and species vulnerability simultaneously across all species and fishery groups within the fishing method.

Total observable captures C_{zg} are estimated across all fishing events per fishery group on an annual basis. Because protected species capture rates refer to relatively infrequent events resulting in individual animal deaths, in the NZSRA and NZMMRA total observable captures are modelled using a Poisson distribution as follows:

$$C_{zg} \sim Poisson(\hat{\lambda}_{zg})$$

(17)

Other error distributions may be appropriate for other implementations of the SEFRA method, e.g., non-target fish bycatch or benthic invertebrate impacts.

Modifying equation (16),

$$\widehat{\lambda'}_{zg} = \sum_{\mathrm{i}} \frac{v_z v_g \Theta'_{zgi}}{k_{zg}} * \epsilon_{zg}$$

(18)

where the ʻ suffix is used to denote parameters referring only to the observed subset of total fishing effort, as follows:

$\widehat{\lambda'}_{zgi}$ is the estimated *observed* captures of all species in species group z associated with fishing group g.

Θ'_{zi} is the observed density overlap of species group z with observed fishing event i. This term is functionally equivalent to the spatial overlap O_{sg} in equation (2), except transformed to represent actual densities across all species in the group rather than probability densities per species, and restricted to observed events rather than all events.

v_z is the species group vulnerability for species group z;

v_g is the fishery group vulnerability for fishery group g;

k_{zg} is the cryptic mortality rate for species group z in fishery group g; and

ϵ_{zg} is an error term associated with the combination of species group *z* and fishery group *g*;

3.2.2 RE-APPLYING MODELLED VULNERABILITES TO SPECIES-LEVEL IMPACT

An integrated Bayesian model fitted to fisheries observer data as in Equation (18) is the best means by which observed capture rates across all fisheries and species can be used to estimate v_z and v_g in a multi-species/multi-fishery risk setting. Subsequently the split vulnerability parameters v_z and v_g are re-combined with estimates of the cryptic mortality multiplier k_{zg} to estimate q_{sg} as in equation (12) (noting $v_s = v_z$ for all species in group z), and combined with live releases and live release survival as in equations (10)–(11) to estimate total fishery-related deaths (hereafter FRDs).

$$I_{sgi} = v_s v_g O_{si} * \mathbb{N}_{si} \tag{19}$$

$$C_{sgi} = \frac{v_s v_g O_{si} * \mathbb{N}_{si}}{k_{zg}} \tag{20}$$

$$D_{sgi} = \frac{v_s v_g O_{si} * \mathbb{N}_{si}}{k_{zg}} * \varkappa_{sg} \tag{21}$$

In these equations impacts can be estimated per individual fishing event (including un-observed fishing events) or combined at any scale to yield spatially explicit estimates of captures and FRDs (with uncertainty) on an individual species and fishery basis, even for species and fisheries for which captures data were insufficient to inform estimates of species catchability on an individual basis. Model diagnostics should include comparisons of observed vs. expected numbers of observed captures, including on a spatially disaggregated basis (e.g. Figure 3.9 below) to inform evaluation of structural model assumptions and to assess the accuracy of spatial data layer inputs.

3.2.3 FROM IMPACT TO RISK

For protected species risk assessments, the estimation of species level impact and risk is as follows. Alternative approaches utilised in fish and benthic habitat risk assessments will be developed separately.

3.2.3.1 BIOLOGICAL POPULATION SIZE

Fishery-related deaths on an annual basis are evaluated as a proportion of the biological population size for each species, N_s. To ensure that risk scores are biologically meaningful, N_s is necessarily applied at the level of a distinct biological population at the scale of a country or region (for protected species) or a distinct stock (for non-target fish). Where and when a proportion of the biological population exists outside of the spatial domain of the risk assessment, biological population N_{si} will differ from available population $\mathbb{N}_{si}$.

3.2.3.2 IMPACT RATIO

Because individual deaths are additive, impacts can be summed across groups, yielding total FRDs at the species level:

$$D_s = \sum_{\mathrm{g}} D_{sg} \tag{22}$$

The *impact ratio U* is defined as the proportion of the total biological population killed by fishing effort each year, either at a fishery group level or collectively for all fishery groups at the species level:

$$U_{sg} = \frac{D_{sg}}{\mathrm{N}_s} \tag{23}$$

$$U_s = \sum_{\mathrm{g}} U_{sg} = \sum_{\mathrm{g}} \frac{D_{sg}}{\mathrm{N}_s} \tag{24}$$

… U_s is therefore analogous to exploitation rate *U* in fisheries.

Note that combining equations (3), (7) and (9) (where $N = \mathbb{N}$, i.e., neglecting or correcting for seasonal migrations that change available population size) implies

$$\frac{D_{sg}}{N_s} = q_{sg} O_{sg} \varkappa_{sg} = U_{sg} \tag{25}$$

Summing across all fishery groups as in equation (25) yields:

$$U_s = \sum_{g} q_{sg} O_{sg} \varkappa_{sg} \tag{26}$$

The power of this formulation is that so long as species catchability q_{sg} can be estimated by some means other than equation (3) (and adjusting for variable seasonal presence of the species in question within the spatial domain) it becomes possible to estimate impact levels (and subsequently risk), *even for species for which both population size N_s and total observable captures C_{sg} are unknown*.

Equation (26) becomes very important in the application of the SEFRA method to very rare species (because captures are too rarely observed to estimate C_{sg} with any statistical power), or to species for which no observer data is available to estimate capture rates, or to species for which population size is unknown (e.g., seabirds for which colonies are inaccessible to survey; deepwater fish; many cetaceans). Alternative means of estimating q are under development for application of the SEFRA method to deepwater fish (Sibanda et al. 2016), analogous to similar approaches applied overseas (Zhou et al. 2009, 2011). In data-poor situations relative catchability *q* between species can also be intuited from first principles and expert knowledge (with uncertainty) or estimated by analogy with more data-rich applications conducted for similar species elsewhere.

3.2.3.3 RISK RATIO

Under the SEFRA framework, 'risk' is defined as the estimated species-level fisheries impact as a proportion of a defined impact sustainability threshold, i.e.,

$$R_s = \widehat{U}_s / U_s \tag{27}$$

Because intuitively the ability of a species to sustain impacts is related to its biological productivity, the chosen threshold U_s will vary accordingly, i.e., analogous to a target exploitation rate U_{msy} for fish or to PBR approaches commonly applied to marine mammals (Wade 1998). Where impacts are generally expressed as an annual exploitation rate (i.e., fish) or a proportional spatial impact per unit time (benthic habitats) we have adopted the term 'Maximum Impact Sustainability Threshold' or MIST, first proposed in the planned implementation of the SEFRA method for fish (Roux et al. 2015).

Implicit in the choice of threshold U_s (MIST, or PST see below) is a particular population outcome corresponding to a particular level of impact; this relationship between impact and population outcome is established via simulations. Note that because under the SEFRA method output estimates of impact and risk are themselves uncertain, it is necessary that the chosen population outcome used to define the impact threshold U_s (corresponding to $R_s = 1$) is expressed with reference to the level of certainty with which the outcome will be achieved.

For protected species where impacts are more commonly expressed as individual deaths rather than annual exploitation rate, an alternative but mathematically equivalent formulation of equation (27) is:

$$R_s = \frac{D_s}{PST_s} \tag{28}$$

where:

> D_s is *total fishery related deaths* from equation (10), and
>
> PST_s is the *Population Sustainability Threshold* expressed as a number of individual deaths per year and defined with reference to a particular population outcome (see below).

3.2.3.4 POPULATION SUSTAINABILITY THRESHOLD (PST)

For protected species, the SEFRA method defines an impact threshold as a function of maximum population growth rate r_{max} , analogous to the PBR ('potential biological removals') formulation of Wade (1998). Wade (1998) defines PBR as:

$$PBR = \frac{1}{2} r_{max} * N_{min} * f \tag{29}$$

where:

r_{max} is the theoretical unconstrained maximum population growth rate, reflecting biological productivity;

N_{min} is a conservative point estimate (20th percentile) of total population size; and

f is a subjective 'recovery factor' defined to adjust the threshold value to reflect management goals on a per-species basis.

Early implementations of the NZSRA utilised variations on the PBR formulation in the definition of risk, but subsequently refined this approach to the extent that referring to 'PBR' in the NZSRA is now misleading. From the 2017 iteration of the NZSRA and the first MMRA, we coin the term 'Population Sustainability Threshold' or PST, defined as follows:

$$PST = \frac{1}{2} \varphi * r_{max} * N \tag{30}$$

where:

φ (greek letter phi) is an adjustment factor estimated by simulation and defined to ensure that impacts equal to PST (i.e., R = 1) correspond to a defined population stabilisation or recovery objective.

The r_{max} term is estimated from biological and demographic input parameters, the estimation of which will be specific to different taxa, e.g., marine mammals vs. seabirds (see Section 3.3, Model Inputs, below).

For seabirds, earlier iterations of the NZSRA estimated r_{max} from field estimates of adult survival S_A and age at first reproduction *A*, and applying the formulae of Niel & Lebreton (2005), but required subsequent correction arising from estimation bias inherent in this method (Richard & Abraham 2013a, 2013b). Following recent (2016) discovery of errors in simulations used to derive the bias correction parameter, an updated approach was reviewed and approved via the AEWG in 2016 whereby r_{max} is estimated by applying an allometric power relationship between body mass M and taxonomic adult survival S_{tax} (see chapter 8). r_{max} and population size *N* are in turn used to estimate a PST via equation (30).

Ideally within the SEFRA method, biological parameters used in the derivation of r_{max} should be defined as inputs to the fully integrated Bayesian model (including representation of uncertainty for each parameter) instead of estimating r_{max} outside the model and defining a single input distribution. In this way uncertainty from biological input parameters propagates through the model, and output uncertainty can be tracked back to its source including uncertain biological inputs (see Section 3.3, Model Inputs, below).

3.2.4 CONSTRAINING PARAMETER INPUTS USING BIOLOGICAL MONITORING AND OTHER AVAILABLE DATA

Under the SEFRA framework uncertainty is reflected explicitly at every stage, (i.e., using ranges or distributions for every input parameter) and propagates through interim calculations through to output estimates of risk, wherever possible via Bayesian models. A major strength of this approach is that it becomes possible to use data sources other than observed captures to constrain model input parameters or impose priors, and this information then affects subsequent estimation of vulnerability and risk via the integrated model. Where model fits are in conflict with input distributions (e.g., high population survival estimates in conflict with high estimated fisheries mortality rates) the integrated model is forced to estimate what combination of parameter estimates is most plausible and revised parameter estimates are reflected in modified posterior distributions. In this way, where logical constraints on total FRDs can be defined as a function of biological and demographic data (e.g., adult survival S, see below) population monitoring data serve to better estimate population level risk (rather than risk scores being a function of captures data only). Appropriately, the influence of non-captures data on model outputs will be stronger for those species and fisheries for which captures data are poor relative to population or demographic data (as will be the case for example for well-monitored seabird breeding colonies). Conversely, where capture rates are better estimated than demographic parameters, model fits based on captures can inform or constrain poorly informed estimates of population parameters and/or help to direct future population research.

3.2.4.1 CONSTRAINING SEABIRD CAPTURES USING ADULT SURVIVAL

Iterative development of the NZSRA illustrates the power of this approach. In previous iterations (up to Richard & Abraham 2015) there were seabird species for which fisheries risk was estimated to be very high, primarily as a consequence of observed multiple-capture events despite very low levels of observer coverage. This resulted in high (and highly uncertain) estimates of impact and risk for these species, for which the upper bound of the estimate extended to levels that, if actualised, would cause certain population decline. Nonetheless populations of some of these same species (e.g., black petrels, Chatham albatrosses) were observed to be approximately stable, and adult survival was high, suggesting that captures were overestimated in the risk assessment.

This difficulty was overcome in the latest (2017) update of the NZSRA by incorporating biological and population input parameters affecting estimation of the *PST* (i.e., adult survival, age at reproduction, population size) within the integrated model and constraining total FRDs such that the annual death rate cannot exceed maximum annual mortality suggested by the adult survival rate, i.e., [$D < (1 - S)$]. Model fits with this constraint indicated (for Chatham albatrosses) that vulnerability to capture was lower than previously modelled, such that revised estimates of FRDs are now consistent with population trend and mark-recapture data. For black petrels, the updated model suggests that population size *N* is likely to be higher than previously estimated, and/or that live release survival is significant (live release survival was not included in previous iterations of the NZSRA). That the integrated model can use observed capture rates to better estimate population parameters, and vice versa, is a major strength of the method, and provides tangible incentive to invest in population monitoring. Before these data were combined in an integrated model, there was no clear mechanism by which seabird population time series data were used to inform seabird fisheries risk, and risk assessment outputs were sometimes in conflict with population monitoring data.

3.2.5 PST VS. PBR

A key difference from the PBR approach of Wade (1998) is that the conservative population point estimate $N_{\min}$ has been replaced with a realistic estimate of N. Because Bayesian methods allow full statistical consideration of uncertainty in the input estimate of *N* (and other input parameters), the consequences of uncertain population size are now reflected as uncertain risk estimate outputs. Because N appears in equations at multiple stages in the SEFRA method, utilising a biased estimator at the outset not only affects the definition of a sustainable impact level; it also affects estimates of *available population size* and *density overlap*, hence capture rate (equations (6)–(9)) , and the estimation of *vulnerability* from observed captures (equation (18)). For this reason it is preferable to adopt realistic estimates of N (including uncertainty) in the risk assessment stage; conservatism is better incorporated in the choice of a population outcome affecting φ within the PST formulation (equation (30)) or in the risk management stage, distinct from risk assessment.

For the same reason, the PST formulation replaces use of the recovery factor f with an alternate formulation that makes explicit the distinction between statistical uncertainty (a scientific consideration) and management risk aversion (a policy consideration). The previous use of *f* in PBR in equation (29) effectively confused risk assessment and risk management within a single term, such that it was impossible when comparing PBR scores to distinguish between a species with low biological productivity and a less ambitious recovery factor (low r_{max}, high *f*) vs. a species with higher productivity and a more ambitious recovery factor (high r_{max}, low *f*). The PST formulation effectively gets the f out of PBR so that species with comparable risk scores in equation (28) can be expected to have a comparable population outcome, irrespective of management goals. Conservatism in the choice of an appropriate reference outcome is more appropriately addressed in the definition of φ in equation (30), which is constant for all species within a particular assessment model. Defining different management objectives for different species is best addressed outside the risk assessment, or by developing species-specific multi-threat models. Lonergan (2011) in his critique of PBR and related approaches makes this same argument for maintaining the separation between risk assessment and risk management.

The φ parameter in equation (30) is a tuning parameter selected with explicit reference to a population recovery or stabilisation outcome, reflecting a policy decision. . For multi-species protected species risk assessments such as the NZSRA and NZMMRA, Fisheries New Zealand has typically assigned a default value of $\varphi = 0.5$, which implies a population reference outcome as follows: 'for a population in which the mean annual impact is equal to or lower than PST (i.e., $R \leq 1$), the population will recover to and/or stabilise at a mean equilibrium level at or above 75% of its un-impacted status' This formulation assumes a logistic population growth curve (i.e. linear density dependence); the implications of alternate forms of density dependence can be explored via simulation.

In different risk assessments the φ parameter can be adjusted such that R=1 corresponds to a different outcome (i.e. the population stabilising at a higher or lower level) reflecting other management objectives applied to different kinds of taxa (e.g., for non-target fish vs. for protected species).

SEFRA risk score outputs are commonly interpreted to imply that R=1 is a maximum 'acceptable' risk threshold, implying a policy objective. However, risk managers may wish to define different policy objectives for different species within the same multi-species SEFRA model, for example reflecting different species' threat status or different levels of exposure to non-fishery threats. In this case it is better to define maximum acceptable risk thresholds for individual species outside of the SEFRA model (rather than adjusting φ values separately for different species inside the model). This ensures that similar risk scores imply comparable population outcomes for all species in the same assessment, irrespective of species-specific policy goals. In contrast, single-species assessments have no such limitation, so the φ parameter may be adjusted to ensure that R=1 corresponds to a specific policy goal.

Because SEFRA model outputs include full Bayesian representation of uncertainty, it is possible (and in fact necessary) when setting risk reduction goals to specify not only the risk reduction goal, but also how much certainty is required that the goal will be achieved. For example in the update of the Hector's-Maui dolphin Threat Management Plan in 2020, the risk reduction goal for Hector's dolphins was defined as follows: "*With 95% certainty*, commercial fisheries risk will be reduced and maintained at or below a level that allows the population to achieve an equilibrium level at or above 90% of its un-impacted status". (i.e. in this instance $\varphi = 0.2$). Note that the required level of certainty is expressed with reference to impact (the numerator, D, in equation 28) rather than with reference to PST (the denominator). In practice this is achieved by choosing which percentile of the estimated impact posterior is compared against the PST when evaluating performance against a risk reduction target, but this decision does not affect the definition of PST.

Table 3.2: Mean species-level risk estimates, disaggregated by target fisheries. Highlighted cells (increasing yellow-red) identify fisheries that are responsible for an increasing proportion of species-level risk. Target fisheries with zero risk to all species (rounded to two decimal places) are not shown; these include: albacore SLL, minor SLL, jack mackerel trawl, and grey mullet set net. Likewise, species for which mean total risk ratio rounds to zero are not shown.

Species	PBR	Inshore trawl	Flatfish trawl	Bluenose BLL	Hapuka BLL	Minor BLL	Snapper BLL	Small ling BLL	Large ling BLL	Small STN SLL	Large STN SLL	Bigeye SLL	Swordfish SLL	Hoki trawl	Hake trawl	Ling trawl	SBW trawl	Scampi trawl	Squid trawl	Middle depth trawl	Deepwater trawl	Flatfish SN	Shark SN	Minor SN	TOTAL RISK RATIO
Black petrel	437	0.20	0	0.18	0.06	0.04	0.22	0	0	0	0	0.29	0.03	0.01	0	0	0	0.01	0	0.01	0	0	0	0	1.15
Salvin's albatross	3598	0.30	0.03	0	0	0.01	0	0.09	0	0	0	0	0	0.12	0	0.01	0.01	0.08	0	0.09	0.02	0	0	0	0.78
Flesh-footed shearwater	1451	0.29	0.01	0	0.04	0.04	0.19	0	0	0	0	0.02	0.01	0.01	0	0	0	0.03	0	0.01	0	0	0	0	0.67
Westland petrel	350	0.15	0.05	0	0.01	0.01	0	0.02	0	0.04	0	0.01	0	0.07	0.01	0	0	0	0	0.03	0	0.01	0.01	0	0.48
Southern Buller's albatross	1369	0.03	0.01	0	0	0	0	0.02	0	0.04	0.01	0	0	0.14	0.01	0.01	0	0.01	0.05	0.04	0	0	0	0	0.39
Chatham Island albatross	425	0.01	0	0	0.01	0.01	0	0.20	0.01	0	0	0	0	0.02	0	0	0	0	0	0.01	0.06	0	0	0	0.36
New Zealand white-capped albatross	10915	0.15	0.05	0	0	0	0	0	0	0.01	0	0.01	0	0.04	0	0.01	0	0.01	0.03	0.03	0	0	0	0	0.35
Gibson's albatross	496	0	0	0	0	0	0	0	0	0.07	0	0.04	0.19	0	0	0	0	0	0	0	0	0	0	0	0.34
Northern Buller's albatross	1628	0.01	0	0	0	0	0	0.02	0	0.03	0	0.07	0	0.03	0	0	0	0.03	0	0.02	0	0	0	0	0.25
Antipodean albatross	364	0	0	0	0	0	0	0	0	0.05	0	0.02	0.10	0	0	0	0	0	0	0	0	0	0	0	0.20
Otago shag	285	0.01	0.13	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.14
Northern giant petrel	336	0	0	0	0.02	0.01	0.01	0	0	0	0	0	0	0.03	0	0	0	0.01	0	0.01	0.01	0	0	0	0.14
Spotted shag	3710	0.02	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.09
Yellow-eyed penguin	287	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01	0.04	0.01	0.08
Campbell black-browed albatross	1980	0	0	0	0	0	0	0	0	0.02	0	0.01	0	0.01	0	0	0	0	0	0	0	0	0	0	0.08
White-chinned petrel	25626	0	0	0	0	0	0	0.02	0.01	0	0	0	0	0.01	0	0	0	0.01	0.01	0	0	0	0	0	0.05
Northern royal albatross	716	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04
Foveaux shag	207	0	0.03	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.04
Grey petrel	5526	0	0	0	0	0	0	0.01	0	0.01	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0.04
Southern royal albatross	848	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02
Snares Cape petrel	1601	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Little black shag	338	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.01
Fishery group total -- all species		1.19	0.38	0.19	0.14	0.11	0.41	0.39	0.03	0.27	0.02	0.47	0.34	0.49	0.02	0.03	0.02	0.18	0.09	0.24	0.10	0.02	0.05	0.02	5.72

3.2.6 STANDARD RISK ASSESSMENT OUTPUTS

Data inputs and analytical pathways utilised in the current NZSRA are shown in Figure 3.1.

A standard output of the NZSRA, showing risk (with uncertainty) at the species level, is shown in Figure 3.2. Note that species-level outputs are less useful to managers charged with managing risks arising from particular fisheries across multiple species simultaneously. Table 3.2 disaggregates species-level risk by fishery group, and highlights those species for which individual fishery groups are responsible for a substantial portion of species-level risk (more than 0.1 PST). Managers concerned about the fate of a particular bird species read across the row to identify fishery groups generating risk to that species; managers responsible for a particular fishery read down a column to see what bird species are affected, and to what extent.

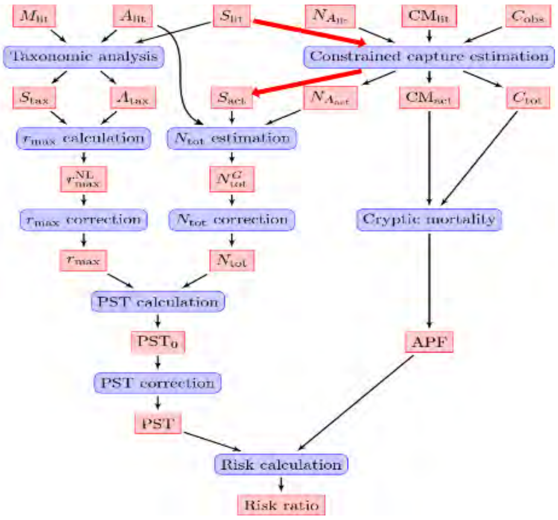


Figure 3.1: Diagram of data inputs and calculation pathways utilised in the current (2017) iteration of the NZSRA. The use of (realised) adult survival estimates to constrain captures and fishery-related deaths is highlighted.

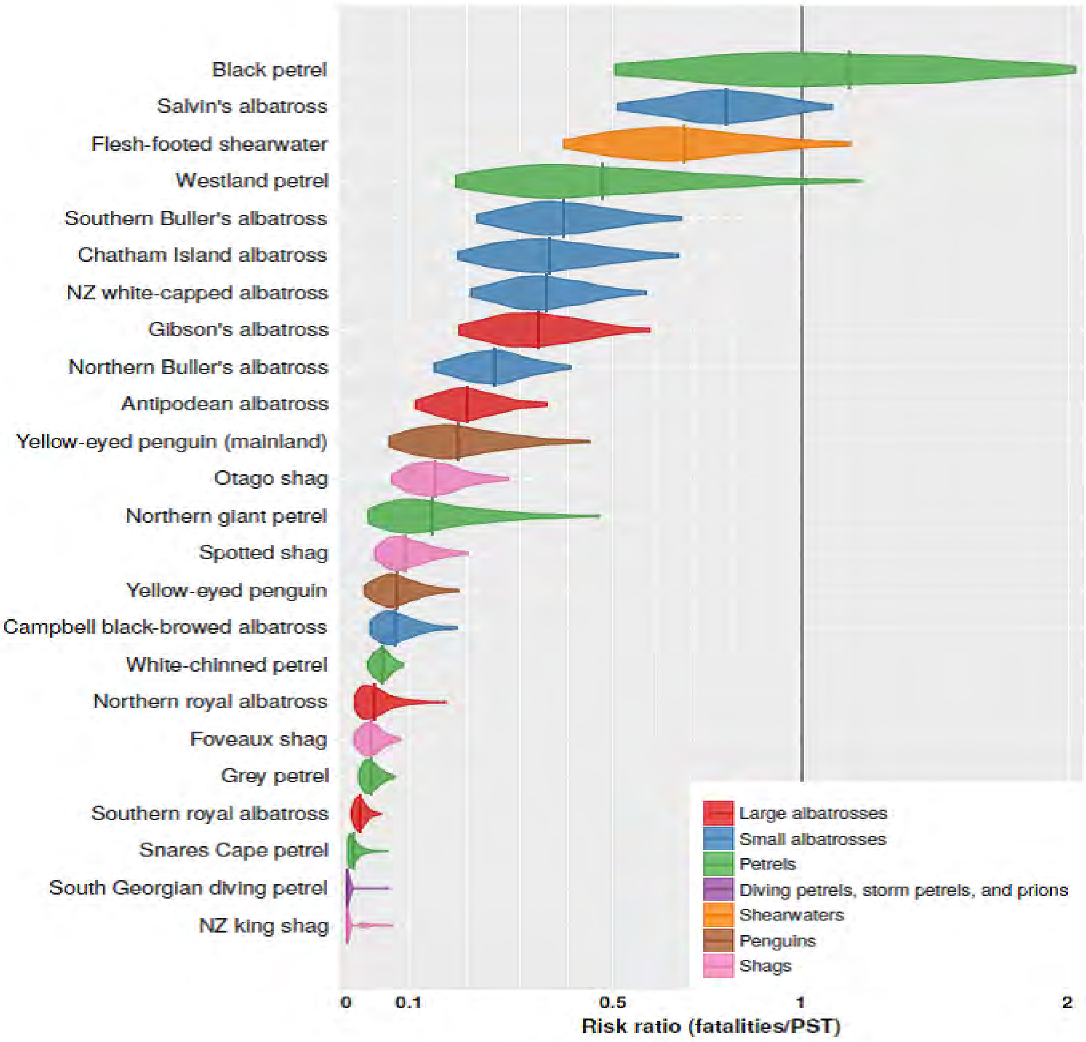
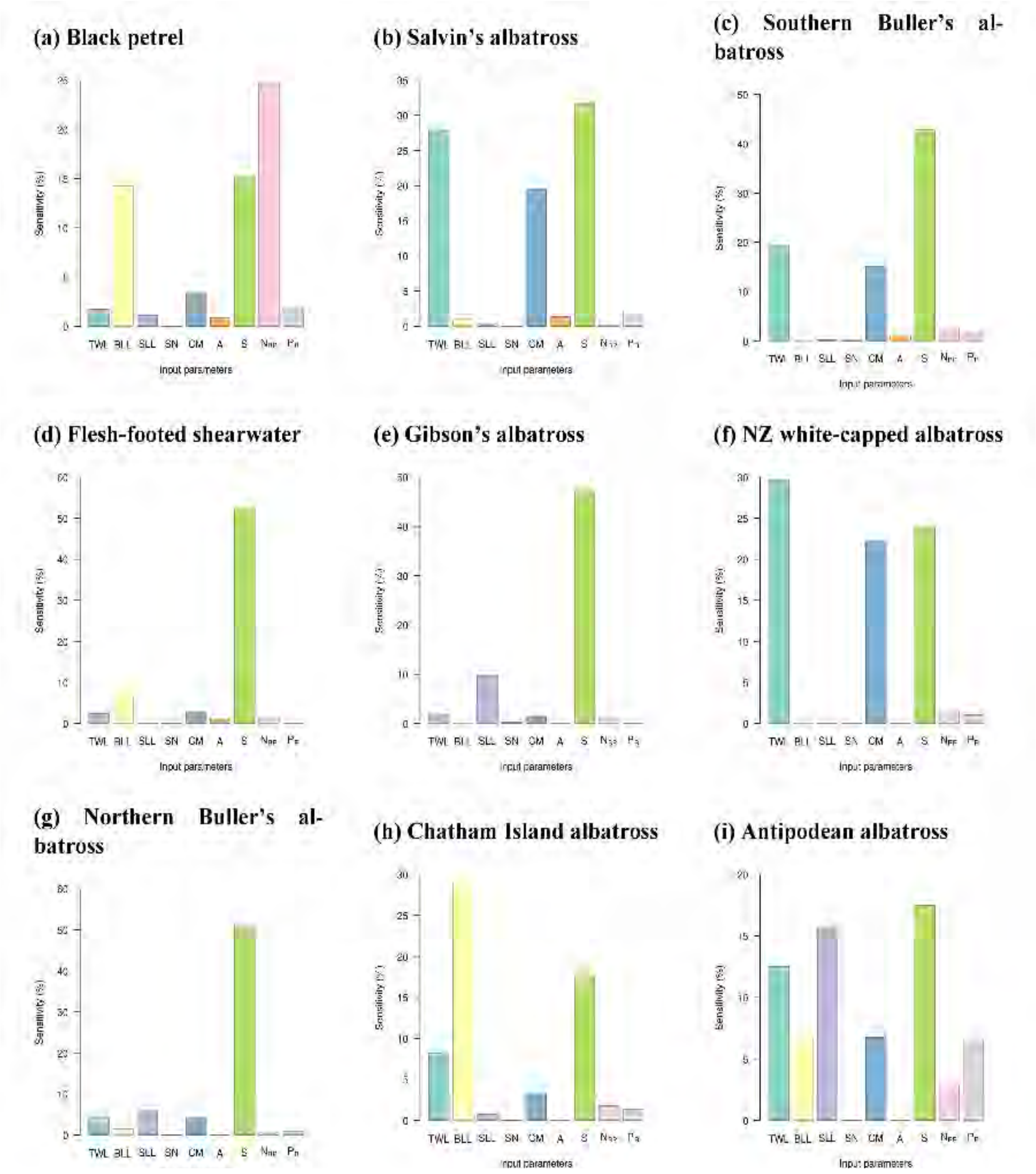


Figure 3.2: Standard species-level output of the NZSRA (from Richard et al. 2017). Species risk is shown on the x axis; the vertical line at R=1 corresponds to the level of *all human-induced mortality* that the species can sustain while still meeting the population recovery

Because Table 3.2 disaggregates mean risk; the representation of uncertainty is lost. For this reason Table 3.2 should always be considered simultaneously with Figure 3.2 rather than in isolation. New work is underway to create a customised query tool to disaggregate and estimate impact and risk, including uncertainty, according to any user-defined criterion without loss of information. Since 2013 the NZSRA has included the results of sensitivities designed to track the propagation of uncertainty from input parameters through to resultant uncertainty of output estimates of risk; an example (from 2015) is shown in Figure 3.3. These figures have proven to be highly valuable to inform research prioritisation model inputs.



(a) Black petrel

(b) Salvin's albatross

(c) Southern Buller's albatross

(d) Flesh-footed shearwater

(e) Gibson's albatross

(f) NZ white-capped albatross

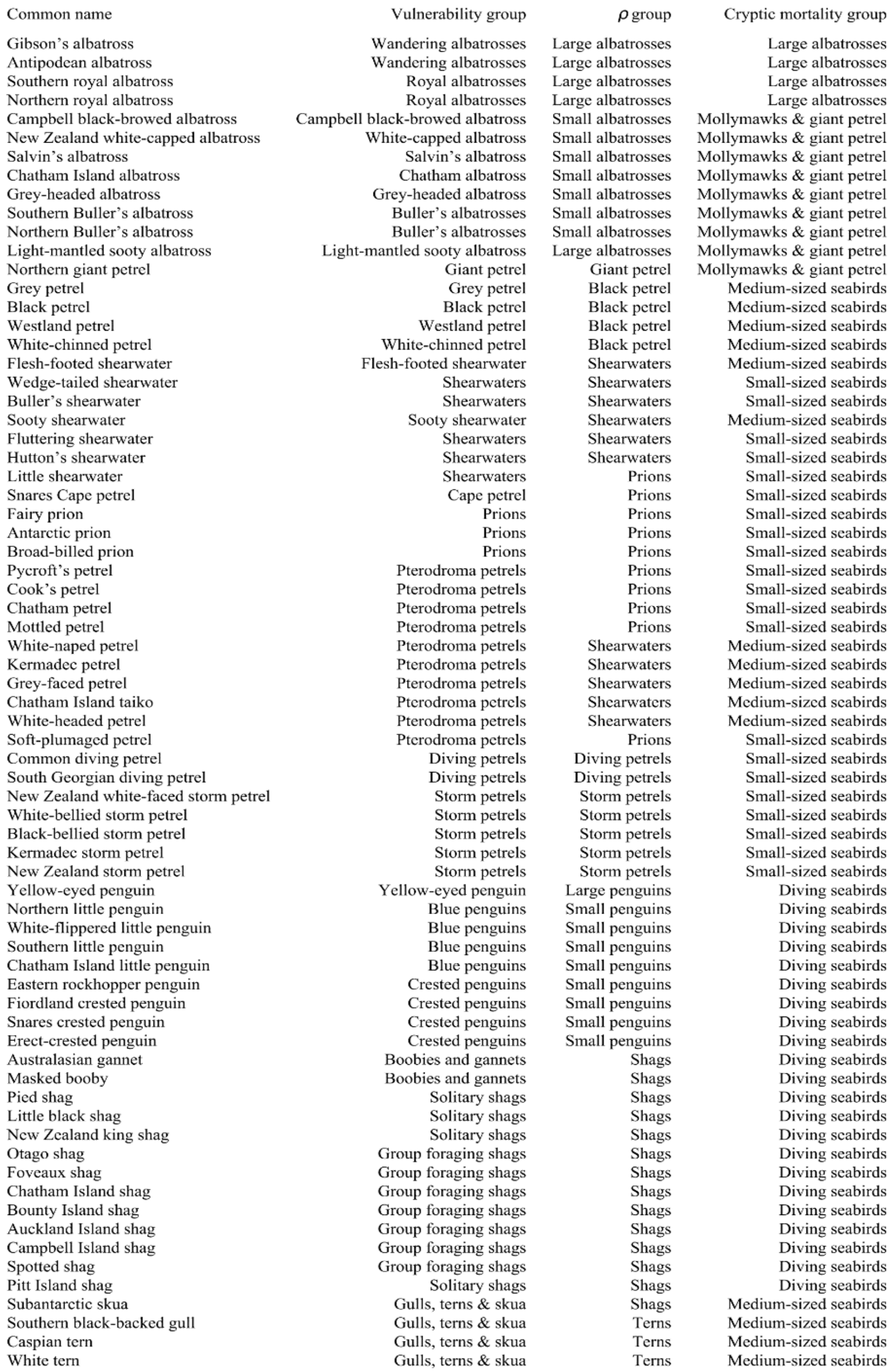
(g) Northern Buller's albatross

(h) Chatham Island albatross

(i) Antipodean albatross

Figure 3.3: Example plot showing the propagation of uncertainty from uncertain input parameters through to output estimates of risk for at-risk species in the NZSRA. Parameters contributing to high levels of output uncertainty become a priority for future research. Note that this figure derives from an out of date version of the NZSRA (Richard et al. 2015) and is provided as an example only. Legend is as follows: TWL = vulnerability in trawl fisheries; BLL = vulnerability in bottom-longline fisheries; SLL = vulnerability in surface-longline fisheries; SN = vulnerability in set-net fisheries; CM = cryptic mortality; A = age at first reproduction; S = adult survival; N_{BP} = breeding population size; P_B = proportion breeding.

Table 3.3: Species (vulnerability) groups and cryptic mortality groups used in the 2017 iteration of the NZSRA.



Common name	Vulnerability group	ρ group	Cryptic mortality group
Gibson's albatross	Wandering albatrosses	Large albatrosses	Large albatrosses
Antipodean albatross	Wandering albatrosses	Large albatrosses	Large albatrosses
Southern royal albatross	Royal albatrosses	Large albatrosses	Large albatrosses
Northern royal albatross	Royal albatrosses	Large albatrosses	Large albatrosses
Campbell black-browed albatross	Campbell black-browed albatross	Small albatrosses	Mollymawks & giant petrel
New Zealand white-capped albatross	White-capped albatross	Small albatrosses	Mollymawks & giant petrel
Salvin's albatross	Salvin's albatross	Small albatrosses	Mollymawks & giant petrel
Chatham Island albatross	Chatham albatross	Small albatrosses	Mollymawks & giant petrel
Grey-headed albatross	Grey-headed albatross	Small albatrosses	Mollymawks & giant petrel
Southern Buller's albatross	Buller's albatrosses	Small albatrosses	Mollymawks & giant petrel
Northern Buller's albatross	Buller's albatrosses	Small albatrosses	Mollymawks & giant petrel
Light-mantled sooty albatross	Light-mantled sooty albatross	Large albatrosses	Mollymawks & giant petrel
Northern giant petrel	Giant petrel	Giant petrel	Mollymawks & giant petrel
Grey petrel	Grey petrel	Black petrel	Medium-sized seabirds
Black petrel	Black petrel	Black petrel	Medium-sized seabirds
Westland petrel	Westland petrel	Black petrel	Medium-sized seabirds
White-chinned petrel	White-chinned petrel	Black petrel	Medium-sized seabirds
Flesh-footed shearwater	Flesh-footed shearwater	Shearwaters	Medium-sized seabirds
Wedge-tailed shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Buller's shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Sooty shearwater	Sooty shearwater	Shearwaters	Medium-sized seabirds
Fluttering shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Hutton's shearwater	Shearwaters	Shearwaters	Small-sized seabirds
Little shearwater	Shearwaters	Prions	Small-sized seabirds
Snares Cape petrel	Cape petrel	Prions	Small-sized seabirds
Fairy prion	Prions	Prions	Small-sized seabirds
Antarctic prion	Prions	Prions	Small-sized seabirds
Broad-billed prion	Prions	Prions	Small-sized seabirds
Pycroft's petrel	Pterodroma petrels	Prions	Small-sized seabirds
Cook's petrel	Pterodroma petrels	Prions	Small-sized seabirds
Chatham petrel	Pterodroma petrels	Prions	Small-sized seabirds
Mottled petrel	Pterodroma petrels	Prions	Small-sized seabirds
White-naped petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Kermadec petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Grey-faced petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Chatham Island taiko	Pterodroma petrels	Shearwaters	Medium-sized seabirds
White-headed petrel	Pterodroma petrels	Shearwaters	Medium-sized seabirds
Soft-plumaged petrel	Pterodroma petrels	Prions	Small-sized seabirds
Common diving petrel	Diving petrels	Diving petrels	Small-sized seabirds
South Georgian diving petrel	Diving petrels	Diving petrels	Small-sized seabirds
New Zealand white-faced storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
White-bellied storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Black-bellied storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Kermadec storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
New Zealand storm petrel	Storm petrels	Storm petrels	Small-sized seabirds
Yellow-eyed penguin	Yellow-eyed penguin	Large penguins	Diving seabirds
Northern little penguin	Blue penguins	Small penguins	Diving seabirds
White-flippered little penguin	Blue penguins	Small penguins	Diving seabirds
Southern little penguin	Blue penguins	Small penguins	Diving seabirds
Chatham Island little penguin	Blue penguins	Small penguins	Diving seabirds
Eastern rockhopper penguin	Crested penguins	Small penguins	Diving seabirds
Fiordland crested penguin	Crested penguins	Small penguins	Diving seabirds
Snares crested penguin	Crested penguins	Small penguins	Diving seabirds
Erect-crested penguin	Crested penguins	Small penguins	Diving seabirds
Australasian gannet	Boobies and gannets	Shags	Diving seabirds
Masked booby	Boobies and gannets	Shags	Diving seabirds
Pied shag	Solitary shags	Shags	Diving seabirds
Little black shag	Solitary shags	Shags	Diving seabirds
New Zealand king shag	Solitary shags	Shags	Diving seabirds
Otago shag	Group foraging shags	Shags	Diving seabirds
Foveaux shag	Group foraging shags	Shags	Diving seabirds
Chatham Island shag	Group foraging shags	Shags	Diving seabirds
Bounty Island shag	Group foraging shags	Shags	Diving seabirds
Auckland Island shag	Group foraging shags	Shags	Diving seabirds
Campbell Island shag	Group foraging shags	Shags	Diving seabirds
Spotted shag	Group foraging shags	Shags	Diving seabirds
Pitt Island shag	Solitary shags	Shags	Diving seabirds
Subantarctic skua	Gulls, terns & skua	Shags	Medium-sized seabirds
Southern black-backed gull	Gulls, terns & skua	Terns	Medium-sized seabirds
Caspian tern	Gulls, terns & skua	Terns	Medium-sized seabirds
White tern	Gulls, terns & skua	Terns	Medium-sized seabirds

3.3 MODEL INPUTS

3.3.1 STRUCTURAL INPUT: SPECIES VULNERABILITY GROUPS

Conceptually, species vulnerability is the probability that an individual animal encountering a fishing event will be captured or fatally injured in that encounter. Vulnerability includes both catchability (animals captured alive or dead) and cryptic mortality. Species are assigned to *species vulnerability groups* (hereafter *species groups*) within which physical and behavioural characteristics are assumed to be similar, such that a single vulnerability score (per fishery group) can be assigned per species group. In this way observed capture rates for abundant and/or commonly observed species serve to inform the estimation of catchability and vulnerability for all species in the same group, even species which for captures are rarely or never observed. Where species groups are not used (i.e., capture rates for every species are modelled independently) statistical estimation of the species vulnerability for rarely captured species is unconstrained (such that for example in Richard et al. (2011), vulnerability and total risk scores for rare bird species varied by more than two orders of magnitude). Species vulnerability groups currently applied in the NZSRA are shown in Table 3.3. Species groups applied in the NZMMRA are shown in Table 3.4.

Assigning species to species groups should be done with care, informed by expert knowledge of species behaviour influencing fishery interactions, to ensure that superficial physiological or taxonomic similarity within the group does not conceal significant behavioural differences between species that result in real differences in vulnerability.

Group assignments should be examined with reference to model diagnostics (e.g., Figures 3.6 and 3.9, below) and redefined as necessary to improve model fits. For example in Richard & Abraham (2013b) royal albatrosses and wandering albatrosses were grouped together. In 2013, visual examination of observed vs. expected capture patterns for these species revealed that the model was over-estimating capture rates for royal albatrosses and under-estimating capture rates of wandering albatrosses, evidently reflecting behavioural differences in the way these species react to fishing vessels. When the species group was subsequently split (Richard & Abraham 2015), visual fits improved markedly and the model estimated a significantly higher vulnerability for wandering albatrosses than for royal albatrosses.

Table 3.4: Species (vulnerability) groups used in the first (2017) iteration of the NZMMRA.

Species group	Common name
Whales	Antarctic blue whale
	Fin whale
	Pygmy blue whale
	Sei whale
	Humpback whale
	Southern right whale
	Sperm whale
	Bryde's whale
	Antarctic minke whale
	Pygmy right whale
	Dwarf minke whale
	Pygmy sperm whale
Blackfish	Killer whale Type A
	Long-finned pilot whale
	Short-finned pilot whale
	False killer whale
Beaked whales	Cuvier's beaked whale
	Shepherd's beaked whale
	Southern bottlenose whale
	Gray's beaked whale
	Spade-toothed whale
	Dense-beaked whale
	Andrews' beaked whale
	Hector's beaked whale
	Strap-toothed whale
Dolphins	Southern right whale dolphin
	Bottlenose dolphin
	Common dolphin
	Dusky dolphin
	Hourglass dolphin
	Hector's dolphin
	Māui dolphin
Pinnipeds	Southern elephant seal
	New Zealand sea lion
	New Zealand fur seal

3.3.2 FISHERY GROUPS

Non-target species capture rates are modelled separately within each of several broadly defined *fishing methods*. The NZSRA defines four such fishing methods: trawls, bottom longlines, surface longlines, and set nets; the NZMMRA defines also a fifth group, purse seines. All fishing effort is assigned to *fishery groups* within which the gear configuration and vessel behaviour is sufficiently consistent that species vulnerability estimates can be estimated and applied uniformly to all effort in the fishery group. Fishery groups are nested subsets of fishing methods.

Fishery group assignments should be informed by expert knowledge and based upon vessel characteristics known to affect non-target species interactions and capture rates, and defined with reference to variables universally stored in fishing effort databases (or otherwise recoverable such that all fishing events can be unambiguously assigned to groups). Variables used to distinguish between fishery groups are nearly always proxies for other underlying vessel characteristics, such that the means by which fishery groups are defined with reference to available data should utilise specific expert knowledge and should be investigated and iteratively adjusted with reference to the underlying data.

To illustrate, in the seabird risk assessment, trawl fishery groups are distinguished on the basis of vessel size and target species (as a proxy for gear configuration), and of on-board offal processing capability (i.e., affecting seabird attraction) (Table 3.5). As with the assignment of species to species vulnerability groups, there is an inherent trade-off between increased specificity in group assignments vs. decreased statistical power arising from fewer observed captures per group, such that fishery group assignments should be made also with reference to the underlying availability of data (to ensure adequate data in each group). For this reason groups in the NZMMRA are more broadly defined than in the NZSRA (i.e., because there are fewer marine mammal captures than seabird captures).

Table 3.5: Fishing methods and fishery groups used in current (2017) iteration of the NZSRA

Method	Group	Target species and vessel size
Trawl	Small inshore <17m	Targeting inshore species (including flatfish), or targeting middle-depth species (principally hoki, hake, or ling) on vessels less than 17m length
	Small inshore <28m	Targeting inshore species (including flatfish), or targeting middle-depth species (principally hoki, hake, or ling) on vessels more than 17m length and less than 28m length
	Southern blue whiting	Targeting southern blue whiting
	Scampi	Targeting scampi
	Mackerel	Targeting mackerel (primarily jack mackerel species)
	Squid	Targeting squid
	Large processor	Targeting middle-depth species, vessel longer than 28m, processing fish on board
	Large fresher	Targeting middle-depth species, vessel longer than 28m, with no processing on board
	Deepwater	Targeting deepwater species (principally orange roughy or oreos)
Bottom longline (BLL)	Bluenose	Targeting bluenose, and vessel less than 34m length
	Snapper	Targeting snapper, and vessel less than 34m length
	Ling and ribaldo	Targeting ling or ribaldo, and vessel less than 34m length
	Other small BLL vessels	Not targeting snapper, bluenose, ling or ribaldo, and vessel less than 34m length
	Large vessels without IWL	Vessel over 34m, without integrated weight line
	Large vessels with IWL	Vessel over 34m, with integrated weight line
Surface longline (SLL)	Swordfish	Targeting swordfish, and vessel less than 45m length
	Other small SLL vessels	Not targeting swordfish, and vessel less than 45m length
	Large vessels	Vessel 45m or longer
Set net	Set net	All set-net fishing
Trawl	Small inshore <17m	Targeting inshore species (including flatfish), or targeting middle-depth species (principally hoki, hake, or ling) on vessels less than 17m length

Fishery group assignments utilised in the current iteration of the NZSRA are shown in Table 3.5. Fishery group assignments utilised in the current iteration of the NZMMRA are shown in Table 3.6.

By its nature mitigation uptake is expected to reduce fishery group vulnerability; therefore vessels consistently utilising different mitigation configurations should be assigned to different fishery groups. For example, in Table 3.6 trawl fishery vessels using Sea Lion Exclusion Devices (SLEDs) are assigned to a different group from vessels not employing SLEDs.

Where mitigation uptake is uneven or unverifiable across a fleet, and/or not recorded in a standardised format in fishing effort databases, fishery group vulnerability will be poorly estimated and the effectiveness of the mitigation to reduce species risk will not be quantifiable in risk assessment outputs. Standardised mitigation reporting (in contrast to qualitative recording e.g., in observer logbooks) and the ability to verify uptake (e.g., via electronic monitoring) will increase the utility of the SEFRA method to detect mitigation efficacy and inform risk management decisions.

Table 3.6: Fishing methods and fishery groups used in the first (2017) iteration of the NZMMRA.

Method	Fishery	Annual Effort	Observed effort
Bottom longline	BLL	37 567	65 157
Purse seine	PS	1 285	1 481
Surface longline	SLL	2 611	18 299
Set net	SN	20 557	4 823
Trawl	Pelagic trawl	2 349	17 991
	Pelagic trawl (SLED)	547	1 645
	Squid trawl	1 415	20 913
	Squid trawl (SLED)	799	8 533
	Inshore trawl	48 340	9 522
	Other trawl	29 100	105 764

3.3.3 SPECIES INPUTS: SPATIAL DISTRIBUTIONS

To inform the calculation of *overlap* between species and fishing effort in equations (1) and (2), the spatial distribution of each species is mapped throughout the spatial domain of the risk assessment. Species distribution layers are defined such that the value in a particular cell represents the probability that an individual animal, selected at random from the population, is present in that cell at the moment of the fishing event. For each species the value of all cells sums to 1 across the spatial domain.

Because overlap is estimated per event (rather than based on cell-aggregated summaries of fishing effort) cell size is computationally unimportant but is necessarily consistent across all species so that the resulting vulnerability estimates are likewise comparable between species.

The New Zealand seabird and marine mammal risk assessments utilise species distribution maps assembled from multiple sources, including mapped distributions from vessel-based and aerial surveys, satellite tracking data, foraging ‘hotspots’ delineated using expert knowledge, density gradients as a function of distance from breeding colonies, and expert-based distributions assembled via ‘Delphi’ workshop methods. An example species distribution, for Gibson’s albatross, is shown in Figure 3.4.

To map the distributions of species for which direct observation and/or tracking is not feasible, spatial habitat models may be employed using spatially comprehensive environment data (e.g., SST, bathymetry, turbidity) as a proxy for species distribution. Successful methods of relating species distribution to underlying environmental data include the application of subjectively defined Relative Environmental Suitability models for widely distributed

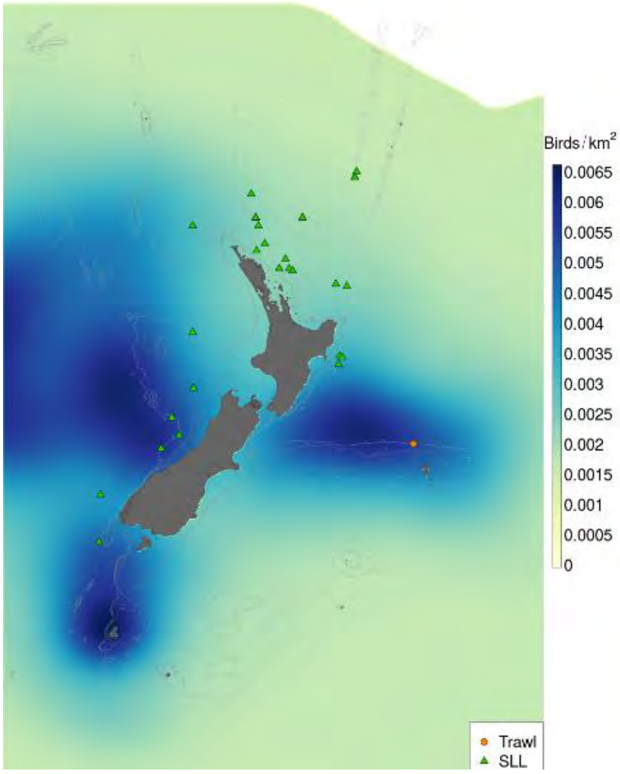


Figure 3.4: Spatial distribution layer (derived from global tracking data) for Gibson’s albatross. Capture events are also shown.

marine mammals (Kaschner et al. 2006, 2011) or the use of sophisticated multivariate statistical methods such as Boosted Regression Trees or State-Space modelling, fitted to fisheries or trawl survey catch data (Leathwick et al. 2006, Pinkerton et al. 2010). Note however that by nature spatial habitat models map the full range of the *potential species habitat*, which may be substantially larger than the actual realized distribution, especially where the actual distribution reflects historical range contraction associated with population decline, or complex behavioural patterns or lifecycle movements that cannot be captured using environmental proxies, as is often the case for marine mammals and seabirds.

In general, fish or invertebrate distributions will more often rely on environmental proxies, whereas protected species distributions will more often rely on tracking or aerial census. MPI is currently progressing work to improve estimation of marine mammal distributions as inputs to the NZMMRA (projects PRO2014-01 and PRO2017-10).

It is generally difficult to include statistical uncertainty in spatial data represented as maps; early iterations of the NZSRA either assumed that species distribution maps were precisely known, or represented uncertainty as a simple binary sensitivity between alternative maps. A superior approach is to subjectively define a normal distribution around the estimation of the overlap term O_{sg} in equation (2), with a CV reflecting the degree of confidence in the underlying spatial distributions. For example the confidence assigned to animal distributions defined from tracking studies or aerial surveys >> habitat distribution modelling >> maps derived from subjective expert knowledge. By applying spatial uncertainty to the O_{sg} term in equation (2) rather than the θ_{sg} term in equation (5), spatial uncertainty from species maps is not confounded with population size uncertainty affecting estimates of actual species density.

3.3.3.1 SEASONALLY VARIABLE DISTRIBUTIONS

Seasonally variable species distributions (e.g., for migratory species) can be used at whatever level of seasonal resolution the distribution data will support, without loss of statistical power.

The NZSRA currently applies two spatial distributions per year, i.e., breeding season and non-breeding season distributions, with the duration of each season defined individually for each species at the scale of months. In contrast, the CCSBT global risk assessment, which relied primarily on electronic tracking data to define global seabird distributions, split tracking data into four seasons (summer/autumn/winter/spring) for all seabird species alike.

Because catchability and vulnerability are estimated as a function of overlap across all fishing events simultaneously regardless of year or season, defining a higher number of seasonal distributions does not result in a loss of statistical power. The underlying assumption is that interaction rate is proportional to encounter rate regardless of season (i.e., vulnerability is constant throughout the year). However where seasonally variable animal behaviour results in changed vulnerability (e.g., if nesting seabirds target fishing boats more aggressively during the chick-rearing period) then it may be useful instead to estimate vulnerability in each season separately (i.e., using q_{zt} in place of q_z in equation (27)). Seasonally variable q should only be considered where sufficient data are available in each season to inform the estimation, and by testing the model's ability to discern seasonally variable vulnerability using model diagnostics (Figures 3.6 and 3.9 below).

3.3.3.2 TRANSIENT OR SEASONALLY ABSENT SPECIES

For highly migratory species that leave the spatial domain of the risk assessment entirely (e.g., New Zealand seabirds that leave the EEZ outside the breeding season), spatial distribution layers are not modified and species overlap O from equations (1) and (2) remains unchanged. Instead, the changed encounter rate is reflected by recording the proportional change to *available population size* as in equation (5).

3.3.4 BIOLOGICAL POPULATION SIZE

Risk is necessarily estimated with reference to a biologically meaningful estimate of population size N. Applications of the SEFRA method in New Zealand to date have estimated species level risk at the scale of the New Zealand breeding population (i.e., not considering transient species and not differentiating between local sub-populations) except in particular instances where locally important sub-populations have been specifically identified, and captures can be unambiguously assigned to that local population. To illustrate, in the NZSRA, since Richard & Abraham (2013b), risk to the small mainland population of yellow-eyed penguins is assessed separately from that to the large Snares Island population, and in the NZMMRA risk is estimated separately for Māui vs. Hector's dolphins. Note however that because vulnerability is an inherent property of the species and is estimated at the species group level in equation (18), there is no loss of statistical power if impact and risk is subsequently disaggregated and applied at the scale of smaller subpopulations. This option will be applied to regional subpopulations of Hector's dolphins in the review of the Threat Management Plan (see Chapter 6), and may be applied to other coastal marine mammal species in the next iteration of the NZMMRA.

For protected species populations, input estimates of biological population size N should utilise the most recent available estimates, e.g., derived from population census, mark-recapture, genetic mark-recapture, or other methods. Because captures are estimated with reference to the entire vulnerable population, estimates derived from breeding colony census must be scaled upwards to also include non-breeders, or, as in the NZSRA, the breeding and non-breeding populations are estimated separately and assigned their own spatial distributions, which are subsequently combined. All input distributions are defined using priors reflecting estimated uncertainty.

3.3.5 AVAILABLE POPULATION SIZE

The use of *available population size* $\mathbb{N}_i$ in equations (5)-(7) recognises that the number of individuals actually present in the spatial domain of the risk assessment at the moment of fishing event *i* may be different than the size of the biological population N against which impacts are evaluated.

The means by which available population size is reflected in the current NZSRA is by estimating, for every migratory species, the proportion of breeding and non-breeding seabirds that are present within the domain of the NZSRA in the breeding and non-breeding seasons. To illustrate, for a migratory bird species for which half of the population is absent from New Zealand waters during the non-breeding season, the available population $\mathbb{N}_i$ = 0.5 N, and expected number of captures associated with fishing events during the non-breeding is correspondingly reduced. In situations where an entire population leaves the spatial domain of the risk assessment on a seasonal basis, $\mathbb{N}_i$= 0 in that period.

Seasonal adjustments of this nature are necessary because the estimation of vulnerability occurs across all fishing events simultaneously. In the example of a migratory bird that is seasonally absent, if the $\mathbb{N}_i$ adjustment were not used, the model would nonetheless 'expect' captures on observed fishing events in the period when the bird is absent, and the effect of the recorded zero capture events would then depress the estimated vulnerability $q_z q_g$, leading to underestimation of capture rates and risk in the period when the bird is once again present.

Where animals present in New Zealand are merely a subset of a single globally distributed population (e.g., many cetacean species) the notion of a 'New Zealand population' may have no biological meaning; in these instances risk should be estimated with reference to the full global population, for which the presence of only a subset of that population in New Zealand waters at any given time is represented by estimating a permanently lower *available population size* $\mathbb{N}_i$ (i.e., $\mathbb{N}_i < N$ for all i).

Note also that in some instances it is possible to have an available population size $\mathbb{N}_i$ that is higher than the biological population N, for example if biological risk is evaluated with reference to a small local population, but observed capture rates reflect the presence of abundant transient individuals from other breeding populations outside the spatial domain of the risk assessment. This was the case for giant petrels in early iterations of the NZSRA (Waugh et al. 2009, Richard et al. 2011) in which giant petrel risk was artificially inflated because all captures were originally assumed to originate from a very small local population despite the presence of transient birds from an abundant overseas population.

Proportional adjustments in available population sizes for breeding and non-breeding populations in the current NZSRA are shown in Table 3.7. Similarly, for wide-ranging marine mammal species in the NZMMRA it is necessary to estimate what proportion of the population is present in the New Zealand domain at a given time.

Table 3.7: Biological and seasonally adjusted (i.e., non-breeding season) available population sizes applied in the 2017 iteration of the NZSRA.

Common name	Scientific name	Breeding period		Biological N	% staying in NZ
		Start	End		
Gibson's albatross	*Diomedea antipodensis gibsoni*	-	-	24 200	-
Antipodean albatross	*Diomedea antipodensis antipodensis*	-	-	17 900	-
Southern royal albatross	*Diomedea epomophora*	-	-	41 800	-
Northern royal albatross	*Diomedea sanfordi*	-	-	35 200	-
Campbell black-browed albatross	*Thalassarche impavida*	Aug	May	81 400	50
New Zealand white-capped albatross	*Thalassarche cauta steadi*	Nov	Aug	457 000	50
Salvin's albatross	*Thalassarche salvini*	Sep	Apr	139 000	10
Chatham Island albatross	*Thalassarche eremita*	Aug	May	17 700	2.5
Grey-headed albatross	*Thalassarche chrysostoma*	Sep	May	29 100	20
Southern Buller's albatross	*Thalassarche bulleri bulleri*	Dec	Aug	52 500	2.5
Northern Buller's albatross	*Thalassarche bulleri platei*	Oct	Jun	62 600	2.5
Light-mantled sooty albatross	*Phoebetria palpebrata*	Sep	Jun	32 200	20
Northern giant petrel	*Macronectes halli*	Aug	Feb	14 900	75
Grey petrel	*Procellaria cinerea*	Feb	Nov	190 000	2.5
Black petrel	*Procellaria parkinsoni*	Oct	Jul	17 900	0.5
Westland petrel	*Procellaria westlandica*	Mar	Dec	12 100	2.5
White-chinned petrel	*Procellaria aequinoctialis*	Nov	May	922 000	20
Flesh-footed shearwater	*Puffinus carneipes*	Sep	May	40 400	0.5
Wedge-tailed shearwater	*Puffinus pacificus*	Oct	May	149 000	0.5
Buller's shearwater	*Puffinus bulleri*	Sep	May	1 330 000	0.5
Sooty shearwater	*Puffinus griseus*	Nov	May	18 500 000	0.5
Fluttering shearwater	*Puffinus gavia*	Jul	Feb	781 000	80
Hutton's shearwater	*Puffinus huttoni*	Sep	Apr	363 000	2.5
Little shearwater	*Puffinus assimilis*	Apr	Nov	493 000	5
Snares Cape petrel	*Daption capense australe*	Nov	Feb	43 800	90
Fairy prion	*Pachyptila turtur*	Mar	Jan	6 420 000	15
Antarctic prion	*Pachyptila desolata*	Nov	Mar	3 170 000	15
Broad-billed prion	*Pachyptila vittata*	Feb	Jan	1 490 000	5
Pycroft's petrel	*Pterodroma pycrofti*	Oct	Apr	8 440	0
Cook's petrel	*Pterodroma cookii*	Sep	Apr	1 050 000	0.5
Chatham petrel	*Pterodroma axillaris*	Nov	Jun	868	0
Mottled petrel	*Pterodroma inexpectata*	Oct	May	1 180 000	0
White-naped petrel	*Pterodroma cervicalis*	Oct	May	178 000	0
Kermadec petrel	*Pterodroma neglecta*	-	-	20 200	-
Grey-faced petrel	*Pterodroma macroptera gouldi*	Mar	Jan	839 000	10
Chatham Island taiko	*Pterodroma magentae*	Sep	May	58	20
White-headed petrel	*Pterodroma lessonii*	Aug	May	993 000	10
Soft-plumaged petrel	*Pterodroma mollis*	Sep	May	13 300	0.5
Common diving petrel	*Pelecanoides urinatrix*	Mar	Jan	2 930 000	20
South Georgian diving petrel	*Pelecanoides georgicus*	Sep	Feb	208	0
New Zealand white-faced storm petrel	*Pelagodroma marina maoriana*	Sep	Apr	5 040 000	0.5
White-bellied storm petrel	*Fregetta grallaria grallaria*	Apr	Aug	3 550	100
Black-bellied storm petrel	*Fregetta tropica*	Oct	May	242 000	50
Kermadec storm petrel	*Pelagodroma albiclunis*	Jun	Dec	175	50
New Zealand storm petrel	*Pealeornis maoriana*	-	-	834	-
Yellow-eyed penguin	*Megadyptes antipodes*	Aug	Apr	6 540	100
Northern little penguin	*Eudyptula minor f. iredalei*	Jul	Feb	21 700	100
White-flippered little penguin	*Eudyptula minor f. albosignata*	Jul	Feb	6 740	100
Southern little penguin	*Eudyptula minor f. minor*	Jul	Feb	21 700	100
Chatham Island little penguin	*Eudyptula minor f. chathamensis*	Jul	Feb	21 700	100
Eastern rockhopper penguin	*Eudyptes chrysocome filholi*	Oct	May	203 000	5
Fiordland crested penguin	*Eudyptes pachyrhynchus*	Jul	Mar	13 300	50
Snares crested penguin	*Eudyptes robustus*	Sep	Feb	137 000	5
Erect-crested penguin	*Eudyptes sclateri*	Sep	Mar	403 000	50
Australasian gannet	*Morus serrator*	Aug	Mar	147 000	20
Masked booby	*Sula dactylatra*	-	-	775	-
Pied shag	*Phalacrocorax varius varius*	-	-	16 000	100
Little black shag	*Phalacrocorax sulcirostris*	Oct	Dec	4 060	100
New Zealand king shag	*Leucocarbo carunculatus*	Mar	Oct	629	100
Otago shag	*Leucocarbo chalconotus*	Aug	Mar	4 390	100
Foveaux shag	*Leucocarbo stewarti*	Aug	Mar	3 210	100
Chatham Island shag	*Leucocarbo onslowi*	Sep	Feb	1 190	100
Bounty Island shag	*Leucocarbo ranfurlyi*	Oct	Dec	412	100
Auckland Island shag	*Leucocarbo colensoi*	Nov	Mar	7 200	100
Campbell Island shag	*Leucocarbo campbelli*	Nov	Feb	7 090	100
Spotted shag	*Stictocarbo punctatus*	-	-	46 900	-
Pitt Island shag	*Stictocarbo featherstoni*	Sep	Feb	1 310	100
Subantarctic skua	*Catharacta antarctica lonnbergi*	Sep	Feb	1 620	50
Southern black-backed gull	*Larus dominicanus dominicanus*	Sep	Mar	6 800 000	100
Caspian tern	*Hydroprogne caspia*	Sep	Jan	3 010	100
White tern	*Gygis alba candida*	Sep	Apr	342	100

3.3.6 FISHING EFFORT DISTRIBUTIONS

Fishing effort is assigned to fishery groups as above and mapped in space. Note that the mathematical estimation of overlap in equation (1) is carried out for each individual fishing event and multiplied by the density of the species group in question at the particular location (equation (5)). Because fishing effort is not summarised spatially before calculating overlap, there is no need to consider the cell size at which fishing effort distributions are aggregated, except for display purposes.

An example fishing effort distribution is shown in Figure 3.. The intersection of the species and effort distributions (Figures 3.4 and 3.5) to estimate overlap is illustrated in Figure 3.6.

In New Zealand, most commercial fishing effort data are reported using spatially precise start and end locations per fishing event. However where fishing effort is reported only within larger statistical areas, it is necessary to assign all fishing events to specific points in space using logical assumptions (e.g., effort randomly distributed within statistical areas, or distributed as a function of proximity to land or ports, etc.). Because spatial overlap influences both the estimation of species and fishery group vulnerability from observed capture rates (equation (18)) and also the subsequent estimation of total captures including in unobserved effort (equation (16)) it is worthwhile to expend effort at the outset to define or model the distribution of fishing effort as accurately as possible.

Translating individual fishing events into mathematical estimates of overlap in equation (1) requires decisions about the units in which effort is expressed, e.g., numbers of deployments vs. length of trawls for trawl fisheries, or numbers of hooks vs. numbers of deployments for longlines. These decisions should be made with care, utilising expert knowledge of seabird-fishery interactions, and informed by exploration of the data to determine what units of effort most effectively model observed capture rates.

Standard units in which effort events are expressed in the MMRA are shown in the legend of Table 3.6. Unsurprisingly effort is expressed with reference to kilometres of net for setnets, and numbers of hooks for longline fisheries; but for trawl fisheries effort is expressed with reference to the number of hauls only, independent of distance or duration (suggesting that most protected species captures occur at

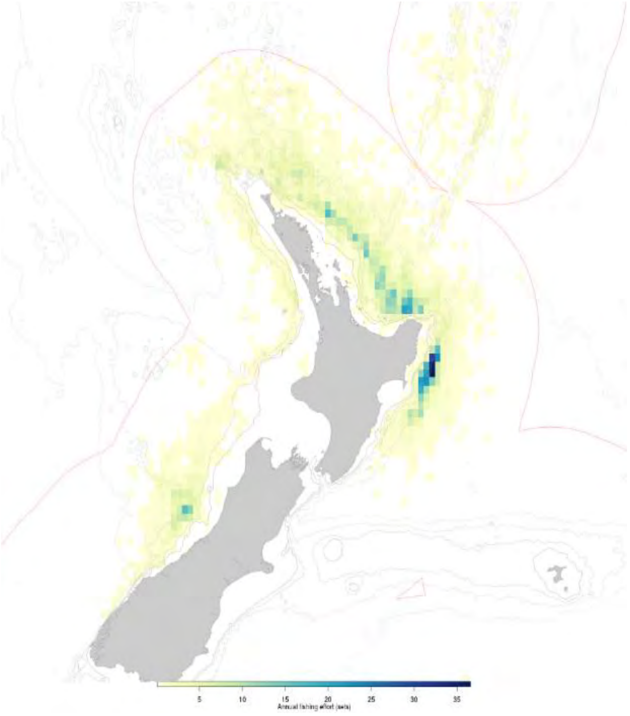


Figure 3.5: Fishing effort spatial distribution for the small (domestic) SLL fishery group not targeting swordfish, 2005–06 to 2014–15.

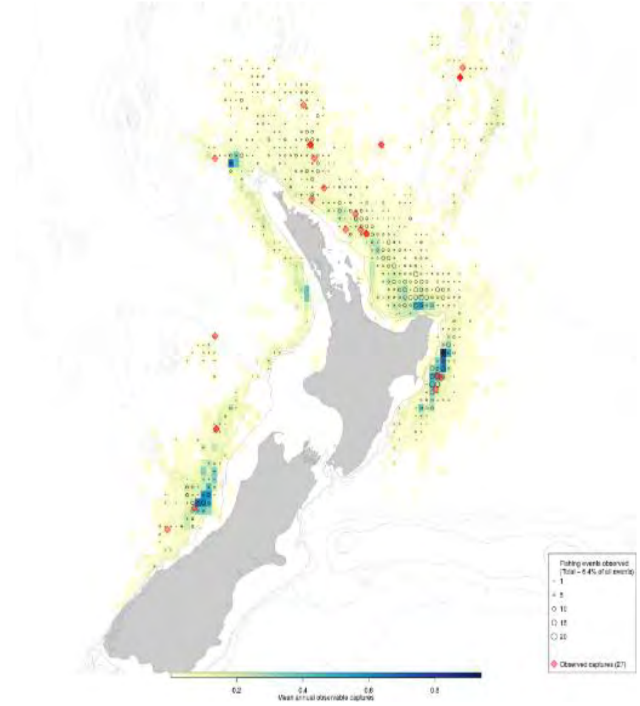


Figure 3.6: Overlap *O* of the distribution of Gibson's albatross (Figure 3.4) with the small (domestic) SLL fishery group (Figure 3.5). Black circles denote observed fishing effort. Capture events are also shown. Captures are expected to occur in space proportional to the intensity of the overlap in that location. Examination of expected vs. observed patterns of capture events in space is a primary diagnostic of model fit. This example fits poorly (i.e., captures occur disproportionately in the north in an area of lower overlap), suggesting the need to re-examine spatial inputs (species distributions) or model structural assumptions (e.g., fishery group definitions).

the time of the set or haul, rather than the tow). In contrast, the SEFRA method applied to fish or benthic impacts will by necessity reflect duration or length of tow for trawl fisheries, except perhaps for midwater or seamount trawl fisheries in which fishers target individual acoustic marks in a highly selective way. These decisions should be informed by experts with knowledge of the operational factors affecting vessel behaviour in the particular fisheries in question, and tested with reference to the data.

3.3.6.1 PARTIALLY OBSERVED FISHING EVENTS

The SEFRA method estimates species and fishery group catchability as a function of observed capture rates and overlap on the observed subset of the fishing effort data (equation (18)). Importantly, in some instances only a subset of a particular fishing event may be observed, effectively reducing observed fishing intensity α'_i in equation (1), (e.g., if a fisheries observer observes only a portion of a longline haul coming on board the vessel, and is off duty or occupied with other duties for the other portion). For this reason it is important that observer databases record what proportion of the event is observed, and that observed capture events distinguish between 'on duty' captures (i.e., caught during the observation period) vs. 'off duty' captures reported independently by the vessel. Whether off-duty captures are included in the estimation of catchability (hence vulnerability and risk) relies on assumptions about the reliability of vessel-reported capture data when an observer is not present to verify. Whether or not off-duty captures are used, accurate estimation is only possible if on-duty vs. off-duty captures are clearly distinguished in fishery databases (not merely in observer comments), and the observed proportion of each fishing event α'_i is recorded.

3.3.6.2 UNIDENTIFIED CAPTURES

Reliance on observed captures data creates a strong imperative to ensure that taxonomic ID by fisheries observers is accurate or subsequently verified by necropsy. However because estimation of species catchability in equation (18) occurs at the species group rather than the individual species level, taxonomic resolution below the level of species group is not required (except for example where observer data is also used to inform species distribution mapping). The taxonomic resolution and reliability of observer data should thus be considered in the stage at which species groups are being defined (i.e., there is no benefit in defining species groups at a finer level of taxonomic resolution than the observed captures can support).

3.3.7 YEARS' OBSERVED FISHING EFFORT DATA USED TO ESTIMATE CATCHABILITY

Because risk assessment approaches are designed for application to data-poor problems, there is an imperative in the estimation of catchability and vulnerability to include as much data as possible. At the same time, implicit in the assignment of fishing events to fishery groups is the underlying assumption that factors affecting capture rates by all fishing events in the same group are similar (or at least indistinguishable) within the fishery group. This assumption is violated in situations where vessels have changed their gear, or adopted mitigation measures, or otherwise changed their at-sea behaviour in ways that would be expected to change the probability of capture and/or cryptic death per encounter with non-target species.

Decisions about which years' data should be used in the estimation of species and fishery group vulnerability in equation (18) should be taken with care, with reference to available data indicative of observed capture rates, and informed by experts with relevant knowledge of fishery gear and at-sea operations and the history of changed practices affecting interactions with non-target species. Where a step-change in capture rates is likely (i.e., corresponding to new gear technology or new imposed regulations) data use should be restricted to the subset of the historical data representing current practice, or fishery data before and after the change should be assigned to different fishery groups. In the latter instance it may be possible to quantify the effect of the change on capture rates empirically, by comparing vulnerability estimates between the groups.

In the update of the NZSRA the AEWG considered as a sensitivity suggestions that deepwater fishery groups should be limited to fishing events post-2010, when new mitigation requirements were imposed and revised offal discard practices were widely implemented. For other fishery groups at present there are not sufficient data to evaluate whether or not capture rates have changed sufficiently to warrant limiting the input fishery data in this way. The time period over which observed capture rate data is used to estimate vulnerability in the NZSRA are shown in Table 3.5. Due to a lack of data, the current

NZMMRA uses the full time period from which data are available.

3.3.7.1 TRACKING FISHERY PERFORMANCE AFFECTING VULNERABILITY OVER TIME

Because of the imperative to include as much data as possible, in the absence of an identifiable step change in fishing practice the SEFRA method is not well suited for tracking changing catchability over time (i.e., indicative of mitigation uptake or voluntarily changed at-sea practices). To detect change of this nature it is necessary to test alternate structural assumptions, i.e., running sensitivities using observer data from different time periods, and comparing the resulting estimates of catchability, vulnerability, and risk. (In contrast, changing spatio-temporal distributions of fishing effort are manifested in overlap rather than vulnerability, so are immediately apparent and easily tracked over time).

Furthermore because vulnerability estimation in equation (18) is integrated across all fishery groups simultaneously and informed by input priors that reflect information other than observed capture rates, changes in the estimated vulnerability can arise from multiple sources other than observed changes in the capture rate in the fishery group in question.

Where tracking changed performance over time in particular fisheries or subsets of fisheries is an imperative, it is necessary to develop dedicated tools for this purpose, i.e., to define particular queries and run sensitivities in which changed outputs arise only from the fishery in question while other inputs are held constant. MPI is progressing work to develop this capability (project PRO2016-06 and SEA2016-30).

3.3.8 YEARS' FISHING EFFORT DATA TO REPRESENT CURRENT EFFORT AND RISK

Once species and fishery group vulnerability have been estimated by the model described in equation (18), there is no longer an imperative to maximise the use of fishing effort data in the subsequent estimation of current impact on a species- and fishery-group-specific basis in equations (5)–(9). Instead, it is important to use the best available proxy for 'current' or expected future fishing effort. Generally the recent past is considered the best proxy for the immediate future, but where fishing effort trends are changing rapidly or future changes can be forecast (e.g., reflecting changed TACs, management boundaries or fleet composition) it may be worthwhile to apply alternative assumptions, or generate hypothetical spatial effort scenarios on a case by case basis.

As a default the NZSRA and NZMMRA use the most recent three years' fishing effort data to approximate the 'current' distribution of effort, and to estimate corresponding 'current' impact and risk.

3.3.9 CRYPTIC MORTALITY

The modelling step of the SEFRA method in equation (18) fits to data indicative of total observable captures. However biological risk is a function of deaths, not captures; the relationship between captures and FRDs is reflected in the estimation of cryptic mortality and live release survival rates in equations (10)–(11). Input parameters to inform these equations are almost always highly uncertain. Often some data may exist for the *live release rate* r_{sg}, but data to better estimate the *cryptic mortality multiplier* k_{sg} and *live release survival rate* L_{sg} are by nature difficult to obtain, generally requiring dedicated research projects. In the absence of data, it is necessary to estimate these parameters outside the model using expert knowledge, reflecting uncertainty as input priors.

Scientists and other technical experts are often reluctant to provide numerical estimates where the answers are highly uncertain, citing lack of data. But failure to explicitly consider cryptic mortality and live release survival within protected species risk assessments constitutes an implicit adoption of extreme values (0 or 1) with absolute certainty; this approach is far less defensible than applying subjective estimates with explicit priors reflecting actual uncertainty. Inclusion of highly uncertain parameters based on expert knowledge serves to illustrate for managers the real consequences of the current lack of knowledge regarding cryptic mortality, and creates positive incentives for fishers, both to modify at-sea behaviour (e.g., improved protected species handling protocols at sea to increase live release survival) and to collect better data so that improved performance is reflected in reduced risk. Furthermore, in an integrated Bayesian multi-species model, ignoring these parameters may force the model to adopt skewed estimates of other important parameters in order to fit model constraints. For these reasons inclusion of even highly subjective parameter estimates in equations (10) and (11) is essential.

Experts who may initially profess their inability to estimate unknown parameters often find that collectively they 'know' far more than they expect, when confronted with the consequences of failing to provide an estimate (i.e., many experts are reluctant to propose a 'correct' estimate but quick to reject one that they 'know' to be 'wrong'). To capture this tendency effectively, highly uncertain subjective estimates are best elicited in a structured workshop setting, or via Delphi methods (e.g., as used in the 2016 NZMMRA).

3.3.9.1 CRYTIC MORTALITY GROUPS

Similar to species vulnerability groups, species are assigned to cryptic mortality groups, reflecting groups of species that are expected to interact with fishing gear in similar ways that will affect cryptic mortality rates. Cryptic mortality groups are more broadly defined than species vulnerability groups.

In the NZSRA, all seabird species are assigned to one of five such groups on the basis of body mass (affecting the amount of forward momentum with which they may be expected to interact with trawl warps and/or wing length that affects likelihood of warp entanglement) and also diving ability; see Table 3.3.

In the NZMMRA, cryptic mortality groups reflect body size and foraging behaviour affecting likely interactions with vessels (e.g., large toothed whales are considered separately from large baleen whales because depredation behaviour may lead to substantially increased entanglement risk in longlines). Cryptic mortality parameters are applied at the level of the five broadly defined fishing methods in Table 3.6.

3.3.9.2 INPUT PARAMETER DISAGGREGATION FOR IMPOROVED ESTIMATION OF CRYTPIC MORTAILITY

Where protected species may interact with fishing vessels in a variety of different ways, refined estimation of cryptic mortality rates is greatly aided by disaggregating the input parameters to distinguish between different types of interactions, to make maximum use of available data. The power of this approach is illustrated below with reference to the NZSRA, for which the most recent iteration estimates and applies different cryptic mortality parameters for each fishery group.

3.3.9.2.1 SEABIRDS IN TRAWL FISHERIES

In the first application of cryptic mortality within the NZSRA, Sharp et al. (2011) disaggregated the estimation of cryptic mortality multiplier k_{sg} in trawl fisheries as follows.

- Captures and/or mortality events are assumed to arise from three types of interaction:
  - Net captures
  - Surface warp strikes (bird resting or hovering at surface is overtaken and potentially entangled/drowned by a moving warp)
  - Aerial warp strikes (a flying bird strikes a warp under its own forward momentum).
- Warp captures vs. net captures are recorded separately by fisheries observers; using these data the estimated proportion of net captures can be estimated separately for each cryptic mortality group and fishery group, and applied to estimate group-specific cryptic mortality rates, as follows:
  - For net captured birds:
    - Live releases are recorded by fisheries observers; these data are used to estimate the live release rate, $r_{sg} - net$.
    - Live release survival $L_{sg} - net$ is estimated subjectively or requires dedicated research projects (e.g., banding or radio-tracking of live released birds)
    - The cryptic mortality multiplier $k_{sg} - net$ (reflecting drowned or injured birds that drop out of the net uncounted) is estimated subjectively or requires dedicated observation.
  - For warp captured birds:
    - All warp captures are assumed to arise from surface warp strikes.
    - No warp captured birds are assumed to be released alive ($r_{sg} - warp$ = 0)
    - The surface strike cryptic mortality multiplier $k_{sg} - surf$ is estimated relative to observed surface captures based on dedicated research projects (e.g., 'corpse catchers) or warp strike observational studies (e.g., Watkins et al. 2010, Abraham 2010)
    - The aerial strike cryptic mortality multiplier $k_{sg} - air$ is estimated relative to surface captures, applying surface: aerial warp strike ratios and subjective estimates of the fate of aerial warp strikes from dedicated observational studies elsewhere (Watkins et al. 2008). These could be productively

updated to also include use of more recent data (e.g., Parker et al. 2013).

The sequence by which disaggregated cryptic mortality parameters for trawl fisheries are combined to generate a total fisheries related deaths multiplier $\varkappa_{sg}$ as in equation (11) is displayed below in Figure 3.7.

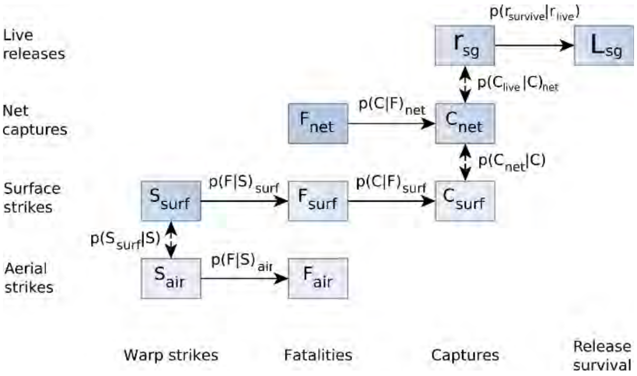


Figure 3.7 Transition probabilities by which the total fisheries-related deaths multiplier $\varkappa_{sg}$ (including live release survival as in equation (10)) is estimated for seabirds in trawl fishery groups.

Cryptic mortality rates consistent with this framework (but without incorporating live releases or distinguishing between different trawl fishery groups) were adopted in the 2013 iteration of the NZSRA (Richard & Abraham 2013b). The full framework was adopted in the 2017 iteration (summarised in Chapter 8). Importantly, the disaggregated cryptic mortality parameters in equation (10) including priors to represent uncertainty are incorporated as separate inputs into the integrated Bayesian multi-species risk model, rather than estimated outside the model and summarised as a single multiplier $\varkappa_{sg}$ from equation (11). In this way posteriors arising from the fitted model will help to refine poorly estimated cryptic mortality or live release parameters and/or to indicate where dedicated research projects may be useful to reduce uncertainty.

Utilisation of a 'corpse catcher' on trawl warps may provide empirical data to better estimate the rate at which fatal surface warp strikes result in an observed capture (i.e., $p(C|D_{surf}$ in Figure 3.7).

3.3.9.2.2 SEABIRDS IN LONGLINE FISHERIES

From the 2013 iteration the NZSRA has applied a total fisheries related deaths multiplier $\varkappa_{sg}$ for all longline fisheries, based on a single dedicated observational study in surface longline fisheries (Brothers et al. 2010). This approach can be substantively improved, e.g., by re-examining the Brothers et al. (2010) dataset to distinguish between species cryptic mortality groups, and by applying distinct assumptions regarding the fate of birds captured on the set vs. the soak vs. the haul (i.e., $k_{sg} - set$ will be higher than $k_{sg} - haul$, and live releases would be applied to haul-captured birds only). Furthermore, the use of these data primarily from global high seas SLL fisheries to estimate cryptic mortality in domestic SLL fishery groups, and the extension of these results also to BLL fishery groups, is untested.

From the 2017 iteration the NZSRA incorporates live release rate (separately for BLL vs. SLL, using New Zealand-specific data) and live release survival (subjectively estimated with high uncertainty). A dedicated research project is in the planning stages using dead geese and ducks as proxies for large and medium seabirds caught on the set, to better estimate $k_{sg} - \boldsymbol{set}$.

Pierre et al. (2015) make specific further recommendations for improvement of cryptic mortality parameter estimation.

3.3.9.3 ESTIMATING CHANGING CRYPTIC MORTALITY OVER TIME

Because cryptic mortality multipliers have a direct and potentially dramatic effect on total FRDs in equations (10) and (11), but are not necessarily reflected in observed capture events by which species vulnerability is estimated in equation (18), it is plausible that changed fishery practices affecting cryptic mortality and/or live release rates may occur without any corresponding change in observed capture rates, hence vulnerability and risk. If such changes are likely then the factors underlying estimation of cryptic mortality need to be examined in a temporally explicit way, so that constant capture rates don't potentially mask substantially changing death rates.

To illustrate, in trawl fisheries, seabird net captures will accrue a fairly low fisheries-related deaths multiplier (likely less than 2) because relatively few diving birds are thought to drown but fall out of the net uncounted ($k_{sg} - net$ is low), and a substantial proportion of flying birds entrapped by the meshes on the outside of the net are released alive and may survive ($r_{sg} - net$ and $L_{sg} - net$ are non-zero). In contrast, warp captures may accrue a very high cryptic mortality rate because: i) surface struck birds dragged underwater and drowned on the warps are only recovered if their bodies are subsequently impaled on a sprag or otherwise entangled in the gear; ii) aerial warp strikes may result in fatal injuries such broken wings, with no mechanism for body recovery leading to a recorded capture; and iii) there are no warp captured birds released alive. For these reasons a capture on the warp implies a higher number of actual deaths, hence greater risk, relative to a capture in the net.

It is therefore possible that changes to seabird mitigation and offal discard practices over time that have the effect of shifting captures from the warp to the net could occur with little to no observable change in estimated capture rate and vulnerability, effectively disguising a substantial reduction in total FRDs and species risk if changes to cryptic mortality and total fisheries related deaths multipliers were considered. This effect may have occurred in some New Zealand deepwater fisheries, for which there is an increasing trend in the proportions of net captures and of live released birds since changed mitigation and offal management practices began to be adopted from around 2005 (Figure 3.8). It is likely that this trend reflects a shift in the species composition of captured birds – away from mollymawk species primarily caught on the warp, and toward medium sized and diving birds, more often caught in the net. The 2017 iteration of the NZSRA applies the observed ratio of net: warp captures for different fishery groups individually, and estimates group-specific fisheries-related deaths multipliers at the level of each combination of cryptic mortality species group x fishery group in Tables 3.3 and 3.5. This will have the likely effect of reducing FRD multipliers for those (well-observed) fisheries and species for which the proportion of net captures has increased relative to warp captures, and increasing the uncertainty associated with FRD multipliers for other poorly observed fishery groups.

A similar modification should be considered in future to distinguish between SLL and BLL fishery groups based on the proportion of captures on the set vs. on the haul.

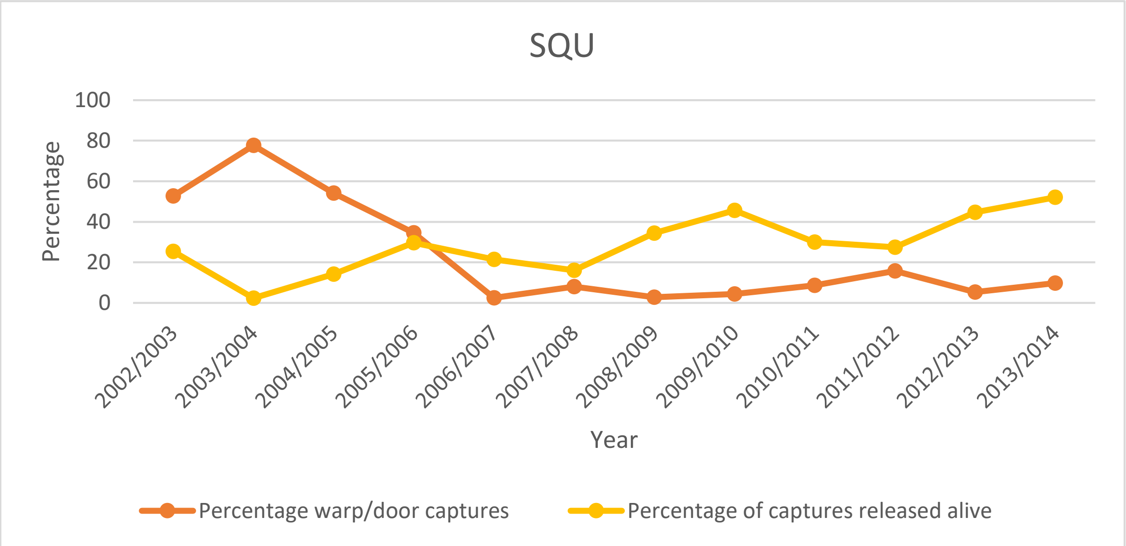


Figure 3.1: Declining proportion of seabird captures on the warp vs. on the net, and increasing proportion of seabirds released alive, in the well-observed squid fishery, corresponding to the implementation of changed practices from approximately 2003–05. Changes of this kind may result in significant reduced cryptic mortality multipliers (hence total fishery-related deaths and risk) even while capture rates remain unchanged.

3.3.10 DEMOGRAPHIC AND BIOLOGICAL PARAMETERS

3.3.10.1 SPECIES BIOLOGICAL INPUTS

Biological parameters are derived from available data or published proxies and defined as input distributions reflecting uncertainty. The SEFRA method applied to protected species requires sufficient biological parameter inputs to inform the estimation of $r_{max,}$, for use in equation (30). In the 2017 iteration of the NZSRA these include age at reproduction and adult survival, both of which are in turn derived from allometric relationships with body mass (Chapter 8). In contrast the NZMMRA uses published literature values for $r_{max,}$; other future applications of the SEFRA framework (e.g., for non-target fish or benthic invertebrates) will use alternative means of representing intrinsic species productivity to derive a MIST as in equation (29).

3.3.10.2 POPULATION DEMOGRAPHIC INPUTS

As described above, population monitoring data may be used to define constraints on total fisheries-related deaths within an integrated model (as in the 2016 NZSRA). In this way the SEFRA method allows utilisation of all available biological, demographic, and fisheries observer data to inform estimates of fisheries impact and risk simultaneously across all fishery groups and species groups.

It is important however to distinguish between the *taxonomic / ideal* biological parameters affecting species productivity and the estimation of $r_{max,}$ (above) vs. *actual/ realised* parameters specific to the impacted population in question. The former inputs represent intrinsic characteristics of the species and may legitimately be sourced from published data from overseas populations, or derived from allometric and life history relationships for the species in question (as in the 2016 NZSRA), or estimated by analogy with similar proxy species. In contrast, demographic parameters used to constrain fisheries impacts must necessarily come from direct observations of the particular impacted population, and must be both reliable and current (i.e., reflective of the same time period over which fisheries effort data are included in the risk assessment). To illustrate, in the 2016 NZSRA, adult survival S appears in both calculation pathways of Figure 3.1, informing the estimation of r_{max} via the left-hand path and constraining total fishery-related deaths via the right-hand path. This model distinguishes between the 'taxonomic' (un-impacted ideal) adult survival S_{tax} affecting estimation of r_{max} on the left, vs. the 'actual/realised' adult survival S_{act} for the impacted population in question, to constrain impact estimates on the right. Using demographic monitoring data to constrain impacts within the Bayesian model is a powerful innovation but should applied cautiously and only using quality data. Adoption of this innovation within the SEFRA method creates powerful incentive to fund and deliver population monitoring research to better inform fisheries risk assessment.

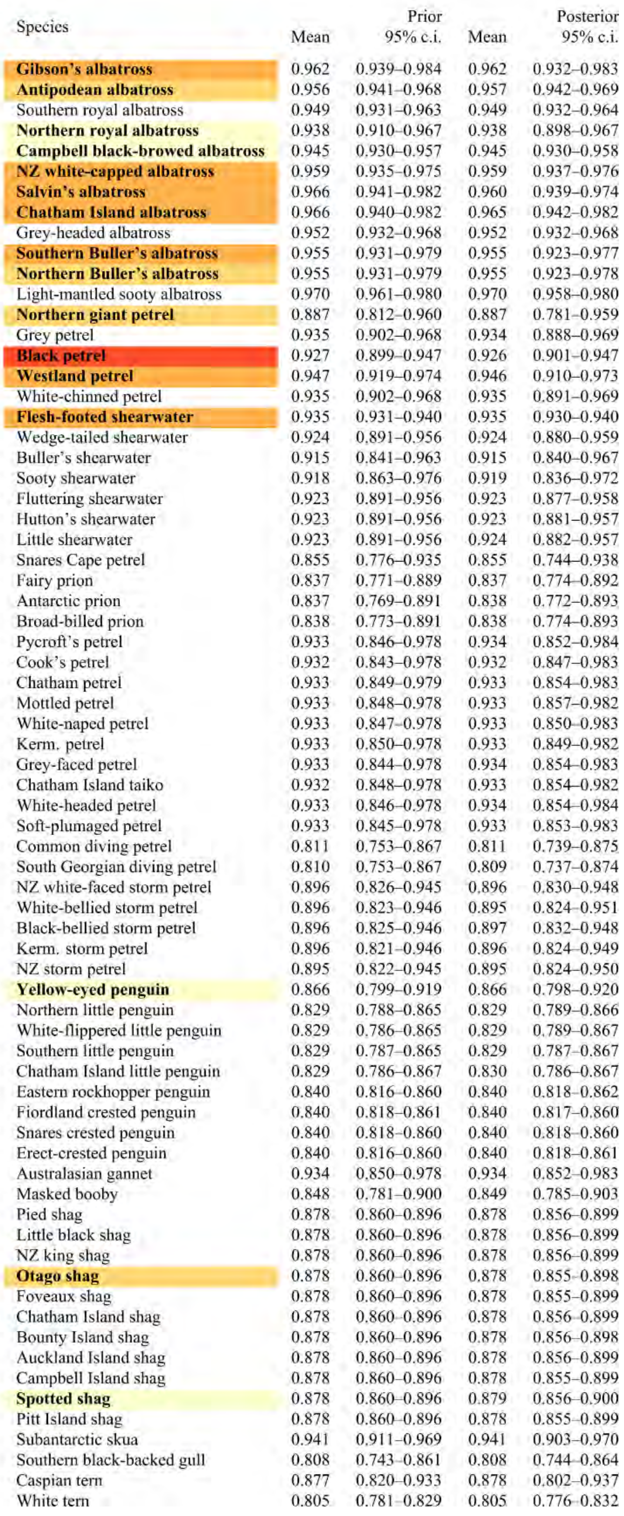
Populations for which adult survival is used to constrain total fishery-related deaths in the 2017 NZSRA, and the source of demographic parameter estimates used to define this constraint, are shown in Table 3.8.

3.3.11 MODEL DIAGNOSTICS

A primary means of testing spatial parameter inputs and structural assumptions and evaluating model fit is to examine spatial patterns of expected vs. observed captures on a species- and fishery-group-specific basis, as in Figure 3.6. These maps should be produced and evaluated routinely for every combination of species group x fishery group that produces substantial risk for any at-risk species (e.g., highlighted in Table 3.2). Where spatial fits are good, observed captures should show the same spatial pattern as the underlying observed overlap. Poor spatial fits should prompt further investigation either of spatial data inputs (i.e., animal distribution layers) or structural assumptions (e.g., species and fishery group definitions, seasonal variation in available population size), which may be iteratively adjusted and re-evaluated until spatial fits improve.

Similarly, expected vs. observed capture estimates should be evaluated across all fishery group x species group combinations simultaneously, as in Figure 3.9. Outliers prompt further investigations.

Table 3.8: Realised adult survival S_{actual}, used to constrain total fishery related deaths in the integrated model of the 2017 NZSRA. Annual fishery-related deaths are constrained to be less than 1 minus adult survival (D < (1-S)). This is a precautionary constraint, allowing that all deaths are attributable to fisheries (i.e., neglecting natural mortality).



Species	Prior		Posterior	
	Mean	95% c.i.	Mean	95% c.i.
Gibson's albatross	0.962	0.939–0.984	0.962	0.932–0.983
Antipodean albatross	0.956	0.941–0.968	0.957	0.942–0.969
Southern royal albatross	0.949	0.931–0.963	0.949	0.932–0.964
Northern royal albatross	0.938	0.910–0.967	0.938	0.898–0.967
Campbell black-browed albatross	0.945	0.930–0.957	0.945	0.930–0.958
NZ white-capped albatross	0.959	0.935–0.975	0.959	0.937–0.976
Salvin's albatross	0.966	0.941–0.982	0.960	0.939–0.974
Chatham Island albatross	0.966	0.940–0.982	0.965	0.942–0.982
Grey-headed albatross	0.952	0.932–0.968	0.952	0.932–0.968
Southern Buller's albatross	0.955	0.931–0.979	0.955	0.923–0.977
Northern Buller's albatross	0.955	0.931–0.979	0.955	0.923–0.978
Light-mantled sooty albatross	0.970	0.961–0.980	0.970	0.958–0.980
Northern giant petrel	0.887	0.812–0.960	0.887	0.781–0.959
Grey petrel	0.935	0.902–0.968	0.934	0.888–0.969
Black petrel	0.927	0.899–0.947	0.926	0.901–0.947
Westland petrel	0.947	0.919–0.974	0.946	0.910–0.973
White-chinned petrel	0.935	0.902–0.968	0.935	0.891–0.969
Flesh-footed shearwater	0.935	0.931–0.940	0.935	0.930–0.940
Wedge-tailed shearwater	0.924	0.891–0.956	0.924	0.880–0.959
Buller's shearwater	0.915	0.841–0.963	0.915	0.840–0.967
Sooty shearwater	0.918	0.863–0.976	0.919	0.836–0.972
Fluttering shearwater	0.923	0.891–0.956	0.923	0.877–0.958
Hutton's shearwater	0.923	0.891–0.956	0.923	0.881–0.957
Little shearwater	0.923	0.891–0.956	0.924	0.882–0.957
Snares Cape petrel	0.855	0.776–0.935	0.855	0.744–0.938
Fairy prion	0.837	0.771–0.889	0.837	0.774–0.892
Antarctic prion	0.837	0.769–0.891	0.838	0.772–0.893
Broad-billed prion	0.838	0.773–0.891	0.838	0.774–0.893
Pycroft's petrel	0.933	0.846–0.978	0.934	0.852–0.984
Cook's petrel	0.932	0.843–0.978	0.932	0.847–0.983
Chatham petrel	0.933	0.849–0.979	0.933	0.854–0.983
Mottled petrel	0.933	0.848–0.978	0.933	0.857–0.982
White-naped petrel	0.933	0.847–0.978	0.933	0.850–0.983
Kerm. petrel	0.933	0.850–0.978	0.933	0.849–0.982
Grey-faced petrel	0.933	0.844–0.978	0.934	0.854–0.983
Chatham Island taiko	0.932	0.848–0.978	0.933	0.854–0.982
White-headed petrel	0.933	0.846–0.978	0.934	0.854–0.984
Soft-plumaged petrel	0.933	0.845–0.978	0.933	0.853–0.983
Common diving petrel	0.811	0.753–0.867	0.811	0.739–0.875
South Georgian diving petrel	0.810	0.753–0.867	0.809	0.737–0.874
NZ white-faced storm petrel	0.896	0.826–0.945	0.896	0.830–0.948
White-bellied storm petrel	0.896	0.823–0.946	0.895	0.824–0.951
Black-bellied storm petrel	0.896	0.825–0.946	0.897	0.832–0.948
Kerm. storm petrel	0.896	0.821–0.946	0.896	0.824–0.949
NZ storm petrel	0.895	0.822–0.945	0.895	0.824–0.950
Yellow-eyed penguin	0.866	0.799–0.919	0.866	0.798–0.920
Northern little penguin	0.829	0.788–0.865	0.829	0.789–0.866
White-flippered little penguin	0.829	0.786–0.865	0.829	0.789–0.867
Southern little penguin	0.829	0.787–0.865	0.829	0.787–0.867
Chatham Island little penguin	0.829	0.786–0.867	0.830	0.786–0.867
Eastern rockhopper penguin	0.840	0.816–0.860	0.840	0.818–0.862
Fiordland crested penguin	0.840	0.818–0.861	0.840	0.817–0.860
Snares crested penguin	0.840	0.818–0.860	0.840	0.818–0.860
Erect-crested penguin	0.840	0.816–0.860	0.840	0.818–0.861
Australasian gannet	0.934	0.850–0.978	0.934	0.852–0.983
Masked booby	0.848	0.781–0.900	0.849	0.785–0.903
Pied shag	0.878	0.860–0.896	0.878	0.856–0.899
Little black shag	0.878	0.860–0.896	0.878	0.856–0.899
NZ king shag	0.878	0.860–0.896	0.878	0.856–0.899
Otago shag	0.878	0.860–0.896	0.878	0.855–0.898
Foveaux shag	0.878	0.860–0.896	0.878	0.855–0.899
Chatham Island shag	0.878	0.860–0.896	0.878	0.856–0.899
Bounty Island shag	0.878	0.860–0.896	0.878	0.856–0.898
Auckland Island shag	0.878	0.860–0.896	0.878	0.856–0.899
Campbell Island shag	0.878	0.860–0.896	0.878	0.855–0.899
Spotted shag	0.878	0.860–0.896	0.879	0.856–0.900
Pitt Island shag	0.878	0.860–0.896	0.878	0.855–0.899
Subantarctic skua	0.941	0.911–0.969	0.941	0.903–0.970
Southern black-backed gull	0.808	0.743–0.861	0.808	0.744–0.864
Caspian tern	0.877	0.820–0.933	0.878	0.802–0.937
White tern	0.805	0.781–0.829	0.805	0.776–0.832

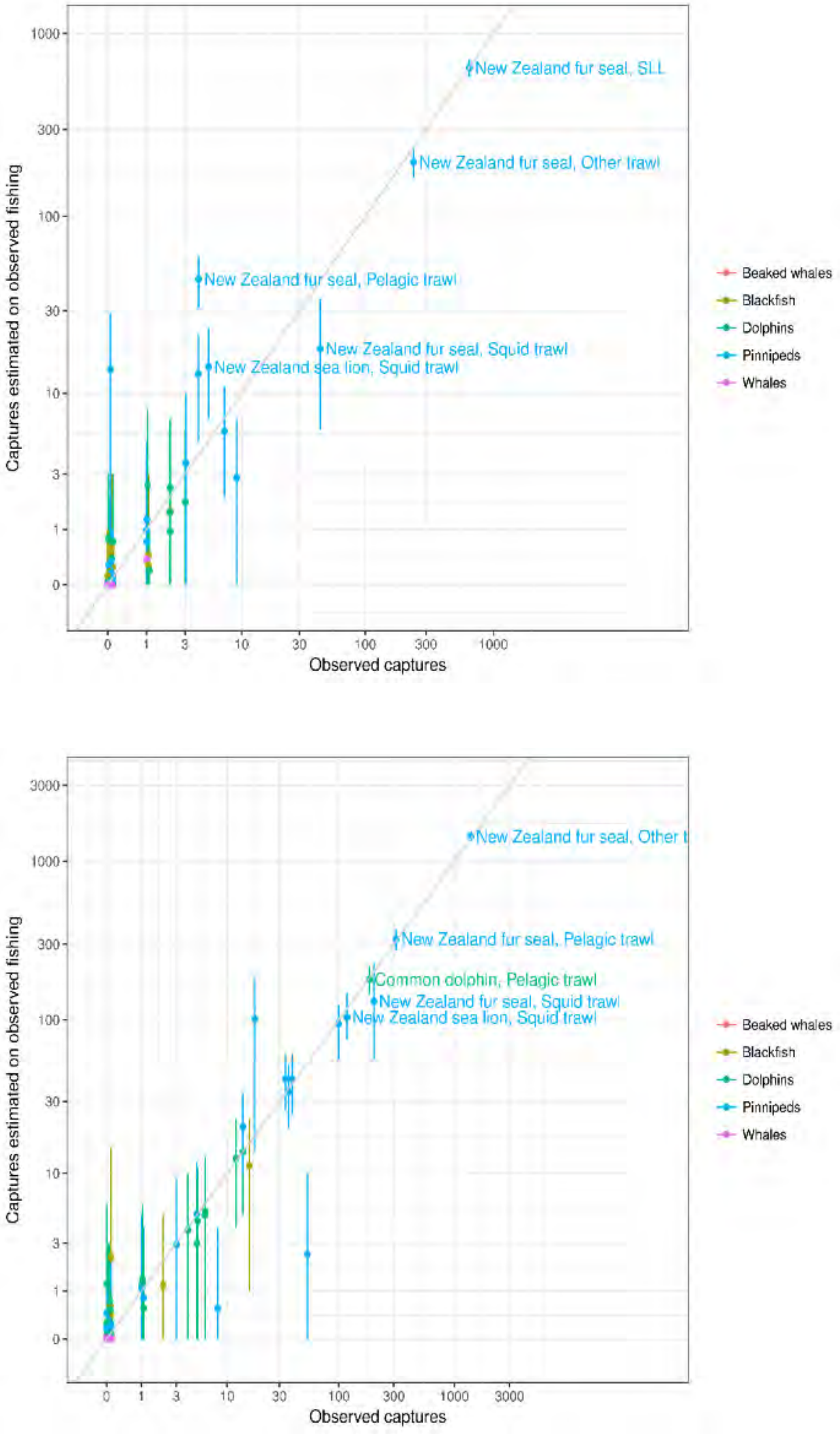


Figure 3.9: Example model diagnostics plot showing observed vs. expected numbers of live captures (top) and dead captures (bottom) for each fishery and species group combination in the 2016 NZMMRA.

3.4 ALTERNATIVE IMPLEMENTATIONS OF THE SEFRA FRAMEWORK

Alternate applications of the SEFRA method are currently planned or in development. To the extent possible, these will be developed to be conceptually and terminologically consistent with the framework described above, noting however that every individual risk assessment will be customised to address the particular nature of the specific problem and to make maximum use of the available data. For these reasons individual risk assessments will develop and apply different specific methodologies as required on a case by case basis.

3.4.1 SPECIES-SPECIFIC SEABIRD AND MARINE MAMMAL ASSESSMENTS

Where the multi-species marine mammal and seabird risk assessments indicate that fisheries risk is likely to be substantial for particular species of interest, separate species-specific implementations may be warranted to enable a more thorough understanding of available data. Species-specific implementations are already in progress for Māui and Hector's dolphins (SEA2016-30 and PRO2017-12) and for New Zealand sea lions in the Subantarctic Islands (PRO2017-10). New projects are under consideration also for New Zealand fur seals and for sea lions at newly established colonies on the New Zealand mainland.

Focusing on a particular species allows structural decisions to be tailored appropriately (i.e., using fishery group definitions or seasonally variable spatial distributions that are tailored to reflect interactions with only the species of interest). These projects will also allow disaggregation of species-level risk outputs to examine risk at a subpopulation level, and to examine sensitivities or evaluate risk management options (e.g., spatial management vs. mitigation vs. effort transition between fishery groups).

Single species SEFRA models also allow consideration of additional data reflecting covariates that may be of particular importance to some species but not to others. To illustrate, project PRO2016-02 will expand on the SEFRA framework to build a multivariate model predicting captures of black petrels and flesh-footed shearwaters by longline fisheries. As in the basic SEFRA method, capture rates are primarily a function of encounter rate, estimated via spatial overlap between species and fisheries. But the expanded model will also incorporate additional covariates thought to particularly affect black petrel and flesh-footed shearwater interactions with fisheries, e.g., moon phase, time of day, and mitigation uptake. In a multi-species model, the effects of these covariates would be diluted and likely impossible to discern. The outputs of PRO2016-02 are expected to provide insight into factors most responsible for driving fisheries captures, to inform the design of risk management options for these important species.

3.4.2 GLOBAL SEABIRD RISK ASSESSMENT

A global (southern hemisphere) seabird risk assessment is in progress to estimate out-of-zone risk to globally distributed New Zealand species. The methodological framework is as described above; available global seabird distributions are as utilised in Waugh et al. (2012). A primary challenge of this work is the poor quality of available observed captures data required to characterise global fishing effort and define meaningful fishery groups (reflecting different fishing behaviour and different levels of mitigation uptake between fleets) and thereby estimate fishery group vulnerability v_g. Species group vulnerability v_z can usefully be applied by proxy from the same or similar species in the NZSRA.

3.4.3 PELAGIC PROTECTED FISH SPECIES

For large, solitary, rare and/or protected fish species generally captured in single-capture events (e.g., pelagic sharks) it is likely that the most effective approach will apply a nearly identical mathematical formulation to that described above for seabirds and marine mammals, so long as population abundance data are available. Genetic mark-recapture methods or genetic half-sibling analyses may prove useful to obtain an estimate of absolute population size.

The primary challenge of a pelagic shark risk assessment under this approach will be to adequately represent highly dynamic spatial distributions in time; this may be achievable by applying sophisticated multi-variate habitat models (e.g., Leathwick et al. 2006, Pinkerton et al. 2010) parameterised using habitat affinity data from satellite tracked individuals, to define seasonal distributions and adjust available population size on a seasonal basis to reflect large-scale movements of pelagic fish species.

Where adequate population data are lacking and only fisheries-dependent data are available to model spatial distributions (e.g., many pelagic sharks), an alternative approach such as that developed by Fu et al. (2016) may be applied.

It is likely that any pelagic protected fish risk assessment could also be usefully extended to marine reptiles (turtles).

3.4.4 NON-TARGET FISH (TRAWL FISHERIES)

An application of the SEFRA method is currently under consideration for non-target fish species captured as bycatch in deepwater trawl fisheries, and for low information inshore fish stocks. Application of the method framework to non-target fish would follow the conceptual framework of the SEFRA method described above, but with substantial modifications of the analytical pathways outlined in equations (1)–(30), reflecting differences in data availability to inform input parameterisation. Application of the SEFRA method to protected species vs. bulk-capture bycatch species follows a similar estimation formulation as in equation 18, but the (relative) knowns and unknowns are reversed. For protected species such as seabirds and marine mammals, population size is generally known with some degree of precision (e.g., from genetic methods, breeding colony census) but capture events are sufficiently rare as to make estimation of catchability and/or vulnerability challenging; thus N and O are used to estimate q. In contrast, for non-target fish species, N is unknown but captures data are generally much richer; thus population size must be estimated from catchability q, which must in turn be estimated by other means (e.g., Zhou et al. 2009, 2011, Sibanda et al. 2016).

Because fishing gear is designed to retain fish, cryptic mortality is unlikely to be as important for bulk captured fish as for protected species, perhaps rendering the distinction between vulnerability and catchability unnecessary and eliminating the need for cryptic mortality multipliers (except for example to reflect small fish escaping through trawl meshes).

At least in trawl fisheries, because captures arise from passive interaction with gear rather than active behavioural attraction to fishing gear (as is the case with seabirds) estimation of q will by necessity include parameters for swept area and probably also a parameter for vertical availability in the water column, distinct from catchability parameters representing capture efficiency within the swept area.

Because fish are actively targeted, and because fish capture and retention in trawls is determined by both species-specific morphological and behavioural characteristics and fishery-group specific gear performance and efficiency, the structural assumptions behind the disaggregation of the vulnerability / catchability parameter into its species-group-specific and fishery-group-specific components is violated; catchability will by necessity be estimated per fishery group x species group combination (q_{sg} not $q_s q_g$).

Because schooling fish are captured in bulk, it will likely be necessary to estimate catchability as the product of two capture estimation models, one for probability of capture per fishing event and a separate model for abundance in those events in which the species is captured.

All of these modifications are under consideration by MPI contracted scientists; preliminary progress is described in Roux et al. (2015) and Sibanda et al. (2016). Subsequent extension to non-target inshore fish will be considered as one available method of the Low Information Stocks Project (LISP), subject to limitations on the ability to accurately estimate spatial distributions.

3.4.5 BENTHIC INVERTEBRATES AND/OR STRUCTURAL HABITATS

The SEFRA method is analogous to and fully compatible with spatially explicit benthic impact assessment methods for example as previously described in Sharp et al. (2009) and developed further by Mormede & Dunn (2012). The primary obstacle to full implementation of the SEFRA method for benthic invertebrates is the inherent difficulty of modelling benthic invertebrate spatial distributions given the sparse and scale-dependent nature of available environmental data to inform habitat models, and poor captures data with which to estimate the relationship between habitat and biology. For this reason the initial implementation of the impact assessment in Sharp et al. (2009) estimated impacts per spatial cell but without reference to the taxonomic composition of the benthic community; hence without an effective ‘population size’ there was no means of defining an impact threshold analogous to the MIST of Roux et al. (2015). Availability of improved high-resolution bathymetric and oceanographic spatial data layers to inform spatial habitat models may make full implementation of SEFRA method increasingly feasible for benthic invertebrate taxa.

Because fishing gear is not designed to retain benthic invertebrates, and damage to benthic habitats occurs regardless of to what extent benthic material is retained, modification of the SEFRA method for bottom fishing impacts will focus exclusively on vulnerability rather than catchability, using swept-area methods, thus eliminating any need to consider cryptic mortality. Growth and recovery factors analogous to the use of r_{max} can be used to model taxon-specific population responses to different spatially explicit impacts, and nett effects on multi-species composition (as in Mormede & Dunn 2012, Pitcher et al. 2016), to inform some objective basis to define a maximum impact threshold, analogous to MIST for non-target fish. Alternatively, impact can be mapped spatially using the overlap approach with traits-based vulnerability estimation (Sharp et al. 2009, Roux et al. 2016).

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Chapter 4: *Snapshot* of Chapter 4 - New Zealand sea lion (*Phocarctos hookeri*)

New Zealand sea lion
(*Phocarctos hookeri*)

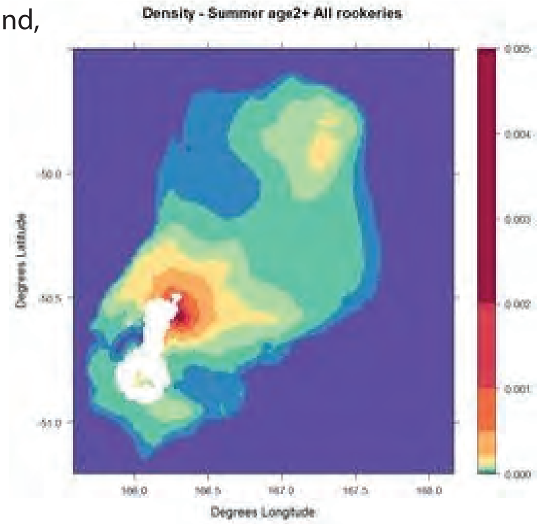
Threatened -
Nationally vulnerable (DOC 2019)

1. THE ISSUE IN BRIEF

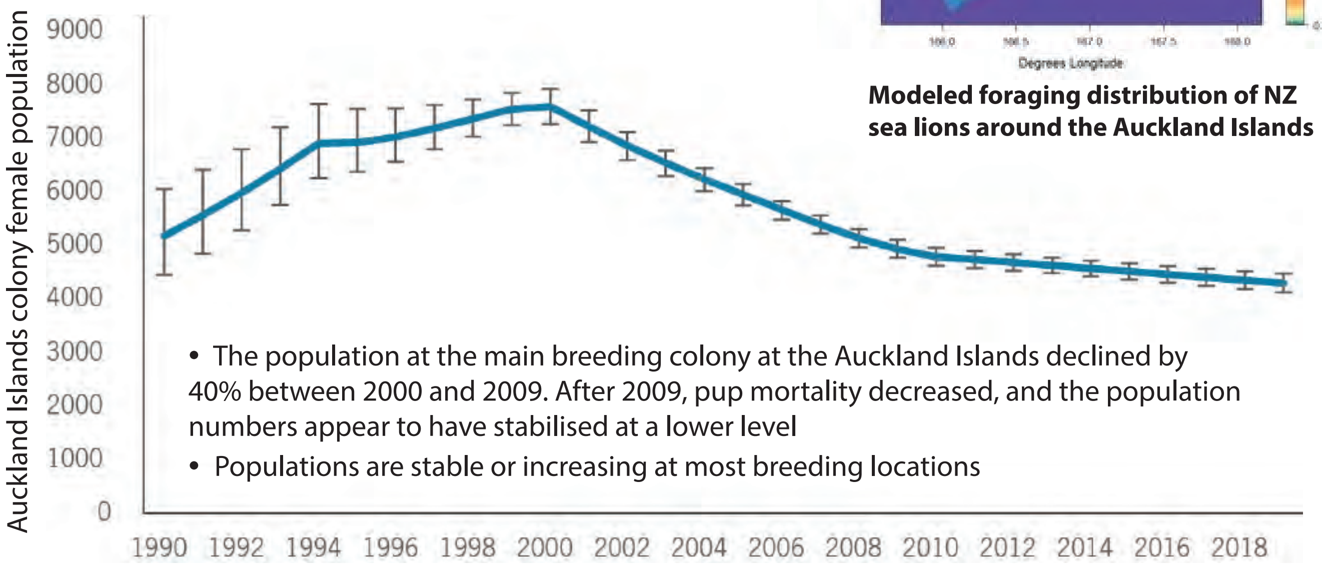
- The New Zealand sea lion (*Phocarctos hookeri*) is a pinniped, breeding only in New Zealand, classed as 'Nationally vulnerable' by the Department of Conservation
- The population of the main breeding colony at the Auckland Islands has declined from a peak in 2000
- Like all marine mammals, NZ sea lions are protected under the Marine Mammals Protection Act 1978 and the Fisheries Act 1996. NZ sea lions are managed under a Threat Management Plan (2017–2022)
- Potential threats to this species include human disturbance (on the mainland), direct and indirect effects of fisheries (for adults and sub-adults, see boxes 4 and 5), diseases, and possible climate effects

2. DISTRIBUTION AND ABUNDANCE

- New Zealand sea lions were once present throughout New Zealand, primarily in the southern regions, prior to human settlement
- Currently there are three recognised breeding colonies, on the Auckland Islands, Campbell Island, and Stewart Island, and recently established breeding sites on the Southland coast
- Population estimates are based on demographic models informed by annual pup counts and mark-recapture data
- Sea lions can roam up to 200 km away from the colony during foraging trips



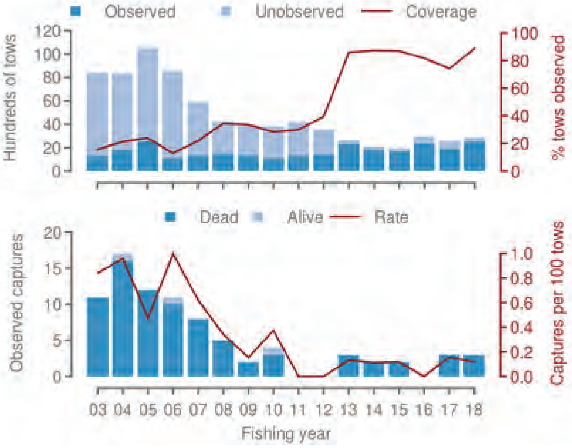
Modeled foraging distribution of NZ sea lions around the Auckland Islands

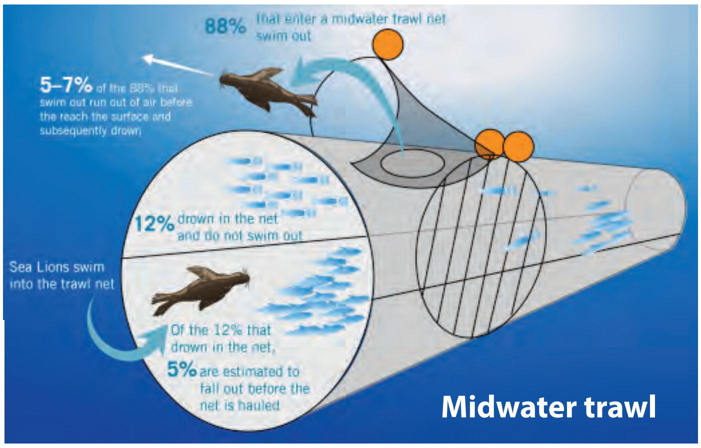


- The population at the main breeding colony at the Auckland Islands declined by 40% between 2000 and 2009. After 2009, pup mortality decreased, and the population numbers appear to have stabilised at a lower level
- Populations are stable or increasing at most breeding locations

3. FISHERIES INTERACTIONS

- Sea lions can enter trawl nets during fishing operations, and may drown in the net
- Sea Lion Exclusion Devices (SLEDs) enable sea lions to exit the net, reducing the risk of drowning. They were developed, and ultimately fully adopted from 2008. SLEDs are used in trawl fisheries near the Auckland Islands and Campbell Island



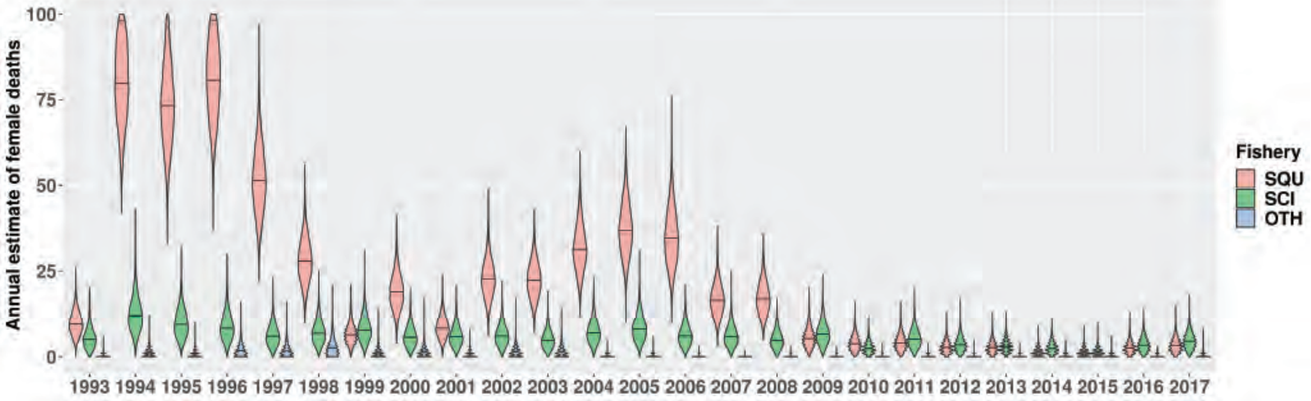


- Capture rates in relevant fisheries declined and stabilised after the full adoption of SLEDs (**see scheme above**). Observer coverage in squid fisheries has increased up to near 100%
- The Auckland Islands squid fishery is closed if the regulated mortality limit for NZ sea lions is reached

Figures on left: fishing effort and observer coverage (above) and observed captures of NZ sea lion (below) in the Squid 6T trawl fishery 2003–2018

4. SEA LION CAPTURE TRENDS IN TRAWL FISHERIES

- Sea lion captures are estimated for different trawl fisheries using a risk assessment model (see Chapter 3)



Estimated NZ sea lion (females only) annual deaths 1993–2017 in trawl fisheries targeting squid (red); scampi (green); and all other trawls (blue) around the Auckland Islands

- In the 1990s, relatively high captures are estimated to have occurred in trawl fisheries targeting squid. Estimated captures declined over time, as the fishing effort decreased
- From around 2007, sea lion deaths in the squid fishery decreased further, ranging between 1 and 5 deaths per year, after the universal adoption of standardised SLEDs
- Cryptic deaths, i.e., sea lions that exit via the SLED but nonetheless die as a consequence of the interaction, are estimated separately and included in the count of annual deaths (see figure above)

5. ONGOING RESEARCH

- Threat Management Plan (2017–2022) in place.
- Colony monitoring and pup counts are updated annually. SQU and SBW fisheries are highly observed. Spatial risk assessment can be updated annually using fishing overlap with sea lion distribution
- Work is in progress to monitor the new breeding sites along the Southland coast, and investigate indirect effects of fishing, diseases, and climate variability

4 NEW ZEALAND SEA LION (*PHOCARCTOS HOOKERI*)

Status of chapter	Observer data and capture estimates have been updated for AEBAR 2021.
Scope of chapter	This chapter describes: the biology of New Zealand sea lions (NZSL; *Phocarctos hookeri*), the nature and extent of potential interactions with fisheries, means of estimating fisheries impacts and population-level risk, management of fisheries interactions, and priorities for future work.
Area	Auckland Islands, Campbell Island, and nearby sub-Antarctic waters over the continental shelf. Stewart Island and nearby coastal waters. Otago and the Catlins Coast and nearby coastal waters.
Focal localities	Areas with potential for significant fisheries interactions include the Auckland Islands Shelf, the Campbell Plateau, Stewart Island, and the southern and south-eastern coasts of the South Island.
Key issues	Improved understanding of the effects of fishing in the context of non-fishery threats and environmental variability; improved understanding of spatio-temporal distributions affecting interaction rates with fishing effort, with a focus on the Dundas Island and Figure of Eight Island breeding populations, and outside the summer season; improved understanding of the risk factors and population consequences of *Klebsiella pneumoniae*-infection and other causes of death for pups at the Auckland Islands and Campbell Island; improved understanding of the causes and population consequences of nutritional stress for the Auckland Islands and Campbell Island colonies; improved understanding of potential anthropogenic barriers to growth of South Island mainland and Stewart Island breeding populations; cryptic mortality in trawls employing Sea Lion Exclusion Devices (SLEDs).
Emerging issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries potentially interacting with South Island and Stewart Island colonies. Improved understanding of the potential indirect effects of fishing on prey availability, in the context of climate variability. Management of public interactions with recovering South Island and Stewart Island populations.
Fisheries New Zealand research (current)	PRO2017-08C *Factors affecting New Zealand sea lion pup survival*; PMM2018-05B *Estimate spatial distributions for South Island NZSL to assess potential fisheries overlap and risk (including aquaculture)*. PMM2019-09: *Update Campbell Island NZSL PST (Population Sustainability Threshold) estimation;* ZBD2018-05: *Environmental variability, regime shifts, and ecosystem function in the sub-Antarctic*.
New Zealand government research (current)	DOC Marine Conservation Services Programme (CSP): INT2017-02 *Identification of marine mammals, turtles and protected fish captured in New Zealand fisheries*; INT2019-01 *Observing commercial fisheries*; INT2019-03 *Characterisation of marine mammal interactions*; POP2018-03 *New Zealand Sea Lion: Auckland Islands pup count;* MIT2014-01 *Protected species engagement project.*
Related chapters/issues	Chapter 5: New Zealand fur seals.

4.1 CONTEXT

The management of fisheries impacts on New Zealand sea lions is legislated under the Marine Mammals Protection Act (MMPA) 1978 and the Fisheries Act (FA) 1996.

The Minister of Conservation gazetted the New Zealand sea lion as a threatened species in 1997. All marine mammal species are designated as protected species under s.2 (1) of the FA. In 2005, the Minister of Conservation approved the Conservation General Policy, which specifies in Policy 4.4 (f) that "*Protected marine species should be managed for their long-term viability and recovery throughout their natural range.*" The Department of Conservation's (DOC) Regional Conservation Management Strategies outline specific policies and objectives for protected marine species at a regional level. New Zealand's sub-Antarctic islands, including Auckland Islands and Campbell Island, were inscribed as a World Heritage area in 1998.

Fisheries New Zealand manages fishing-related mortality of New Zealand sea lions under s.15 (2) of the FA. Under that section, the Minister of Fisheries "*may take such measures as he or she considers are necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality".*

The relevant National Fisheries Plan for the management of incidental captures of New Zealand sea lions is the National Fisheries Plan for Deepwater and Middle-depth Fisheries Part 1A (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand sea lions is Environmental Outcome 8: *Manage deepwater and middle-depth fisheries to avoid, remedy, or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened, and protected species*.

Specific objectives for the management of incidental captures of New Zealand sea lions will be outlined in the fishery-specific chapters of the National Deepwater Plan for the fisheries with which New Zealand sea lions are most likely to interact. These fisheries include sub-Antarctic trawl fisheries for arrow squid, southern blue whiting, and scampi.

The New Zealand sea lion population is monitored by pup counts at the main breeding colonies, the largest of which are at the Auckland Islands. The number of sea lion pups born at the Auckland Islands declined nearly 50% between 1998 and 2009 and appears to have stabilised thereafter. In 2014, following the third-lowest pup count on record, the Minister of Conservation and the Minister for Primary Industries requested that DOC and MPI work to develop a New Zealand sea lion/rāpoka Threat Management Plan (TMP). The process to develop the TMP involved a number of workstreams, including: a workshop to understand causes of pup mortality for sea lions at the Auckland Islands; two multi-day workshops, attended by a panel of independent experts, to inform a multi-threat risk assessment (Roberts 2015, Debski & Walker 2016); and inaugural meetings of the New Zealand sea lion/rāpoka Forum and Advisory Groups in early 2017. The TMP was finalised in 2017 (Department of Conservation & Ministry for Primary Industries 2017).

The TMP reflects the female New Zealand sea lion demographic population models and multi-threat risk assessment for the Auckland Islands described by Roberts & Doonan (2016) and recognises that no single identified threat in isolation was responsible for the population decline observed there since 2000. Population recovery would benefit from mitigation of multiple threats at the four main breeding sites (Department of Conservation & Ministry for Primary Industries 2017). The TMP commits to two objectives:

1) halt the decline of the New Zealand sea lion population within 5 years and
2) ensure the New Zealand sea lion population is stable or increasing within 20 years, with the ultimate goal of achieving 'Not Threatened' status.

The TMP outlines a work programme toward achievement of the plan's objectives, to be reviewed every five years. An overview of the TMP and identified workstreams, including research priorities, are reproduced in Figures 4.1 and 4.2.

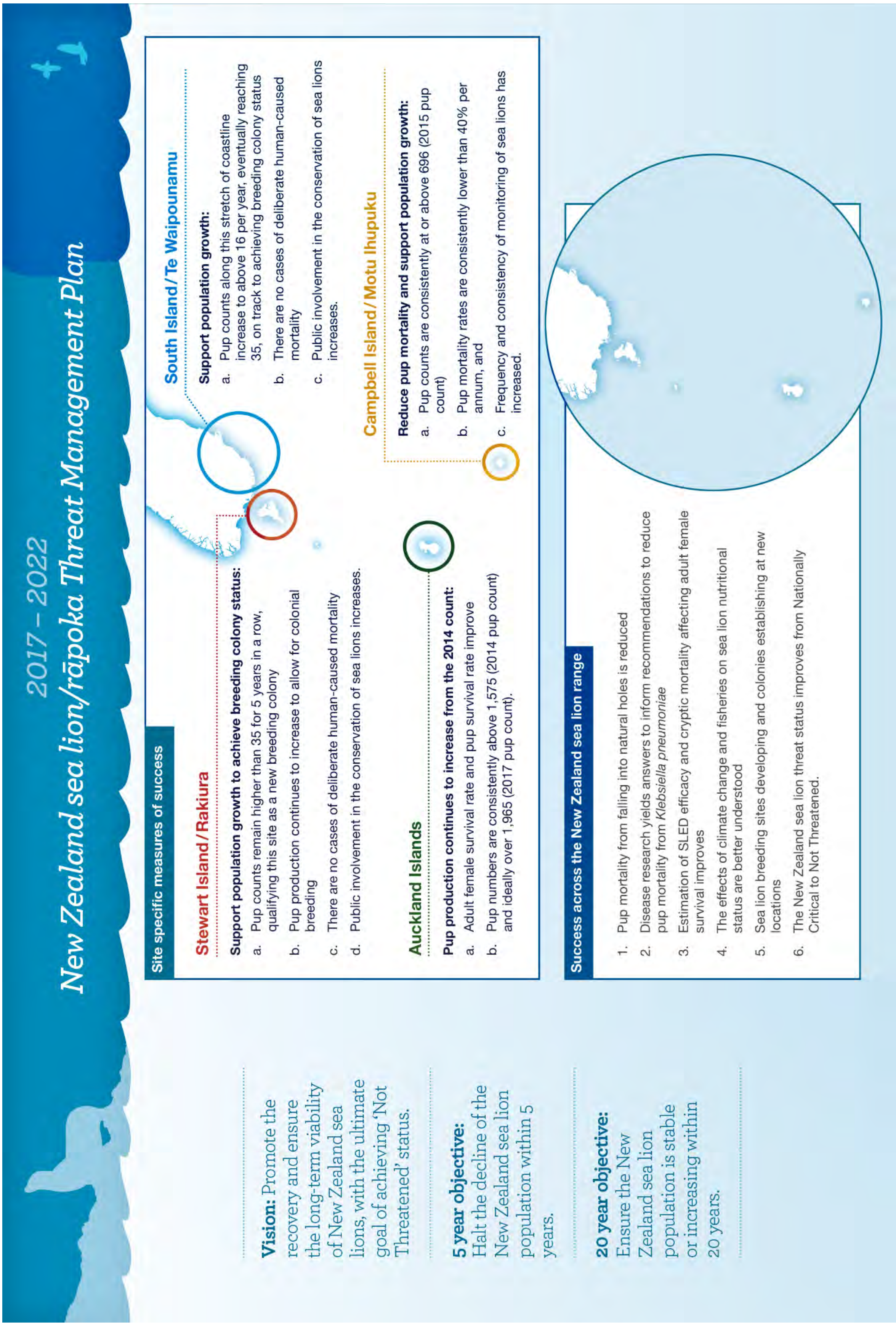
4.2 BIOLOGY

4.2.1 TAXONOMY

The New Zealand sea lion (*Phocarctos hookeri*, Gray 1844) is one of only two species of otariid (eared seals, including fur seals and sea lions) native to New Zealand, the other being the New Zealand fur seal (*Arctocephalus forsteri*, Lesson 1828). The New Zealand sea lion is New Zealand's only endemic pinniped, in terms of the breeding distribution (noting that males haul out at Macquarie Island—an Australian sub-Antarctic island—but there is no breeding colony there).

4.2.2 HISTORICAL DISTRIBUTION

Before the arrival of humans in New Zealand, New Zealand sea lions ranged around the North and South islands of New Zealand and the Chatham Islands (Rawlence et al. 2016). Pre-European remains of New Zealand sea lions have been identified from at least 47 archaeological sites, ranging from Stewart Island to North Cape, with most occurring in the southern half of the South Island (Smith 1989, 2011, Childerhouse & Gales 1998, Gill 1998). Analysis of Holocene remains indicated that breeding sea lions once occurred around north-west Nelson, and that South Island and Chatham Island subpopulations were genetically distinct from contemporary New Zealand sea lions. These subpopulations became extinct shortly after the arrival of Polynesian settlers (Collins et al. 2014a, 2014b, Rawlence et al. 2016).



2017 – 2022

New Zealand sea lion/rāpoka Threat Management Plan

Vision: Promote the recovery and ensure the long-term viability of New Zealand sea lions, with the ultimate goal of achieving 'Not Threatened' status.

5 year objective: Halt the decline of the New Zealand sea lion population within 5 years.

20 year objective: Ensure the New Zealand sea lion population is stable or increasing within 20 years.

Site specific measures of success

Stewart Island/Rakiura

Support population growth to achieve breeding colony status:

a. Pup counts remain higher than 35 for 5 years in a row, qualifying this site as a new breeding colony
b. Pup production continues to increase to allow for colonial breeding
c. There are no cases of deliberate human-caused mortality
d. Public involvement in the conservation of sea lions increases.

Auckland Islands

Pup production continues to increase from the 2014 count:

a. Adult female survival rate and pup survival rate improve
b. Pup numbers are consistently above 1,575 (2014 pup count) and ideally over 1,965 (2017 pup count).

South Island/Te Waipounamu

Support population growth:

a. Pup counts along this stretch of coastline increase to above 16 per year, eventually reaching 35, on track to achieving breeding colony status
b. There are no cases of deliberate human-caused mortality
c. Public involvement in the conservation of sea lions increases.

Campbell Island/Motu Ihupuku

Reduce pup mortality and support population growth:

a. Pup counts are consistently at or above 696 (2015 pup count)
b. Pup mortality rates are consistently lower than 40% per annum, and
c. Frequency and consistency of monitoring of sea lions has increased.

Success across the New Zealand sea lion range

1. Pup mortality from falling into natural holes is reduced
2. Disease research yields answers to inform recommendations to reduce pup mortality from *Klebsiella pneumoniae*
3. Estimation of SLED efficacy and cryptic mortality affecting adult female survival improves
4. The effects of climate change and fisheries on sea lion nutritional status are better understood
5. Sea lion breeding sites developing and colonies establishing at new locations
6. The New Zealand sea lion threat status improves from Nationally Critical to Not Threatened.

Figure 4.1: Threat management and population recovery objectives specific to four different New Zealand sea lion breeding populations, from the New Zealand sea lion Threat Management Plan (Department of Conservation & Ministry for Primary Industries 2017).

2017 – 2022
New Zealand sea lion/rāpoka Threat Management Plan

Vision: Promote the recovery and ensure the long-term viability of New Zealand sea lions, with the ultimate goal of achieving 'Not Threatened' status.

Partnership: The principles of mātauranga Māori will be woven throughout all four workstreams to achieve the vision of the Threat Management Plan.

Workstreams

	Engagement	Direct mitigation	Targeted research	Evaluation
	Coordinating and implementing an engagement campaign that facilitates a positive and accepted expansion of the range of sea lions.	Actions to reduce the impacts of key threats to sea lions are determined annually.	To address key threats to sea lions applied research will be determined annually.	To evaluate progress against the NZSL TMP objectives, monitoring of the sea lion breeding population is required.
2017/2018	• Establish the New Zealand sea lion/ rāpoka Forum and Advisory Group • Establish a New Zealand sea lion community liaison officer to coordinate and implement a recovery strategy for South Island/ Te Waipounamu sea lions. • Develop the broader engagment campaign	• 'Planks for Pups' programme at the Auckland Islands is maintained and expanded upon • Develop a strategy to reduce pup mortality from natural holes at Campbell Island • Establish a Technical Advisory Group to review the SQU6T Operational Plan	• Complete second season of disease research programme on the Auckland Islands • Female sea lion nutritional stress and diet studies • Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk assessment (sea lions) • Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries • Review of the potential impacts of aquaculture on sea lions, with relevance to Port Pegasus sea lion ecology	2017/2018 Monitoring, including tagging, pup counts and tag re-sights, will occur at: • Auckland Islands • Campbell Island/Motu Ihupuku • South Island/Te Waipounamu • Stewart Island/Rakiura
2018–2021	Prioritisation of direct mitigation and targeted research undertaken annually based on the evaluation results and recommendations from the Forum and Advisory Groups.			
2022	Review against the NZSL TMP 20 year objectives			

Figure 4.2: Workstreams identified in the New Zealand sea lion Threat Management Plan (Department of Conservation & Ministry for Primary Industries 2017).

Subsistence hunting on the South Island and subsequent commercial harvest from outlying islands of New Zealand sea lions for skins and oil resulted in population decline and contraction of the species range (Gales 1995, Childerhouse & Gales 1998, Nagaoka 2001, 2006). Despite the historic reduction in population size and range contraction as a result of subsistence hunting and commercial harvest, the New Zealand sea lion population does not display low genetic diversity at microsatellite loci and thus does not appear to have suffered effects of genetic drift and inbreeding depression (Robertson & Chilvers 2011).

4.2.3 CURRENT DISTRIBUTION

Currently, most New Zealand sea lions are found in the New Zealand sub-Antarctic, with individuals ranging to the New Zealand South Island and Macquarie Island. New Zealand sea lion breeding colonies[1] are highly localised, with most pups being born at the Auckland Islands and Campbell Island (Wilkinson et al. 2003, Chilvers 2008). At the Auckland Islands, there are three extant breeding colonies: Enderby Island (at Sandy Bay), Dundas Island, and Figure of Eight Island. On Campbell Island there is one breeding colony at Davis Point, another colony at Shoal Point, and an increasing number of non-colonial breeders (Wilkinson et al. 2003, Chilvers 2008, Maloney et al. 2009, Maloney et al. 2012, McNutt et al. 2020). Breeding at the Auckland Islands represents 68–79% of the pup production for the species, with the remaining 21–32% occurring on Campbell Island (based on concurrent pup counts in 2008, 2010, 2015, 2018, 2019, and 2020; see Figure 4.3). Numbers of breeding sea lions at the new Stewart Island colony and at haul-out sites on the South Island are comparatively low but may be expected to increase steadily if these recolonisation events continue successfully.

Intermittent sea lion pup sightings have been reported at Port Pegasus, Stewart Island since the 1990s. In 2011, a pup survey and tagging programme was initiated, with 16 pups tagged. Breeding success at the Stewart Island location has increased steadily since that time, with 55 pups tagged in 2018 in the standard survey area, plus another 7 pups tagged outside the survey area (Boren 2018). In 2018, after 5 consecutive years with more than 35 pups being counted, the Stewart Island population was officially recognised as the third New Zealand sea lion breeding colony. The latest pup count for Stewart Island was 48 pups in 2020 (47 at Port Pegasus and one at Ulva Island; DOC unpublished data).

Successful sea lion breeding has also been observed on the Otago Coast, South Island, beginning with a single female that arrived in 1992 and gave birth in 1993 (McConkey et al. 2002). Pup production at this location increased slowly, to 7 pups in 2013, followed by a more rapid increase, to 21 pups in 2020 (see Figure 4.3).

On land, New Zealand sea lions can travel long distances and ascend hills. They are found in a variety of habitats including grass fields, exposed rock, and dense bush and forest; breeding colonies are usually on large sandy beaches (Gales 1995, Augé et al. 2012), though not everywhere, e.g., at Campbell Island. In early summer, colonial breeding sea lions are spatially constrained in the vicinity of colony locations. Following the end of the females' oestrus cycle in late January, adult and sub-adult males disperse throughout the species range, whereas the dispersal of females (both breeding and non-breeding) is more restricted both during and subsequent to the breeding season (Marlow 1975, Robertson et al. 2006, Chilvers & Wilkinson 2008).

4.2.4 FORAGING ECOLOGY

Foraging studies have been conducted on known populations of lactating female New Zealand sea lions, i.e., from Enderby Island, Dundas Island, and Figure of Eight Island (all in the Auckland Islands group) (Chilvers et al. 2005b, 2006, 2013, Chilvers & Wilkinson 2009); Stewart Island; and the Otago Peninsula (see Augé et al. 2011a, 2014, Chilvers et al. 2011). Leung et al. (2012, 2013b, 2014b) investigated foraging by juvenile New Zealand sea lions at Enderby Island in contrast with juvenile animals at Otago Peninsula (Leung et al. 2013a), and in mother-yearling pairs at Enderby Island (Leung et al. 2014a). A comprehensive analysis of spatial foraging patterns of Auckland Islands females used all the available satellite telemetry data to characterise spatial foraging patterns and estimate spatial overlap, annual deaths, and population risk from all Auckland Islands commercial trawl fisheries (Large et al. 2019). This assessment estimated foraging distributions that primarily represent the summer foraging of breeding-age females from the Sandy Bay colony on Enderby Island. Further tracking is planned to collect data

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[1] DOC (2009) defines colonies as 'haul-out sites where 35 pups or more are born each year for a period of 5 years or more.' Haul-out sites are defined as 'terrestrial sites where New Zealand sea lions occur but where pups are not born, or where fewer than 35 pups are born per year over 5 consecutive years.'

from sea lions at Dundas Island and/or Figure of Eight Island, and to prioritise tracking data outside summer months. Analyses of satellite-tracked individuals from Campbell Island is in preparation (Lea et al. in prep, Lea et al. in press).

Previous analyses of sea lion foraging indicate that females from Enderby Island forage primarily over the Auckland Islands continental shelf and its northern edge, and that individuals show strong foraging site fidelity both within and across years. Satellite tagging data from lactating females at the Auckland Islands shows that the mean return distance travelled per foraging trip is 423 ± 43 km (n = 26), which is greater than that recorded for any other sea lion species (Chilvers et al. 2005b). While foraging, about half of the time was spent submerged, with a mean dive depth of 130 ± 5 m (max. 597 m) and mean dive duration of 4 ± 1 minutes (max. 14.5 minutes; Chilvers et al. 2006). Both juvenile female and male sea lions foraged to the north of the Auckland Islands, but the mean distance travelled per foraging trip was shorter in females (99 ± 12 km, n = 19) compared with males (184 ± 25 km, n = 12), and the mean maximum distance from the colony for males (93 ± 10 km) was about twice that for females (51 ± 5 km; Leung et al. 2012). A study of seven dependent yearling New Zealand sea lions (Leung et al. 2013b) found that dive depth was negatively related with animal mass (lighter sea lions dived to greater depths), but in juvenile (2–5 years old) New Zealand sea lions, diving ability (dive depth, dive duration, and bottom time per dive) improved with both mass and age, and five-year-old male New Zealand sea lions had similar dive capability to adult females (Leung et al. 2014b). New Zealand sea lions, like most pinnipeds, may use their whiskers to help them locate and capture prey at depths where light does not penetrate (Marshall 2008, Hankel et al. 2010). Leung et al. (2014a) found no evidence that yearling New Zealand sea lions were developing foraging skills through observational learning of maternal behaviours in a study of seven mother-yearling partnerships at Enderby Island.

A recent review of studies conducted on female New Zealand sea lions suggests a continuum of foraging behaviour between benthic foraging vs. mesopelagic foraging modes (Roberts et al. 2018, Lea et al. in press). An earlier study suggested that individual animals may tend to specialise in one or the other foraging mode (Chilvers & Wilkinson 2009). In that study benthic divers had fairly consistent dive profiles, reaching similar depths (120 m on average) on consecutive dives in relatively shallow water, presumably to feed on benthic prey. Mesopelagic divers, by contrast, exhibited more varied dive profiles, undertaking both deep (over 200 m) and shallow (less than 50 m) dives over deeper water. Benthic divers tended to forage further from their breeding colonies, making their way to the north-eastern limits of Auckland Islands Shelf, whereas mesopelagic divers tended to forage along the north-western edge of the shelf over depths of approximately 3000 m (Chilvers & Wilkinson 2009). Meynier et al. (2014), employed fatty acid analyses of blubber samples and found that fatty acid profiles were different in primarily benthic diving vs. primarily mesopelagic diving lactating New Zealand sea lions, suggesting a different utilisation of prey resources such that, though prey species taken were similar for both dive modes, the proportion of particular prey differed between the two modes. In addition, Chilvers (2017) found that the composition of stable isotopes obtained from both blood serum and whiskers differed between benthic vs. mesopelagic foraging sea lions at the Auckland Islands. Further, Meynier et al. (2014) found that the body condition index (the residual between the measured and predicted body mass from the mass-length regression provided by Childerhouse et al. 2010a) was significantly greater in meso-pelagic divers than in benthic divers.

The differences in dive profiles have further implications for the estimated aerobic dive limits (ADL; Gales & Mattlin 1997, Chilvers et al. 2006), defined as the maximum amount of time that can be spent underwater without increasing blood lactate concentrations (a byproduct of anaerobic metabolism). If animals exceed their ADL and accumulate lactate, they must surface and go through a recovery period to aerobically metabolise the lactate before they can undertake subsequent dives. Chilvers et al. (2006) estimated that lactating female New Zealand sea lions at the Auckland Islands exceed their ADL on 69% of all dives, a much higher proportion than most other otariids (which exceed their ADL for only 4–10% of dives, Chilvers et al. 2006). Auckland Islands sea lions that exhibit benthic diving profiles are estimated to exceed their ADL on 82% of dives, compared with 51% for meso-pelagic divers (Chilvers 2008).

Chilvers et al. (2006) and Chilvers & Wilkinson (2009) suggested that the long, deep-diving behaviour, the propensity to exceed their estimated ADL, and differences in physical condition and age at first reproduction from animals at Otago together indicate that females from the Auckland Islands may be foraging at or near their physiological limits. However, Bowen (2012) suggested a lack of relationship between surface time and anaerobic diving would seem to indicate that ADL has been underestimated. Further, given a number of studies of diving behaviour were conducted during early lactation when the demands of offspring are less than they would be later in lactation, Bowen (2012) considered it unlikely that females are operating at or near a physiological limit.

Adult females at Otago are generally heavier for a given age, breed earlier, undertake shorter foraging trips, and have shallower dive profiles compared with females from the Auckland Islands (Table 4.1). These observed differences may reflect differences in habitat (including prey availability) between the Auckland Islands and the Otago Peninsula, a founder effect, or a combination of these or other factors. Similarly, Leung et al. (2013a) compared foraging characteristics in juvenile (2–3 years old) female New Zealand sea lions at Enderby Island and Otago Peninsula. Overall, females at Otago were heavier (3-year-old mean 96 kg) than females at Enderby (3-year-old mean 72 kg) and exhibited shorter mean foraging trip distance (19 km at Otago, 103 km at Enderby), shallower mean dive depth (15 m at Otago, 69 m at Enderby), and shorter mean dive duration (1.8 min at Otago, 3.2 min at Enderby). Leung et al. (2013a) concluded that the Auckland Islands are a less optimal habitat compared with Otago.

Satellite telemetry studies collected data during the 2019–20 summer field season to characterise the foraging distribution and dive behavior of breeding females in the Catlins coast mainland population (DOC unpublished data). Similar work may continue in future field seasons.

New evidence from satellite tracked individuals at Campbell Island (Lea et al. in press) and from analysis of sea lion prey including a dedicated ocean survey (Roberts et al. 2018) suggests that sea lions at the sub-Antarctic islands may suffer from periods of low prey availability and may be forced to forage at the limits of their physiological capabilities by low prey availability over the shelf. This would make these populations particularly susceptible to environmental variability affecting availability of preferred prey (Roberts et al. 2018).

The foraging of lactating females at Port Pegasus, Stewart Island was recently characterised by Chilvers (2018), describing their foraging characteristics as intermediate between Auckland Islands and Otago Peninsula females with respect to dive depth, dive duration, and body mass. Satellite telemetry data indicated that nearly all foraging was within 50 km of the tagging site at Port Pegasus (Roberts 2017a).

Table 4.1: Comparison of selected characteristics between adult female New Zealand sea lions from the Auckland Islands and those from the Otago Peninsula (Augé et al. 2011a, 2011b, Chilvers et al. 2006, Chilvers 2018, Roberts & Doonan 2016). Data are means ± s.e. (where available).

Characteristic	Auckland Islands	Stewart Island	Otago Peninsula
Reproduction at age 4	19% of females (95 % CI = 16 –23 %)	Unknown	> 85% of females
Average mass at 8–13 years of age	112 kg	Unknown	152 kg
Foraging distance from shore	102.0 ± 7.7 km (max = 175 km)	45.0 ± 4.1 km (max = 38 km)	4.7 ± 1.6 km (max = 25 km)
Time spent foraging at sea	66.2 ± 4.2 hrs	14.9 ± 1.4 hrs	11.8 ± 1.5 hrs
Dive depth	129.4 ± 5.3 m (max = 597 m)	59.6 ± 7.0 (max ≥ 250 m)	20.2 ± 24.5 m (max = 389 m)
Dives estimated to exceed ADL	68.7 ± 4.4%	35.1 ± 3.3%	7.1 ± 8.1%

New Zealand sea lions are generalist predators with a varied diet that includes marine mammal prey (New Zealand fur seal *Arctocephalus forsteri*), seabirds (yellow-eyed penguin *Megadyptes antipodes*, blue penguin *Eudyptula minor*, southern rockhopper penguin *Eudyptes chrysocome*, southern royal albatross *Diomedea epomophora*), elasmobranchs (rough skate *Raja nasuta*), teleost fish (e.g., opalfishes *Hemerocoetes* spp., hoki *Macruronus novaezelandiae*, red cod *Pseudophycis bachus*, jack mackerels *Trachurus* spp., barracouta *Thyrsites atun,* southern blue whiting *Micromesistius australis*); cephalopods (e.g., octopus *Enteroctopus zelandicus* and *Macroctopus maorum*, squid *Nototodarus sloanii*); crustaceans (e.g., lobster krill *Munida gregaria*); and other invertebrates (e.g., salps) (Cawthorn et al. 1985, Moore & Moffat 1992, Bradshaw et al. 1998, Childerhouse et al. 2001, Lalas et al. 2007, Moore et al. 2008, Meynier et al. 2009, Augé et al. 2012, Lalas et al. 2014, Lalas & Webster 2014, Morrison et al. 2017). The three main methods used to assess New Zealand sea lion diets involve analyses of stomach contents, scats, and regurgitate, and the fatty acid composition of blubber (Meynier et al. 2008). Stomach contents of incidentally captured animals tend to be biased towards the target species of the fishery concerned (e.g., squid in the Auckland Islands squid fishery), whereas scats and regurgitates are biased towards less digestible prey (Meynier et al. 2008). Stomach, scat, and regurgitate approaches tend to reflect only recent prey (Meynier et al. 2008). By contrast, analysis of the fatty acid composition of blubber provides a longer-term perspective on diets ranging from weeks to months (although individual prey species are not identifiable). Fatty acid analysis suggests that the diet of female New Zealand sea lions at the Auckland Islands tends to include proportionally more arrow squid and hoki and proportionally fewer red cod than for male New Zealand sea lions, and that lactating and non-lactating females do not differ in their diet (Meynier et al. 2008, Meynier 2010). Within a sample of lactating female New Zealand sea lions, Meynier et al. (2014) used fatty acid analyses to show that the diet of benthic diving and meso-pelagic diving animals consisted of similar prey, though different mass contributions for each prey species.

Previous assessments have identified considerable spatial (comparing colonies) and temporal (inter-annual and seasonal) variation in the diet composition of New Zealand sea lions. For instance, jack mackerel and barracouta were identified as the main prey of the Otago Peninsula population (Augé et al. 2012), though were less prevalent in winter and spring when inshore species dominated diet composition (Lalas 1997) and were infrequent prey of the Auckland Islands population (Childerhouse et al. 2001, Stewart-Sinclair 2013). A long-term diet assessment of the Sandy Bay colony at the Auckland Islands (1994–95 to 2012–13) identified a decrease in the occurrence of large-sized prey (e.g., *Enteroctopus zealandicus*) and an increasing trend in small-sized prey (e.g., opalfishes, rattails, and *Octopus* spp.) (Childerhouse et al. 2001, Stewart-Sinclair 2013).

Teeth from individual sea lions at the Auckland Islands that were archived at Massey University and Te Papa Tongarewa were used to estimate trophic histories over an extended historical period. Graham et al. (2019) analysed 396 samples from the annual growth bands found in 22 sea lion teeth (19 females and 3 males) dating from 1935 to 2005. Nitrogen isotope (δ15N) data indicate an animal's trophic ecology and changes in their foraging strategies. It was found that the male sea lions consistently forage at a higher trophic level than the females. The δ15N values of the 19 females reveal aspects of their foraging ecology and physiology. At a broad scale, there is considerable variation between individuals, suggesting variable foraging strategies. A maternal or lactation signal was observed in almost all teeth samples. This signal occurs because as the pup consumes the mother's milk its isotope value will be one trophic level higher than its mother. In general, the lactation signal declined for most of the individuals in the first year, and by year two it was only present in two individuals born in 1943 and 1994. An increase in trophic level occurs after age five, which coincides with the age at first breeding, but again there is inter-individual variation. The δ13C dataset for female New Zealand sea lions shows an overall decreasing temporal trend, with notable decreases pre-1960 and post-1990. Changes in primary productivity affect the δ13C values at the base of the food web and this signal has been shown to propagate up the food web. Overall, in periods of higher productivity the δ13C values increase (Laws et al. 1995, Schell et al. 1998, Graham et al. 2010). This suggests that during the 1940–60s and late 1990–early 2000s there was either a) a decrease in productivity around the Auckland Islands where the female sea lions forage (i.e., shift in ocean conditions) and/or b) the females shifted their main foraging strategy (e.g., benthic vs. mesopelagic related to available prey). A higher sample size would be required from the earlier time period to resolve the timing of these isotopic signals because they may relate to ecosystem changes potentially affecting fish stocks or other species.

4.2.5 REPRODUCTIVE BIOLOGY

New Zealand sea lions exhibit marked sexual dimorphism; adult males are darker in colour and much larger than adult females (Walker & Ling 1981, Cawthorn et al. 1985). Cawthorn et al. (1985) and Dickie (1999) estimated the maximum age of males and females to be 21 and 23 years, respectively; Childerhouse et al. (2010b) reported a maximum estimated age for females of 28 years. Females can become sexually mature as early as age two and may give birth the following year. However, at the Auckland Islands most females do not breed until they are six years old (Childerhouse et al. 2010b, Roberts & Doonan 2016); at Otago Peninsula most females breed by age four (Roberts & Doonan 2016). Males generally reach sexual maturity at age four, but because of their polygynous colonial breeding strategy (i.e., males actively defend territories and mate with multiple females within a harem) they are only able to successfully breed at 7–9 years old, once they have attained sufficient physical size to compete successfully with other males (Marlow 1975, Cawthorn et al. 1985). At the Auckland Islands, the reproductive rate in females increases rapidly between the ages of 3 and 7, reaching a plateau until the age of approximately 15 and declining rapidly thereafter, with the maximum recorded age at reproduction being 26 years (Breen et al. 2016, Childerhouse et al. 2010a, Chilvers et al. 2010). Chilvers et al. (2010) estimated from tagged sea lions that the median lifetime reproductive output of a female New Zealand sea lion at the Auckland Islands was 4.4 pups, and 27% of all females that survive to age 3 never breed. Analysis of tag-resighting data from female New Zealand sea lions on Enderby Island indicates the average probability of breeding is approximately 0.30–0.35 for prime-age females that did not breed in the previous year (ranges reflect variation relating to the definition of breeders) and 0.65–0.68 for prime-age females that did breed in the previous year (MacKenzie 2011).

New Zealand sea lions are strongly philopatric (i.e., they return to breed at the same location where they were born, although more so for females than males). Breeding is highly synchronised and starts in late November when adult males establish territories (Robertson et al. 2006, Chilvers & Wilkinson 2008). Pregnant and non-pregnant females appear at the breeding colonies in December and early January, with pregnant females giving birth to a single pup in late December before entering oestrus 7–10 days later and mating again (Marlow 1975). Twin births and the fostering of pups in New Zealand sea lions are rare (Childerhouse & Gales 2001). Shortly after the breeding season ends in mid-January, the harems break up with the males dispersing offshore and females often moving away from the rookeries with their pups (Marlow 1975, Cawthorn et al. 1985).

Pup birth weight is 8–12 kg and is highly variable between years; parental care is restricted to females (Walker & Ling 1981, Cawthorn et al. 1985, Chilvers et al. 2006). Females remain ashore for about ten days after giving birth before alternating between foraging trips lasting approximately two days at sea and returning for about one day to suckle their pups (Gales & Mattlin 1997, Chilvers et al. 2005b). New Zealand sea lion pup growth rates at the Auckland Islands are lower than those reported for other sea lion species and may be linked to a relatively low concentration of lipids in the females' milk during early lactation (Chilvers 2008, Riet-Sapriza et al. 2012). Riet-Sapriza et al. (2012) also found that there was a temporal (year and month) effect on milk quality, reflecting individual sea lion characteristics and environmental factors, and that maternal body condition was positively correlated with milk lipid concentration, energy content, and milk protein concentration: lactating females in good condition produced more energy-rich milk than did relatively lean females. Pups are weaned after about 10–12 months (Marlow 1975, Gales & Mattlin 1997).

4.2.6 POPULATION BIOLOGY

For New Zealand sea lions, the overall size of the population is indexed using estimates of the number of pups that are born each year (Chilvers et al. 2007). Moderately reliable pup counts have been made at Auckland Islands colonies since the 1960s (e.g., Falla et al. 1976, and see a review by Childerhouse & Gales 1998, and summary of estimates in table 1 of Breen et al. 2016), though these were intermittent, and reliable counts were not made across all known Auckland Islands colonies in the same year prior to 1995. Since 1995, DOC has conducted mark-recapture and pup census counts at each of the main breeding colonies at the Auckland Islands, using a consistent methodology, to estimate annual pup production (i.e., the total number of pups born each year, including dead and live animals; Robertson & Chilvers 2011). Pup censuses have been less frequent for other colonies, including the large population at Campbell Island (Maloney et al. 2012).

For the Auckland Islands population, the data show a decline in pup production from a peak of 3021 in 1997–98 to a low of 1501 ± 16 pups in 2008–09 (Chilvers & Wilkinson 2011, Robertson & Chilvers 2011; Table 4.2 and see Figure 4.3), with the largest single-year decline (31%) occurring between the 2008 and 2009 counts.

Since 2009, estimated pup production at the Auckland Islands appears to have stabilised, fluctuating without trend between roughly 1600 and 1800 pups in most years. The most recent estimate of pup production for the Auckland Islands population was 1740 pups in 2020, of which 289 were at Sandy Bay and 1399 were at Dundas Island (Table 4.2 and Figure 4.3).

The total New Zealand sea lion population size (including pups) at the Auckland Islands has been estimated using Bayesian population models (Breen et al. 2003, 2016, Breen & Kim 2006a, 2006b, Roberts & Doonan 2016). Although other abundance estimates are available (e.g., Gales &

Fletcher 1999), for the Auckland Islands population, estimates derived from the integrated models are preferred because they take into account a variety of age-specific factors (breeding, survival, maturity, fisheries incidental captures), as well as data on the resighting of tagged animals and pup production estimates (Table 4.3).

When using demographic models to predict future population trends, the future trajectory of the Auckland Islands population is highly dependent on the time period of demographic rates used to generate forward projections. For instance, negative population growth (λ = 0.959; 95% credible interval = 0.955–0.963) was estimated when using the demographic rates for the period of declining pup production (between 1999 and 2009). But increasing (λ = 1.087; 95% credible interval = 1.069–1.105) or approximately stable (λ = 0.989; 95% credible interval = 0.985–0.993) trajectories were produced when using demographic rates for the prior period of growth (until 1999) or relatively stability (since 2009) was used (Roberts 2019).

At locations outside the Auckland Islands, breeding sea lions have only established more recently, so their expected population trajectories may be generally characterised by initial population growth followed by eventual stabilisation as populations approach local habitat limits, or by alternating periods of population growth and decline reflecting variable environmental conditions.

At Campbell Island, recorded pup production has grown from very low levels in the early 1990s up to 734 pups in 2018 (Boren 2018) and was 595 pups in 2020 (although note that comparability may have been affected by exceptionally high pup mortality rate in the latest year; see below) (McNutt et al. 2020). Estimates of pup production at Campbell Island increased sharply in the period from 1990 to 2010 (i.e., including during the period of steepest decline at the Auckland Islands) but there has been some variation in the timing and methodology of these surveys, and one of the breeding colonies has moved over time. The later surveys in 2003, 2008, 2010, and 2015 were considered to be of sufficient quality to inform a simple population estimate (Roberts & Doonan 2016) and a comparable methodology was used to estimate pup production in 2018, 2019, and 2020. Early pup mortality (i.e., in the first few months of life) at Campbell Island has been relatively high in all recent census years, including: 1998 (31%), 2003 (36%), 2008 (40%), 2010 (55%), 2015 (58%), 2018 (23%), 2019 (54%), and 2020 (81%, the highest recorded at any New Zealand sea lion breeding site) (see McNally et al. 2001, Childerhouse et al. 2005, Maloney et al. 2009, 2012, Childerhouse et al. 2015a, Boren 2018, Foo & Weir 2019, McNutt et al. 2020). Multiple consecutive years of high pup mortality rates can be expected to have detectable impacts on adult population size and future pup production as the affected cohorts reach maturity and recruit into the breeding population.

For the Otago coast, annual pup production has increased from 0 in the 1995 breeding season to 21 in the 2020 season (Figure 4.3). Sea lions at Otago are of special interest because they highlight the potential for establishing new breeding colonies; the Otago coast breeding population originated with a single pregnant female (McConkey et al. 2002). The TMP identifies that the viability of new colony locations on the New Zealand South Island is of particular importance for the restoration of New Zealand sea lions to non-threatened status.

Sea lions have established at Stewart Island, where pup census estimates have been made since 2011, about 3–4 months after the probable pupping period. Stewart Island pup counts have increased from 16 pups in 2011 to 48 pups in 2020 (Chilvers 2014, DOC unpublished data, Roberts & Doonan 2016; Figure 4.3). From 2018, the Stewart Island population was formally recognised as a new breeding colony, after 5 consecutive years in which annual pup production was estimated to have exceeded 35 births (Department of Conservation & Ministry for Primary Industries 2017).

4.2.7 THREATS TO SEA LIONS

Known anthropogenic sources of direct mortality to New Zealand sea lions include, historically: subsistence hunting and commercial harvest (Gales 1995, Childerhouse & Gales 1998); and pup entrapment in rabbit burrows prior to rabbit eradication from Enderby Island in 1993 (Gales & Fletcher 1999). On Stewart Island and the South Island, sea lions encounter human disturbance (including attacks by dogs), vehicle strikes, and deliberate shooting on South Island New Zealand (Gales 1995). Incidental captures in fisheries may affect both sub-Antarctic and mainland populations (see section 4.4). Scientific research may also pose a threat, e.g., there is a risk of accidental death arising from the use of anesthesia (Lynch et al. 1999), and disturbance and handling of animals may create other risks to animal health. Other anthropogenic effects may indirectly affect New Zealand sea lion populations, but for which the actual level of impact is presently unclear, include potential trophic competition between New Zealand sea lions and fisheries (Robertson & Chilvers 2011, Bowen 2012, Roberts et al. 2018; see below); effects of organic and inorganic pollutants, including polychlorinated biphenyls (PCBs), dichlorodiphenyltrichloroethane (DDT), and heavy metals such as mercury and cadmium (Baker 1999, Robertson & Chilvers 2011); and impacts of casual or organised eco-tourism.

Very high rates of pup mortality observed at Campbell Island are mainly due to pups drowning in wallows (Lea et al. 2018) or dying of exposure or starvation arising from adverse weather conditions (McNutt et al. 2020). The magnitude of these impacts may reflect that a substantial proportion of Campbell Island sea lions may be breeding in locations with sub-optimal conditions (i.e., muddy beach substrate and high exposure to extreme weather events). Similarly, high rates of pup mortality from holes and storms are not typically observed elsewhere. Other sources of natural mortality that may occur in all locations include predation by white pointer sharks (Cawthorn et al. 1985, Robertson & Chilvers 2011), starvation of pups if they become separated from their mothers (Walker & Ling 1981, Castinel et al. 2007), and male aggression towards females and pups (Wilkinson et al. 2000, Chilvers et al. 2005a).

4.2.7.1 DISEASE

Epizootic diseases can be a significant threat to New Zealand sea lion populations; for example, *Campylobacter*, which is thought to have killed 1600 pups (53% of pup production) and at least 74 adult females on the Auckland Islands in 1997–98 (Wilkinson et al. 2003, Robertson & Chilvers 2011).

More recently, *Klebsiella pneumoniae* killed 33% and 21% of new pups at the Auckland Islands in 2001–02 and 2002–03, respectively (Wilkinson et al. 2006), and 55% of pups between 2009 and 2014 (Roe et al. 2014). A hypermucoviscous (highly-sticky) strain of *K. pneumoniae* was isolated from a number of pups that died in field seasons 2005–06 to 2009–10 (Roe 2011). In this period, disease-related mortalities occurred late in the field season relative to the period 1998–99 to 2004–05 and were still occurring up to the end of sampling (Castinel et al. 2007, Roe 2011). *K. pneumoniae* was found to have caused, on average, 60% of pup deaths annually at Enderby Island between 2013 and 2018 (Table 4.4, with likely more continuing mortality following pup dispersal and the cessation of the summer monitoring season (Michael et al. 2019)). By comparison, less common causes of pup death over this time included starvation (14.8%), trauma or asphyxiation (9.9%), and other infections (7%) (Michael et al. 2019).

The 1998 epizootic event may also have affected the fecundity of the surviving pups, reducing their breeding rate relative to other cohorts (Gilbert & Chilvers 2008), though the pupping rate estimate for this cohort is likely to have been negatively biased by particularly high tag shedding rates for individuals tagged in that year (Roberts et al. 2014a).

Table 4.2: Pup census estimates for all known breeding populations of New Zealand sea lions since 1994–95. Years with no census estimates were left blank (i.e., blanks do not necessarily indicate that no pups were born at that location in that year). See table 1 of Breen et al. (2016) for a summary of counts from years prior to 1990 and the review by Childerhouse & Gales 1998. (Continued on next page)

Pupping season	Annual pup census estimate					
	Auckland Islands			Campbell Island	Otago coast	Stewart Island
	Dundas Island	Sandy Bay	All			
1990		434				
1991		429				
1992	1 934	489				
1993	1 870	432				
1994						
1995	1 837	467	2 518		0	
1996	2 017	455	2 685		1	
1997	2 260	509	2 975		0	
1998	2 373	477	3 021		2	
1999	2 186	513	2 867		1	
2000	2 163	506	2 856		1	
2001	2 148	562	2 859		3	
2002	1 756	403	2 282		3	
2003	1 891	488	2 516	385	3	
2004	1 869	507	2 515		3	

Pupping season	Annual pup census estimate					
	Auckland Islands			Campbell Island	Otago coast	Stewart Island
	Dundas Island	Sandy Bay	All			
2005	1 587	441	2 148		4	
2006	1 581	422	2 089		7	
2007	1 693	437	2 224		4	
2008	1 635	448	2 175	583	6	
2009	1 132	301	1 501		5	
2010	1 369	385	1 814	681	6	
2011	1 089	378	1 550		6	16
2012	1 248	361	1 684		6	25
2013	1 491	374	1 940		6	26
2014	1 213	290	1 575		4	32
2015	1 230	286	1 576	696	8	36
2016	1 347	321	1 727		15	31
2017	1 549	349	1 965		16	41
2018	1 397	332	1 792	734	18	55
2019	1 295	319	1 679	705	17	52
2020	1 398	289	1 740	595	21	48

Table 4.3: Pup production and population estimates of New Zealand sea lions from the Auckland Islands. Pup production data are direct counts or mark-recapture estimates from Chilvers et al. (2007), Robertson & Chilvers (2011), Chilvers (2012a), and Childerhouse et al. (2014, 2015b, 2016), noting that counts of dead pups began later in 2013 and 2014 and this is likely to have led to a negative bias in estimates for these years. Standard errors apply only to the portion of pup production estimated using mark-recapture methods. Mature female and total female population estimates are from the base case model by Roberts (2019). Year refers to the second calendar year of a breeding season (e.g., 2010 refers to the 2009–10 season). (Continued on next page)

Year	Pup production estimate		Mature female population size		Total female population size	
	Mean	Standard error (for mark recapture estimates)*	Median	95% confidence interval	Median	90% confidence interval
1995	2 518	21	3 151	2 834–3 505	6 920	6 373–7 552
1996	2 685	22	3 369	3 067–3 703	7 027	6 559–7 560
1997	2 975	26	3 602	3 317–3 913	7 183	6 793–7 622
1998	3 021	94	3 819	3 559–4 106	7 363	7 034–7 723
1999	2 867	33	3 976	3 746–4 232	7 544	7 247–7 848
2000	2 856	43	4 098	3 889–4 328	7 591	7 269–7 929
2001	2 859	24	3 817	3 640–4 013	7 218	6 925–7 515
2002	2 282	34	3 582	3 426–3 755	6 863	6 598–7 124
2003	2 518	38	3 391	3 253–3 545	6 536	6 294–6 767
2004	2 515	40	3 239	3 114–3 381	6 233	6 009–6 445
2005	2 148	34	3 096	2 978–3 231	5 949	5 740–6 146
2006	2 089	30	2 952	2 839–3 079	5 662	5 481–5 833
2007	2 224	38	2 813	2 704–2 936	5 390	5 224–5 555
2008	2 175	44	2 688	2 581–2 808	5 129	4 966–5 295
2009	1 501	16	2 578	2 473–2 692	4 931	4 769–5 101
2010	1 814	36	2 484	2 379–2 596	4 786	4 617–4 961
2011	1 550	41	2 466	2 373–2 571	4 733	4 575–4 898
2012	1 684	22	2 444	2 354–2 545	4 681	4 530–4 834
2013**	1 940	50	2 416	2 328–2 517	4 626	4 479–4 774
2014**	1 575	19	2 384	2 292–2 486	4 569	4 424–4 717
2015	1 576		2 355	2 262–2 457	4 512	4 363–4 661
2016	1 727		2 327	2 232–2 428	4 456	4 304–4 610
2017	1 965		2 299	2 202–2 402	4 401	4 242–4 561
2018	1 792		2 271	2 172–2 377	4 346	4 181–4 517
2019	1 679		2 244	2 141–2 355	4 293	4 120–4 473
2020	1 740					

* Calculated as the sum of standard errors associated with estimates for Sandy Bay and Dundas Island (estimates for other rookeries from direct count rather than mark-recapture).
** Field season began later in these years and pups that died early in the pupping period were unlikely to have been included in pup production estimates.
*** Roberts & Doonan (2016) estimated 11 755 for the entire species.

Table 4.4: Annual proportions of necropsied New Zealand sea lion (*Phocarctos hookeri*) pups at Sandy Bay, Enderby Island at the Auckland Islands that were attributed to *Klebsiella pneumoniae* infection, for field seasons between 2013–14 and 2017–18 (Michael et al. 2019).

Field season	Total necropsies	Number (and percentage) attributed to *K. pneumoniae* infection
2013–14	69	48 (70%)
2014–15	58	41 (71%)
2015–16	33	13 (39%)
2016–17	75	40 (53%)
2017–18	49	29 (59%)
Total	284	171 (60%)

4.2.7.2 INDIRECT/ TROPHIC EFFECTS OF FISHING

It is possible that indirect fisheries effects may have population-level consequences for New Zealand sea lions. Such indirect effects may include competition for food resources between various fisheries and New Zealand sea lions (Robertson & Chilvers 2011, Roberts et al. 2018). To determine whether resource competition is present and is having a population-level effect on New Zealand sea lions, research has sought to identify if there are resources in common for New Zealand sea lions and the various fisheries within their preferred foraging range, and to what extent those resources are limiting. Diet studies have revealed some overlap in the species consumed by New Zealand sea lions and those caught in fisheries within the range of New Zealand sea lions, particularly hoki and arrow squid (Cawthorn et al. 1985, Childerhouse et al. 2001, Meynier et al. 2009). Meynier et al. (2014) analysed energy and amino acid content of prey and determined that the selected prey species contained all essential amino acids and were of low to medium energy levels. This study concluded that given low energy densities of prey, sea lions may be able to sustain energy requirements, but not necessarily store energy reserves and, thus, sea lions may be sensitive to factors that negatively affect trophic resources. Meynier (2010) also developed a bio-energetic model and used it to estimate that roughly 17 871 t of prey are consumed by New Zealand sea lions at per year. This is about 30% of the annual harvest of arrow squid, and about 15% of the hoki harvested annually by the fisheries in the sub-Antarctic between 2000 and 2006; note however that later research suggests that squid and hoki do not constitute the major portion of sea lion diet (Roberts et al. 2018)

Comparison of the temporal and spatial distributions of sea lion prey, sea lion foraging, and of historical fishing extractions may help to identify the mechanisms whereby resource competition might occur (Bowen 2012), but the potential trophic effects of fishing on sea lions are likely to be complicated due to complex food web interactions. Multi-species models may help to assess the extent to which resource competition can impact on sea lion populations. Roberts et al. (2018) investigated the abundance and distribution of sea lion prey species, including via a dedicated trawl survey, and suggested that the Auckland Islands sea lion subpopulation has endured a protracted period of nutritional stress, such that during unfavourable periods this population may have been limited by the availability of key prey. However, conclusions regarding the extent to which this may reflect indirect fisheries effects are inconclusive pending a more thorough understanding of sea lion diet and foraging behaviour under different environmental conditions, and the relative impacts of other threats including episodic climate-induced mortality events and/or disease.

4.2.7.3 CLIMATIC AND/OR FISHERIES-RELATED DRIVERS OF NUTRITIONAL LIMITATION

Temporally coincident changes in annual abundance, spatial distribution, and/or reproductive success have been observed in different ecosystem components at sub-Antarctic latitudes — including New Zealand sea lions, Antipodean albatrosses, and demersal and pelagic fish communities. These observations suggest that climatic variability at decadal scales (sometimes labeled ‘regime shifts’) may affect ecosystem productivity in these systems, in turn affecting critical demographic rates for sub-Antarctic islands sea lions and resulting decadal-scale population dynamics (see below). For instance, a retrospective analysis of trawl survey data from the Campbell Plateau found evidence for a decade-long period of very low hoki abundance at depths foraged by New Zealand sea lions (Roberts et al. 2018). Long-term shifts in the catch rates of arrow squid around the Auckland Islands and the Stewart-Snares shelves were estimated from commercial trawl data and appear to be correlated with changes in primary production through time (Hurst et al. 2012).

Fisheries New Zealand is progressing new research (project ZBD2018-05) to investigate the evidence for regime shifts in the marine ecosystem of the sub-Antarctic and adjacent areas, likely climatic drivers, and potential ecological consequences as reflected in the productivity of fish and megafauna species. This project aims to identify climate and productivity indices for monitoring changes to the ecosystem that affect New Zealand sea lions and other focal ecosystem components.

4.2.8 RELATING DEMOGRAPHIC RATES TO DRIVERS OF POPULATION CHANGE

Over several years, various demographic assessments have been conducted to identify the proximate demographic causes of observed population and pup production trends at the Auckland Islands (see MacKenzie 2011, Roberts et al. 2014a, Roberts & Doonan 2016, Roberts 2017b, Roberts 2019). Roberts et al. (2014a) concluded that that the substantial decline in pup production between 1999 and 2009 was a consequence of low pupping rates during this period (including occasional years with very low rates), a declining trend in cohort survival to age two (pup survival) and to age five (juvenile survival) since the early 1990s, and relatively low adult survival (age 6–14) from 1999–2000 to 2010–11 (Figure 4.4.) In particular, very low pup survival rates at the Sandy Bay colony in 2005–2007, if they are indicative of similar processes occurring also at other breeding locations (i.e., Dundas Island and Figure of Eight Island), are likely to have compromised breeder numbers and pup production in later years (Roberts & Doonan 2016).

The subsequent change in the observed pup production trajectory (from declining in 1999–2009 to stable in 2009–2019) appears to have been driven by increased juvenile survival (ages 2–5) and increased adult survival (ages 6–14) and by a slightly increased pupping rate, rather than by a significant increase in pup survival, which remains lower than was observed before the period of population decline (Roberts 2019). This work suggests that further improvement in the observed trend may not be possible without an increase in pup survival rates. The extent to which pup survival is affected by the disease *K. pneumoniae* (see below) on an ongoing basis remains a priority for future research. New modelling work was completed in 2020 (Edwards & Roberts 2021, Roberts et al. 2021) to evaluate the extent to which pup morphometric data (including mass and condition), collected annually by the DOC field team at the Auckland Islands, can be used as a covariate to improve demographic model predictions and better understand potential drivers of observed demographic rates affecting population trends (e.g., first year survival of pups and annual pupping rate). A correlative assessment was conducted to identify the causes of varying demographic rates at Sandy Bay, for which hypothetical models developed with expert consultation were used as a framework for testing relationships between demographic rate estimates, biological observations (e.g., diet composition, maternal body condition, or pup mass), and candidate drivers of population change (e.g., changes in prey availability, disease-related pup mortality, or direct fishery-related mortalities) (Roberts & Doonan 2014).

Climate indices including Interdecadal Pacific Oscillation (IPO) and sea surface height (SSH) were well correlated with the occurrence of an array of key prey species in scats (Childerhouse et al. 2001, Stewart-Sinclair 2013). A weak, though significant, positive correlation was identified between maternal body condition and pup mass in seasons from 1990–91 to 2004–05. In this time period, pup mass at three weeks appeared to have been a good predictor of cohort-specific survival to age two, though there was no relationship with cohorts born 2004–05 to 2009–10, for which survival estimates were consistently low despite high pup mass (Figure 4.5). A correlation between cohort survival to age two and the rate of pup mortalities attributed to *K. pneumonia* infection late in the field season (Castinel et al. 2007, Roe 2011) was consistent with disease-related mortality affecting a decline in pup/yearling survival after 2004–05. Survival at ages 2–5 years (juveniles) or ages 6–14 years (adults) were not correlated with the estimated level of fishery interactions in the Auckland Islands arrow squid (SQU 6T) trawl fishery (Thompson et al. 2011). However, from 1998–99 to 2003–04 survival at ages 6–14 years was negatively correlated with the survival of pups born in the previous year, suggesting that the high energetic costs of lactation may compromise maternal survival (Roberts & Doonan 2014).

In most cases observations were available only for short time periods and longer series would be required to identify a causative relationship. However, broad changes in diet composition (e.g., an increased prevalence of small-sized prey species), reduced maternal body condition, and depressed pupping rates, are all consistent with a sustained period of nutritional stress negatively affecting the productivity of New Zealand sea lions at the Auckland Islands.

In addition, disease-related mortality of pups since 2005–06 (Roe 2011) has caused a decline in pup/yearling survival, which may further compromise breeder numbers at the Auckland Islands in the immediate future. It has been suggested that nutritional stress can be expected to predispose the population to higher rates of disease mortality, such that pups in poorer condition may be expected to have higher rates of death from disease. However, preliminary blubber depth data up to the 2014 field season indicated that pups dying from *Klebsiella* infection were in comparatively good condition relative to pups dying from other causes (preliminary data from W. Roe, unpublished, see Figure 4.6). As such, if poor body condition once predisposed pups to death from *Klebsiella* infection, then this no longer appears to be so. The relative contributions of *Klebsiella* infection, nutritional stress, or other factors potentially affecting first year mortality of pups are unclear, these are being investigated in ongoing research (e.g., Edwards & Roberts 2021, Roberts et al. 2021).

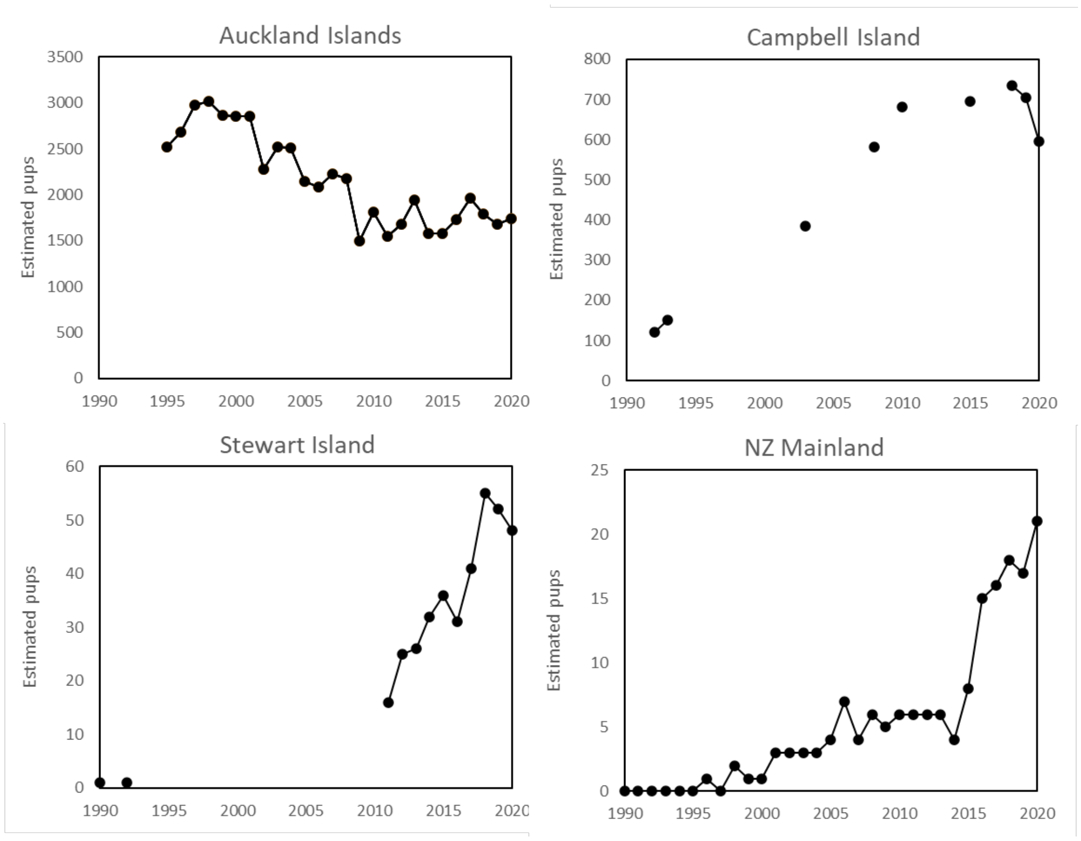

Figure 4.3: Annual sea lion pup count estimates from breeding sites (DOC unpublished data, McNutt et al. 2020, Melidonis & Childerhouse 2020). Note that the y-axis scale is different in each figure. Where count methodology was not consistent between adjacent years, annual point estimates are not joined by solid lines in the figures (e.g., some years in the Stewart Island and Campbell Island figures). Note that because the location of the Campbell Island breeding colony appears to be changing, it is possible that the 2020 pup production estimate is biased low (McNutt et al. 2020).

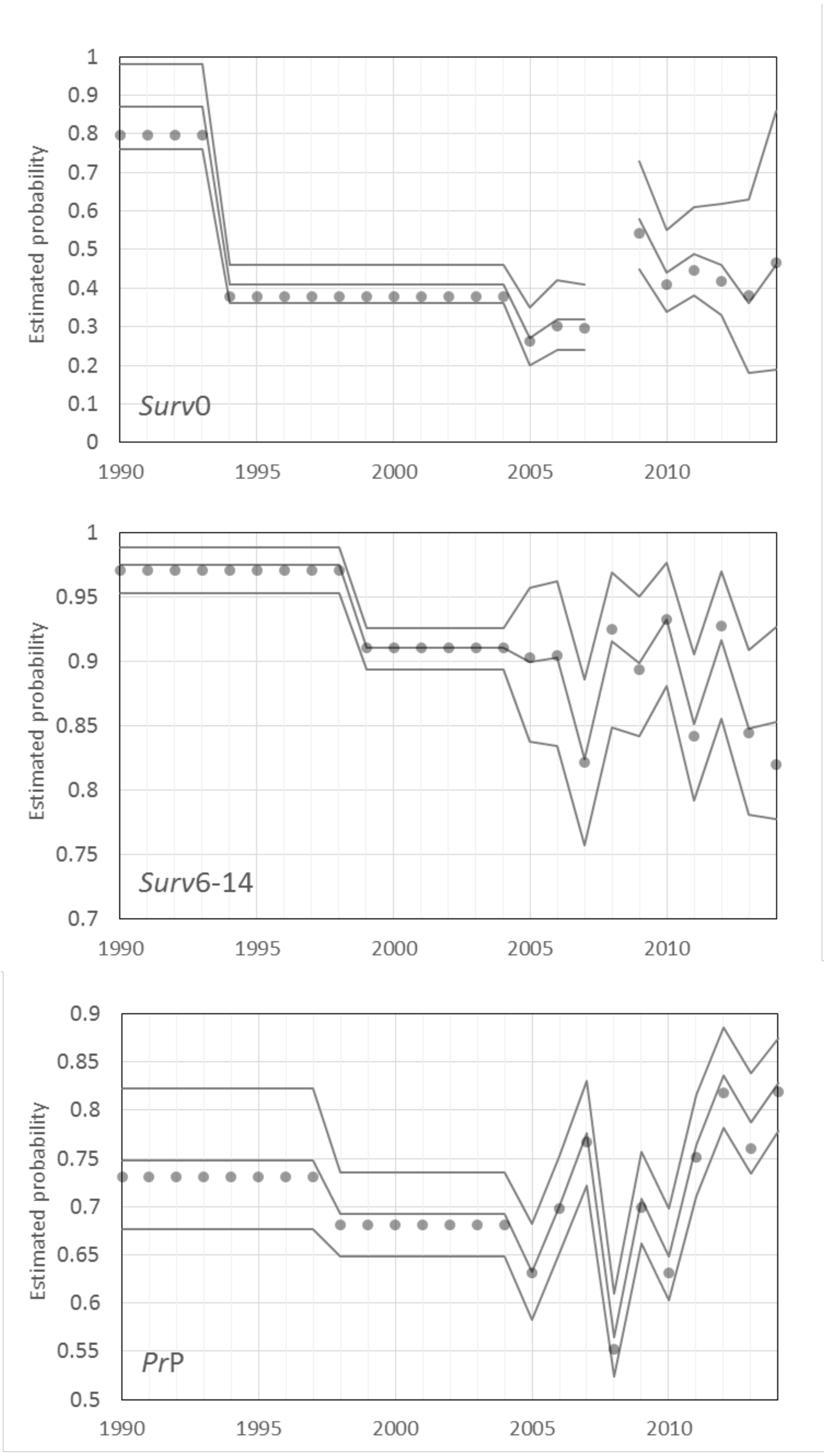

Figure 4.4: Annual estimates of pup survival to age 1 year(top), annual survival at age 6–14 years (middle), and annual probability of pupping (bottom) of female New Zealand sea lions at the Auckland Islands; points are point estimates; lines are median estimates and 95% c.i. (Roberts & Doonan 2016). Note that terminal estimates of very high pupping rates (> 80%) may be implausible; it is likely that annual data collection favouring Sandy Bay rather than Dundas Island is forcing the model to explain increased pop production as a consequence of increased pupping rate rather than increased female population size at Dundas Island. Collecting mark-recapture data from Dundas Island would address this potential issue but will face logistical obstacles.

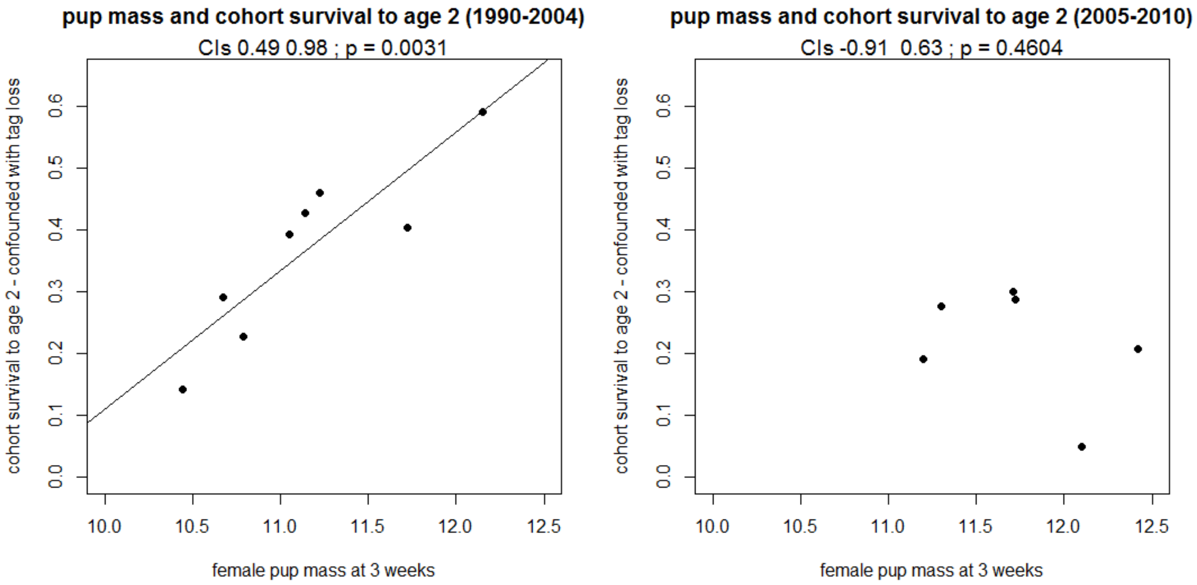

Figure 4.5: Pup mass of females and demographic modelling estimate of cohort survival to age 2; survival estimates confounded with tag loss rate; regression line shown for correlations significant at the 5% level.

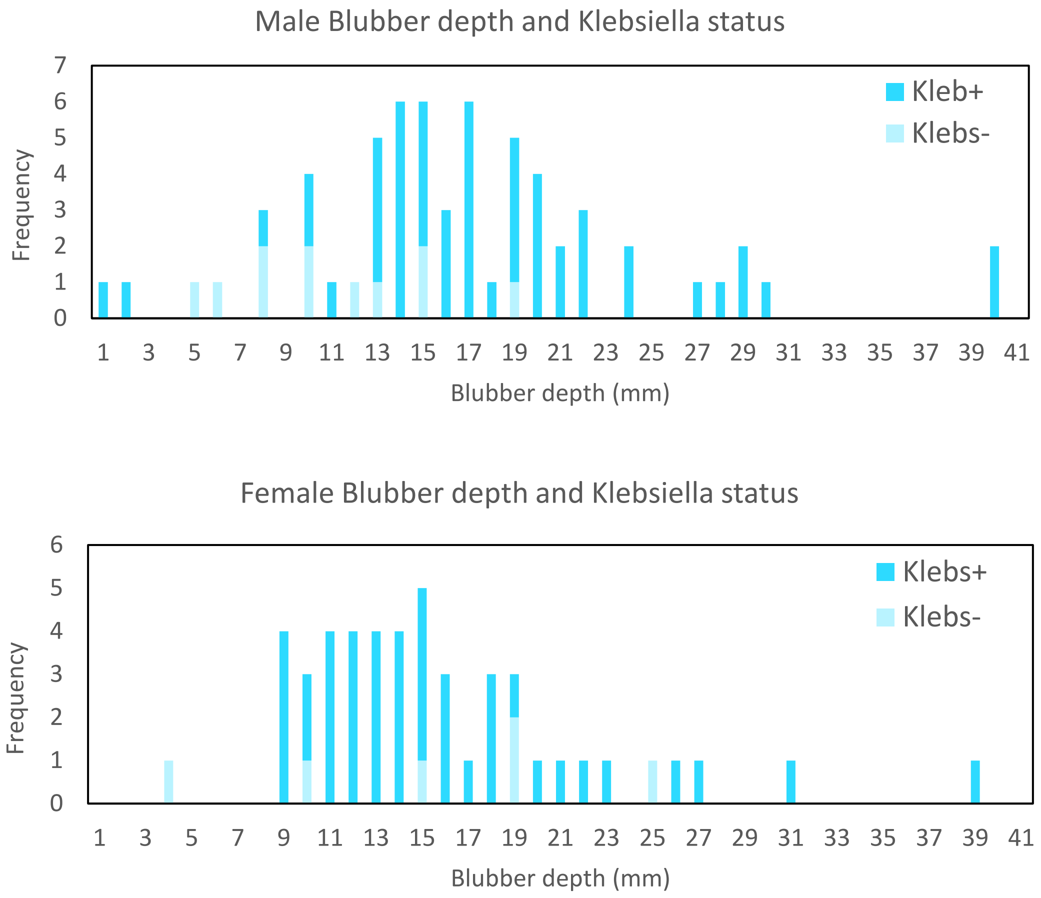

Figure 4.6: Blubber depth of necropsied Auckland Islands sea lion pups for which *Klebsiella pneumoniae* was or was not identified as the cause of death based on histology and/or tissue culture (preliminary data from W. Roe unpublished). All bodies were sampled in February, in the 2006–07 to 2009–10 and 2013–14 field seasons.

4.2.9 CONSERVATION THREAT STATUS CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2010). The threat status of New Zealand sea lions has been assessed under two threat classification systems, the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2010) and the New Zealand Threat Classification System (Townsend et al. 2008).

In 2015, the IUCN updated the Red List status of New Zealand sea lions, listing them as Endangered, on the basis of a projected decline in breeders over three generations (calculated to be 32 years) exceeding a 50% reduction (estimated to be 72%), assuming a linear extrapolation of the observed rate of decline in pup production at the Auckland Islands between 1997–98 and 2008–09 (Chilvers 2015). In 2013, the New Zealand Threat Classification status for New Zealand sea lions was changed from At Risk, Range Restricted[2] to Nationally Critical under criterion C[3] (with a Range Restricted qualifier) based on the same observations of declining population trend at the Auckland Islands (Baker et al. 2016).

In 2019 the New Zealand Threat Classification status for New Zealand sea lions was updated to Nationally Vulnerable (Baker et al. 2019) reflecting that the formally declining population trend at the Auckland Islands has been approximately stable since 2010, and populations are increasing or stable at all other locations.

4.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Reviews of fisheries interactions among pinnipeds globally can be found in Woodley & Lavigne (1991), Read et al. (2006), Katsanevakis (2008), and Moore et al. (2009). Because New Zealand sea lions are endemic to New Zealand, the global understanding of fisheries interactions for this species is outlined under state of knowledge in New Zealand.

4.4 STATE OF KNOWLEDGE IN NEW ZEALAND

New Zealand sea lions interact with some trawl fisheries, sometimes resulting in incidental capture and death of the sea lion in the net. Observed trawl fishery interactions are confined to sub-Antarctic waters (Figure 4.7); particularly the two trawl fisheries around the Auckland Islands – the arrow squid fishery (SQU 6T) and the scampi fishery (SCI 6A). Male sea lions are caught in the southern blue whiting fishery near Campbell Island (SBW 6I) and occasional mostly male captures occur at the Stewart-Snares shelf in trawl fisheries targeting mainly arrow squid (SQU 1T; Thompson & Abraham 2010, Thompson et al. 2011, 2013).[4] New Zealand sea lions can forage to depths of 600 m but mainly overlap with trawling at depths of 180–220 m for trawls targeting arrow squid, 250–600 m for trawls targeting spawning southern blue whiting, and 350–550 m for trawls targeting scampi (Tuck 2009, Fisheries New Zealand 2020).

There is seasonal variation in the overlap between New Zealand sea lions and the target species fisheries (Table 4.5) Breeding male sea lions in the Auckland Islands area are ashore between November and January with occasional trips to sea, then migrate away from the area (Robertson et al. 2006). Breeding females are in the Auckland Islands area year-round, coming ashore for up to 10 days to give birth during December and January and then alternately foraging at sea (for about 2 days) and suckling their pup ashore (about 1.5 days; Chilvers et al. 2005b). The SQU 6T fishery currently operates between December and June, peaking between February and May, whereas the SQU 1T fishery operates between December and May, peaking between January and April, before the squid spawn. The SBW 6I fishery operates in August and September, peaking in the latter month, when the fish aggregate to spawn. The SCI 6A fishery typically operates between May and October.

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[2] A taxon is listed as 'Range Restricted' if it is confined to specific substrates, habitats, or geographic areas of less than 1000 km^2 (100 000 ha); this is assessed by taking into account the area of occupied habitat of all subpopulations (Townsend et al. 2008).

[3] A taxon is listed as 'Nationally Critical' under criterion C if the population (irrespective of size or number of subpopulations) has a very high (rate of) ongoing or predicted decline; greater than 70% over 10 years or three generations, whichever is longer (Townsend et al. 2008).

[4] See Fisheries plenary report (Fisheries New Zealand 2020) https://www.mpi.govt.nz/news-and-resources/science-and-research/fisheries-research.

Table 4.5: Monthly distribution of New Zealand sea lion activity and seasonal activity of trawl fisheries in which captures may occur.

New Zealand sea lions	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Breeding males	Dispersed at sea or at haulouts		At breeding colony			Dispersed at sea or at haulouts						
Breeding females	At sea			At breeding colony		At breeding colony and at-sea foraging and suckling						
New pups				At breeding colony								
Non-breeders	Dispersed at sea, at haulouts, or at breeding colony periphery											
Major fisheries	**Sep**	**Oct**	**Nov**	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**
Squid				Stewart-Snares shelf		Auckland Islands Shelf and Stewart-Snares shelf						
Southern blue whiting	Pukaki Rise and Campbell Rise											Bounty Islands
Scampi	Auckland Islands								Auckland Islands			

4.4.1 DIRECT ESTIMATION OF FISHERIES CAPTURES

Incidental captures of New Zealand sea lions are recorded by fisheries observers and used to estimate total capture rates including in unobserved fishing events.

From 2007 to 2017, in fisheries with low numbers of observed captures, total captures were estimated using simple ratio estimates; these included Auckland Islands scampi (SCI 6A) fishery, other Auckland Islands trawl fisheries, and the Stewart-Snares shelf fisheries (Thompson et al. 2013, Abraham & Berkenbusch 2017). Observed annual captures by target fishery are shown in Tables 4.6 and 4.7. Modeled total capture estimates for the Auckland Islands trawl fisheries have subsequently been replaced by spatially explicit estimation methods (Large et al. 2019; see below).

Model estimates by Abraham & Berkenbush (2017) for the Campbell Island and Snares-Stewart shelf fisheries are given in Table 4.6. Observed and estimated New Zealand sea lion captures and capture rates in the SBW 6I fishery have been highly variable. Following the 2012–13 season in which 21 male sea lion captures were observed in a very short period (17 dead and 4 released alive), the fishing industry took immediate action in consultation with the Crown to mitigate sea lion mortalities including 100% use of Sea Lion Exclusion Devices (SLEDs; see below). Since that time 100% of tows have been observed (Table 4.6); annual captures have ranged from 2 to 6 sea lions (annual average under 3).

For the SQU 6T and SBW 6I fisheries, in which the majority of historical captures have been observed, early models suggested that the rate at which sea lions interacted with trawl nets was influenced by a number of factors, including year, distance from the colony, tow duration, time of day, and change of tow direction (Smith & Baird 2005). Subsequently, Smith & Baird (2007a), Thompson et al. (2013), and Abraham & Berkenbusch (2017) applied Bayesian models using these and other categorical covariates to estimate total capture levels thereafter, with greater than 85% observer coverage since the 2012–13 fishing year.

For fisheries with observer coverage approaching 100%, statistical captures estimation that simply scales up from the observed to unobserved fishing effort became largely unnecessary. However, early in the same period during which observer coverage was increasing (i.e., 2001–02 to 2007–08), the SQU 6T fishery also transitioned to widespread adoption of SLEDs, a mitigation device designed to allow sea lions entering the trawl net to exit via the SLED and survive. Unsurprisingly, following the introduction of SLEDs to the SQU 6T fishery in 2001–02, both the observed and estimated numbers of New Zealand sea lion captures declined (Table 4.8 and 4.9). However, since the universal adoption of a single 'standard' SLED design in 2006–07, model estimates of *interaction rates* — i.e., the number of sea lions entering the net but potentially exiting again via the SLED — became increasingly uncertain over time, because the interaction rate was confounded in the model with a corresponding and inversely correlated estimate of SLED efficacy, both of which were effectively unobservable. In the most recent such models for Auckland Islands sea lions (Abraham & Berkenbusch 2017), the estimated interaction rates were effectively unbounded, and model estimates in particular years became unstable as new years of data were added. For this reason, from 2017 Fisheries New Zealand discontinued these models for Auckland Islands sea lions and applied a new approach under which *interactions, captures*, and *deaths* are estimated separately, combining an application of the spatial risk assessment (SEFRA) method described in Chapter 3 with a separate means of estimating cryptic mortality (i.e., unobservable deaths) as a function of the observable captures in trawls employing SLEDs. The outcomes of this new body of research are described below.

Table 4.6: Sea lion captures in Campbell Island trawl fisheries targeting southern blue whiting (SBW) and in Stewart-Snares shelf trawl fisheries targeting squid (SQU 1T) hoki, hake, ling and all other middle depth target species, between 2002–03 and 2019–20 (Protected species bycatch (protectedspeciescaptures.nz)). Annual fishing effort (total number of tows), observer coverage (percentage of tows observed), number of observed sea lion captures (both dead and alive), observed capture rate (captures per 100 tows), the estimation method used (model or ratio estimate), and the mean number of estimated sea lion captures (with 95% confidence interval, c.i.) (see Thompson et al. 2013 and 2016 for details). Data for subsequent years are provided by Fisheries New Zealand. * Standardised SLED designed in this year.

Fishing year	Fishing	Observed captures			Estimated captures		
	All effort	% observed	Number	Rate	Method	Mean	95% c.i.
Campbell Island/Antipodes Island/Pukaki Rise SBW							
2002–03	638	43	0	0.0	Model	1	0-3
2003–04	740	33	1	0.4	Model	3	1-9
2004–05	870	39	2	0.6	Model	5	2-13
2005–06	624	35	3	1.4	Model	10	3-22
2006–07*	630	36	3	1.3	Model	15	6-30
2007–08	816	41	5	1.5	Model	8	5-14
2008–09	1185	25	0	0.0	Model	1	0-7
2009–10	1111	36	11	2.8	Model	24	15-37
2010–11	1171	37	6	1.4	Model	15	8-25
2011–12	951	70	0	0.0	Model	1	0-4
2012–13	790	100	21	2.7	Model	21	21-21
2013–14	785	100	2	0.3	Model	2	2-2
2014–15	666	100	6	0.9	Model	6	6-6
2015–16	437	100	3	0.7			
2016–17	526	100	0	0.0			
2017–18	455	100	2	0.4			
2018–19	749	100	0	0.0			
2019–20	348	100	1	0.3			
Stewart-Snares shelf							
2002–03	6978	15	0	0.0	Model	3	0-7
2003–04	7403	16	1	0.1	Model	4	1-8
2004–05	8377	21	3	0.2	Model	7	3-11
2005–06	7267	15	1	0.1	Model	4	1-8
2006–07	6218	20	1	0.1	Model	3	1-6
2007–08	5026	29	1	0.1	Model	3	1-6
2008–09	4268	29	0	0.0	Model	1	0-4
2009–10	4886	35	1	0.1	Model	2	1-5
2010–11	4312	29	0	0.0	Model	1	0-4
2011–12	4393	35	1	0.1	Model	2	1-4
2012–13	4212	70	1	0.0	Model	2	1-4
2013–14	4066	58	0	0.0	Model	1	0-3
2014–15	4045	56	1	0.0	Model	1	0-3
2015–16	2903	66	1	0.1			
2016–17	3566	54	0	0.0			
2017–18	3610	73	1	0.0			
2018–19	4426	73	0	0.0			
2019–20	4700	74	0	0.0			

Relatively high observed capture rates of sea lions in the SQU 6T fishery before 2002, with moderate fishing observer coverage and highly variable total effort, suggested that substantial numbers of captures may be occurring. Observer coverage levels increased substantially in the SQU 6T fishery in the years since the 2002–03 season. Observer coverage ranged from 28–45% between the 2002–03 and 2011–12 fishing seasons and achieved high

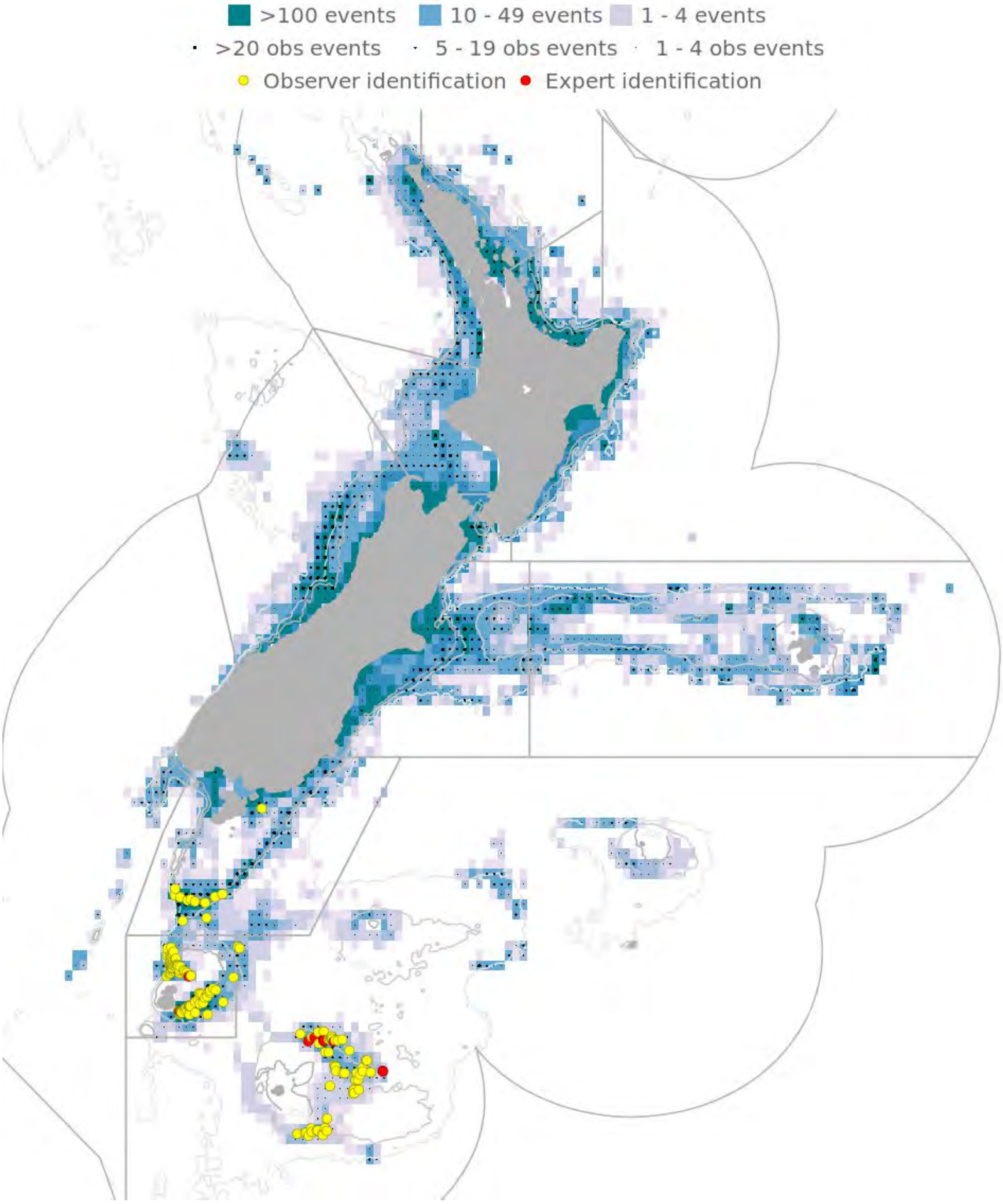

Figure 4.7: Spatial distribution of trawl fishing effort and observed New Zealand sea lion captures, 2002–03 to 2019–20 (https://protectedspeciescaptures.nz/PSCv6/). Fishing effort density is mapped into 0.2-degree cells, in blue and green. The corresponding level of fisheries observer coverage is indicated by the superimposed black dots; observed captures are indicated in yellow or red.

Table 4.7: Sea lion captures in Auckland Islands trawl fisheries, for trawls targeting scampi (SCI) and trawls targeting all species other than scampi and squid, from 1992–93 to 2016–17 (from Large et al. 2019). Data for subsequent years are provided by Fisheries New Zealand. Columns denote annual fishing effort (total number of tows), observer coverage (percentage of tows observed), and number of observed sea lion captures (combined for male and female sea lions, including both live and dead captures). Corresponding estimates of total fisheries deaths are shown in Figure 4.11 and Table 4.9.

Fishing year	SCI trawl			Other trawl		
	All effort	Observed (%)	Captures	All effort	Observed (%)	Captures
1992–93	835	18	3	195	18	0
1993–94	1 314	21	0	308	7	0
1994–95	1 349	4	0	492	7	0
1995–96	1 312	5	2	411	6	1
1996–97	1 227	16	1	296	4	0
1997–98	1 109	12	0	688	17	0
1998–99	1 255	2	0	525	10	0
1999–00	1 383	5	0	751	13	0
2000–01	1 419	6	4	577	7	0
2001–02	1 603	10	0	590	4	0
2002–03	1 351	11	0	532	13	1
2003–04	1 363	12	3	289	17	0
2004–05	1 275	0	–	169	7	0
2005–06	1 331	9	1	38	16	0
2006–07	1 328	8	1	38	5	0
2007–08	1 327	7	0	146	45	0
2008–09	1 457	4	1	120	50	0
2009–10	940	10	0	77	68	0
2010–11	1 401	15	0	131	37	0
2011–12	1 247	10	0	57	30	0
2012–13	1 093	12	0	60	43	0
2013–14	850	6	0	203	23	0
2014–15	548	0	–	224	31	0
2015–16	1 414	5	0	140	26	0
2016–17	1 677	21	0	170	51	0
2017–18	1 728	17	2	146	58	0
2018–19	1 637	21	1	102	61	0
2019–20	1 405	28	0	62	60	0

Table 4.8: Sea lion captures in Auckland Islands trawl fisheries targeting squid, shown separately for bottom trawl and midwater trawl gear configurations, from 1992–93 to 2016–17 (from Large et al. 2019). Data for subsequent years are provided by Fisheries New Zealand. Columns denote annual fishing effort (total number of tows), observer coverage (percentage of tows observed), and number of observed sea lion captures (separately for female and male sea lions, including both live and dead captures. Corresponding estimates of total fisheries deaths are shown in Figure 4.11 and Table 4.9. This table has not been updated for the 2021–22 edition due to the unavailability of the data at time of publication but will be updated in the subsequent edition. (continued next page)

Fishing year	Bottom trawl effort				Midwater trawl effort			
	All effort	Observed (%)	Female captures	Male captures	All effort	Observed (%)	Female captures	Male captures
1992–93	86	10	0	0	568	33	3	2
1993–94	0	–	2	1	3 226	7	0	1
1994–95	0	–	1	2	2 633	7	3	2
1995–96	721	0	0	0	3 747	15	10	3
1996–97	0	–	2	7	2 177	25	7	12
1997–98	242	19	2	2	1 219	24	2	9
1998–99	89	33	1	0	313	41	3	1
1999–00	455	15	1	0	751	50	12	12
2000–01	173	99	6	4	410	99	16	13
2001–02*	498	21	2	0	1 149	40	12	7
2002–03*	738	34	2	1	728	23	5	3
2003–04*	1 452	17	3	1	1 142	47	11	1
2004–05*	1 375	21	5	2	1 318	39	0	2

Fishing year	Bottom trawl effort				Midwater trawl effort			
	All effort	Observed (%)	Female captures	Male captures	All effort	Observed (%)	Female captures	Male captures
2005–06*	1 905	13	3	0	554	55	7	0
2006–07*	732	43	2	1	585	38	4	0
2007–08*	634	43	2	2	631	50	1	0
2008–09+	1 068	34	1	1	857	46	0	0
2009–10+	1 026	23	2	0	162	41	1	0
2010–11+	1 218	30	0	0	365	49	0	0
2011–12+	973	34	0	0	308	78	0	0
2012–13+	813	83	3	0	214	100	0	0
2013–14+	477	83	2	0	260	87	0	0
2014–15+	328	92	0	0	305	84	1	0
2015–16+	822	87	0	0	543	100	0	0
2016–17+	1 090	67	2	0	204	78	1	0
2017–18+	987	88	2	0	143	100	0	0
2018–19+	712	96	3	4	94	88	0	0

* denotes years in which SLEDS were deployed on a variable proportion of trawls, in the absence of a standard design or systematic inspection and audit programme.

\+ denotes years in which SLEDs were deployed universally on all trawls, with a standard design and a systematic inspection and audit programme.

4.4.2 SPATIAL FISHERIES RISK ASSESSMENT FOR AUCKLAND ISLANDS SEA LIONS

The widespread introduction of SLEDs in the SQU 6T trawl fishery created the need for a new modelling approach in which *interactions, captures,* and *deaths, including cryptic deaths,* can be estimated separately, and under which estimation of the interaction rate is not confounded by the SLED efficacy rate. These terms are defined clearly here to avoid confusion, noting that before the adoption of the new modelling approach in 2019, the same terms may have been applied in a less consistent way.

Captures are sea lions captured in nets and brought on deck (both dead and alive). Captures necessarily exclude the animals that exit trawls through the SLED, as well as bodies that are recovered in a decomposed state hence presumed to be already dead at the time that the body entered the net (Smith & Baird 2007b, Thompson & Abraham 2010, Thompson et al. 2013).

Interactions in the SQU 6T fishery are defined as the number of sea lions that enter the net alive and would have been captured if no SLED had been used. Until 2017 interactions were estimated using a statistical model fitting to observed capture rates both before and after the deployment of SLEDs, with an additional term to approximate the presumed level of 'SLED efficacy', i.e., the proportion of interactions in which the sea lion exits via the SLED and survives (Thompson et al. 2013). For trawl fisheries that do not deploy SLEDs, the number of estimated interactions is equivalent to the number of estimated captures.

Deaths include both observable captures (excluding animals released alive and presumed to survive) and also *cryptic deaths*, i.e., animals which are not recovered on board the vessel or otherwise observable (i.e., 'captured') even in the presence of a fisheries observer but are nonetheless expected to die as a direct consequence of their interaction with the fishing gear.

Prior to the introduction of SLEDs there was no feasible mechanism by which sea lion bodies could be accounted for if lost or unable to be observed, such that *interactions* = *captures* and *captures* $\geq$ *deaths* (i.e., *cryptic deaths* = 0). After the successful introduction of SLEDs, the modelling approach used previously became increasingly ill-suited to estimating sea lion deaths due to uncertainties about the rate at which sea lions were exiting via the SLED, and the potential for cryptic mortality. From 2019 an adaptation of the SEFRA approach outlined in Chapter 3, was used instead, in which each of the critical rates are estimated empirically. See Large et al. (2019) for a full description of the spatial risk modelling for Auckland Islands sea lions; key outputs of this work are summarised below.

4.4.2.1 SPATIAL FORAGING DISTRIBUTION MODELLING

Satellite telemetry data indicative of spatial foraging patterns for Auckland Islands sea lions were compiled and analysed to predict the most likely foraging track per trip (i.e., removing implausible location outliers). Tracks were then used to estimate the spatial density of female sea lions (estimated separately for adults and juveniles). Due to the high density of the available telemetry data, and the incomplete spatial coverage of prey availability information, the best fitting spatial models used simple geographic covariates (latitude, longitude, depth, and distance to colony) rather than true habitat variables. Groomed satellite telemetry data and the resulting combined density layer, scaled for the relative abundance of adults and juveniles, are shown in Figure 4.8.

Note that to the extent that the three different breeding colonies (Sandy Bay, Dundas Island, and Figure of Eight Island) may exhibit distinct spatial foraging patterns, there may be spatial biases in the estimated spatial density used to inform the risk assessment, reflecting that the majority of the data were collected at Sandy Bay. Furthermore, all available data were collected in summer; the seasonal bias will not affect estimation of fisheries risk in the squid fishery (which occurs in summer) but may create additional uncertainty with respect to fishing in other seasons (e.g., scampi trawl fisheries). Additional tracking studies are planned to address these potential imbalances.

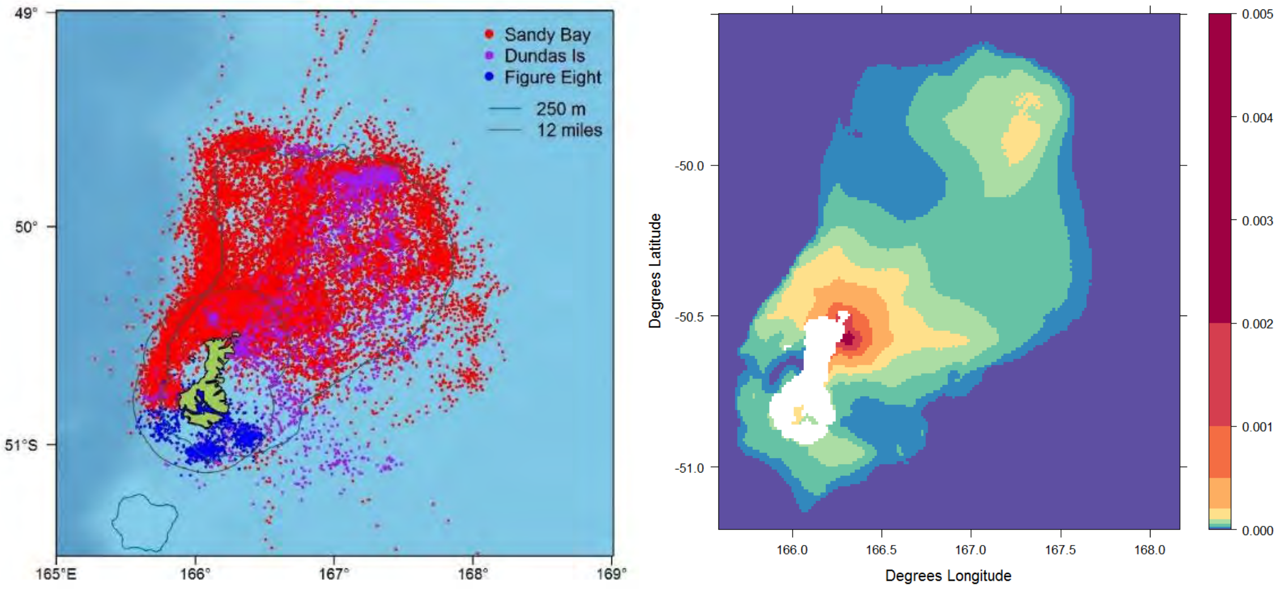

Figure 4.8: Groomed and filtered fix locations for female sea lions tagged at three Auckland Islands breeding colonies (left) and estimated spatial density (on a relative scale; all cell values sum to 1) of all female Auckland Islands sea lions (age 2+), as used in spatial risk models (right). Reproduced from Large et al. (2019).

4.4.2.2 SEA LION CATCHABILITY IN COMMERCIAL TRAWL FISHERIES

Under the SEFRA method, encounters between sea lions and fishing effort are proportional to the spatial overlap between the sea lion distribution and the distribution of fishing effort. Catchability is the probability of capture per encounter.

Catchability was estimated separately in eight different trawl fishery groups, as follows. First, fishery groups were divided by target fishery, i.e., squid target fisheries vs. scampi target fisheries vs. ‘other deepwater trawl’. Next, within the squid fishery, catchability was estimated separately for ‘bottom trawl’ vs. ‘midwater trawl’ gear configurations (noting that both gear configurations are actually deployed in contact with the sea floor, but the length and headline height of the nets differs between them). Finally, each squid fishery gear type was divided into three categories relating to the deployment of SLEDs as follows:

- no SLED deployed (all effort prior to the 2001 season) and a declining proportion of effort thereafter

- non-standard SLED: SLEDs deployed during the years 2001–2007, during which the design of the SLED had not been standardised and there was no systematic audit to ensure proper deployment

- standard SLED: from 2008 onward, all fishing effort used a standardised SLED design subject to verification under a systematic inspection process (Cleal et al. 2007)

Sea lion catchability in these eight fishery groups is shown in Figure 4.9. This figure indicates that catchability may be highest in 'other trawl' fisheries (e.g., hoki trawl), but this result is highly uncertain and has almost no contribution to actual risk because spatial overlap is very low (i.e., there is almost no fishing effort in this category occurring near the Auckland Islands). Of the fisheries that do overlap with Auckland Islands sea lions, catchability is estimated to be highest in scampi trawls.

Comparing catchability estimates among squid fishery groups reveals important patterns. In both the midwater and bottom trawl gear configurations, the effectiveness of SLEDs at reducing catchability is clear, but this effect was only realised after SLED designs were standardised and audited, from 2008. When catchability in midwater trawls vs. bottom trawls is compared, it appears that without the use of SLEDs, sea lions are more likely to be captured in midwater gear, but, with SLEDs, captures are more likely in bottom trawl gear. These results imply that in a given encounter with fishing effort, a sea lion is more likely to enter a midwater net than a bottom trawl net but is also more likely to exit successfully from a midwater net via the SLED. This may be related to the higher headline height of midwater nets relative to bottom trawl nets; note however that these indications are uncertain (the confidence intervals overlap).

4.4.2.3 ESTIMATED CAPTURES, DEATHS, AND POPULATION-LEVEL RISK OVER TIME

Because spatially resolved fishing effort data are available from 1993, by applying the estimated catchabilities in Figure 4.9 it is possible to estimate historical changes in fisheries captures over time, including the effect of changing effort levels, changing spatial fishing effort patterns, and changing sea lion population sizes. Cryptic mortality in trawls employing SLEDs is estimated separately by Meyer (2019; see below); risk reflects fisheries deaths as a proportion of population size, with an implied population impact limit (which is a policy decision). Figures 4.10a and 4.10b. show that estimated sea lion deaths in the squid fishery peaked in the early to mid-1990s, declined to a low in 1999 reflecting greatly reduced fishing effort levels, increased again as effort levels increased to a lower peak in 2006, then declined dramatically reflecting the universal adoption of standardised SLEDs, which reduced sea lion catchability from the 2009 season onward (Large et al. 2019). Cumulative impacts across all trawl fisheries are shown in Figure 4.11 and Table 4.9. Note that the analysis does not include the include the 1980s period of relatively high squid fishery effort, when SLEDs were not used, and annual mortalities were likely to be high relative to the following, assessed period (Large et al. 2019).

These figures suggest that since the universal adoption of standardised SLEDs by the squid fishery in 2008, scampi target fisheries rather than squid target fisheries may now be responsible for the largest proportion of commercial fisheries risk to Auckland Islands sea lions; note however that this conclusion is uncertain due to possible spatial and seasonal biases in the spatial data informing this model.

Table 4.9: Estimated deaths of female Auckland Islands sea lions in trawl fisheries targeting squid, scampi, and other target species, from 1992–93 to 2016–17, from the SEFRA fisheries risk model by Large et al. (2019). Squid trawl fishery estimates combine both the midwater and bottom trawl fishery groups and include cryptic mortality in trawls utilising SLEDs, as estimated by Meyer (2019). This figure has not been updated for the 2021–22 edition due to the unavailability of the data at time of publication but will be updated in the subsequent edition.

Fishing year	Squid trawl		Scampi trawl		Other trawl	
	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.
1992–93	10	5–16	5	1–10	0	0–2
1993–94	82	61–108	12	4–21	1	0–4
1994–95	74	54–97	9	4–17	1	0–3
1995–96	83	62–108	8	3–16	1	0–6
1996–97	51	36–70	6	2–12	1	0–5
1997–98	28	18–39	7	2–13	2	0–8
1998–99	6	3–12	8	3–15	1	0–4
1999–00	19	12–28	6	2–11	1	0–5
2000–01	8	4–14	6	2–12	0	0–3
2001–02*	23	14–32	6	2–12	1	0–5
2002–03*	22	14–32	5	1–10	1	0–5
2003–04*	31	21–44	7	2–13	0	0–1
2004–05*	37	25–51	8	3–15	0	0–1

Fishing year	Squid trawl		Scampi trawl		Other trawl	
	Median	95% c.i.	Median	95% c.i.	Median	95% c.i.
2005–06*	35	22–50	6	2–12	0	0–1
2006–07*	17	10–25	6	2–12	0	0–0
2007–08*	17	10–26	5	1–10	0	0–1
2008–09+	5	2–11	7	2–13	0	0–1
2009–10+	4	1–8	2	0–6	0	0–1
2010–11+	4	1–9	5	12–0	0	0–1
2011–12+	3	0–6	3	1–8	0	0–1
2012–13+	3	0–6	3	0–7	0	0–1
2013–14+	1	0–4	2	0–6	0	0–1
2014–15+	1	0–4	1	0–4	0	0–2
2015–16+	2	0–6	3	1–8	0	0–1
2016–17+	3	1–7	5	1–9	0	0–2

* denotes years in which SLEDS were deployed on a variable proportion of squid target trawls, in the absence of a standard design or systematic inspection and audit programme.

\+ denotes years in which SLEDs were deployed universally on all squid target trawls, with a standard design and a systematic inspection and audit programme.

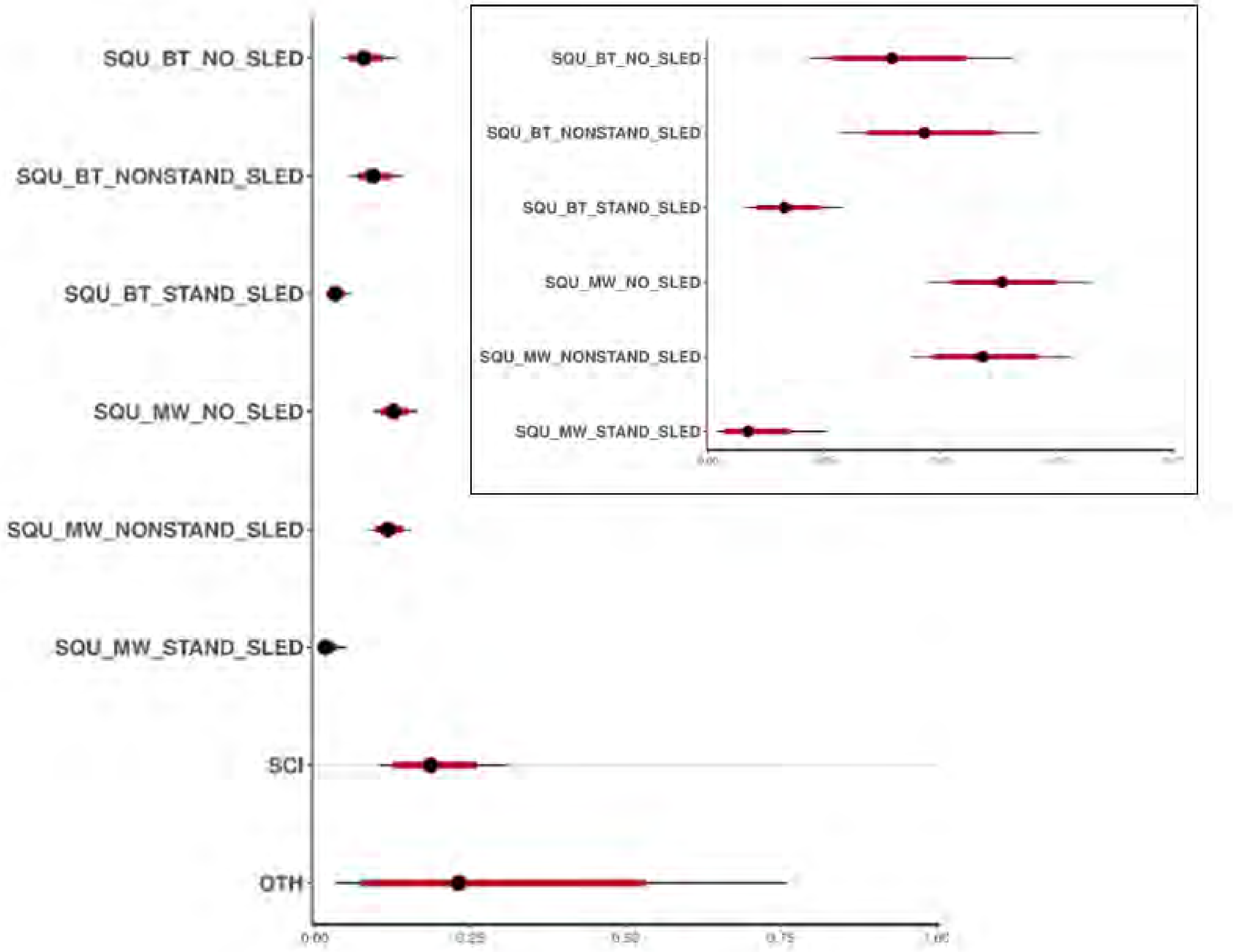

Figure 4.9: Estimated catchability of female New Zealand sea lions in commercial trawl fishery groups: 'SQU_BT_NO_SLED' = bottom trawls targeting southern arrow squid without a sea lion exclusion device (SLED), 'SQU_BT_NONSTAND_SLED' = bottom trawls targeting squid with a SLED in the period before SLED designs were standardised and universally audited; 'SQU_BT_STAND_SLED' = bottom trawls targeting squid with a SLED using a standardised configuration; other groups containing 'MW' instead of 'BT', were as above except that a midwater trawl was used, 'SCI' = scampi trawl, 'OTH' = trawls targeting all other species at the Auckland Islands. Posteriors for fishery groups targeting southern arrow squid are also shown in an embedded plot, with the x-axis rescaled to make outputs easier to read. Reproduced from Large et al. (2019).

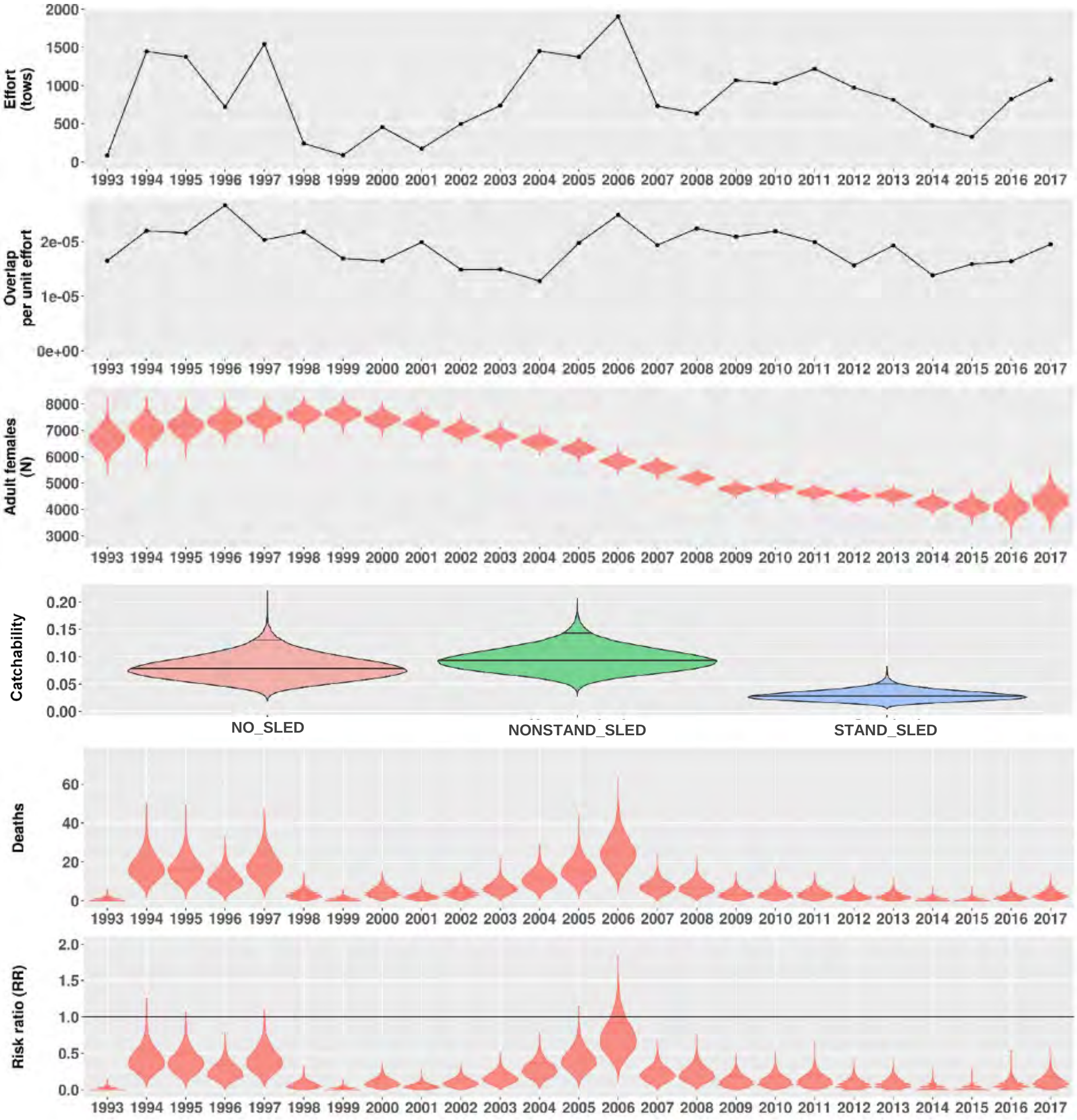

Figure 4.10a: Time series spatial risk model outputs for female Auckland Islands sea lions in the 'bottom trawl' fishery group targeting southern arrow squid (SQUBT): total effort; spatial overlap per unit effort; population size (females only); catchability; annual deaths (females only); and risk ratio. Note that the risk ratio in this model run assumed a calibration coefficient (ϕ) of 0.1. (Large et al. 2019).

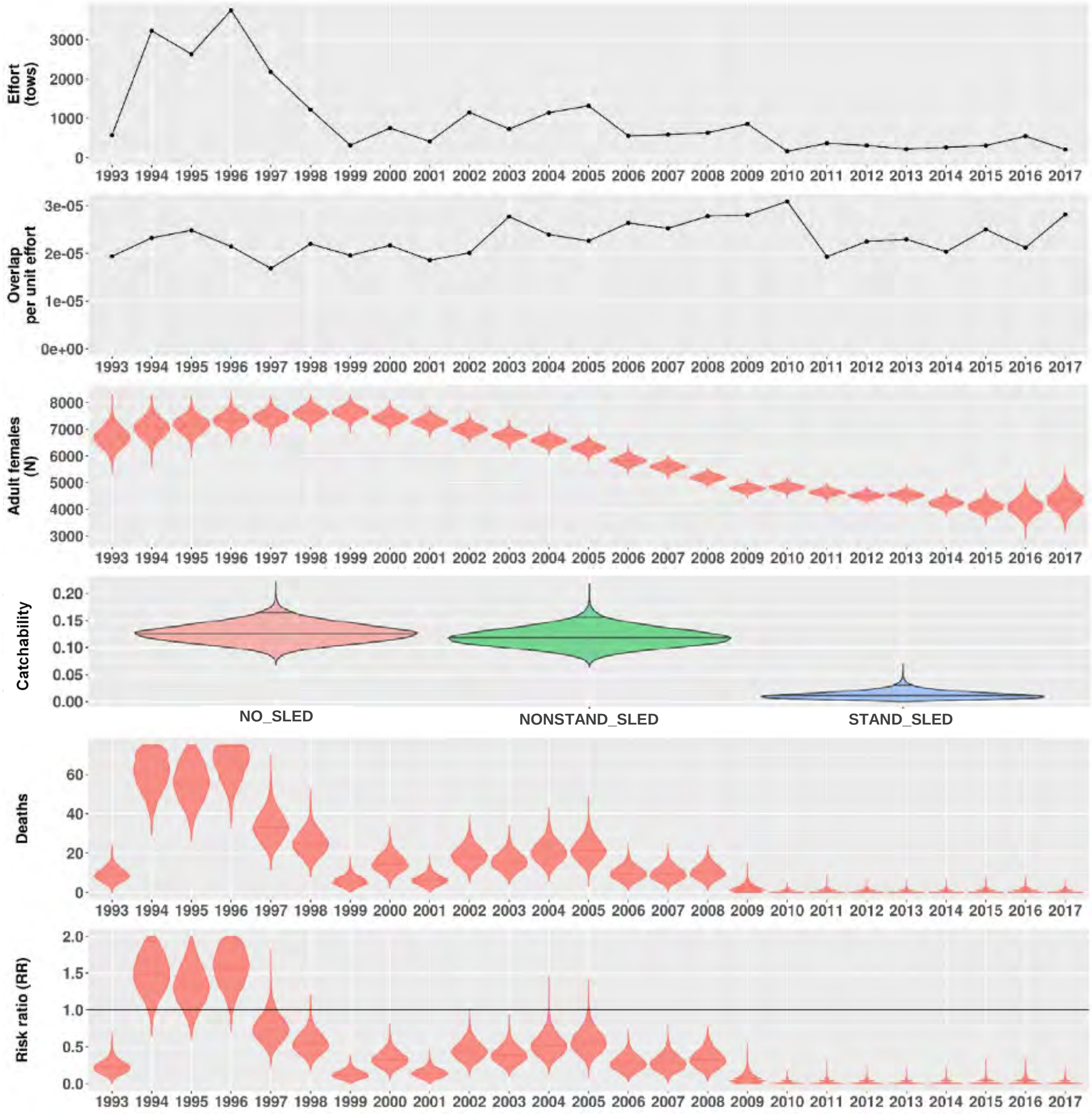

Figure 4.10b: Time series spatial risk model outputs for female Auckland Islands sea lions in the 'midwater trawl' fishery group targeting southern arrow squid (SQUBT): total effort; spatial overlap per unit effort; population size (females only); catchability; annual deaths (females only); and risk ratio. Note that the risk ratio in this model run assumed a calibration coefficient (ϕ) of 0.1. (Large et al. 2019).

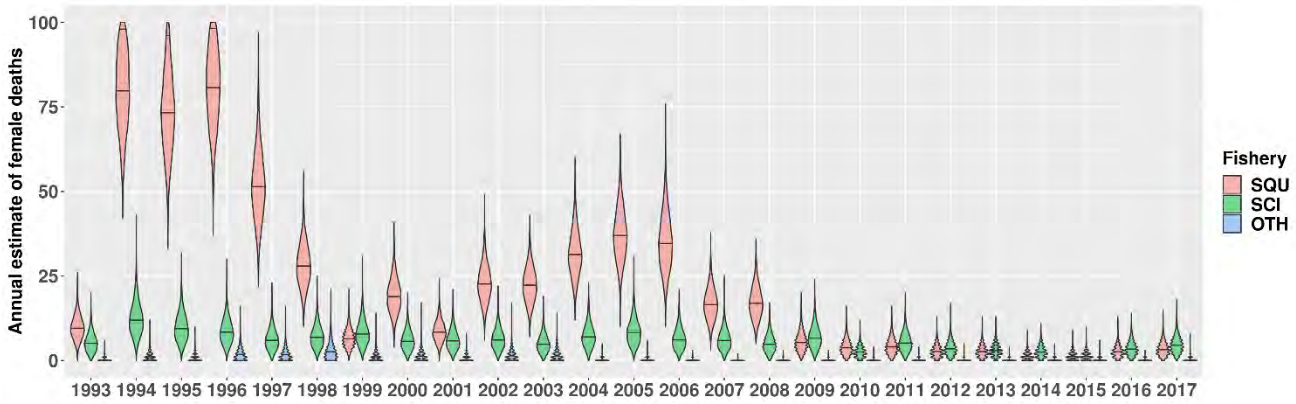

Figure 4.11: Estimated annual deaths of female Auckland Islands sea lions in commercial trawl fisheries from 1992–93 to 2016–17, aggregated by target species: 'SQU' = southern arrow squid, 'SCI' = scampi, 'OTH' = trawls targeting all other species. (Large et al. 2019).

4.4.3 SEA LION EXCLUSION DEVICE (SLED) DEVELOPMENT AND USE

In 2004, the Minister of Fisheries requested that Squid Fishery Management Company, government agencies, and other interested stakeholders work collaboratively to develop a plan of action to determine SLED efficacy. In response, an independently chaired working group (the SLED Working Group) was established to develop an action plan to determine the efficacy of SLEDs, with a particular focus on the survivability of New Zealand sea lions that exit the nets via the exit hole in the SLED. The group undertook a number of initiatives, most notably the standardisation of SLED specifications (including grid spacing) across the fleet (DOC CSP project MIT 2004/05; Cleal et al. 2007) and the establishment of an underwater video monitoring programme to help understand the fate of New Zealand sea lions that exit the net via the SLED. White light and infra-red illuminators were tested. Sea lions were observed outside the net on a number of occasions, but only one fur seal and one New Zealand sea lion were observed exiting the net via the SLED (on tows when white light illumination was used). The footage contributed to understanding of SLED performance but established that video monitoring was only suitable for tows using midwater gear, because the camera view was often obscured on tows where bottom gear was used (Middleton & Banks 2008, Middleton 2019a). The SLED Working Group was disbanded in early 2010.

Figures 4.12 and 4.13 illustrate the SLED design and the way in which New Zealand sea lions interact with the trawl gear and the SLED itself.

4.4.4 CRYPTIC MORTALITY OF SEA LIONS IN TRAWLS WITH SLEDS

SLEDs are effective in allowing most New Zealand sea lions to exit a trawl (see Figure 4.12), but occasionally a sea lion does not exit and is drowned and retained in the net. These are recorded as observed captures. However, there remains some uncertainty about the fate of sea lions that are not retained in the net, some of which may nonetheless die as a consequence of the interaction. Interactions that result in unobservable deaths are termed '*cryptic mortality*'. Sources of cryptic mortality are best understood by categorising four potential outcomes of a sea lion entering a trawl:

i. exits the net via SLED and survives (survivor);
ii. dies in net and is retained (observable capture);
iii. dies in the net but the body is subsequently lost without being recovered on the vessel ('body non-retention');
iv. exits the SLED but is at the limit of its 'breath hold' and drowns before reaching the surface ('post-escape drowning').

Collectively, points iii and iv constitute cryptic mortality. Previously, a fifth potential outcome had been defined, i.e., v) 'exits the net but dies from head injuries sustained during interaction with the SLED' ('mild traumatic brain injury', or MTBI). However, upon review of the process by which sea lions interact in trawls with SLEDs, it was judged that the effect of MTBI will be to affect the rate at which a sea lion exits (or does not exit) the net, thereby affecting the likelihood of other outcomes (e.g., capture or post-escape drowning). Therefore, MTBI does not constitute a separate outcome in itself (see Figure 4.13).

The following section describes research undertaken to estimate various components potentially contributing to cryptic mortality. In 2019, this information was integrated into a Bayesian estimation model to estimate cryptic mortality empirically (Meyer 2019), the results of which are summarised separately below.

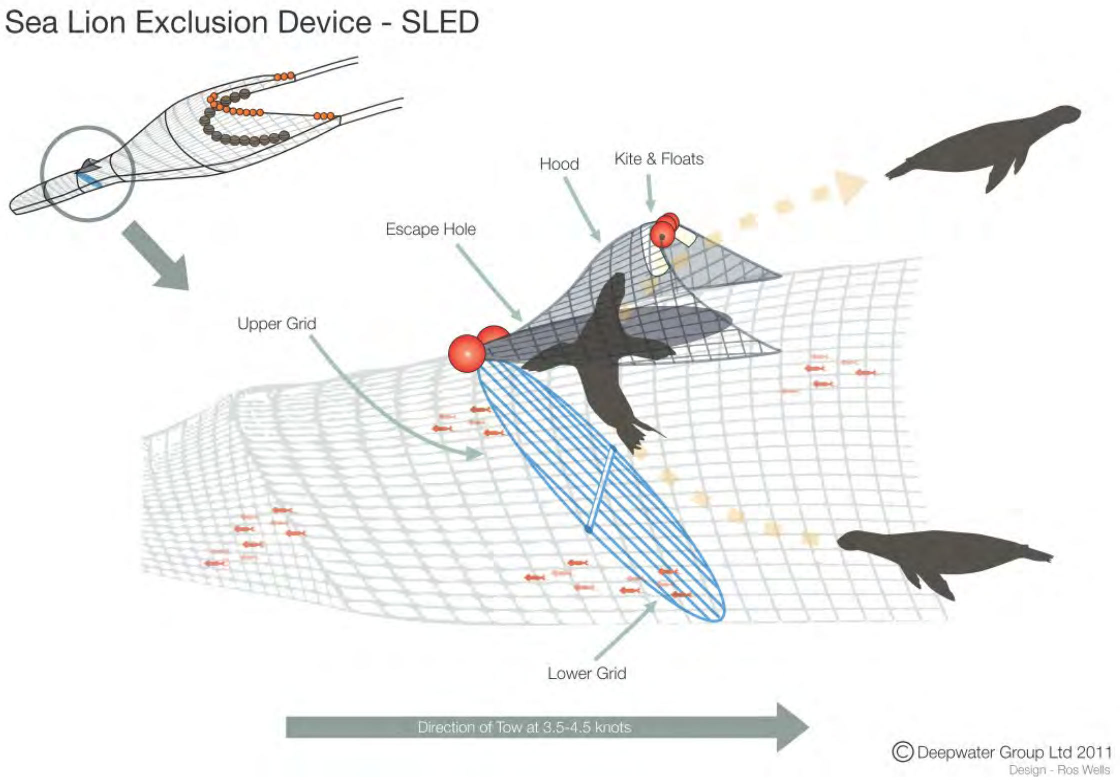

Figure 4.12: Diagram of a New Zealand sea lion exclusion device (SLED) inside a trawl net. Image courtesy of the Deepwater Group Ltd.

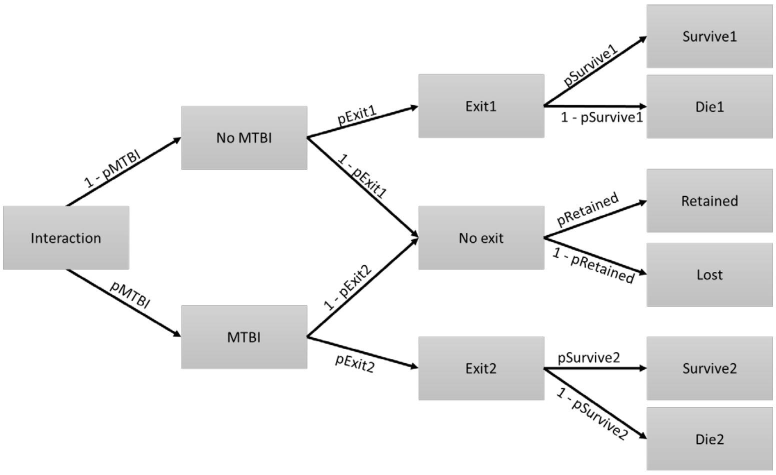

Figure 4.13: State transition process for New Zealand sea lions interacting with trawl nets that have SLEDs deployed, as developed under consultation with AEWG (November 2018). Boxes are categorical states; variables atop of arrows denote transition probabilities. MTBI is 'mild traumatic brain injury'. See section 4.4.4.4 for details.

4.4.4.1 MILD TRAUMATIC BRAIN INJURY

Sea lions may strike their heads on the SLED grid. To look for evidence of injury arising from such collisions, necropsies were conducted on animals recovered from cover net trials (see below) and on those incidentally caught and recovered from vessels operating in the SQU 6T, SQU 1T, and SBW 6I fisheries. All several hundred New Zealand sea lions returned for necropsy died as a result of drowning rather than physical trauma from interactions with the trawl gear including the SLED grid (Roe & Meynier 2010, Roe 2010). Necropsies were designed to assess the nature and severity of trauma sustained during capture and to infer the survival prognosis had those animals been able to exit the net (Mattlin 2004). However, problems associated with this approach limited the usefulness of the results. For example, sea lions had to be frozen on vessels and stored for periods of up to several months before being thawed for 3–5 days to allow necropsy. Roe & Meynier (2010) concluded that this freeze-thaw process created artefactual lesions that mimic trauma but, particularly in the case of brain trauma, could also obscure real lesions. Further, two reviews in 2011 concluded that the lesions in retained animals may not be representative of the injuries sustained by animals that exit a trawl via a SLED (Roe & Meynier 2010, Roe 2010).

Notwithstanding the limitations of the necropsy data in assessing trauma for previously frozen animals, it was possible to determine that none of the necropsied animals sustained sufficient injuries to the body (excluding the head) to compromise survival (Roe & Meynier 2010, Roe 2010). However, the potential for head trauma arising from impacts with the SLED grid could not be ruled out as a potential contributing factor to an animal’s death (Roe & Meynier 2010, Roe 2010).

Abraham (2011) used biomechanical modelling, euphemistically referred to as ‘crash-test-dummy’ modelling, to quantify the likelihood of a sea lion experiencing physical trauma sufficient to render the animal insensible (and therefore likely to drown) arising from a collision with a SLED grid. This work used video footage of Australian fur seals interacting with comparable trawl exclusion devices (Wilcox 2008, Lyle & Wilcox 2011) to estimate (for sea lions) the likelihood of a head-first impact, the speed of impact, the angle of impact relative to individual SLED grid bars and relative to the grid plane, and the location of impact on the grid. The risk of MTBI was then assessed by biomechanical testing and modelling across a range of plausible and ‘worst-case’ impact scenarios (Ponte et al. 2010, 2011) and combined in a simulation-based probabilistic model (Abraham 2011). In the base case model, 2.7% of sea lions entering the trawl were estimated to experience MTBI; in the most extreme sensitivity the estimate was as high as 8.2%. These results indicate that rates of death by MTBI for New Zealand sea lions interacting with the SLED grid are likely to be low. It is thought that animals affected by MTBI may be more likely than uninjured animals to remain and drown in the net, where they will be counted among observed captures unless the body is subsequently lost from the net. For this reason, MTBI may influence the rate at which sea lions exit or drown, but MTBI is not in itself a source of cryptic mortality.

4.4.4.2 BODY NON-RETENTION

From first principles and considering SLED design (Figure 4.12) it seems unlikely that body non-retention rates are high, because:

i) the escape opening of SLEDs employed in New Zealand fisheries is at the top of the net, whereas drowned pinnipeds are observed to be negatively buoyant;
ii) forward-facing hoods are designed to allow exit for actively swimming animals and retain passive or inert bodies due to the forward motion of the net; and
iii) hood floats are designed to close the escape opening in the event that the trawl net becomes inverted (turns upside down) or when the net reaches the surface of the water.

Preliminary results of SLED monitoring trials in overseas jurisdictions support the conclusion that drowned pinnipeds are likely to be lost, and thereby not counted among observed captures, in trawls employing SLEDs. Overseas researchers with first-hand knowledge of the operation of these devices were consulted in the process of parameterising the cryptic mortality model (i.e., representing assumptions about body non-retention as a model input prior). Informed by expert input, the Aquatic Environment Working Group estimated that between 1 and 10% of drowned sea lion carcasses may be subsequently lost from the net (i.e., a uniform prior on pRetention of 0.90 – 0.99).

4.4.4.3 POST-ESCAPE DROWNING

Between 1999–2000 and 2002–03, an experimental approach was taken to estimate interaction rates and SLED efficacy rates, by intentionally capturing animals as they exited the escape hole of a SLED. Cover nets were added over the escape holes of some SLEDs and sea lions were restrained in these nets after they exited the SLED. An underwater video camera was deployed in 2001 to assess the behaviour and the likelihood of post-exit survival of those animals that were retained in the cover nets (Wilkinson et al. 2003, Mattlin 2004). Due to low sample sizes and ambiguous interpretation of necropsy results, this work was judged to be inconclusive (Roe 2010). Re-analysis of the video data in 2019 indicated that at least some of the animals were conscious and active at the time that they exited the net, but the number of observations was too low to draw any quantitative conclusions.

In 2019, data on sea lion dive behavior and trawl characteristics in the SQU 6T fishery were used to simulate the outcome of dives in which sea lions interact with SLED-equipped trawls, to estimate the probability of post-escape drowning (Middleton 2019b). This study used electronic telemetry data indicative of sea lion dive behavior under 'normal' foraging conditions to characterise critical rates such as: i) how long a sea lion can remain conscious underwater (euphemistically labeled 'maximum breath-hold' although pinnipeds actually expel their air before diving; oxygen is stored in the blood); ii) descent speed; iii) horizontal swimming speed; and iv) ascent speed. These data were available from sea lions tagged specifically at the Auckland Islands (Crocker et al. 2001, Chilvers et al. 2006).

The simulation tracked the fate of sea lions as they passed through the net using a time step whereby every minute underwater was subtracted from the animal's remaining 'breath-hold' time (i.e., time at which the animal will run out of oxygen and become unconscious). The study used video data of Australian fur seals in nets equipped with comparable exclusion devices, to estimate the likelihood in each minute that an animal inside the net will exit via the SLED (Lyle & Wilcox 2008). Animals that contribute to cryptic mortality are those that: fail to exit before becoming unconscious and drown and are retained in the net; exit the net with sufficient time to reach the surface survive; exit the net, but with too little conscious time remaining to reach the surface, and are presumed to drown outside the net.

In the simulation base case, roughly 7% of animals exiting the net are estimated to nonetheless drown, and the ratio of total deaths to deaths that occur inside the net was estimated at 1.4 (range 1.2–1.5 depending on what proportion of the animals successfully exit, which varied between bottom trawl and midwater trawl nets.) These outputs were used to inform the definition of priors for the Bayesian estimation of cryptic mortality from all sources (Meyer 2019).

4.4.4.4 BAYESIAN ESTIMATION OF SEA LION CRYPTIC MORTALITY IN TRAWLS USING SLEDS

New research was completed in 2019 integrating all available information indicative of the fate of sea lions that enter trawls equipped with SLEDs, in a Bayesian modelling framework (Meyer 2019). Models constructed under this project used a state transition matrix of different possible states for each sea lion that enters the net. The probability of each state transition was estimated by fitting to observed captures data or was influenced by the priors (which were estimated outside the model or informed by expert knowledge).

The state transition matrix used to estimate cryptic mortality in sea lions is reproduced in Figure 4.13. Transition probabilities were informed as follows:

- the number of sea lions entering the net (annual interactions) were estimated outside the model (separately for midwater and bottom trawl fishery groups) as a function of spatial overlap between fishing effort and sea lion distribution (Large et al. 2019);
- the probability of a sea lion suffering MTBI was informed by 'crash-test-dummy' modelling described above, mean pMTBI = 0.027 (from Abraham 2011);
- the probability that a sea lion exits the net was fitted to observed captures — mean pExit = 0.57 for bottom trawls and 0.88 for midwater trawls;
- the probability that an exiting sea lion survives (i.e., does not drown before reaching the surface) was informed by simulation as described above (Middleton 2019b) — mean pSurvive = 0.94 for both bottom trawl and midwater trawl;

- the probability that the body of a sea lion that drowns in the net is retained was estimated by the AEWG, informed by expert discussion and observations of comparable exclusion devices for Australian fur seals (in which all drowned carcasses were retained in the net). Mean pRetention = 0.945 for both bottom trawl and midwater trawl.

Combining these estimates, the model estimates that the cryptic mortality multiplier is 1.15 in bottom trawls (95% c.i. 1.05–1.31) and 1.60 for midwater trawls (95% c.i. 1.20–2.63). These estimates were used to convert estimated captures to deaths in the risk assessment by Large et al. (2019).

Note that the apparent higher catchability in bottom trawls is offset by a lower cryptic mortality, suggesting sea lions are less likely to enter a bottom trawl relative to a midwater trawl, but also less likely to exit successfully via the SLED. Because these two factors work in opposition, the actual risk to sea lions per trawl event is similar between the two gear types.

Sensitivity analyses conducted by Meyer (2019) indicate that the model estimates of cryptic mortality are not highly sensitive to expert-derived assumptions (reflected as priors for the transition probabilities) within plausible ranges. The parameter with the most ability to affect the cryptic mortality multiplier is the probability that drowned carcasses are retained in the net, but major changes are required to this prior (e.g., assuming more than half of all drowned carcasses are lost from the net) to have large effects on the cryptic multiplier. High non-retention rates are not consistent with known hydrodynamic principles or observations of fur seal carcasses retained by comparable exclusion devices in Australia.

4.4.5 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES DEATHS

Consistent with terminology used in the SEFRA methodology (Chapter 3), Fisheries New Zealand has now adopted the term ‘Population Sustainability Threshold’ or PST to denote the number of anthropogenic deaths that a population can sustain while still meeting a defined population recovery or stabilisation outcome, evaluated via simulations using a demographic population model. The choice of reference outcome is a policy decision.

For Auckland Islands sea lions, the likely effect of fisheries impacts at different levels was estimated from a demographic population model informed by mark-recapture observations, annual pup census results, estimated fisheries-related deaths, and the estimated age distribution of lactating females, as described by Roberts & Doonan (2016). The model also supported a quantitative risk assessment to estimate the effects of non-fishery threats (section 4.4.5.1 below). In 2017, additional model runs were carried out under project SEA2026-30, incorporating the newest pup count data from Figure 4.3 (but not including all available mark-recapture data), and were used to update management options for the Operational Plan for the SQU 6T fishery in 2017 (Roberts 2017b).

The population projections by Roberts & Doonan (2016) relied on the untestable structural assumption that future demographic rates would approximate historically observed rates from the preceding 20 years. This period included the period of steepest population decline and subsequent apparent stabilisation, but not the preceding period of population growth. It is likely that changes in demographic rates reflect changing environmental conditions. Because it is not possible to anticipate what environmental conditions are likely to prevail in the future, with unknown potential consequences for sea lion demographic rates, uncertainty of this nature is best addressed with model sensitivities.

In 2019 the Auckland Islands sea lion demographic model of Roberts (2017b) was updated with all available data including mark-recapture information up to the 2018–19 field season (Roberts 2019). To address environmental uncertainty, the updated model was structured to estimate demographic rates separately based on periods of historically increasing (1990 to 1998), decreasing (1999 to 2009), and stable pup production trajectories (2010 to 2019) and to simulate the effects of fisheries mortality on population outcomes under these three regimes, reflecting that it is likely that decadal-scale climatic variability affects critical sea lion demographic rates, but future climate conditions cannot be foreseen.

Models also incorporated estimates of SQU 6T fishery related deaths, derived from the outputs of separate research projects estimating historical fishery captures (Large et al. 2019) and cryptic mortality levels relating to the use of sea lion exclusion devices (Meyer 2019).

The base case model produced good fits to pup census, mark-resighting, and age distribution observations. Model estimates indicate that the observed change in pup production trajectory in 2009 (from decreasing to stable) was driven by increased annual survival at age groups 2–5 years and 6–14 years, and a slightly increased annual pupping rate, rather than by improved first-year 'pup' survival, which was unchanged relative to the period of decline.

The base model estimated a current population size of 4293 females in 2019 (95% CI = 4120–4473), which was subsequently used in the estimation of the PST.

A new PST criterion was defined by Fisheries New Zealand, SEFRA approach (Chapter 3). A calibration coefficient (ϕ) of 0.1 was selected as the base case value by Fisheries New Zealand, such that annual impacts equal to the PST are consistent with a stable population size at 95% of the un-impacted level. At this level of ϕ, a female-only PST of 26 individuals was estimated. In comparison, Large et al. (2019) estimated 2.8 (median) or 5.4 (upper 95% CI) actual female deaths in the squid fishery from 2013 to 2017.

Assuming future fisheries mortality equal to the PST (at ϕ = 0.1), i.e., 26 female deaths per year, model projections estimated a mature female population size in 2025 of between 95.0% (95% CI = 94.7–95.2%) and 96.1% (95% CI = 95.8–96.3%) of what would have occurred in the absence of fishery mortality (depending on the future population growth scenario). Under the 'stable' (recent) demographic rate scenario, future deaths consistent with recent estimated levels would result in a population size in 2025 of between 99.0% (95% CI = 99.0–99.1%) and 99.5% (95% CI = 99.5–99.5%) of un-impacted levels, depending on whether the upper 95% CI or median of recent annual deaths was assumed, respectively.

Note that because this modelling framework does not account for density dependence, these estimates of future population status will be lower than would be estimated from a generic application of the PST formulation using *Rmax* (equation 30 in Chapter 3). In that formulation, applying a value of (ϕ = 0.1) yields an impacted population that is 5% lower than the un-impacted population in the long term (at equilibrium) because population productivity increases to compensate as the population is reduced below carrying capacity.

4.4.5.1 MULTI-THREAT QUANTITATIVE RISK ASSESSMENT FOR AUCKLAND ISLANDS SEA LIONS

In 2016 a quantitative risk assessment estimating the potential impacts of both fisheries and non-fishery threats to sea lions was undertaken to inform the development of a Threat Management Plan for the species (Roberts & Doonan 2016). On the advice of the AEWG, for purposes of informing management, this model replaced the previous 'BFG model' that had been used and updated since 2000 (Breen et al. 2003, Breen & Kim 2006a, 2006b).

A panel of national and international independent experts, supported by relevant subject matter advisors, was convened to provide guidance on the level of threats to New Zealand sea lions and review the demographic assessment. The first of two workshops was held from 28 April to 1 May 2015. It built on previous discussions at a pup mortality workshop held in 2014, but considered all threats to all sea lion age groups. The initial stage of the risk assessment model – the demographic assessment – was completed in advance of the first workshop, for the panel to review and provide recommendations for model improvements (Debski & Walker 2016).

Separate demographic assessment models were developed for females at the Auckland Islands and Otago Peninsula, integrating information from mark-recapture observations, pup census, and the estimated age distribution of lactating females (Auckland Islands only). With respect to the Auckland Islands assessment, good fits were obtained to all three types of observation and the model structure and parameter estimates appeared to be a good representation of demographic processes that have affected population decline there (primarily low pup survival and low adult survival). The Otago Peninsula assessment made use of a much smaller number of observations; however, the assessment still produced good estimates of all key demographic rates, with much higher pup survival relative to the Auckland Islands population (Roberts & Doonan 2016).

A two-stage assessment of the effects of threats was undertaken where the consequences of removing the effects of a threat were estimated in terms of the population growth rate of mature individuals in 2037. This used threat-specific mortality estimates at age (provided by MPI/DOC subsequent to two dedicated TMP workshops, see Debski & Walker 2016), in which:

1. 'triage' projections were undertaken for all assessed threats using the upper bound estimates of threat-related mortality to screen out threats that had little effect on projected growth rate;

2. 'best-estimate' projections were undertaken using the best estimate of threat-specific mortality for all threats that passed through the triage stage (Roberts & Doonan 2016).

The triage of the risks posed to New Zealand sea lions was conducted to limit the number of risks to be included in the more detailed Markov chain Monte Carlo (MCMC) modelling. To do this, a simple model was used to assess the upper bound, or worst case scenario, of the threat by predicting the response of the population to that threat being removed. The results of this triage are not considered to be the best estimate of the risks posed to the New Zealand sea lions, but a mechanism to reduce the list of the threats to those that have the largest influence.

Triage model run projection outputs for the Auckland Islands using the final model are shown in Figure 4.14 and for the Otago Peninsula population in Figure 4.15. The black line in Figures 4.14 and 4.15 indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately.

The effects of removing the threats that act on pups (i.e., *K. pneumoniae*, hookworm, wallows[5]) have a delayed effect on the size of the mature population of sea lions. This is because the pups that will survive still need time to mature before they are included in the modelled mature female population (Roberts & Doonan 2016, Debski & Walker 2016).

Removal of the upper bound of *Klebsiella* risk creates the largest change in population size over the 20-year time period (2017–37), however the population reacts more quickly to the removal of the upper bound of estimated trawl interactions because this acts directly on the mature females. The ratio of mature female population in 2037 compared with 2017 is 1.30 when *Klebsiella* is removed, and 1.24 when trawl interactions are removed (Roberts & Doonan 2016). The independent panel considered that some of the upper bounds used in the triage process were unlikely to be realistic and should be treated with caution (Debski & Walker 2016).

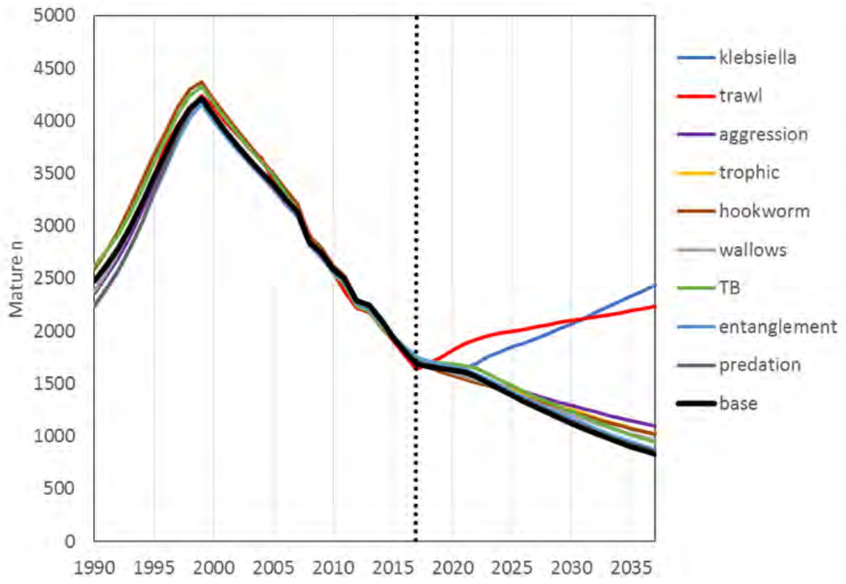

Figure 4.14: Triage projections of model estimated number of mature individuals (mature n) at the Auckland Islands during 1990–2037, using upper values of threat mortality. The black dotted line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines. Reproduced from Roberts & Doonan (2016).

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[5] Where this report refers to this threat as 'wallows', this includes all types of hole, drop, or barrier that either cause a sea lion pup to be separated from its mother or to drown.

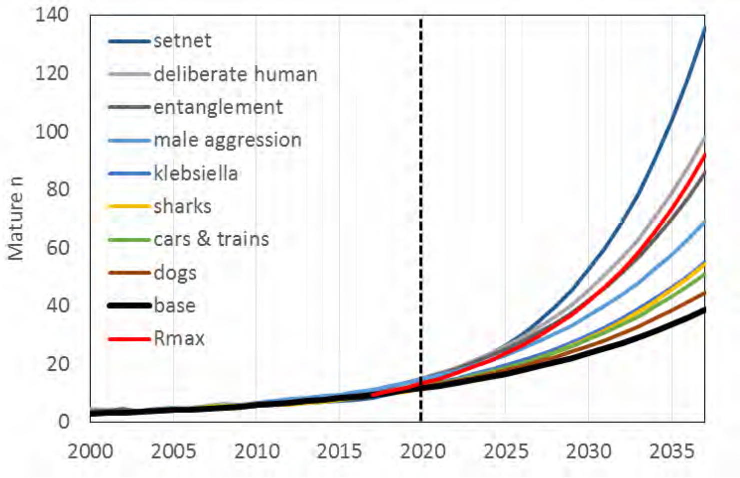

Figure 4.15: Triage projections of model estimated number of mature individuals (mature n) at the Otago Peninsula during 1990–2037, using upper values of threat mortality. The black dashed line indicates the estimated historical trend and population projection based on demographic parameters from the last 10 years. The removal of each single threat is plotted separately as coloured lines except for the red line, which shows population growth at $R_{\max}$ (assumed to be 0.12). Reproduced from Roberts & Doonan (2016).

For the Otago Peninsula model, the removal of upper bounds of some threats produced a very rapidly growing population, higher than the assumed maximum optimal growth rate ($R_{\max}$) (Figure 4.15). This indicates that the upper bounds used for set net and deliberate human threats were probably unrealistically high (Roberts & Doonan 2016, Debski & Walker 2016).

For the Auckland Islands population, best-estimate projections were undertaken for commercial trawl related mortality, *Klebsiella* related mortality of pups, trophic effects (food limitation), pups drowning in wallows, male aggression, and hookworm mortality. These threats were compared with the base run — a continuation of demographic rates since 2005 (λ_{2037} = 0.961, 95% c.i.: 0.890–1.020). A positive growth rate was obtained only with the alleviation of *Klebsiella* (λ_{2037} = 1.005, 95% c.i.: 0.926–1.069). When assuming the most pessimistic view of cryptic mortality (all interactions resulted in mortality and associated death of pups), alleviating the effects of commercial trawl related mortality resulted in an increased population growth rate relative to the base run, but did not reverse the declining trend (λ_{2037} = 0.977, 95% c.i.: 0.902–1.036) (Figure 4.16). The alleviation of trophic effects (food limitation) had the next greatest effect (λ_{2037} = 0.974, 95% c.i.: 0.905–1.038), and all other threats had a minor effect relative to the base run projection (increase in λ_{2037} of less than 0.01) (Figure 4.17, Roberts & Doonan 2016).

For the Otago Peninsula population, similar effects were estimated with the alleviation of any of the threats that passed through triage: commercial set net fishery related mortality, direct human mortality, pollution related mortality, entanglement, and male aggression, relative to the base run projection (λ_{2037} = 1.070, 95% c.i.: 1.053–1.087). Deliberate human mortality was estimated to have the greatest effect on projected population size (λ_{2037} = 1.093, 95% c.i.: 1.075–1.112) (Figure 4.18, Roberts & Doonan 2016).

For the Auckland Islands population, model outputs suggest that if demographic rates used to simulate forward population trajectories (i.e., sampled from the previous 20-year period) are accurate, then the TMP goals would be difficult to achieve with the complete alleviation of a single threat. In this context, the most effective approach to meeting the goals of the TMP may be to spread the management effort across the suite of key perceived threats identified from this assessment.

The population projections are sensitive to assumptions about what demographic rates are being realised in the population, in the context of considerable environmental variability on a decadal scale, with likely effects on critical demographic rates driving population change. A high priority is the development of tools for monitoring the effects of environmental and management drivers on threat-specific mortality and influential demographic rates

(Roberts & Doonan 2016). For example, research to examine factors affecting pup survival (Edwards & Roberts 2021, Roberts et al. 2021) commenced in 2018, and integrative ecosystems research to investigate decadal scale climate variability potentially affecting sea lion demographic parameters is being progressed under a separate contract (ZBD2018-05).

The assessment for some of the key threats to New Zealand sea lions was hampered by incomplete information for estimating threat-specific mortality, e.g., relating to the causes of pup mortality during the entire first year of life and of cryptic mortality. In addition, a lack of demographic observations for the Campbell Island and Stewart Island populations precluded the development of comprehensive quantitative risk assessments for these populations (Roberts & Doonan 2016).

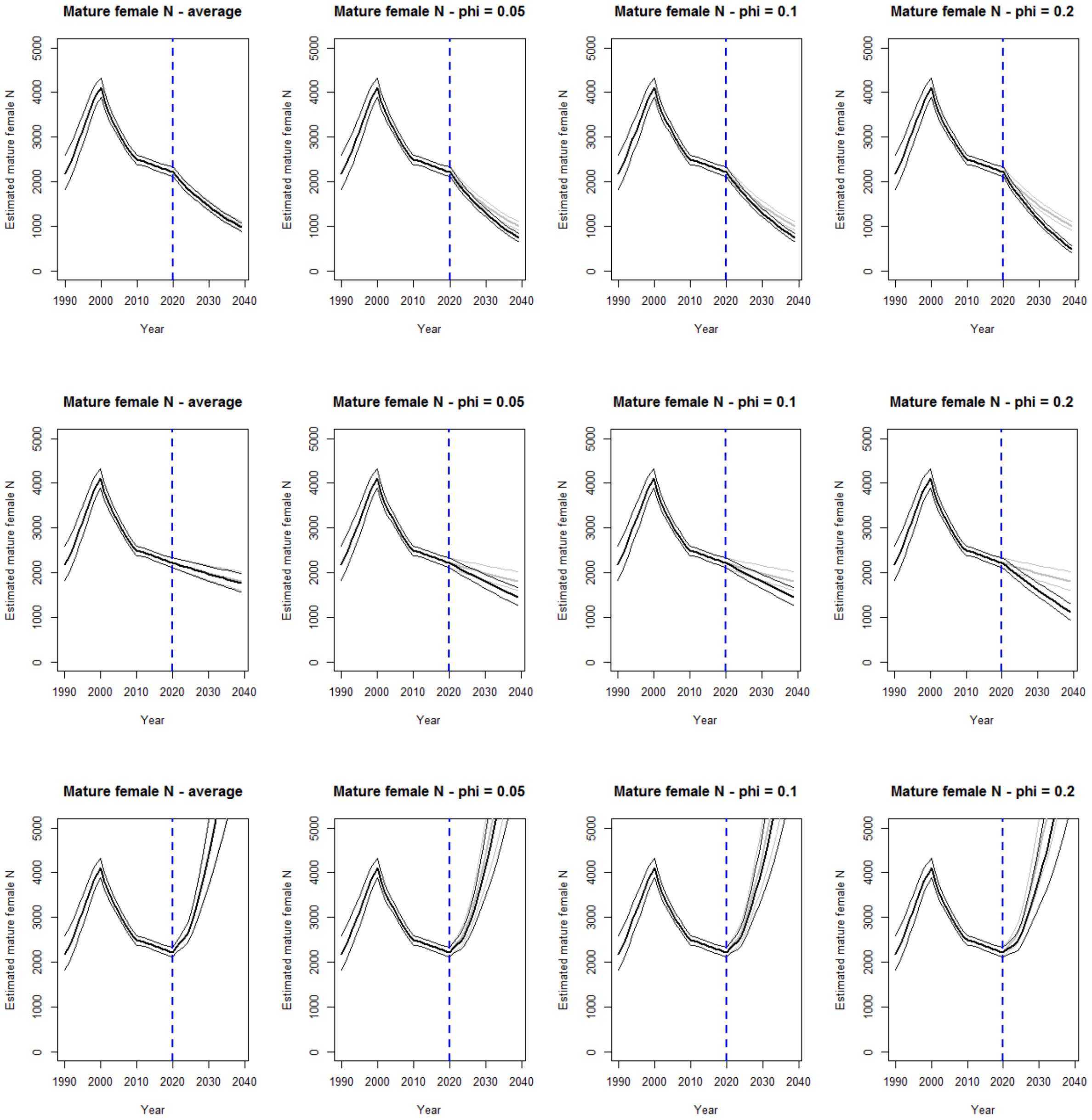

Figure 4.16: Female New Zealand sea lions population projections with alternative scenarios of: demographic rates affecting population growth (i.e., decreasing (top row), stable (middle row), or increasing (bottom row)); and future squid fishery-related deaths (i.e., zero future deaths (grey, shown in all plots for reference), the average of the last five years of estimated deaths, or equal to the population sustainability threshold (PST) assuming alternative values of ϕ ('phi', p. 25 equation 30) (all black)). Reproduced from Roberts (2019).

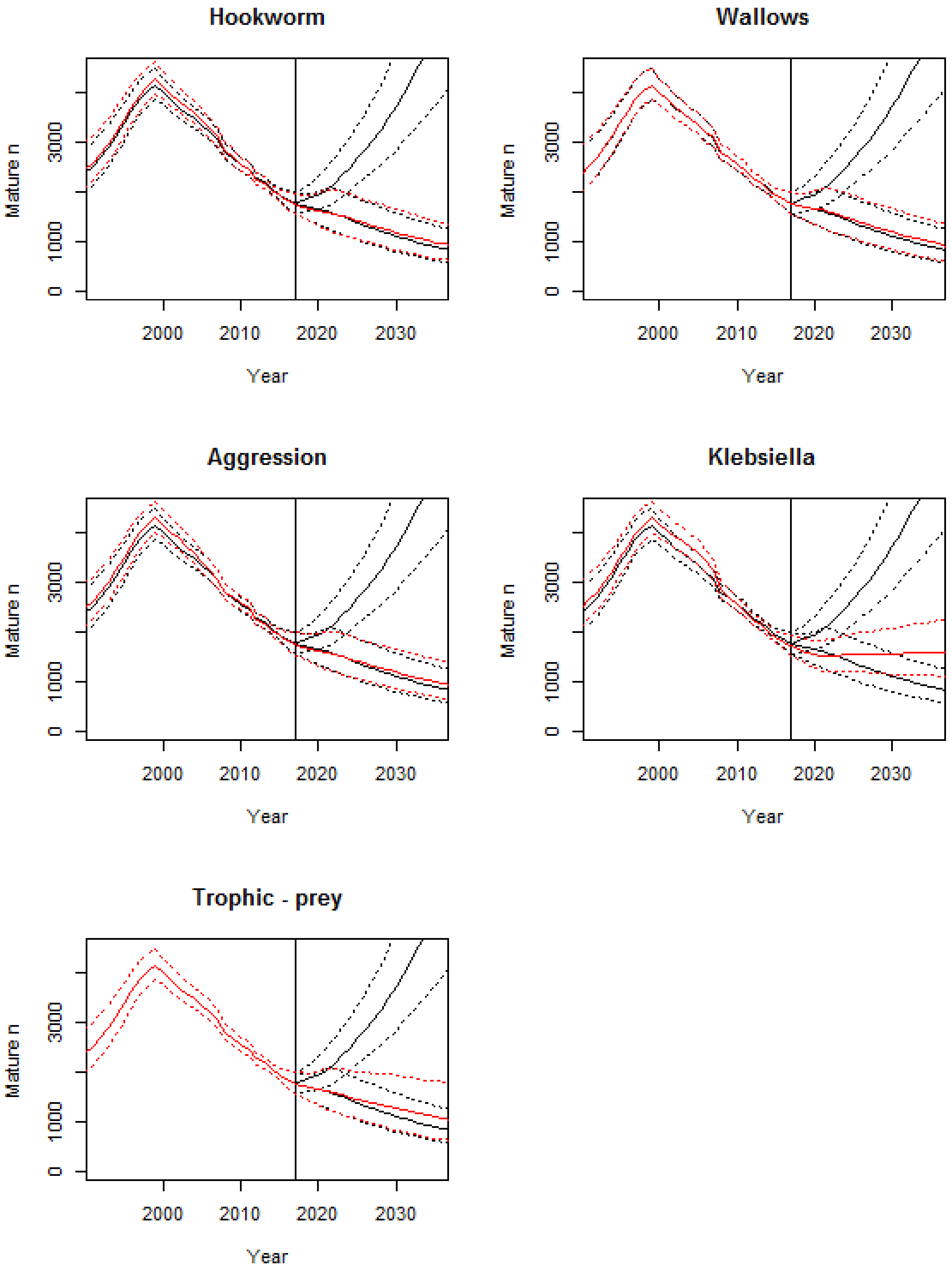

Figure 4.17: Best-estimate projections of mature individuals (mature n) at the Auckland Islands in the period 1990–2037 for all other threat scenarios. Lower black lines are with all threats (base run); upper black lines are with the 'max growth' scenario (1990–93 estimate of Surv0, 1990–98 estimates of Survival 6–14 years of age, and 1990–99 estimate of PrP; red lines are with a threat alleviated. Reproduced from Roberts & Doonan (2016).

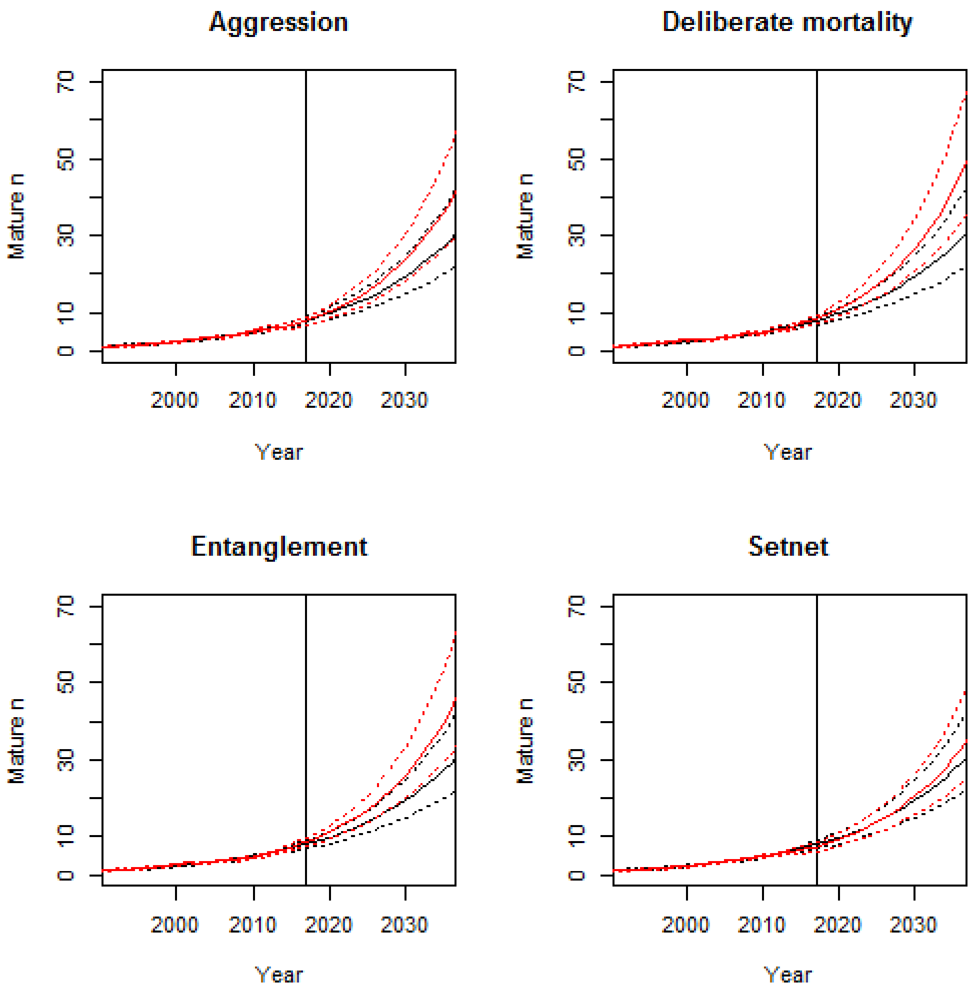

Figure 4.18: Best-estimate projections of mature individuals (mature n) at the Otago Peninsula in the period 1990–2037 for all threat scenarios (from Roberts & Doonan 2016). Black lines are with all threats (base run); red lines are with the threat alleviated. Reproduced from Roberts & Doonan (2016).

4.4.5.2 PBR (NOW PST) ASSESSMENT FOR CAMPBELL ISLAND POPULATION

Following the 2013 season in which a hjgh number of New Zealand sea lions were captured in the Campbell Rise Southern blue whiting fishery (SBW 6I), a review was conducted of potential biological removal (PBR) guidelines and relevant scientific literature to inform the selection of appropriate PBR parameter values for the Campbell Island sub-population (Roberts et al. 2014b). The PBR is a traditional approach to defining a safe level of human-related mortalities of marine mammals, which was originally developed for the US Marine Mammals Protection Act (Wade 1998). It is a precursor to the PST formulation used for most New Zealand protected species; like the PST, PBR relies on R_{max} to represent the species intrinsic population growth rate, but rather than full Bayesian consideration of parameter and modelling uncertainty it uses a conservative point estimate of population size N_{min} and a recovery factor F_R that provides for parameter bias correction as well as reflecting policy decisions regarding the level of acceptable impact (replacing the calibration factor phi in the PST formulation). New work is planned to replace this PBR estimation with an updated PST estimate consistent with the New Zealand SEFRA framework.

The pup census at Campbell Island of 681 pups in 2010 (Maloney et al. 2012) was taken as a robust lower estimate of total pup production. A matrix modelling analysis was conducted to estimate plausible pup to whole-of-population multipliers of 4.5 and 5.5, which were applied to the pup census estimate to calculate N_{min} values of 3065 and 3746. The rate of increase in pup counts from a time series of pup censuses was used as an approximation to whole-of-population growth rate for estimating a credible lower limit of R_{max}. Values of 0.06, 0.08, and 0.10 were used in PBR calculations, with the upper and lower limits considered as plausible bounds for this parameter used in a sensitivity analysis. The Auckland Islands and Campbell Island sea lion breeding populations are likely to be demographically independent, so were assessed as separate subpopulations (Moore & Merrick 2011). A default recovery factor (F_R) of 0.5 was applied, as is used in the USA for stocks of a threatened species with unknown (or not declining) population trajectory (Roberts et al. 2014b).

Prior to 2005–06 the annual number of captures was very low, though capture rate appears to have increased since, with the greatest number of captures in 2012–13 (Table 4.6). Running means of capture levels (3 and 5-year) were also calculated for comparison with PBR estimates. For F_R of 0.5, and the selected estimates of N_{min} (3065) and R_{max} (0.08), the calculated PBR was 61. Estimated captures did not exceed the PBR in any year when the default F_R of 0.5 was used, regardless of which other parameter values used. When the lower F_R of 0.1 was used, the calculated PBR of 12 was exceeded in two years when using a 3-year running mean of captures and in one year with a 5-year running mean of captures. When F_R of 0.2 was used, the calculated PBR of 25 was not exceeded in any year. There has been a very strong bias towards males in observed captures (Thompson et al. 2013). An array of female-only PBRs was estimated by halving the PBR for all animals and was not exceeded by female captures in any year regardless of which combination of parameter values was used (Roberts et al. 2014b).

New work is planned by Fisheries New Zealand under project PMM2019-09 to re-estimate a Campbell Island PST incorporating updated demographic information.

4.4.6 MANAGING FISHERIES RISK

For New Zealand sea lions, efforts to mitigate incidental captures in fisheries have historically focused on the SQU 6T fishery.

Current management reflects previously designated spatial fisheries closures. In 1982 the Minister of Fisheries established a 12-nautical mile exclusion zone around the Auckland Islands from which all fishing activities are excluded (Wilkinson et al. 2003); in 1995, the exclusion zone was replaced with a Marine Mammal Sanctuary with the same controls on fishing (Chilvers 2008). The area was subsequently designated as a Marine Reserve in 2003.

From 1992, the Ministry adopted a maximum allowable level of fisheries-related mortality (MALFiRM; later referred to as a Fisheries Related Mortality Limit, FRML) to set an upper limit on the number of New Zealand sea lions that can be incidentally killed each year in the SQU 6T trawl fishery (Chilvers 2008). If this limit is reached, the fishery is closed for the remainder of the season. The original MALFiRM was calculated using the 'potential biological removals' approach (PBR; Wade 1998) and was used from 1992–93 to 2003–04 (Smith & Baird 2007a). From 2003–04 to 2017–18 the FRML was translated into a fishing effort limit (maximum permitted number of tows) based on assumptions about the interaction rate and SLED efficacy rate, regardless of the number of observed New Zealand sea lion captures. This approach was taken because since the introduction of SLEDs, observed sea lion captures were no longer a reliable index of the number of sea lions interacting with the net, and there was uncertainty about the survival rate of sea lions exiting the net via the SLED.

SLEDs were first deployed on some vessels in the SQU 6T fishing fleet from around 2000. SLED use increased in subsequent years through a development phase in which SLED designs were trialed and modified, followed by a phase in which a single design specification existed but was not mandated or universally adopted on all fishing effort across the fleet. Subsequently the Squid Fishery Management Company in consultation with Fisheries New Zealand mandated a standardised SLED design that would be required for the vessel to receive the 'discount rate' relative to the tow limit applied by the government (set to ensure that estimated mortalities remained below the designated FRML; see section 4.4.6.1, below). From the 2008 season the standardised model Mark 3/13 SLED (Figure 4.12) has been universally employed by all vessels in the SQU 6T fleet. SLED design consistent with these specifications, and SLED deployment during fishing operations, are audited and monitored by Fisheries New Zealand Observers.

From 2017, advice to manage sea lion interactions in this fishery was developed in consultation with the Squid 6T Operational Plan Technical Advisory Group (SqOPTAG), including representatives from government and stakeholder groups as well as technical experts and advisors. Under an Operational Plan adopted in December

2017, Fisheries New Zealand set an FRML for sea lions in the SQU 6T fishery based on estimation of a Population Sustainability Threshold (PST) using a Bayesian population dynamic model (Roberts & Doonan 2016). The PST represents the maximum number of anthropogenic mortalities that the population can sustain while still achieving a defined population objective. For the Auckland Islands sea lion population, the choice of population objective underlying the PST was as follows: 'Fisheries mortalities will be limited to ensure that the impacted population is no more than 5% lower than it would otherwise be in the absence of fishing mortality, with 90% confidence, over five years'. The choice of the population objective is a policy decision.

The SQU 6T Operational Plan was updated in 2019 reflecting the outcomes of the new scientific approach whereby interactions, captures, and deaths (including cryptic mortality) are estimated directly and observed captures are applied toward the adopted FRML without the need for a proxy effort limit. The outputs of the new scientific approach were reviewed and the implications for advice to inform an updated management plan were discussed via the SqOPTAG.

The four-year Squid 6T Operational Plan was adopted in 2019 and will remain in place until 30 September 2023[6]. The Operational Plan defines a new FRML to reflect updated population model outputs, including sensitivities reflecting the likelihood that critical demographic rates for Auckland Islands sea lions are affected by decadal scale climatic variations (Roberts 2019, above). The plan also sets a minimum observer coverage requirement of 90%, to ensure that sea lion captures are recorded and SLEDs are properly deployed.

4.4.6.1 MANAGEMENT SETTINGS IN THE SQUID 6T FISHERY

Before the widespread use of SLEDs, New Zealand sea lions incidentally caught during fishing were usually retained in trawl nets and hauled onboard, allowing observers to gain an accurate assessment of the number of New Zealand sea lion interactions on observed tows in a given fishery. This enabled a robust estimation of the total number of New Zealand sea lions killed by fishing. However, following the introduction of SLEDs, the number of New Zealand sea lions interacting with trawls but exiting via the SLED was unobservable, so the interaction rate was instead estimated statistically. Subsequently, a management setting meant to approximate the interaction rate, i.e., the 'strike rate' was set by the government (along with a second setting, the 'discount rate' representing SLED efficacy, see below) to inform a proxy estimate of potential sea lion fatalities per 100 tows. This proxy estimate was then used to set an effort limit (maximum number of tows) on the operation of the fishery, to ensure that sea lion fisheries mortalities remained below the FRML.

The 'discount rate' was a management setting that approximated SLED efficacy, i.e., the proportion of sea lion interactions in which the sea lion exits the SLED and survives. The management regime for the SQU 6T fishery provided that the discount rate would be applied to all tows in which an approved Mark 3/13 SLED was used and relevant requirements of the Operational Plan met (e.g., notification of intention to fish in SQU 6T and reporting requirements). Discount rates applied between 2003–04 and 2018–19 are given in Table 4.10.

The SLED discount rate was a fisheries management setting and should not be confused with the actual estimated survival rate of New Zealand sea lions exiting the SLED; for example, the discount rate could be set deliberately lower than the actual estimated SLED efficacy rate, reflecting cautious management in the presence of uncertainty.

From 2019 a new science approach was adopted under which sea lion interactions, captures, and deaths (including cryptic mortality) are estimated directly. Under this approach it is now possible to evaluate performance against the FRML using observed captures directly, without the need for an effort proxy and associated SLED discount rate. For this reason, the new Squid 6T Operational Plan does not define a strike rate or discount rate; instead, total captures are monitored by fisheries observers and compared against the FRML as the season progresses. Cryptic deaths are estimated as a proportion of observable deaths, effectively adjusting the capture limit lower to account for sea lion bodies that may not be counted by fisheries observers.

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[6] https://www.fisheries.govt.nz/dmsdocument/38189-squid-6t-operational-plan-2019-2023

Table 4.10: Maximum allowable level of fisheries-related mortality (MALFiRM) or fisheries-related mortality limit (FRML) from 1991 to 2020. Note that direct comparisons among years of the limits in Table 4.10 are not possible because the assumptions underlying the MALFiRM or FRML changed over time.

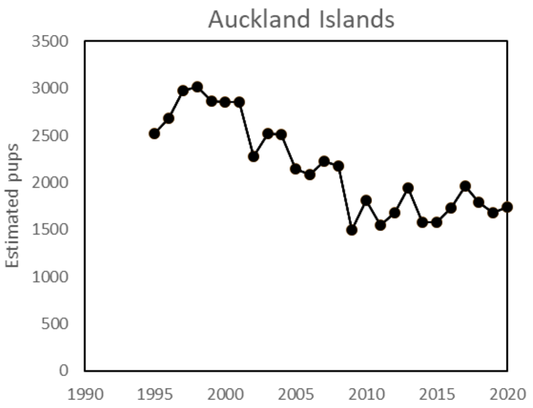
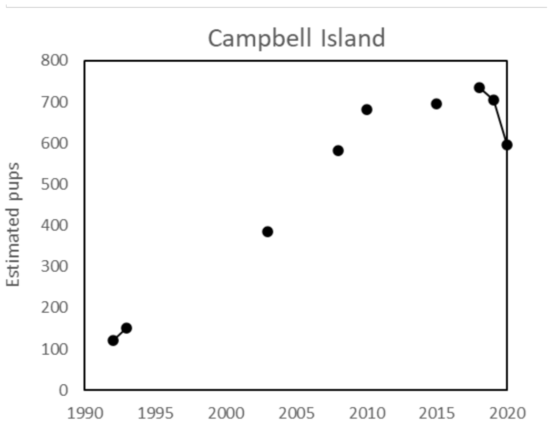
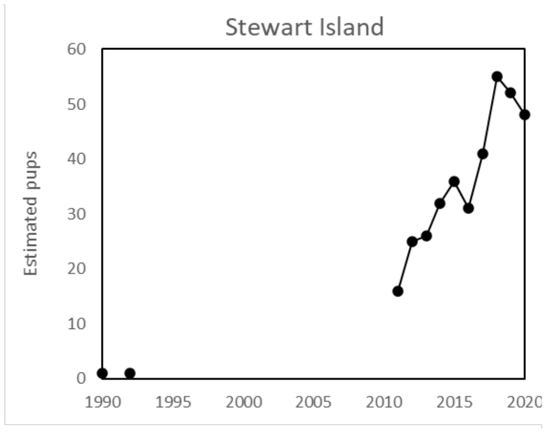
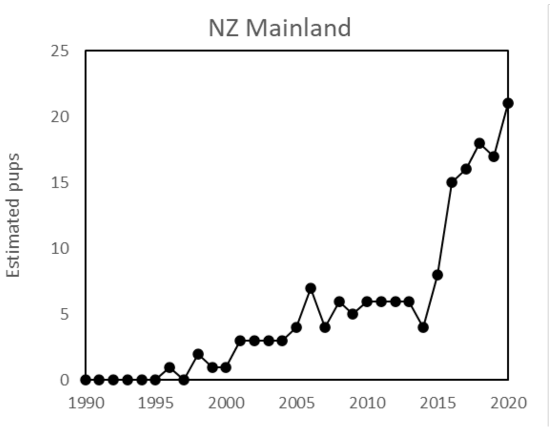
Year	MALFiRM or FRML	Discount rate	Management actions
1991–92	16 (female only)		
1992–93	63		
1993–94	63		
1994–95	69		
1995–96	73		Fishery closed by MFish (4 May)
1996–97	79		Fishery closed by MFish (28 March)
1997–98	63		Fishery closed by MFish (27 March)
1998–99	64		
1999–00	65		Fishery closed by MFish (8 March)
2000–01	75		Voluntary withdrawal by industry
2001–02	79		Fishery closed by MFish (13 April)
2002–03	70		Fishery closed by MFish (29 March), overturned by High Court
2003–04	62 (124)	20%	Fishery closed by MFish (22 March), overturned by High Court FRML increased
2004–05	115	20%	Voluntary withdrawal by industry on reaching the FRML
2005–06	97 (150)	20%	FRML increased in mid-March due to abundance of squid
2006–07	93	20%	
2007–08	81	35%	
2008–09	113 (95)	35%	Lower interim limit agreed due to the decrease in pup numbers
2009–10	76	35%	
2010–11	68	35%	
2011–12	68	35%	
2012–13	68	82%	
2013–14	68	82%	
2014–15	68	82%	
2015–16	68	82%	
2016–17	68	82%	
2017–18	38	75%	
2018–19	38	75%	
2019–20	52	N/A	New approach whereby deaths are estimated directly as a function of captures, eliminating the need for an effort limit and discount rate setting

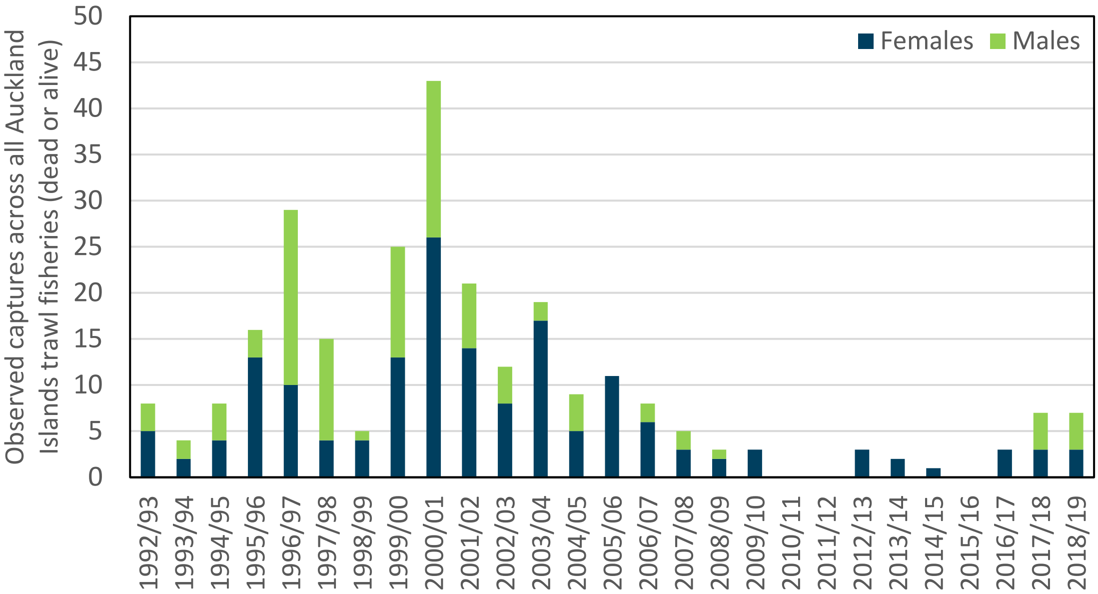
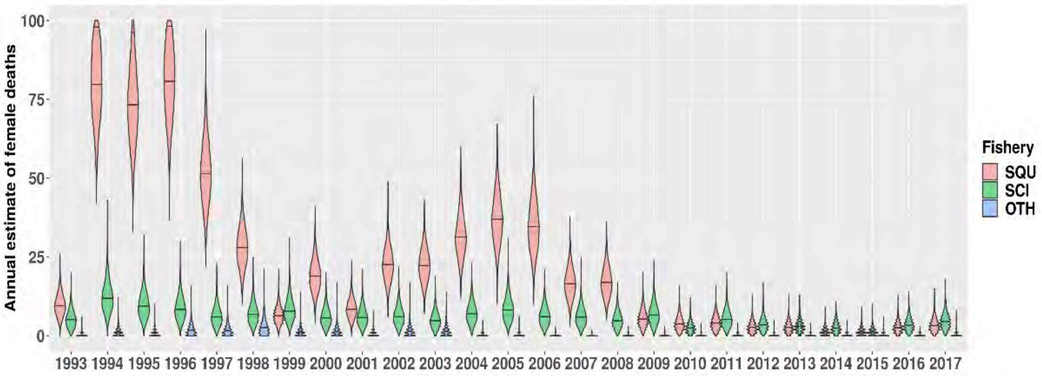
4.4.7 KEY INFORMATION GAPS

The Roberts & Doonan (2016) model and subsequent updates make no assumptions about the current status of the Auckland Islands sea lion population relative to ecological carrying capacity. Previously a review of life-history traits such as pup mass, pup survival, and female fecundity found no evidence for density dependent responses in the Auckland Islands population (Chilvers 2012b). However, a number of indicators of nutritional stress have been identified during the period of population decline, including a temporal shift in diet composition to small-sized prey (Childerhouse et al. 2001, Stewart-Sinclair 2013), low pupping rate/delayed age at first pupping (Childerhouse et al. 2010a, Roberts et al. 2014a), low pup/yearling survival rate (Roberts et al. 2014a), and reduced maternal condition (Riet-Sapriza et al. 2012, Roberts & Doonan 2014) – all of which are common density dependent responses. However, there is no evidence of typical density dependent responses, such as poor pup survival, being alleviated with decreasing population size (Roberts 2019). The underlying environmental causes of the apparent change is unknown; and it is unknown whether similar changes can be expected in future, and on what time scales. For this reason, updates of the demographic model of Roberts (2019) estimated population trajectories under three different hypothetical climatic regimes corresponding to observed periods of growth, decline, and stability in the annual pup production trend, and additional work is underway under project ZBD2018-05 to better understand potential climatic drivers of ecosystem change including potential ‘regime shifts’ affecting sea lions. Analysis of factors affecting pup survival and subsequent effects on demographic rates (e.g., Edwards & Roberts 2021, Roberts et al. 2021) may yield additional insight.

The spatial risk model of Large et al. (2019) relies on a single spatial foraging density layer informed by telemetry data collected primarily in summer, and primarily from breeding females at the Sandy Bay colony. The seasonal bias is appropriate for summer fisheries (i.e., targeting squid) and the sex and age bias is appropriate for a risk model concerned primarily with modelling the effect of fisheries on population reproductive output. However, the seasonal bias toward summer may affect the accuracy of risk estimates for winter fisheries such as scampi, and the relative lack of data from the Dundas Island and Figure of Eight Island colonies may introduce other biases. New data collection is proposed to address these gaps.

4.5 INDICATORS AND TRENDS

Population size	Roberts & Doonan (2016) estimated 11 755 New Zealand sea lions including pups (immediately after pupping) across all populations. It is estimated that there were: 1740 pups born at the Auckland Islands in 2019–20 (Melidonis & Childerhouse 2020); 595 pups born at Campbell Island in 2019–20 (McNutt et al. 2020); 48 pups born at Stewart Island in 2019–20 (47 at Port Pegasus and 1 at Ulva Island; DOC unpublished data); and 21 pups born on the Otago coast in 2019–20 (DOC unpublished data).
Population trend	Estimated annual pup production at the Auckland Islands, Campbell Island, Stewart Island, and New Zealand South Island is shown below. Note that the y-axis scale varies in each plot.    

Threat status	New Zealand: Nationally Vulnerable, Criterion D(1)[7], Range Restricted[8], in 2019[9] IUCN: Endangered, A4bd[10], in 2015
Number of captures	3 observed captures in trawl fisheries in 2016–17 8 observed captures in trawl fisheries in 2017–18 9 observed captures in trawl fisheries in 2018–19 1 observed capture in trawl fisheries in 2019–20
Trends in observed captures (both sexes)	
Trends in estimated deaths (females only). From Large et al. (2019).	

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[7] A taxon is listed as ‘Nationally Vulnerable’ under criterion D1 if it has a large population (5000–20 000 mature individuals) and a moderate to high rate of ongoing or predicted population decline (-30 to -70%) over three generations.

[8] A taxon is listed as ‘Range Restricted’ if it is confined to specific substrates, habitats, or geographic areas of less than 1000 km^2 (100 000 ha); this is assessed by taking into account the area of occupied habitat of all subpopulations (Townsend et al. 2008).

[9] Baker et al. (2019).

[10] A taxon is listed as ‘Vulnerable’ if it is considered to be facing a high risk of extinction in the wild. A3b refers to a reduction in population size (A), based on a reduction of 30% or more over the last 10 years or three generations (whichever is longer up to a maximum of 100 years (3); and when considering an index of abundance that is appropriate to the taxon (b; IUCN 2010).

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Chapter 5: New Zealand fur seal (*Arctocephalus forsteri*) - Technical Summary

New Zealand fur seal
(*Arctocephalus forsteri*)

Not threatened (DOC 2019)

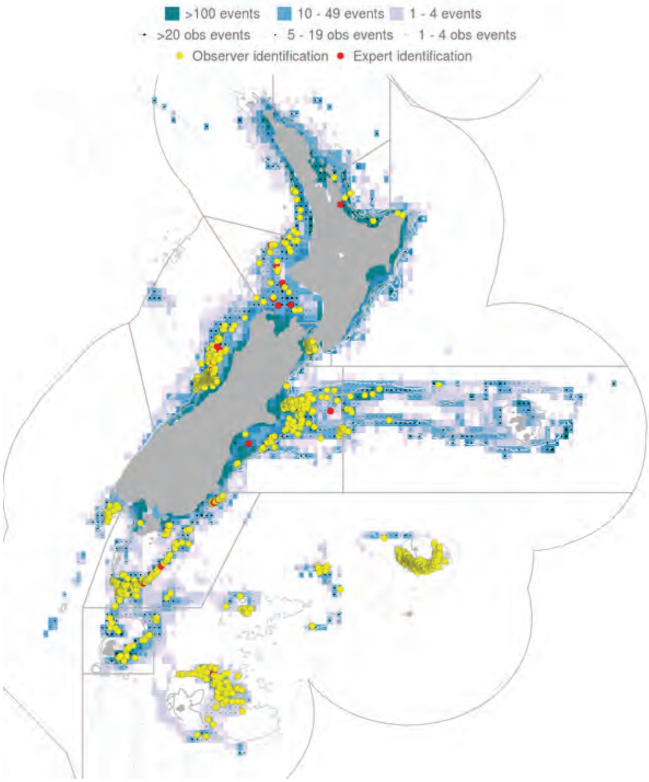
1. THE ISSUE IN BRIEF

- The New Zealand fur seal (*Arctocephalus forsteri*) is a pinniped, endemic to New Zealand
- NZ fur seals are attracted by fisheries operations, which can result in incidental captures and, potentially, deaths
- The NZ fur seal is abundant and classified as 'least concern' by DOC (population levels are increasing)
- Like other marine mammals, the NZ fur seal is rotected under the Marine Mammals Protection Act 1978 and the Fisheries Act 1996 but, because of its favourable conservation status, there is no Population Management Plan in place

2. CAUSES OF DEATH

- Current population estimates stand at nearly 100 000 individuals in the NZ Exclusive Economic Zone and numbers are increasing
- Starvation, stillbirth, suffocation, trampling, drowning, natural predation, and human disturbance are among the causes of pup mortality. Adult mortality causes include predation and fisheries interactions

3. INCIDENTAL CAPTURES - LOCATION



- Between 2002–03 and 2017–18, there were 1691 observed incidental captures of NZ fur seals in trawl fisheries, 408 in surface longline fisheries, 58 in set net fisheries, 2 in bottom longlines fisheries, and 1 in purse seine fisheries

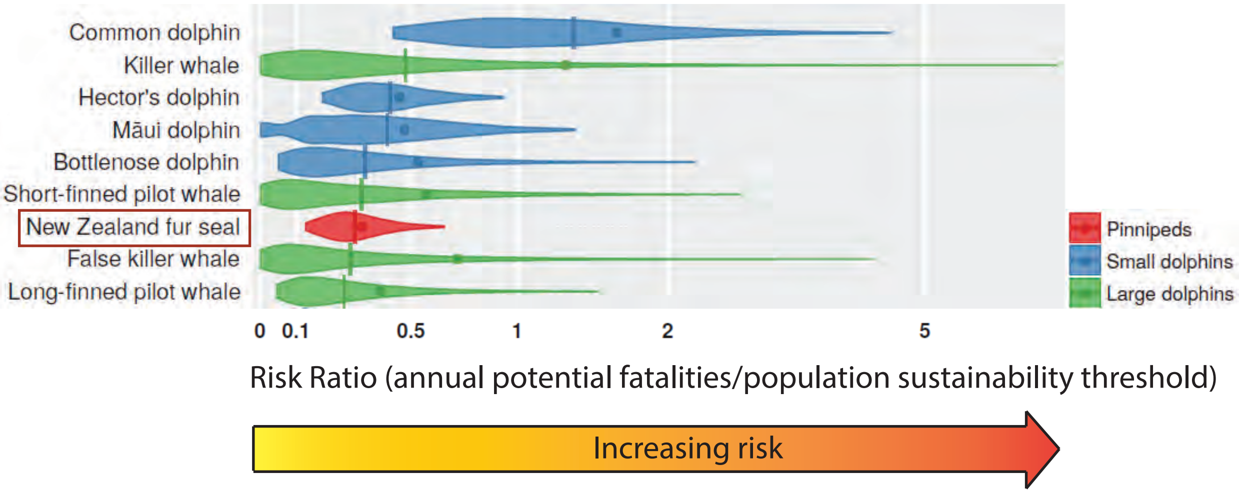
- The trawl fisheries mainly contributing to incidental captures of NZ fur seals are hoki (winter) and southern blue whiting (spring)

- The most common locations for incidental captures have been near Campbell Island and the Bounty Islands, as well as areas off the west coast of the South Island and the Cook Strait

- Observed captures are limited in the inshore trawl fishery, due to the low observer coverage

Map of NZ fur seal captures in NZ trawl fisheries between 2002 and 2018.
Yellow and red dots indicate NZ fur seal capture events, identified by observers and experts, respectively. Blue shades represent the trawl fishing effort

4. RISK ASSESSMENT



The multi-species marine mammal risk assessment (2017, see also Chapter 7) showed that the risk posed by fisheries to the NZ fur seal is comparable to that for several cetaceans, but has smaller uncertainty. However, the level of this risk is unlikely to pose a threat to the NZ fur seal population sustainability

5. INCIDENTAL CAPTURES - HOKI TRAWL FISHERIES

The observer coverage on board hoki trawl vessels has been increasing since 2003, reaching about 35% of the fishery in recent years

Capture rates of NZ fur seal peaked from 2003 to 2005, but have since slowly declined. Total capture numbers have been stable (with small fluctuations) over the last 10 years

Fishing effort (above) and observed captures (below) of NZ fur seals in NZ trawl fisheries

6. ONGOING RESEARCH

- Fishing interactions are considered unlikely to have adverse consequences for NZ fur seals at a national scale
- However, Fisheries New Zealand is trying to characterise the population structure and spatio-temporal foraging distribution of NZ fur seals and estimate commercial fisheries overlap and risk

5 NEW ZEALAND FUR SEAL (*ARCTOCEPHALUS FORSTERI*)

Status of chapter	Observer data and capture estimates have been updated for AEBAR 2021.
Scope of chapter	This chapter describes: the biology New Zealand fur seals (*Arctocephalus forsteri)*; the nature and extent of potential interactions with fisheries; management of fisheries interactions; means of estimating fisheries impacts and population level risk; and remaining sources of uncertainty, to guide future work.
Area	The New Zealand EEZ and Territorial Sea, but primarily in coastal environments extending to the continental slope.
Key locations	Areas with the potential for significant fisheries interactions include waters over or close to the continental shelf surrounding the South Island and southern offshore islands, notably Cook Strait, west coast South Island, Banks Peninsula, Stewart-Snares shelf, Campbell Rise, and the Bounty Islands, and the Bay of Plenty-East Cape. Interactions also occur off the west coast of the North Island.
Key issues	Improved means of estimating fisheries captures and and risk in poorly observed inshore fisheries; improved understanding of population size, structure, and trend on a regional basis; improved understanding of spatio-temporal distributions affecting encounter rates between fur seals and fishing effort.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional sub-population basis, or at finer spatial and temporal scales.
MPI research (current)	*PMM2018-04A: Estimate spatial distributions for at-risk marine mammals to assess fisheries overlap and risk: fur seal; PMM2018-07: Updated spatially explicit fisheries risk assessment for New Zealand marine mammal populations*
New Zealand government research (current)	DOC Marine Conservation Services Programme (CSP): INT2015-02 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*; MIT2014-01 *Protected species bycatch newsletter*.
Related chapters/issues	Chapter 3: Spatially Explicit Fisheries Risk Assessment (SEFRA); Chapter 4 New Zealand Sea Lions.

5.1 CONTEXT

Management of fisheries impacts on New Zealand fur seals is legislated under the Marine Mammals Protection Act (MMPA) 1978 and the Fisheries Act (FA) 1996. Under s.3E of the MMPA or s.14F of the Wildlife Act 1953, the Minister of Conservation, with the concurrence of the Minister for Oceans and Fisheries (formerly the Minister of Fisheries and Minister for Primary Industries), may approve a population management plan (PMP). There is no PMP in place for New Zealand fur seals.

In the absence of a PMP, the Ministry for Primary Industries (MPI) manages fishing-related mortality of New Zealand fur seals under s.15(2) of the FA "*to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality*".

All marine mammal species are designated as protected species under s.2(1) of the FA. In 2005, the Minister of Conservation approved the Conservation General Policy, which specifies in Policy 4.4 (f) that "*Protected marine species should be managed for their long-term viability and recovery throughout their natural range*". DOC's Regional Conservation Management Strategies outline specific policies and objectives for protected marine species at a regional level. Baker et al. (2016) list New Zealand fur seals as Not Threatened in 2009, and the IUCN classification is Least Concern (Chilvers & Goldsworthy 2015).

In 2004, DOC approved the *Department of Conservation Marine Mammal Action Plan for 2005–2010*, which still reflects their need for marine mammal conservation (Suisted & Neale 2004). The plan specifies a number of species-specific key objectives for New Zealand fur seals, of which the following is most relevant for fisheries interactions: "*To control/mitigate fishing-related mortality of New Zealand fur seals in trawl fisheries (including the*

WCSI hoki and Bounty Island southern blue whiting fisheries)". Management of New Zealand fur seal incidental captures aligns with Fisheries 2030 Objective 6: Manage impacts of fishing and aquaculture. Further, the management actions follow Strategic Action 6.2: *Set and monitor environmental standards, including for threatened and protected species and seabed impacts*.

All National Fisheries Plans except those for inshore shellfish and freshwater fisheries are relevant to the management of fishing-related mortality of New Zealand fur seals.

The relevant Fisheries Plan for the management of incidental captures of New Zealand fur seals is the "National Fisheries Plan for Deepwater and Middle-depth Fisheries Part 1A" (the National Deepwater Plan). Under the National Deepwater Plan, the objective most relevant for management of New Zealand fur seals is Environmental Outcome 8: *Manage deepwater and middle-depth fisheries to avoid, remedy or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened and protected species*.

Specific objectives for the management of incidental captures of New Zealand fur seals are outlined in the fishery-specific chapters of the National Deepwater Plan for the fisheries with which New Zealand fur seals are most likely to interact. These fisheries include trawl fisheries for hoki, hake, ling, jack mackerel, and southern blue whiting. The chapters are being reviewed and updated in 2019.

Fisheries New Zealand works closely with the fishing industry to increase awareness amongst the fishing fleet of how to minimise interactions with fur seals and emphasises the importance of adherence to the industry Marine Mammal Operational Procedures (MMOP). These procedures aim to reduce the risk of interactions with marine mammals by requiring that vessels:

- Minimise the length of time the fishing gear is on the surface;
- Remove all pieces of dead fish from the net before shooting the gear;
- Steam away from any congregations of marine mammals before shooting the gear; and
- Appoint a crew member to watch for marine mammal interactions every time the gear is shot or hauled.

Performance in relation to these procedures is audited by Fisheries New Zealand Observers and reported in the Deepwater Annual Review Report.

5.2 BIOLOGY

5.2.1 TAXONOMY

The New Zealand fur seal (*Arctocephalus forsteri*, Lesson 1828) is an otariid seal (Family Otariidae – eared seals, including fur seals and sea lions), one of two native to New Zealand, the other being the New Zealand sea lion (*Phocarctos hookeri*, Gray 1844).

5.2.2 DISTRIBUTION

Pre-European archaeological evidence suggests that New Zealand fur seals were present along much of the east coasts of the North Island (except the less rocky coastline of Bay of Plenty and Hawke's Bay) and the South Island, and, to a lesser extent, on the west coasts, where fewer areas of suitable habitat were available (Smith 1989, 2005, 2011). A combination of subsistence hunting and commercial harvest resulted in contraction of the fur seal range and in a population decline almost to the point of extinction (Smith 1989, Ling 2002, Smith 2005, Lalas 2008, Smith 2011). New Zealand fur seals became fully protected in the 1890s and, with the exception of one year of licensed harvest in the 1950s, have remained protected since that time.

Currently, New Zealand fur seals occur throughout New Zealand waters, predominantly in waters south of 40° S and as far south as Macquarie Island. On land, New Zealand fur seals are distributed around the New Zealand coastline, on offshore islands, and on subantarctic islands (Crawley & Wilson 1976, Wilson 1981, Mattlin 1987). The recolonisation of the coastline by New Zealand fur seals has resulted in the northward expansion of the distribution of breeding colonies and haulouts (Lalas & Bradshaw 2001), and breeding colonies are now present on many exposed rocky areas (Baird 2011). The extent of breeding colony distribution in New Zealand waters is bounded to the north by a very small (space-limited) colony at Gannet Island off the North Island west coast (latitude 38° S), to the east by colonies of unknown sizes at the Chatham Islands group, to the west by colonies of unknown size on Fiordland offshore islands, and to the south by unknown numbers on Campbell Island. Outside New Zealand waters, breeding populations exist in South and Western Australia (Shaughnessy et al.

1994, Shaughnessy 1999, Goldsworthy et al. 2003), with smaller colonies in Tasmania (Gales et al. 2010).

The seasonal distribution of the New Zealand fur seals is determined by the sex and maturity of each animal. Males are generally at the breeding colonies from late October to late January then move to haulout areas around the New Zealand coastline (see Bradshaw et al. 1999), with peak density of males and sub-adult males at haulouts during July–August and lowest densities in September–October (Crawley & Wilson 1976). Females arrive at the breeding colony from November and lactating females remain at the colony (apart from short foraging trips) for about 10 months until the pups are weaned, usually during August–September (Crawley & Wilson 1976).

5.2.3 FORAGING ECOLOGY

Most New Zealand fur seal foraging research in New Zealand has focused on lactating females at Open Bay Islands off the South Island west coast (Mattlin et al. 1998), Otago Peninsula (Harcourt et al. 2002), and Ohau Point, Kaikoura (Boren 2005), using time-depth recorders, satellite-tracking, or very-high-frequency transmitters. Individual females show distinct dive pattern behaviour and may be relatively shallow or deep divers, but most forage at night and in depths shallower than 200 m. At Open Bay Islands, dives were generally deeper and longer in duration during autumn and winter. Females dived to at least 274 m (for a 5.67 min dive in autumn) and remained near the bottom at over 237 m for up to 11.17 min in winter (Mattlin et al. 1998). Females in some locations undertook longer dive trips, with some to deeper waters, in autumn (in over 1000 m beyond the continental shelf; Harcourt et al. 2002).

The relatively shallow dives and nocturnal feeding observed during summer suggests that seals feed on pelagic and vertical migrating prey species (for example, arrow squid, *Nototodarus sloanii*). Conversely, the deeper dives and increased number of dives in daylight during autumn and winter suggest that prey species at this time may include benthic, demersal, and pelagic species (Mattlin et al. 1998, Harcourt et al. 2002). The deeper dives enabled seals to forage along or off the continental shelf (within 10 km) of the studied colony (at Open Bay Islands). These deeper dives may be demersal or to depths in the water column where spawning hoki are concentrated.

Methods to analyse New Zealand fur seal diets have included investigation of freshly killed animals (Sorensen 1969), scats, and regurgitates (e.g., Allum & Maddigan 2012). Fish prey items can be recognised by the presence of otoliths, bones, scales, and lenses, while cephalopods are indicated by beaks and pens. Foraging modes appear to vary between specific individuals, and distinct diets may be apparent in the scats and regurgitations of males vs. females vs. juveniles from the same colony. These analyses can be biased, however, particularly if only one collection method is used, and this limits fully quantitative assessment of prey species composition.

Dietary studies of New Zealand fur seals have been conducted at colonies in Nelson-Marlborough, on the west coast South Island, at Otago Peninsula, Kaikōura, Banks Peninsula, Snares Islands, and off Stewart Island, and summaries are provided by Carey (1992), Harcourt (2001), Boren (2010), and Baird (2011).

New Zealand fur seals are opportunistic foragers and, depending on the time of year, method of analysis, and location, their diet includes at least 61 taxa (Holborow 1999) of mainly fish (particularly lanternfish (myctophids) in all studied colonies except Tonga Island (in Golden Bay; Willis et al. 2008), as well as anchovy (*Engraulis australis*), aruhu (*Auchenoceros punctatus*), barracouta (*Thrysites atun*), hoki (*Macruronus novaezelandiae*), jack mackerel (*Trachurus* spp.), pilchard (*Sardinops sagax*), red cod (*Pseudophycis bachus*), red gurnard (*Chelidonichthys kumu*), silverside (*Argentina elongate*), sprat (*Sprattus* spp.), and cephalopods (octopus (*Macroctopus maorum*), squid (*Nototodarus sloanii*, *Sepioteuthis bilineata*)). For example, myctophids were present in Otago scats throughout the year (representing offshore foraging), but aruhu, sprat, and juvenile red cod were present only during winter-spring (Fea et al. 1999). Medium-large arrow squid predominated in summer and autumn. Jack mackerel species, barracouta, and octopus were dominant in winter and spring. Prey such as lanternfish and arrow squid rise in the water column at night, the time when New Zealand fur seals exhibit shallow foraging (Harcourt et al. 1995, Mattlin et al. 1998, Fea et al. 1999).

Recent foraging and dietary studies include one on male fur seal diets by Lalas & Webster (2014) and one on lactating females by Meynier et al. (2013). Arrow squid was the most important dietary item in fur seal scats and regurgitations sampled from male fur seals at The Snares during February 2012 (Lalas & Webster 2014). Meynier et al. (2013) assess the trophic and spatial overlap between fur seals from two different South Island locations with local fisheries using analyses of dietary fatty acids, stable isotope signals, and telemetry. Lactating females from the east coast rookery at Ohau Point fed on oceanic prey in summer and females from the west coast rookery at Cape Foulwind fed on benthic or coastal prey over the continental shelf in summer and winter. The west coast females spent 50% of their at-sea time in winter in and near the Hokitika Canyon, where the winter spawning hoki fishery operates.

5.2.4 REPRODUCTIVE BIOLOGY

New Zealand fur seals are sexually dimorphic and polygynous (Crawley & Wilson 1976); males may weigh up to 160 kg, whereas females weigh up to about 50 kg (Miller 1975, Mattlin 1978a, 1987, Troy et al. 1999). Adult males are much larger around the neck and shoulders than females and breeding males are on average 3.5 times the weight of breeding females (Crawley & Wilson 1976). Females are philopatric and are sexually mature at 4–6 years, whereas males mature at 5–9 years (Mattlin 1987, Dickie & Dawson 2003). The maximum age recorded for New Zealand fur seals in New Zealand waters is 22 years for females (Dickie & Dawson 2003) and 15 years for males (Mattlin 1978a).

New Zealand fur seals are annual breeders and generally produce one pup after a gestation period of about 10 months (Crawley & Wilson 1976). Twinning can occur and females may foster a pup (Dowell et al. 2008), although both are rare. Breeding animals come ashore to mate after a period of sustained feeding at sea. Breeding males arrive at the colonies to establish territories during October–November. Breeding females arrive at the colony from late November and give birth shortly after. Peak pupping occurs in mid-December (Crawley & Wilson 1976).

Females remain at the colony with their newborn pups for about 10 days, by which time they have usually mated. Females then leave the colony on short foraging trips of 3–5 days before returning to suckle pups for 2–4 days (Crawley & Wilson 1976). As the pups grow, these foraging trips are progressively longer in duration. Pups remain at the breeding colony from birth until weaning (at 8–12 months of age).

Breeding males generally disperse after mating to feed and occupy haulout areas, often in more northern areas (Crawley & Wilson 1976). This movement of breeding adults away from the colony area during January allows for an influx of sub-adults from nearby areas. Little is described about the ratio of males to females at breeding colonies (Crawley & Wilson 1976), or the reproductive success. Boren (2005) reported a fecundity rate of 62% for a Kaikōura colony, based on two annual samples of between about 5 and 8% of the breeding female population. This rate is similar to the 67% estimated by Goldsworthy & Shaughnessy (1994) for a South Australian colony.

Newborn pups are about 55 cm long and weigh about 3.5 kg (Crawley & Wilson 1976). Male pups are generally heavier than female pups at birth and throughout their growth (Crawley & Wilson 1976, Mattlin 1981, Chilvers et al. 1995, Bradshaw et al. 2003b, Boren 2005). Pup growth rates may vary by colony (see Harcourt 2001). The proximity of a colony to easily accessible rich food sources will vary, and pup condition at a colony can vary markedly between years (Mattlin 1981, Bradshaw et al. 2000, Boren 2005). Food availability may be affected by climate variation, and pup growth rates probably represent variation in the ability of mothers to provision their pups from year to year. The sex ratio of pups at a colony may vary by season (Bradshaw et al. 2003a, 2003b, Boren 2005), and in years of high food resource availability, more mothers may produce males or more males may survive (Bradshaw et al. 2003a, 2003b).

5.2.5 POPULATION BIOLOGY

Historically, the population of New Zealand fur seals in New Zealand was thought to number above 1.25 million animals (possibly as high as 1.5 to 2 million) before the extensive sealing of the early 19th century (Richards 1994). Present day population estimates for New Zealand fur seals in New Zealand are dated, few, and highly localised. In the most comprehensive attempt to quantify the total New Zealand fur seal population, Wilson (1981) summarised population surveys of mainland New Zealand and offshore islands undertaken in the 1970s and estimated the population size within the New Zealand region at between 30 000 and 50 000 animals. Since then, several authors have suggested a population size of ~100 000 animals (Taylor 1990, see Harcourt 2001), but this estimate is very much an approximation and its accuracy is difficult to assess in the absence of comprehensive surveys.

Fur seal colonies provide the best data for consistent estimates of population numbers, generally based on pup production in a season (see Shaughnessy et al. 1994). Data used to provide colony population estimates of New Zealand fur seals have been, and generally continue to be, collected in an ad hoc fashion. Regular pup counts are made at some discrete populations. A 20-year time series of Otago Peninsula colony data is updated, maintained, and published primarily by Chris Lalas (assisted by Sanford (South Island) Limited), and the most recent published estimate is 20 000–30 000 animals (Lalas 2008). Lalas & MacDiarmid (2014) applied a logistic growth model, using established parameters, to 13 years of pup production estimates from colonies at Oamaru south to Slope Point, and indicated the 2009 population was at 95% of the asymptote of 19 600 animals (plausible range of 13 000–28 800). In this region, 90% of the population growth occurred over 24–27 years; and the growth rate was faster in seasons up to 1998, than in later years.

Similar population growth rates occurred at Kaikōura, where the population expanded by 32% per annum over the years 1990–2005 (Boren et al. 2006). An estimate of 600 pups was reported for 2005 (Boren 2005), 1508 (s.e. = 28) pups were estimated for 2009, and 2390 (s.e. = 226) pups for 2011 (L. Boren, DOC, pers. comm.).

Since 1991, the Department of Conservation has monitored New Zealand fur seal pup production at three breeding colonies on the West Coast, at Cape Foulwind, Wekakura Point, and Taumaka (Open Bay Islands) (see Best 2011). A DOC-commissioned project is underway to compile the tag, measurement, and mark-recapture data from these colonies and create a New Zealand fur seal database (Roberts & Best 2016). The data have been made available by the scientists who complete the fieldwork, most recently by Hugh Best, who coordinates the population monitoring programme, DOC Regional and District staff, Tai Poutini Papatipu Runanga, and the trustee owners of Taumaka me Popotai. Once the database has been through a quality assurance process, it will be made publically available. The pup production estimates for these colonies are derived using direct counts of dead pups and mark recapture methodology undertaken in the last week of January each year. At Taumaku Island, the largest of the Open Bay Islands and the most southern of these three colonies, approximately 800 pups are marked each year, and the first 100 pups of each sex are weighed and measured. At Cape Foulwind, approximately 200 pups are marked each year, and the first 50 of each sex are weighed and measured. At the most northern of the three colonies, Wekakura Point, approximately 500 pups are marked and 75 of each sex are weighed and measured.

Other studies of breeding colonies generally provide estimates for one or two seasons, but many of these are more than 10 years old. Published estimates suggest that populations have stabilised at the Snares Islands after a period of growth in the 1950s and 1960s (Carey 1998) and increased at the Bounty Islands (Taylor 1996), Nelson-Marlborough region (Taylor et al. 1995), Kaikōura (Boren 2005), Otago (Lalas & Harcourt 1995, Lalas & Murphy 1998, Lalas 2008, Lalas & MacDiarmid 2014), and near Wellington (Dix 1993).

For many areas where colonies or haulouts exist, count data have been collected opportunistically (generally by Department of Conservation staff during their field activities) and thus data are not often comparable because counts may represent different life stages, different assessment methods, and different seasons (see Baird 2011). Known breeding locations (as at October 2012) are summarised in the NABIS supporting lineage document for the '*Breeding colonies distribution of New Zealand fur seal*' layer[1].

Baker et al. (2010) conducted an aerial survey of the South Island west coast from Farewell Spit to Puysegur Point and Solander Island in 2009, but their counts were quite different, i.e., lower than ground counts collected at a similar time at the main colonies (Mellina & Cawthorn 2009). This discrepancy was thought to be a result mainly of the survey design and the nature of the terrain. However, the aerial survey confirmed the localities shown by Wilson (1981) of potentially large numbers of pups at sites such as Cascade Point, Yates Point, Chalky Island, and Solander Island.

Population numbers for some areas, especially more isolated ones, are not well known. The most recent counts for the Chatham Islands were collected in the 1970s (Wilson

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[1] http://www2.nabis.govt.nz/LayerDetails.aspx?layer=Breeding%20colonies%20distribution%20of%20New%20Zealand%20fur%20seal.

1981), and the most recent reported for the Bounty Islands were made in 1993–94. Taylor (1996) reported an increase in pup production at the Bounty Islands since 1980, and estimated that the total population was at least 21 500, occupying over 50% of the available area. Information is sparse for populations at Campbell Island, the Auckland Islands group, and the Antipodes Islands

Little is reported about the natural mortality of New Zealand fur seals, other than reports of sources and estimates of pup mortality for some breeding colonies. Estimates of pup mortality or pup survival vary in the manner in which they were determined and in the number of seasons they represent and are not directly comparable. Each colony will be affected by different sources of mortality related to habitat, location, food availability, environment, and year, as well as the ability of observers to count all the dead pups (may be limited by terrain, weather, or time of day).

Reported pup mortality rates vary: 8% for Otago Peninsula pups up to 30 days old and 23% for pups up to 66 days old (Lalas & Harcourt 1995); 20% from birth to 50 days and about 40% from birth to 300 days for Taumaka Island, Open Bay Islands pups (Mattlin 1978b); and in one year, 3% of Kaikōura pups before the age of 50 days (Boren 2005). Starvation was the major cause of death, although stillbirth, suffocation, trampling, drowning, predation, and human disturbance also occur. Pup survival of at least 85% was estimated for a mean 47-day interval for three Otago colonies, incorporating data such as pup body mass (Bradshaw et al. 2003b), though pup mortality before the first capture effort was unknown. Other sources of natural mortality for New Zealand fur seals include predators such as sharks and New Zealand sea lions (Mattlin 1978b, Bradshaw et al. 1998).

Human-induced sources of mortality include: fishing, for example, entanglement or capture in fishing gear; vehicle-related deaths (Lalas & Bradshaw 2001, Boren 2005, Boren et al. 2006, 2008); and mortality through shooting, bludgeoning, and dog attacks. New Zealand fur seals are vulnerable to certain bacterial diseases and parasites and environmental contaminants, though it is not clear how life-threatening these are. The more obvious problems include tuberculosis infections, *Salmonella*, hookworm enteritis, phocine distemper, and septicaemia (associated with abortion) (Duignan 2003, Duignan & Jones 2007). Low food availability and persistent organohalogen compounds (which can affect the immune and the reproductive systems) may also affect New Zealand fur seal health.

Various authors have investigated fur seal genetic differentiation among colonies and regions in New Zealand (Lento et al. 1994, Robertson & Gemmell 2005). Lento et al. (1994) described the geographic distribution of mitochondrial cytochrome *b* DNA haplotypes. Robertson & Gemmell (2005) described low levels of genetic differentiation (consistent with homogenising gene flow between colonies and an expanding population) based on genetic material from New Zealand fur seal pups from seven colonies. One aim of the latter work is to determine the provenance of animals captured during fishing activities, through the identification and isolation of any colony genetic differences.

In 2015–16, Gooday et al. (unpub., 2016) conducted trials of unmanned aerial vehicle (UAV) technology combined with thermal imaging in the Ohau Point fur seal colony, as part of an investigation into non-invasive population sampling. They found aerial surveys using a T320 19 mm infrared camera were successful in detecting fur seals in open areas and distinguishing them from rocks, but they were unsuccessful in areas of high canopy cover (> 80%). Ground surveys were also conducted using a higher resolution Optris PL450^{TM} infrared camera and detected more fur seals than paired photographs during cooler times of the day (morning and evening). In the Ohau Stream where seal pups visit the waterfall, the Optris PL450^{TM} detected pups hiding in the forested areas better than the naked eye but was less effective when they were swimming or if they had recently left the water. The Optris PL450^{TM} is currently under development to be mounted to the UAV, which is expected to increase aerial counts dramatically. Gooday et al. (unpub., 2016) concluded that thermal imagery has the potential to become an effective and widely used tool for ecological population surveys.

5.2.6 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2014). The risk of extinction for New Zealand fur seals has been assessed under two threat classification systems: the New Zealand Threat Classification System (Townsend et al. 2008) and the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2014).

In 2008, the IUCN updated the Red List status of New Zealand fur seals, listing them as Least Concern on the basis of their large and apparently increasing population size (Chilvers & Goldsworthy 2015). In 2010, DOC updated the New Zealand Threat Classification status of all New Zealand marine mammals (Baker et al. 2016). In the revised list, New Zealand fur seals were classified as Not Threatened with the qualifiers increasing (Inc) and secure overseas (SO) (Baker et al. 2016).

5.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

New Zealand fur seals are found in both Australian and New Zealand waters. Overall abundance has been suggested to be as high as 200 000, with about half of the population in Australian waters (Goldsworthy & Gales 2008). However, this figure is very much an approximation, and its accuracy is difficult to assess in the absence of comprehensive surveys.

Pinnipeds are caught incidentally in a variety of fisheries worldwide (Read et al. 2006). Outside New Zealand waters, species captured include: New Zealand fur seals, Australian fur seals, and Australian sea lions in Australian trawl and inshore fisheries (e.g., Shaughnessy 1999, Norman 2000); Cape fur seals in South African fisheries (Shaughnessy & Payne 1979); South American sea lions in trawl fisheries off Patagonia (Dans et al. 2003); and seals and sea lions in United States waters (Moore et al. 2009).

5.4 STATE OF KNOWLEDGE IN NEW ZEALAND

New Zealand fur seals are attracted to feeding opportunities offered by various fishing gears. Anecdotal evidence suggests that the sound of winches as trawlers haul their gear acts as a cue. The attraction of fish in a trawl net, on longline hooks, or caught in a set net provide opportunities for New Zealand fur seals to interact with fishing gear, which can result in capture and, potentially, death via drowning

Most captures occur in trawl fisheries and New Zealand fur seals are most at risk from capture during shooting and hauling (Shaughnessy & Payne 1979), when the net mouth is within diving depths. Once in the net some animals may have difficulty in finding their way out within their maximum breath-hold time (Shaughnessy & Davenport 1996). The operational aspects that are associated with New Zealand fur seal captures on trawlers include factors that attract the New Zealand fur seals, such as the presence of offal and discards, the sound of the winches, vessel lights, and the presence of 'stickers' in the net (Baird 2005). It is considered that New Zealand fur seals are at particular risk of capture when a vessel partially hauls the net during a tow and executes a turn with the gear close to the surface. At the haul, New Zealand fur seals often attempt to feed from the codend as it is hauled and dive after fish that come loose and escape from the net (Baird 2005).

Factors identified as important influences on the potential capture of New Zealand fur seals in trawl gear include the year or season, the fishery area, gear type and fishing strategies (often specific to certain nationalities within the fleet), time of day, and distance to shore (Baird & Bradford 2000, Mormede et al. 2008, Smith & Baird 2009). These analyses did not include any information on New Zealand fur seal numbers or activity in the water at the stern of the vessel because of a lack of data. Other influences on New Zealand fur seal capture rate (of Australian and New Zealand fur seals) may include inclement weather and sea state, vessel tow and haul speed, increased numbers of vessels and trawl frequency, and potentially the weight of the fish catch and the presence of certain bycatch fish species (Hamer & Goldsworthy 2006). This Australian study found similar mortality rates for tows with and without Seal Exclusion Devices (see also Hooper et al. 2005). The use of fur seal exclusion devices is not required in New Zealand fisheries.

The spatial and temporal overlap of commercial fishing grounds and New Zealand fur seal foraging areas has resulted in New Zealand fur seal captures in fishing gear (Mattlin 1987, Rowe 2009). Most fisheries with observed captures occur in waters over or close to the continental shelf. Because the topography around much of the South Island and offshore islands slopes steeply to deeper waters, most captures occur close to colonies and haulouts. Locations of captures by trawl vessels and surface longline vessels are shown in Figures 5.1 and 5.2. Winter hoki fisheries attract New Zealand fur seals off the west coast South Island and in Cook Strait between late June and September (Table 5.1). In August–October, New Zealand fur seals are caught in southern blue whiting effort near the Bounty Islands and Campbell Island. In September–October captures may occur in hoki and ling fisheries off Puysegur Point on the south-western coast of the South Island.

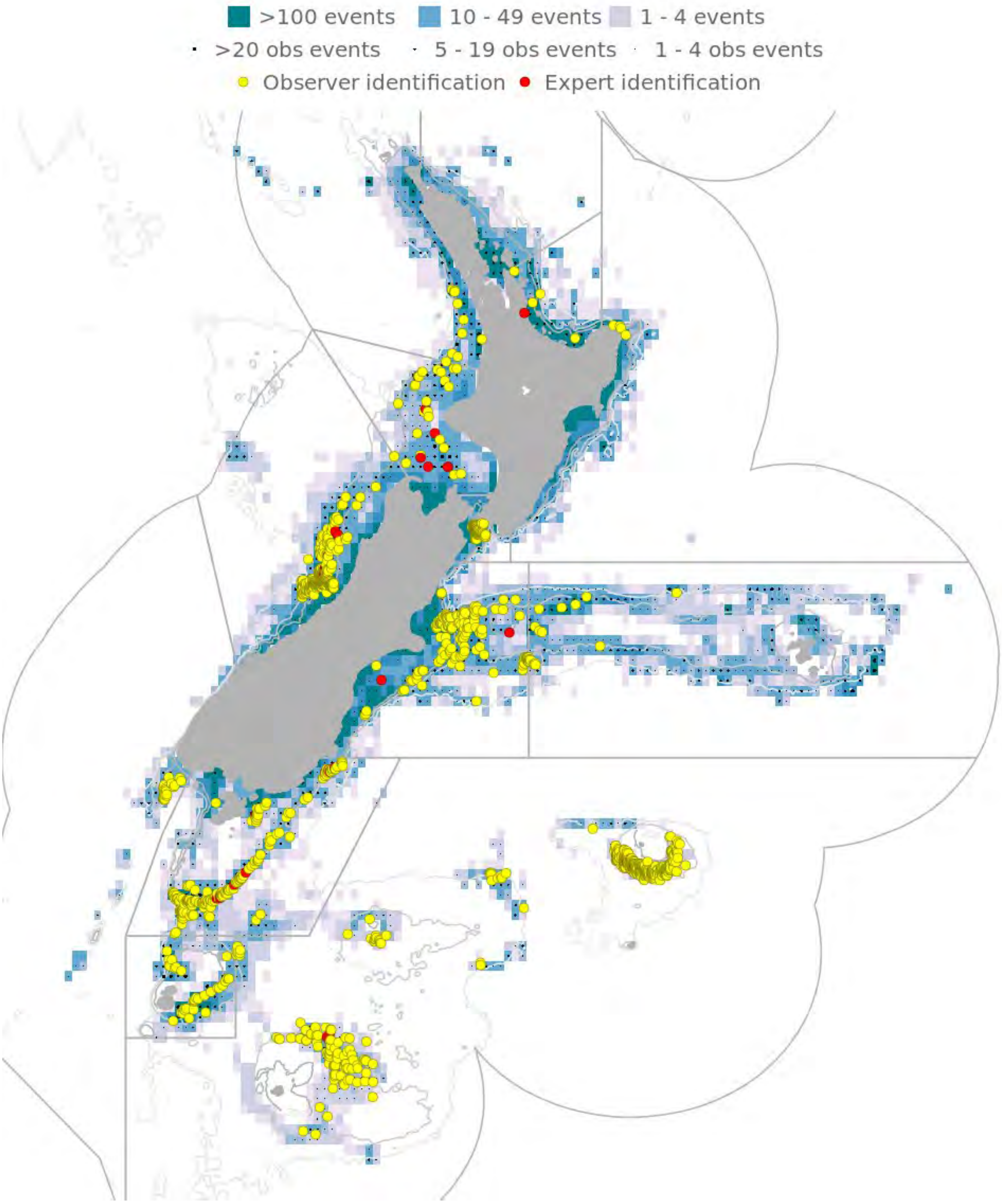

Figure 5.1: Distribution of trawl fishing effort and observed New Zealand fur seal captures, 2002–03 to 2019–20 (for more information see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by yellow and red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell.

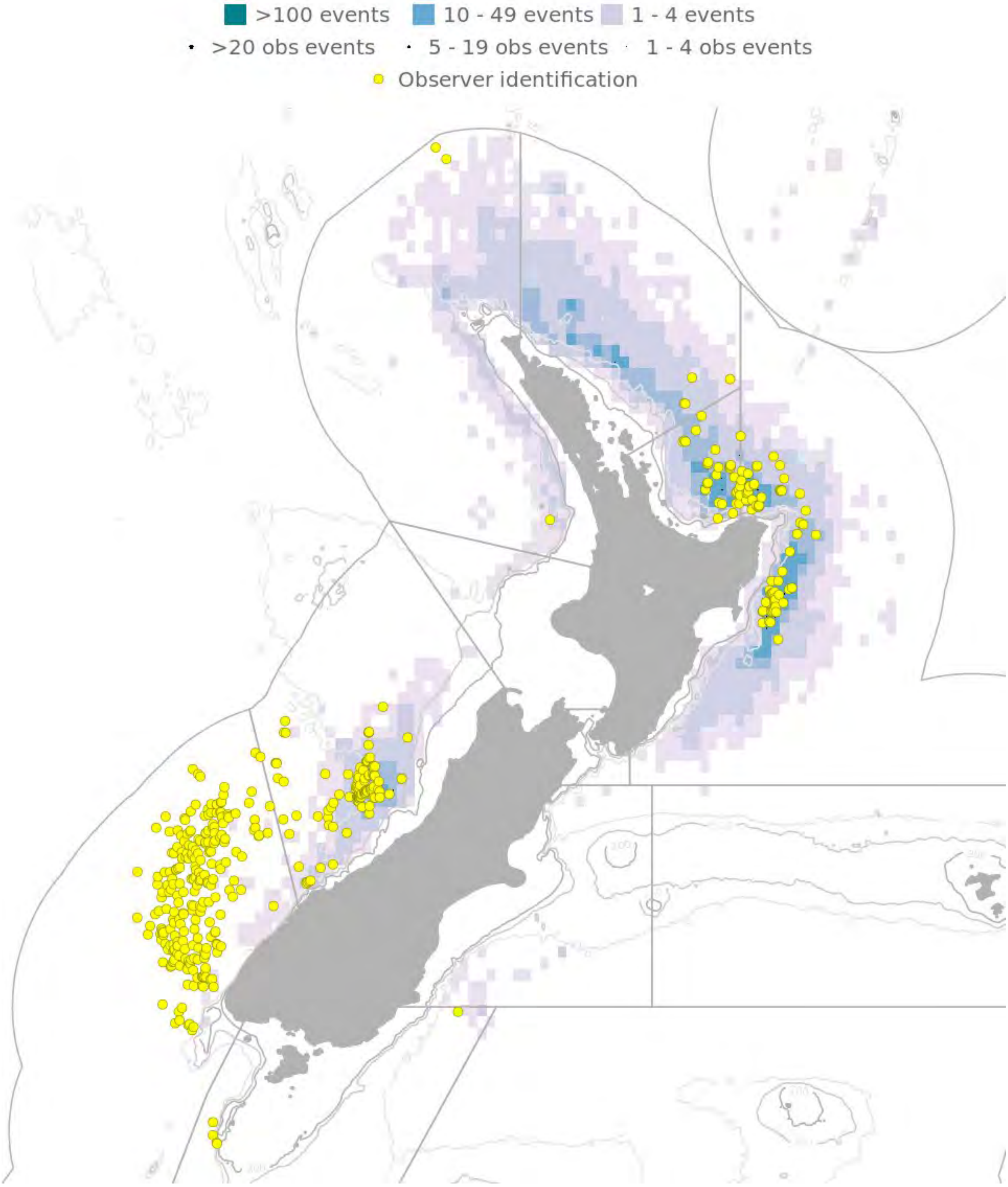

Figure 5.2: Distribution of surface-longline fishing effort and observed New Zealand fur seal captures, 2002–03 to 2019–20 (for more information see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Fishing effort is mapped into 0.2-degree cells, coloured to represent the amount of effort. Observed fishing events are indicated by black dots, and observed captures are indicated by yellow dots. Fishing effort is shown for sets with latitude and longitude data, where three or more vessels and three or more companies or persons fished within a cell. For these years, 90.4% of the effort is shown.

Captures are also reported from the Stewart-Snares shelf fisheries that operate during summer months, mainly for hoki and other middle depths species and squid, and from fisheries throughout the year on the Chatham Rise though captures have not been observed east of longitude 180° on the Chatham Rise.

Captures were reported from trawl fisheries for species such as hoki, hake (*Merluccius australis*), ling (*Genypterus blacodes*), squid, southern blue whiting, jack mackerel, and barracouta (Baird & Smith 2007, Abraham et al. 2010b). Between 1 and 3% of observed tows targeting middle-depths fish species catch New Zealand fur seals compared with about 1% for squid tows and under 1% of observed tows targeting deepwater species such as orange roughy (*Hoplostethus atlanticus*) and oreo species (for example, *Allocyttus niger*, *Pseudocyttus maculatus*) (Baird & Smith 2007). The main fishery areas that contribute to the estimated annual catch of New Zealand fur seals (modelled from observed captures) in middle depths and deepwater trawl fisheries are Cook Strait hoki, west coast South Island middle-depths fisheries (mainly hoki), western Chatham Rise hoki, and the Bounty Islands southern blue whiting fishery (Baird & Smith 2007, Thompson & Abraham 2010). Captures on longlines occur when the New Zealand fur seals attempt to feed on the fish catch during hauling. Most New Zealand fur seals are released alive from surface and bottom longlines, typically with a hook and short snood or trace still attached.

5.4.1 QUANTIFYING FISHERIES INTERACTIONS

Observer data and commercial effort data have been used to characterise fur seal incidental captures and estimate the total catches (Baird & Smith 2007, Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Abraham et al. 2017). This approach is currently applied using information collected under DOC project INT2013-01 and analysed under MPI project PRO2013-01 (Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017). The analytical methods used to estimate capture numbers across commercial fisheries vary depending on the quantity and quality of the data, i.e., total numbers of observed captures and the representativeness of the observer coverage. Initially, stratified ratio estimates were provided for the main trawl fisheries, starting in the late 1980s, after scientific observers reported 198 New Zealand fur seal deaths during the July to September west coast South Island spawning hoki fishery (Mattlin 1994a, 1994b). In subsequent years, ratio estimation was used to estimate New Zealand fur seal captures in the Taranaki Bight jack mackerel fisheries and Bounty Platform, Pukaki Rise, and Campbell Rise southern blue whiting fisheries, based on observed catches and stratified by area, season, and gear type (Baird 1994).

In the last 10 years, model-based estimates of captures have been developed for all trawl fisheries in waters south of 40° S (Baird & Smith 2007, Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017). These models use fisheries observer data and fishing effort data in a hierarchical Bayesian model that includes season and vessel-season random effects and other covariates (for example, day of fishing year, time of day, tow duration, distance from shore, gear type, target) to model variation in capture rates among tows. This method compensates in part for the lack of representativeness of the observer coverage and includes the contribution from correlation in the capture rate among tows by the same vessel. The method is limited by the very large differences in the observed and non-observed proportions of data for the different vessel sizes; most observer coverage is on larger vessels that generally operate in waters deeper than 200 m. The operation of inshore vessels in terms of the location of effort, gear, and the vessel behaviour is only poorly understood compared with the deepwater fisheries. Nonetheless, following changes to reporting requirements, data collection is improving such that inshore trawl effort (not including flatfish trawl effort) is now included in the captures estimation modelling (Thompson et al. 2012, see also description of the Trawl Catch Effort Return, TCER, in use since 2007–08, in Chapter 11 on benthic effects).

Since 2005, there has been a downward, then relatively flat trend in estimated capture rates and total annual estimated captures of New Zealand fur seals in trawl fisheries (Smith & Baird 2009, Thompson & Abraham 2010, Abraham & Thompson 2011, Thompson et al. 2011, Thompson et al. 2012, Abraham et al. 2017; Figure 5.3). This may reflect bycatch reduction efforts undertaken by vessels (see Section 5.4.2) combined with a reduction in fishing effort since the late 1990s. Simultaneous with this decrease in effort is an increase in fisheries observer coverage, especially since 2007. In 2014–15, about 17% of the 78 696 tows were observed, with a capture rate of 0.93 fur seals per 100 tows, to give an annual mean total of 486 captures (95% c.i.: 299–876) (Table 5.2, Figure 5.3).

Observed and estimated capture rates include animals that are released alive; 13% of 1420 observed trawl captures in the 2002–03 to 2014–15 fishing years were recorded as released alive by the observer.

Ratio estimation was used to calculate total captures in longline fisheries by target fishery fleet and area (Baird 2008) and across all fishing methods (Abraham et al. 2010b). New Zealand fur seal captures in surface-longline fisheries have been generally observed in waters south and west of Fiordland, but also in the Bay of Plenty and off East Cape. Estimated surface-longline captures range from 299 (95% c.i.: 199–428) in 2002–03 to 32 (14–55) in 2006–07 (Table 5.2). These capture rates include animals that are released alive; 5.6% of observed surface-longline captures from 2002–03 to 2014–15 were live releases (Abraham et al. 2017).

Captures of New Zealand fur seals have also been recorded in other fisheries; 39 in set nets, 2 in bottom-longline fisheries and 1 from purse seine fisheries from 2002–03 to 2014–15 (Abraham et al. 2017). Because observer data are too sparse and/or unrepresentative to support the estimation method, capture estimation models are not produced for these fisheries. Captures associated with recreational fishing activities are poorly known (Abraham et al. 2010a)

5.4.2 MANAGING FISHERIES INTERACTIONS

The population level impact of direct fisheries mortalities on the New Zealand fur seal population remains somewhat uncertain. However, fishing interactions are considered unlikely to have adverse consequences for New Zealand fur seals at the scale of the entire New Zealand population on the basis of the following evidence: i) the estimated level of bycatch relative to overall New Zealand fur seal abundance; ii) the apparently increasing population and range; and iii) the low threat status assigned to this species by both the New Zealand and IUCN threat classification processes. However, fisheries impact and risk may be higher at the scale of particular colonies or affecting regional subpopulations.

Management has focused on encouraging vessel operators to alter fishing practices to reduce captures, and monitoring captures via the observer programme. A marine mammal operating procedure (MMOP) has been developed by the deepwater sector to reduce the risk of marine mammal captures and is currently applied to trawlers greater than 28 m LOA.

Table 5.1: Monthly distribution of New Zealand fur seal activity and the main trawl and longline fisheries with observed reports of New Zealand fur seal incidental captures.

New Zealand fur	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug
Breeding males	Disp	At breeding colony				Dispersed at sea or at haulouts						
Breeding females	At sea		At breeding colony		At breeding colony and at-sea foraging and suckling							
New pups	At sea			At breeding colony								
Non-breeders	Dispersed at sea, at haulouts, or breeding colony periphery											
Major fisheries	**Sep**	**Oct**	**Nov**	**Dec**	**Jan**	**Feb**	**Mar**	**Apr**	**May**	**Jun**	**Jul**	**Aug**
Hoki trawl		Chatham Rise and Stewart-Snares Shelf								Cook Strait, west coast South Island, Puysegur		
Squid				Stewart-Snares shelf		Auckland Islands and Stewart-Snares shelf						
Southern blue whiting	Pukaki and Campbell Rise											Bounty Islands
Scampi	Mernoo Bank (Chatham Rise) and Auckland Islands											

Southern bluefin tuna longline		South-west South Island	

Table 5.2: Fishing effort and observed and estimated New Zealand fur seal captures in trawl and surface-longline fisheries by fishing year in the New Zealand EEZ (Abraham & Berkenbusch 2019, and see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). For each fishing year, the table gives the total number of tows or hooks; the observer coverage (the percentage of tows or hooks that were observed); the number of observed captures (both dead and alive); the capture rate (captures per hundred tows or per thousand hooks); the estimation method used (model or ratio); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Abraham & Berkenbusch 2019. Estimated captures for trawl fisheries are from a previous version of the data (2019v1).

Fishing year	Fishing effort		Observed captures		Estimated captures	
	All effort	% observed	Number	Rate	Mean	95% c.i.
Trawl fisheries						
2002–03	130 119	5.3	68	0.99	927	646-1 307
2003–04	120 819	5.4	90	1.37	914	646-1 286
2004–05	120 430	6.4	199	2.58	1 579	1 170-2 123
2005–06	109 944	6	143	2.16	1 019	734-1 432
2006–07	103 314	7.7	74	0.94	660	469-916
2007–08	89 531	10.1	142	1.57	737	552-993
2008–09	87 549	11.2	72	0.74	493	353-682
2009–10	92 893	9.7	72	0.8	487	353-668
2010–11	86 078	8.7	73	0.98	551	374-819
2011–12	84 418	11.1	83	0.89	452	323-632
2012–13	83 837	14.8	121	0.98	600	413-904
2013–14	85 110	15.6	159	1.2	379	297-492
2014–15	78 765	17.2	127	0.94	479	352-653
2015–16	78 029	16.6	109	0.84	375	275-521
2016–17	78 173	17.6	79	0.58	–	–
2017–18	74 243	20.1	80	0.54	–	–
2018–19	70 924	19.6	65	0.47	–	–
2019–20	65 994	23.6	60	0.38	–	–
Surface-longline fisheries						
2002–03	10 769 838	20.4	56	0.026	324	177–549
2003–04	7 386 429	21.8	40	0.025	189	105–308
2004–05	3 682 695	21.3	20	0.026	101	51–178
2005–06	3 691 329	19.1	12	0.017	74	30–141
2006–07	3 739 912	27.8	10	0.01	54	22–105
2007–08	2 245 439	18.8	10	0.024	63	27–117
2008–09	3 115 633	30.1	22	0.023	81	42–140
2009–10	2 995 264	22.1	19	0.029	109	54–193
2010–11	3 188 179	21.2	17	0.025	92	45–163
2011–12	3 099 877	23.5	40	0.055	180	102–294
2012–13	2 876 932	19.5	21	0.037	147	72–262
2013–14	2 550 814	30.7	57	0.073	196	120–300
2014–15	2 413 386	30.1	37	0.051	165	93–273
2015–16	2 358 541	13.7	3	0.009	65	16–146

2016–17	2 094 236	16.5	32	0.093	155	89–244
2017–18	2 292 381	12.9	12	0.041	94	37–176
2018–19	2 056 736	9.4	47	0.243	267	140–468
2019–20	2 000 759	9.8	14	0.071	185	89–341

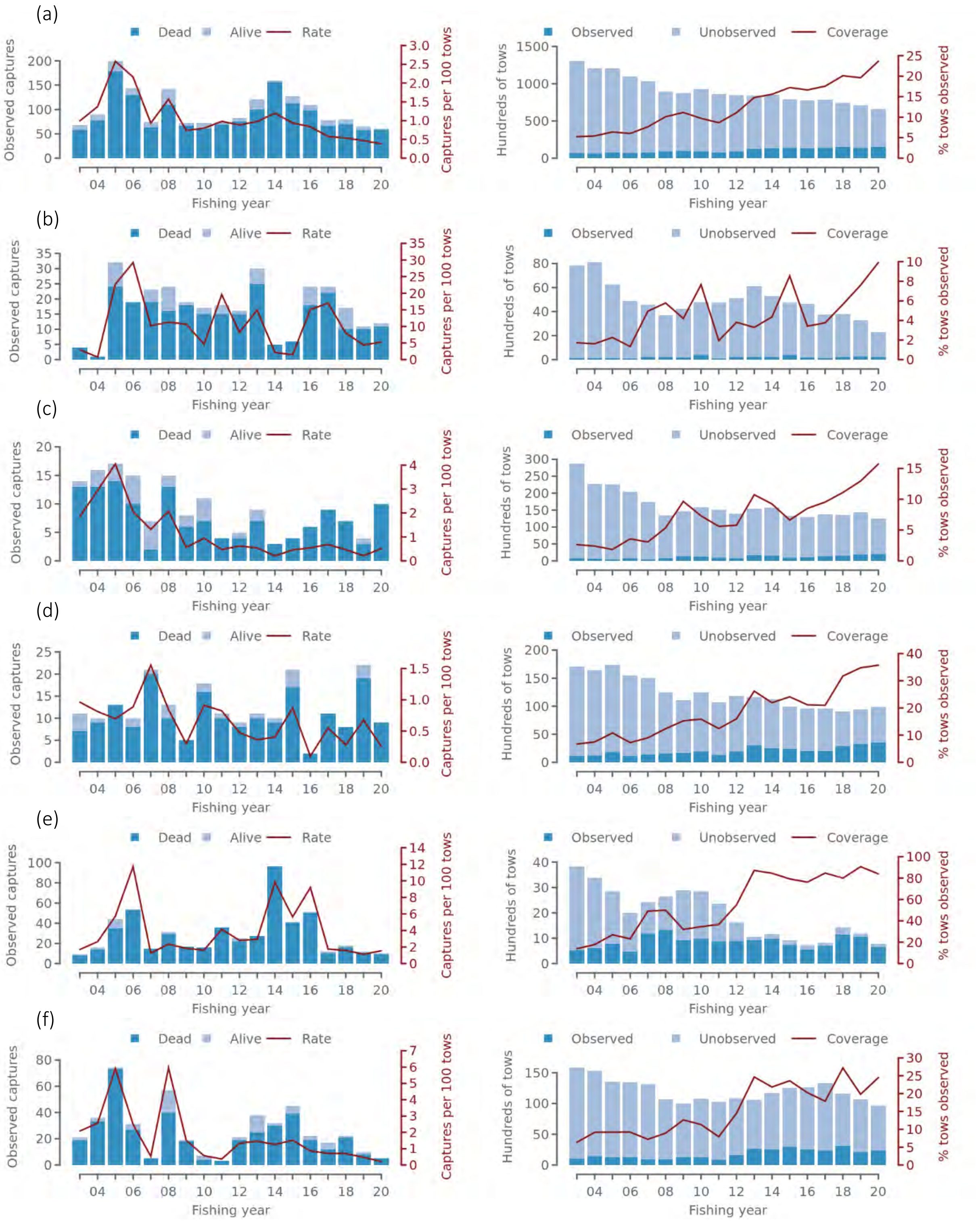

Figure 5.3. Observed captures of New Zealand fur seals (dead and alive) in trawl fisheries, the capture rate (per hundred tows), and the amount of total and observed effort by fishing year for regions with more than 50 observed captures since 2002–03: (a) New Zealand's EEZ; (b) the Cook Strait area; (c) the East Coast South Island area; (d) the Stewart-Snares shelf area; and (e) the subantarctic area; and (f) the West Coast South Island area (Abraham et

al. 2017, and see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Abraham & Thompson (2011).

The management includes a number of mitigation measures supported by annual training; these include managing offal discharge, refraining from shooting the gear when New Zealand fur seals are congregating around the vessel, and the introduction of 'trigger' points whereby if two fur seals are captured within 24 hours, or five seals over seven days, then the following procedure is triggered:

1. Advise vessel manager
2. Record capture event including location of capture in ship's log
3. Ensure gear failures are addressed with the gear either onboard or at a depth > 50 m
4. Report capture to Deepwater Group either directly or via shore management.

The major focus of the MMOP is to reduce the time gear is at or near the surface when it poses the greatest risk. MPI, via observers, monitors and audits vessel performance against this procedure (see the MPI National Deepwater Plan for further details). Research into methods to minimise or mitigate New Zealand fur seal captures in commercial fisheries has focused on fisheries in which New Zealand fur seals are more likely to be captured (trawl fisheries; see Clement and Associates 2009). Finding ways to mitigate captures has proved difficult because the animals are free swimming, can easily dive to the depths of the net when it is being deployed, hauled, or brought to the surface during a turn, and are known to actively and deliberately enter nets to feed. Further, any measures also need to ensure that the catch is not greatly compromised, either in terms of the amount of fish or their condition. Possible fish loss is one potential drawback of using seal exclusion devices (see Rowe 2007). Adhering to current risk mitigation methods (e.g., MMOP) will help to minimise the level of impacts; however, bycatch rates are still expected to fluctuate depending on fleet deployment, New Zealand fur seal abundance, and local feeding conditions.

5.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Uncertainty about the size of the New Zealand fur seal population limits our ability to estimate the effects of direct fisheries mortalities on fur seals at the scale of the New Zealand population. Potential impacts on specific colonies are best addressed via spatially explicit methods (below). The provenance of New Zealand fur seals caught during fishing is presently unknown. Improved research to understand foraging distributions in relation to colony locations is in progress (PMM2018-04A). In addition, genetics research may help to assign bycaught animals to a specific colony (Robertson & Gemmell 2005).

5.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017, the first iteration of a New Zealand Marine Mammal Risk Assessment (NZMMRA) was complete (Abraham et al. 2017) applying a partial implementation of the Spatially Explicit Fisheries Risk Assessment (SEFRA) method formerly applied for New Zealand seabirds and described in Chapter 3.

In the risk assessment outputs fur seals are the seventh-highest at-risk species of marine mammal from New Zealand commercial fisheries. Fisheries risk to fur seals is attributable primarily to 'other trawl' fisheries (i.e., primarily targeting hoki and southern blue whiting), and secondly to set net fisheries. Estimated annual potential fishery-related deaths for fur seals by fishery group are shown in Figure 5.4.

The estimated cumulative fisheries risk score for fur seals ranges from approximately 0.2 to 0.6 (Figure 5.5), consistent with colony observations indicating a general trend of increasing population size in recent years. Note that unlike the NZSRA, the NZMMRA does not utilise population monitoring results directly in the risk assessment to inform or constrain total fishery related deaths to be consistent with observed adult survival rates. Introducing this constraint is a priority when a full implementation of the SEFRA framework is delivered for all marine mammal species (PMM2018-07).

An independent external review of the SEFRA method (Lonergan et al. 2017) noted that the reliability and specific applicability of the previous NZMMRA is limited by its reliance on species spatial distributions derived from expert knowledge in which animal densities are assigned to discrete spatial blocks using a Delphi approach. The reviewers recommended that the MMRA should be updated using more reliable species spatial distributions as these become available. Input data layers reflecting finer-scale spatial and seasonal patterns are likely to be especially important for coastal and/or colony-associated species such as fur seals. Where sightings or satellite telemetry data are available, it is likely that these can be used to parameterise predictive spatial foraging models fitted to continuous environmental variables using multivariate statistical approaches, to estimate spatio-temporal species distributions in a more rigorous way. This work has recently been completed to improve available distribution models for cetaceans (under contract PRO2014-01) and for Māui and Hector's dolphins (PRO2017-12). This work is in progress for Auckland Island sea lions (PRO2017-09), for Stewart Island/ South Island sea lions (PMM2018-04B); and for New Zealand fur seals (PMM2018-04A). Because fur seals show sex-specific movement patterns, it is likely that this work will consider male and female distributions and mortalities separately, given that male and female deaths are likely to have very different implications for the population response of harem-breeding mammals

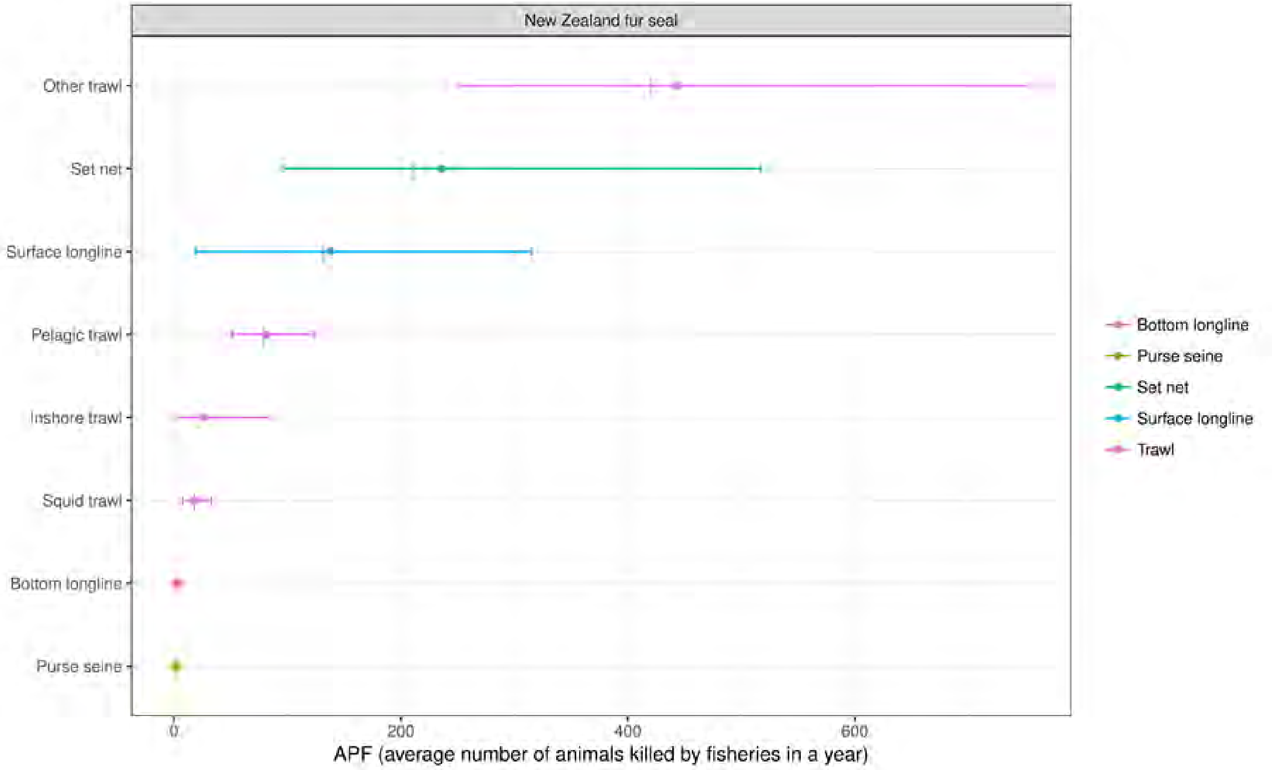

Figure 5.4: Preliminary estimates of annual potential fishery-related deaths of fur seals by fishery group, as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

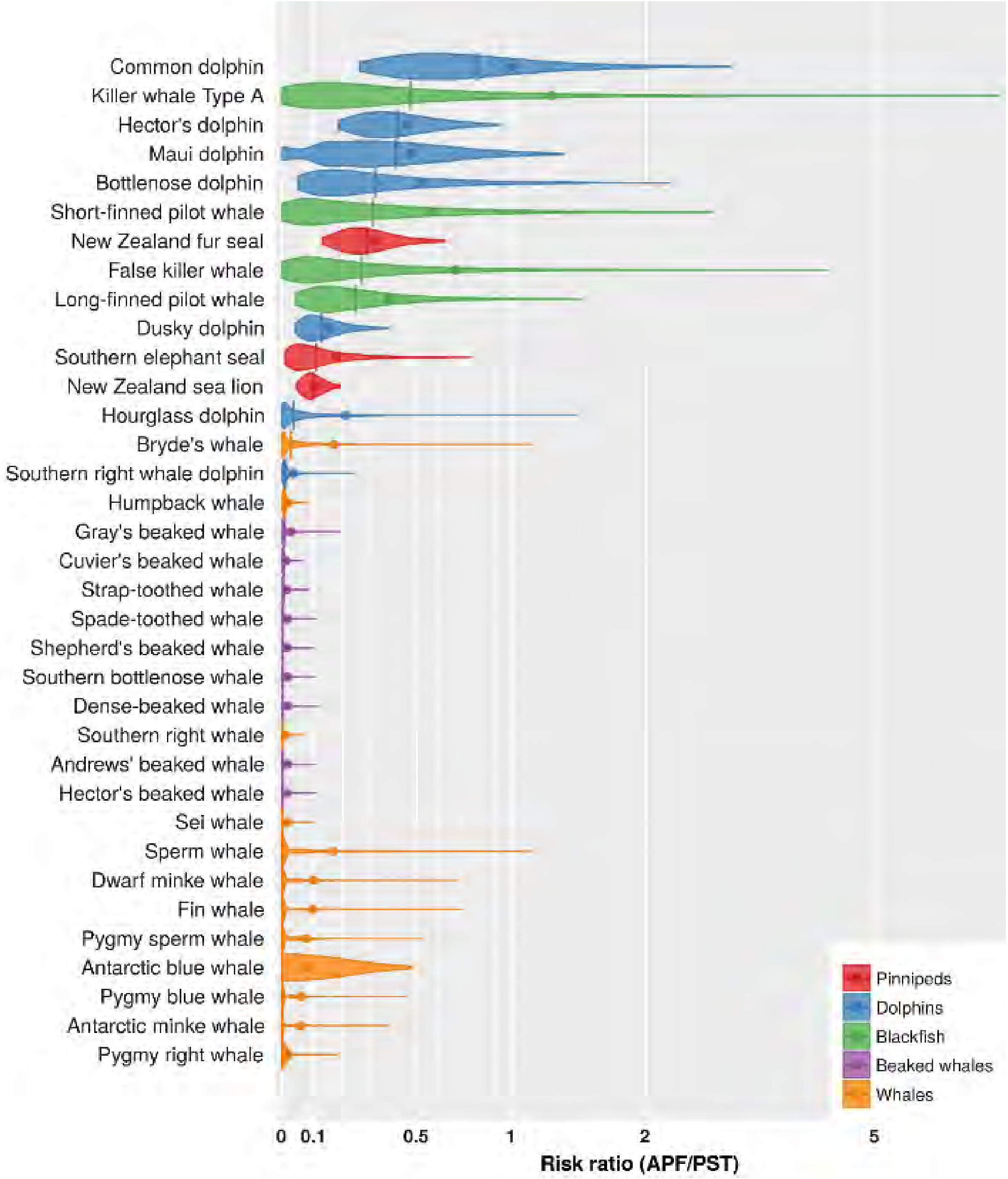

Figure 5.5: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Taxonoic groups are colour coded.

5.4.5 SOURCES OF UNCERTAINTY

Any measure of the effect of New Zealand fur seal mortality from commercial fisheries on New Zealand fur seal populations requires adequate information on the size of the populations at different colonies. Although there is reasonable information about where the main New Zealand fur seal breeding colonies occur, colony size and population dynamics are poorly understood. At present, the main sources of uncertainty are the lack of consistent data on: abundance by colony and in total; population demographic parameters; and at-sea distribution (which would ideally be available at the level of a colony or wider geographic area where several colonies are close together) (Baird 2011).

Collation and analysis of existing data, such as that for the west coast South Island, would fill some of these gaps; there is a 20-year time series of pup production from three west coast South Island colonies, a reasonably long data series from the Otago Peninsula, and another from Kaikōura. Maximum benefit could be gained through the use of all available data, as shown by the monitoring of certain colonies of New Zealand fur seals in Australia to provide a measure of overall population stability (see Shaughnessy et al. 1994, Goldsworthy et al. 2003).

Fur seals may forage in waters near a colony or haulout, or may range widely, depending on the sex, age, and individual preferences of the animal (Baird 2011). It is not known whether the New Zealand fur seals around a fishing vessel are from colonies nearby. Some genetic work is proposed to test the potential to differentiate between colonies so that in the future New Zealand fur seals drowned by fishing gear may be identified as being from a certain colony (Robertson & Gemmell 2005).

The low to moderate levels of observer coverage in some fishery-area strata add uncertainty to the total estimated captures. However, the main source of uncertainty in the level of bycatch is the paucity of information from the inshore fishing fleets, which use a variety of gears and methods. Recent increases in observer coverage enabled fur seal capture estimates to include inshore fishing effort. Further increases in coverage, particularly for inshore fisheries, would provide better data on the life stage, sex, and size of captured animals, as well as samples for fatty acid or stable isotope analysis to assess diet and to determine provenance. Information on the aspects of fishing operations that lead to capture in inshore fisheries would also be useful as input to designing mitigation measures.

5.5 INDICATORS AND TRENDS

Population size	Unknown, but potentially ~100 000 in the New Zealand EEZ.[2]
Population trend	Increasing at some mainland colonies but unknown for offshore island colonies. Range is thought to be increasing.
Threat status	New Zealand: Not Threatened, Increasing, Secure Overseas, in 2013.[3] IUCN: Least Concern, in 2015.[4]
Number of interactions	375 estimated captures (95% c.i.: 275–521) in trawl fisheries in 2015–16[6] 24 estimated captures (95% c.i.: 8–49) in surface-longline fisheries in 2015–16[6] 80 observed captures in trawl fisheries in 2017–18[6] 12 observed captures in surface-longline fisheries in 2017–18[6] 949.3 estimated annual potential fatalities (APF) (95% c.i.: 949.3–1 406.5) [7]
Trends in interactions[6]	Trawl fisheries: 

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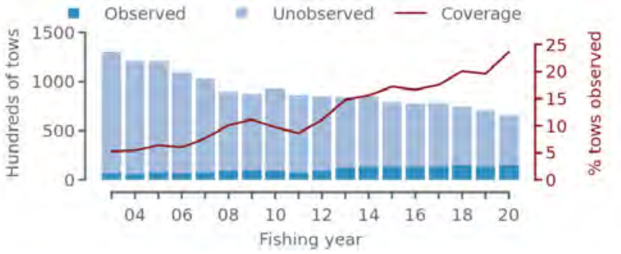
[2] Taylor (1990), Harcourt (2001).

[3] Baker et al. (2016).

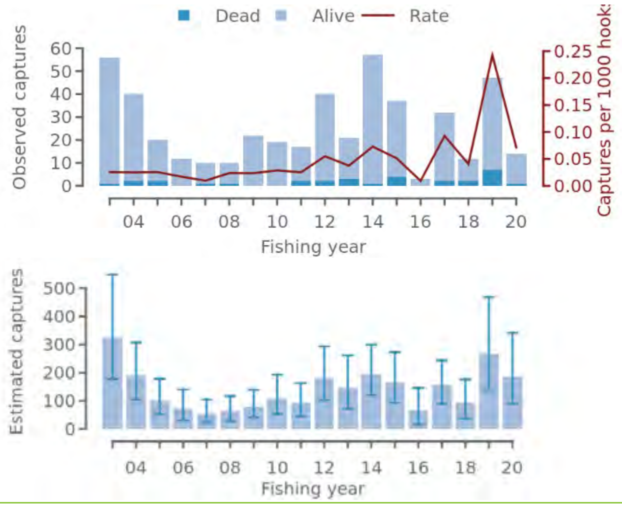
[4] Chilvers & Goldsworthy (2015).

[6] For more information, see: https://protectedspeciescaptures.nz/PSCv6/

[7] Abraham et al. (2017).



Surface-longline fisheries:



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Chapter 6: Hector's and Māui dolphin - Technical Summary

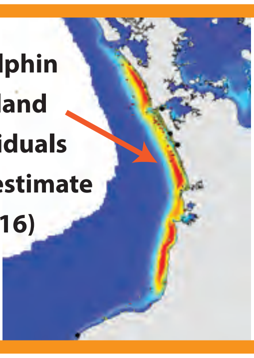
Hector's and Māui dolphins are found only in New Zealand

Both at risk from multiple threats, including fishing and disease

1. THE ISSUE IN BRIEF

- Hector's and Māui dolphin (*Cephalorhyncus hectori*), comprising the South Island subspecies referred to as Hector's dolphin (*C. h. hectori*) and the North Island subspecies known as Māui dolphin (*C. h. maui*), are endemic to the coastal waters of New Zealand.
- Hector's and Māui dolphin are nationally vulnerable and critical (DOC 2019), respectively, and protected under the Marine Mammals Protection Act 1978 and the Fisheries Act 1996. Threats are managed through a Threat Management Plan
- Hector's and Māui dolphins can drown when entangled in fishing gear, and new research has identified other potential threats to their conservation, including diseases (see e.g, box 3)

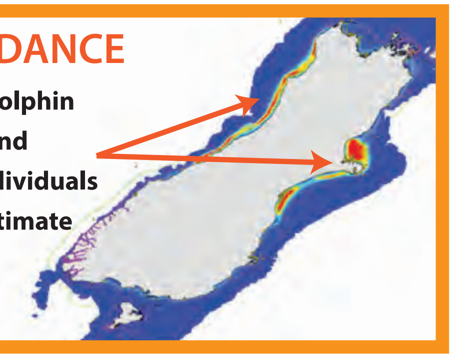
2. DISTRIBUTION AND ABUNDANCE



Māui dolphin
North Island
63 individuals
(model estimate 2015-2016)

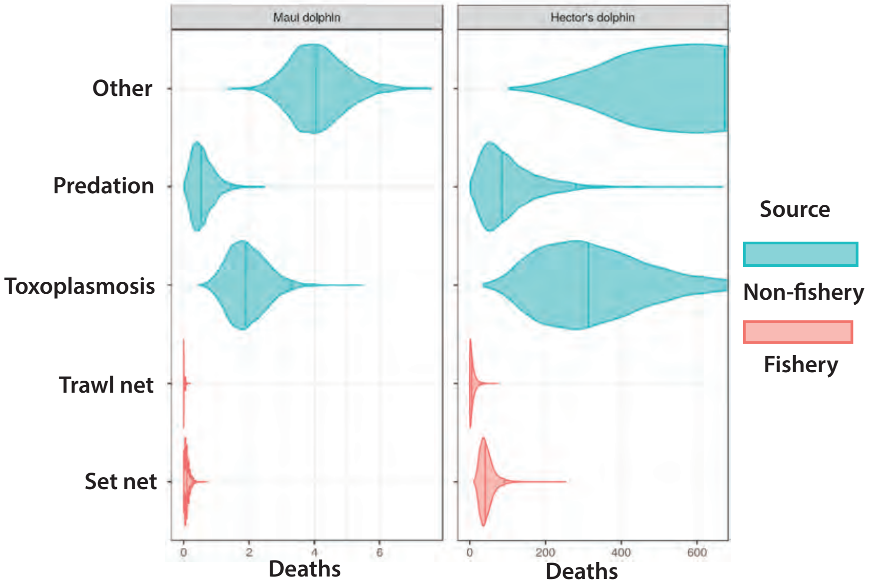
Dolphin abundance is estimated from aerial surveys and genetic census

Dolphin distribution is estimated using spatial habitat models fitted to survey data



Hector's dolphin
South Island
14,594 individuals
(model estimate 2016)

3. MAIN THREATS



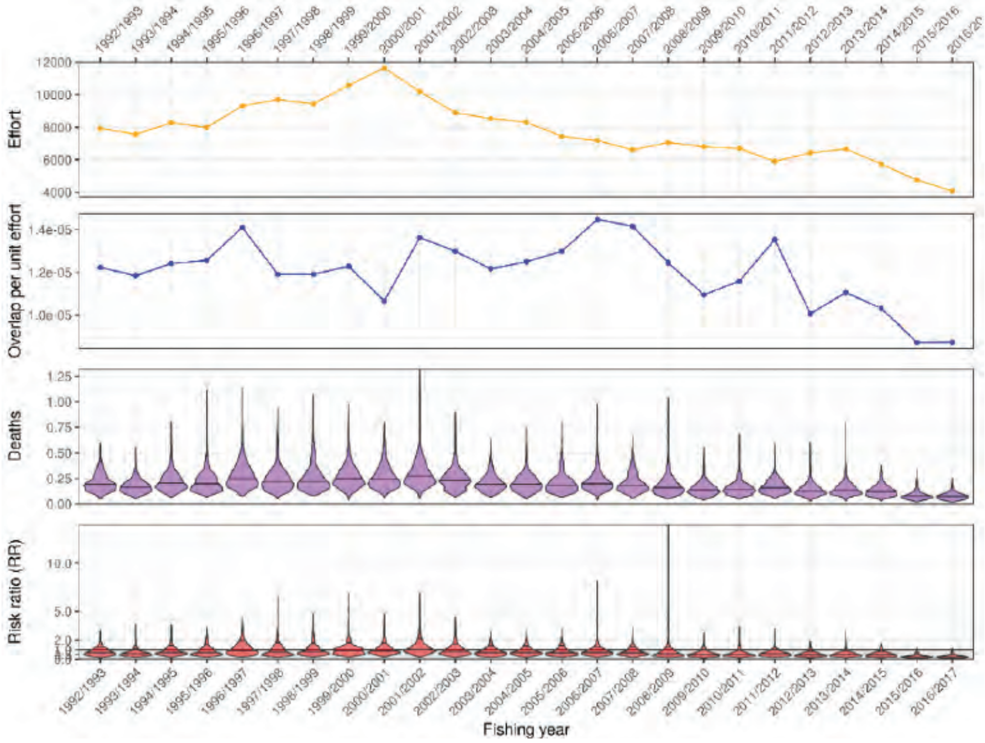
- A spatially explicit risk assessment is used to estimate the risk from different threats (see Chapter 3)
- Some risk of death comes from fishing interactions (commercial set nets and, to a lesser extent, commercial trawl nets)
- Experts estimate that mining and oil exploration activities may also affect Hector's and Māui dolphin
- Toxoplasmosis, a parasitic infection spread to native wildlife by cats, has been recently highlighted as a major threat (see box 6)

Outputs of the multi-threat risk assessment model (Roberts et al. 2019). It is important to note that commercial fisheries deaths (set net and inshore trawl) are based on fisheries observer data and have been estimated with high certainty. Toxoplasmosis deaths have been estimated from necropsy results, which relies on the relative detectability of dolphin carcasses that have died from various causes, resulting in uncertainty that may not be reflected in the ranges above. All mortalities not accounted for in the remaining categories are in "other"

4. ONGOING MULTI-THREAT RISK ASSESSMENT

- Estimates different levels of risk to each subpopulation from multiple threats (including fishing, disease, etc.)
- Provides more accurate estimates than single-risk assessments
- Allows scientists to estimate when, where and how many e.g., fishing- or disease-related deaths occur
- Highlights e.g., the risk of Hector's and Māui dolphin interactions with set nets or encounters with a disease

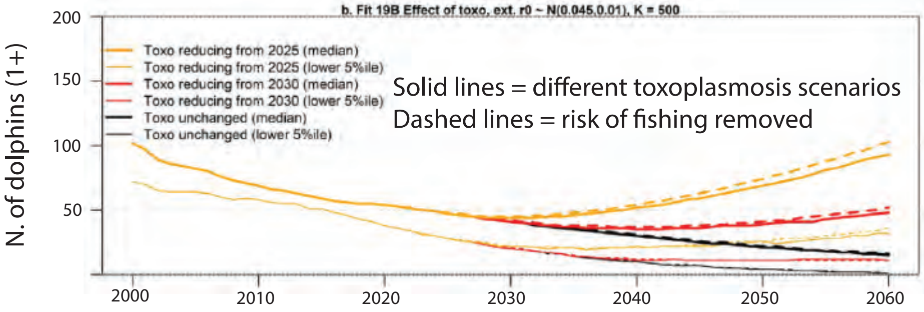
5. MAUI DOLPHIN AND FISHERIES INTERACTIONS



- Fishing gear and area restrictions led to a reduction in fisheries interactions, and therefore fishing-related Māui dolphin deaths, since a peak in 2000-2001
- Additional restrictions to fishing have recently been announced to further reduce risk
- However, limiting fisheries risk alone may not be enough to halt the decline of Māui dolphins (see box 6)

This figure shows historical decrease of set net fisheries effort (and its overlap with dolphins), leading to a corresponding decline in risk of death due to entanglement (Roberts et al. 2019)

6. TOXOPLASMOSIS



Solid lines = different toxoplasmosis scenarios
Dashed lines = risk of fishing removed

- The models demonstrate that risks from fishing and toxoplasmosis both need to be managed to allow Māui dolphins to recover

Maui dolphin population projections from an individual-based demographic model fitted to genetic mark-recapture data (Cooke et al. 2019). Solid lines represent projections of population numbers under different toxoplasmosis scenarios, dashed lines represent the effects of removing fisheries risks under each scenario

7. ONGOING RESEARCH

- Ongoing research is investigating the threat posed by toxoplasmosis, a disease to which some marine mammals may be particularly sensitive
- For these species to recover, other potential threats (e.g. other diseases or climate change) and novel technologies for population monitoring (aerial surveys through drones) are currently being assessed

6 HECTOR’S DOLPHIN (*CEPHALORHYNCHUS HECTORI HECTORI)* AND MĀUI DOLPHIN (*C. H. MAUI)*

Status of chapter	This chapter has not been updated for AEBAR 2021.
Scope of chapter	This chapter briefly summarises: the biology, foraging ecology, population structure, abundance, and spatial distribution of Hector’s and Māui dolphins (*Cephalorhynchus hectori hectori and C. h. maui*); fisheries and non-fisheries threats to Hector’s and Māui dolphins; means of estimating fisheries impacts and subpopulation level risk; population demographic modelling; management of fisheries risk; and identified priority research questions, to guide future work.
Area	West coast North Island; all coastal areas of South Island.
Focal localities	Hector’s and Māui dolphin habitat includes nearshore waters, mostly in locations with high water turbidity, around the full extent of the South Island and the west coast of the North Island. Hector’s and Māui dolphins are also occasionally sighted around the north and east coasts of the North Island.
Key issues	The following issues are identified as key areas for further investigation: improved estimation of Hector’s and Māui dolphin spatio-temporal density affecting spatial overlap with fisheries in low-dolphin-density locations, e.g., North Coast South Island (NCSI), South Coast South Island (SCSI),and Kaikōura; estimation of population status and trajectory at subpopulation scales; improved population size estimates for the North Coast South Island Hector’s dolphin subpopulation; improved estimation of cryptic mortality in set nets; improved understanding of factors potentially affecting dolphin catchability in different types of fishing gears (for example low-headline-height trawl nets); options for fisheries mitigation.
Emerging issues	The following issues are identified as areas of emerging importance for future work: Improved carcass recovery and data capture from bycaught, beach-cast, and/or at-sea recovered carcasses to better understand non-fishery causes of death including from disease; improved understanding of the impact of toxoplasmosis on dolphin subpopulations; effects of other diseases such as brucellosis; improved understanding of potential biases arising from the use of beach-cast carcasses to understand threats; improved understanding of factors affecting reproductive success in different subpopulations (e.g., effects of fishing or climatic variability on dolphin prey and/or habitat); establishment of ongoing population monitoring for priority subpopulations; improved understanding of dolphin movements affecting connectivity between subpopulations.
Fisheries New Zealand research (current)	SEA2019-21 *Characterisation of DOC Hector’s and Māui dolphin incidents data*; SEA2019-22 *Reanalysis of Banks Peninsula Hector’s dolphin demographic data*; SEA2019-27 *Hector’s dolphin trawl-deployed acoustics feasibility study*; PRO2019-11 *Historical reconstruction and characterisation of spatially explicit historical set net fishing;* PMM2018-07 *Updated spatially explicit fisheries risk assessment for New Zealand marine mammal populations*
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2017-03 *Identification of marine mammal, turtle and protected fish captured in New Zealand fisheries*; INT2018-03 *Improvement in observer photograph protocols and photograph curation*; INT2019-03 *Characterisation of marine mammal interactions*; POP2019-01 *Investigation of electronic device options to assess distribution, diving, and foraging behaviour of Hector’s dolphins*; MIT2018-01 *Protected species engagement project*; MIT2019-01 *Dolphin dissuasive device mitigation in inshore fisheries*. Additional work being undertaken by DOC: Genetic sampling and necropsy (where suitable) of any retained Hector’s and Māui dolphin carcasses; Validation of public sightings of Māui dolphins, and Hector’s dolphins at the top of the South Island; Epigenetic aging of Hector’s and Māui dolphins; Abundance estimate of Māui dolphins; Toxoplasmosis literature review; Toxoplasmosis research programme – currently being defined; Analysis of North Coast South Island Hector’s genetic samples – not contracted at present. At DOC conservancy level there is a programme to evaluate acoustic data collected from C-pods.

Other research[1]	Otago University: Long-term study of Hector's dolphins at Banks Peninsula, including distribution, abundance, survival, reproduction, movement, and feeding ecology. Abundance and distribution of Hector's dolphins on Otago coast, Porpoise Bay. Effects of tourism and aquaculture. Auckland University: Population monitoring of Māui dolphins; genetics of Hector's and Māui dolphin subpopulations. Novel drone technologies for studying and monitoring dolphin populations (NGO partnership). Massey University: Necropsy of recovered Hector's and Māui dolphin carcasses; disease threats to dolphins.
Related chapters/issues	Chapter 3 (SEFRA); Chapters 4–5 (sea lions and fur seals); Chapter 7 (common dolphins)

6.1 CONTEXT

Hector's and Māui dolphin[2] (*Cephalorhynchus hectori*), comprising the South Island subspecies referred to as Hector's dolphin (*C. h. hectori*) and the North Island subspecies known as Māui dolphin (*C. h. maui*), is endemic to the coastal waters of New Zealand. Like most other small cetaceans, the species is vulnerable to fishing-related mortality, particularly from set net fisheries (e.g., Read et al. 2006, Reeves et al. 2020, Geijer & Read 2013), in locations where fisheries and dolphins overlap.

Hector's and Māui dolphin was gazetted as a 'threatened species' by the Minister of Conservation in 1999 and is defined as a 'protected species' according to part 1, s2(1) of the Fisheries Act 1996 and s2(1) of the Marine Mammals Protection Act (MMPA) 1978. Management of fisheries impacts on Hector's and Māui dolphins is legislated under both these acts. The MMPA 1978 allows for the approval of a population management plan for any protected species, within which a maximum allowable level of fishing-related mortality may be imposed. For threatened species, this level 'should allow the species to achieve non-threatened status as soon as reasonably practicable, and in any event within a period not exceeding 20 years' (MMPA 1978, p.11). If a population management plan has been approved, the Fisheries Act 1996 requires that all reasonable steps be taken to ensure that the maximum allowable level of fishing-related mortality is not exceeded, and the Minister may take other measures necessary to further avoid, remedy, or mitigate any adverse effects of fishing on the relevant protected species. In the absence of a population management plan, 'the Minister may, after consultation with the Minister of Conservation, take such measures as he or she considers are necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species, and such measures may include setting a limit on fishing-related mortality' (Fisheries Act 1996, p.66).

No population management plan has been produced for either Hector's or Māui dolphins, and no maximum allowable level of fishing-related mortality has been set. Human-induced threats to Hector's and Māui dolphins are instead managed through a Threat Management Plan (TMP); first developed jointly by the Department of Conservation (DOC) and the former Ministry of Fisheries (MFish) in 2007. The TMP is not a statutory document, but a management plan identifying human-induced threats to the populations and outlining strategies to mitigate those threats. The TMP is reviewed approximately every 5 years. A review of the Māui portion of the TMP undertaken in 2012 provided a comprehensive overview of information relating to the biology, distribution, threats to, and management of Māui dolphins (MPI & DOC 2012). This review was informed by a spatially explicit, semi-quantitative risk assessment conducted using an expert panel, applying an early modification of the SEFRA method (Chapter 3), to identify, analyse, and evaluate all threats to Māui dolphins (Currey et al. 2012).

A full review of the TMP was undertaken in 2019, including a comprehensive review of new science as well as a collaborative stakeholder engagement process to inform the policy aspects of the TMP, including articulation of a vision statement and population-level goals and objectives.

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[1] Du Fresne et al. (2012) compiled a bibliography of all Hector's and Māui dolphin research completed since 2003 (http://www.doc.govt.nz/documents/science-and-technical/drds332entire.pdf).

[2] In this document, 'Hector's dolphin(s)' refers to the South Island subspecies (*Cephalorhynchus hectori hectori*), and 'Māui dolphin(s)' refers to the North Island subspecies (*C. hectori maui*). 'Hector's and Māui dolphin(s)' refers to both subspecies collectively (*C. hectori*). This approach is taken to avoid confusion and enable distinction between the South Island subspecies and the species as a whole.

Stakeholder workshops were attended by iwi and hapū, commercial and recreational fishing industry representatives, fishers, scientific experts, dolphin advocates and environmental NGOs, local government representatives, dolphin tourism business representatives, and interested members of the public. The review proposed a new vision statement to guide the TMP:

New Zealand's Hector's and Māui dolphin populations are resilient and thriving throughout their natural range.

To achieve this vision the review recommended adoption of a set of long- and medium-term goals. One of the goals is to:

Ensure known human-caused threats are managed within levels that allow subpopulations to thrive and recover.

To operationalise this goal within the definition of the Population Sustainability Threshold (PST; see Chapter 3), population outcomes were proposed for Māui dolphins and for each Hector's dolphin subpopulation, corresponding to a maximum impact that the subpopulation can sustain while still achieving the defined objective. The population outcomes thereby help to define specific measurable metrics by which to reduce the impact of particular threats (e.g., fishing).

Stakeholders discussed the need for population outcomes to reflect the urgent conservation status of Māui dolphins in particular, and to consider the specific circumstances of small or reproductively isolated Hector's dolphin subpopulations. Note however that the choice of population outcome itself is a policy decision, reflecting a societal value judgment not a scientific assessment.

The following population outcomes were proposed:

- *Māui dolphins: Human impacts are managed to allow the population to increase to a level at or above 95 percent of the maximum number of dolphins the environment can support.*
- *Hector's dolphins: Human impacts are managed to allow each subpopulation to increase to a level at or above 90 percent of the maximum number of dolphins the environment can support.*

A population outcome of 95 percent for Māui dolphins (with high certainty, see footnote below) means that human-induced deaths need to be as near as practicable to zero.

The population of Hector's dolphins is much larger than the Māui dolphin population. Therefore, the level of impact that Hector's dolphin subpopulations can sustain will be higher while still allowing the population to achieve a defined population objective, expressed in terms of maintaining average population size at or above a very high proportion of the maximum number of dolphins the environment can sustain[3].

The 2019 review was informed by a more comprehensive spatially explicit risk assessment including fisheries and non-fishery threats to Hector's and Māui dolphins, and demographic population models for separate regional subpopulations (Roberts et al. 2019a). The risk assessment incorporated updated estimates of population size, demographic parameters affecting population growth and recovery potential (r^{max}), and improved estimates of the distribution of the dolphins to better estimate spatial overlap with threats, adapting methods described in Chapter 3. This information was used to reassess the risk of commercial fishing, recreational set net fishing, and non-fishing-related threats for the Hector's and Māui dolphin local and subpopulations, and to evaluate the effectiveness of current and new potential management measures and monitoring programmes to address those threats. As at June 2020, decisions on the revised Hector's and Māui dolphin Threat Management Plan are still pending.

6.2 BIOLOGY

6.2.1 TAXONOMY

Hector's and Māui dolphin (also recognised as the South and North Island Hector's dolphin) are designated as subspecies in acknowledgement of their common ancestral connections, but there are current differences in

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[3] Note that with respect to particular impacts (e.g., fisheries), where decision makers wish to ensure that a population objective is achieved with high certainty, this is achieved by comparing the upper 90th or 95th percentile of the impact estimate against the PST, rather than by adjusting the definition of the PST itself. For example, in advice to inform the update of the dolphin TMP, evaluation against the fisheries impact objective used the 95th percentile estimate of fisheries impact.

morphology and genetics as a result of the North Island dolphins being isolated from the South Island around the time of the last glacial period about 15 000 years ago (Pichler et al. 2001, Baker et al. 2002). Due to the similar appearance of both Hector's and Māui dolphins, genetic markers are the only way to identify which subspecies an individual belongs to. The species is classified within the *Cephalorhyncus* genus of dolphins, which includes three other species found in the Southern Hemisphere (Heaviside's dolphin found off South Africa and Namibia, the Chilean dolphin found in the coastal waterways of Chile, and the Commerson's dolphin found in Argentina, the Falkland Islands, and the Kerguelen Islands).

6.2.2 REPRODUCTIVE BIOLOGY

Information from incidentally captured or stranded Hector's dolphins indicates that Hector's and Māui dolphins reach sexual maturity around 5–9 years old. The dolphins appear to live until at least their mid-20s based on mark-recapture and necropsy data (Gormley 2009, Rayment et al. 2009b, Webster et al. 2009). These estimates are used in a Bayesian assessment integrating information from ageing and maturity data, and a novel invariant based on body length at maturity relative to asymptotic length, which indicated that the age at which 50% of animals are mature is 6.91 y (95% credible interval = 5.82–8.24) (Edwards et al. 2018).

Breeding occurs in summer, during which larger aggregations of dolphins engage in high levels of activity associated with their multi-mate breeding system (Slooten et al. 1993). There is competition amongst males to mate with the few females in oestrus, and males move between different groups of dolphins to increase their mating opportunities (Slooten 1991, Slooten et al. 1993). Females give birth to a single calf during the summer and will come into oestrus again about2 years later at around the time the previous calf is weaned (Dawson 2019). Calves are born large relative to the mother (neonatal length 60–75 cm; Slooten & Dawson 1994). Calves grow rapidly in the first few years and reach adult size at around five years old (Webster et al. 2010).

Hector's and Māui dolphins are typically found in small groups of 1–14 individuals (Slooten et al. 2006, Rayment et al. 2010, 2011b, Oremus et al. 2012). Mean group sizes appear to be larger when estimated from boat-based surveys (e.g., Webster et al. 2009, Oremus et al. 2012) compared with aerial surveys (e.g., Slooten et al. 2006, Rayment et al. 2010) possibly due to the species' boat-positive behaviour (e.g., Dawson et al. 2004). Webster et al. (2009) found that Hector's dolphin groups were highly segregated by sex, with 91% of groups of up to five individuals being all male or all female; similar patterns of sex segregation are not apparent in Māui dolphins (Oremus et al. 2012). Although often associated with mother-calf pairs outside the breeding season, males play no role in calf rearing, but females often form nursery groups, comprising either a single mother-calf pair or small aggregations of mother-calf pairs (Bräger 1999, Webster et al. 2009, Oremus et al. 2012).

6.2.3 FORAGING ECOLOGY

Miller (2014) and Miller et al. (2013) investigated the diet and feeding ecology of Hector's and Māui dolphins through the examination of diagnostic prey remains in the stomachs of 63 incidentally captured and beach-cast animals and stable isotope analyses. They concluded that Hector's dolphins take a wide variety of prey throughout the water column (in total 29 taxa were recorded), but that the diet is dominated by a few midwater and demersal species. The diets of Hector's dolphins from the South Island west and east coasts were significantly different, due largely to the high prevalence of javelin fish (*Lepidorhynchus denticulatus*) on the west coast, and a greater prevalence of demersal prey species on the east coast (Miller et al. 2013). Nonetheless red cod (*Pseudophycis bachus*) was the most abundant prey species by mass on both coasts. Red cod comprised 37% of the total dietary mass and may be particularly important to east coast South Island (ECSI) females (60% of the dietary mass of 19 individuals). Five other taxa — arrow squid (*Nototodarus* sp.), ahuru (*Auchenoceros punctatus*), sprat (*Sprattus* sp.), sole (*Peltorhamphus* sp.), and stargazer (*Crapatulus* sp.) — together comprised 30% of the total dietary mass from all 63 stomachs. Prey items ranged from an estimated 0.5–60.8 cm in length, but the majority were less than 10 cm in length, indicating that for the larger fish species, predation focuses on juveniles. Weir (2018) reconstructed the mean lengths of the main prey species compiled by Miller et al. (2013) as follows: red cod: 17.9±10.1 cm; arrow squid 17.1±9.4 cm; sprat 10.4±2.1 cm; stargazer 10.2±4.1 cm; ahuru 8.3±3.3 cm; and sole 4.4±4.0 cm.

Only two samples were derived from Hector's and Māui dolphins off the west coast North Island (WCNI), containing red cod, ahuru, sole, and flounder (*Rhomboselea* sp.; Miller et al. 2013).

The stomachs of the six smallest dolphins in the total sample (standard length under 90 cm) contained only milk; a single specimen (at 99 cm long) contained milk and remains of arrow squid in the stomach; but specimens longer than 107 cm did not contain milk (Miller et al. 2013).

Although demersal fish account for the majority of dolphin diet by number and by mass, Hector's dolphins are also occasionally seen foraging near the sea surface on small fish including sprat, pilchard (*Sardinops neopilchardus*), and yellow-eyed mullet (*Aldrichetta forsteri*; Miller et al. 2013), sometimes in association with white-fronted terns (*Sterna striata*; Bräger 1998).

Hector's dolphins have been observed foraging in association with demersal trawlers at Banks Peninsula, presumably targeting the fish disturbed but not captured by the trawl net (Rayment & Webster 2009). New work is underway under Fisheries New Zealand project SEA2019-27 to investigate options using vessel-deployed hydrophone arrays to better characterise Hector's dolphin interactions with trawl fishing operations.

6.2.4 DISTRIBUTION AND MOVEMENT

Historically, Hector's dolphins were distributed throughout the coastal waters around most of the South Island and around large parts of the North Island. There are several different Māori names for these dolphins depending on the iwi or hapū in the region they were observed, with the dolphins regularly appearing in local narratives around New Zealand. This reflects the common presence of these dolphins throughout coastal waters in pre-European times (McGrath submitted).

Hector's dolphins occur in highest densities off the west coast of the South Island (WCSI) between Jackson Bay and Kahurangi Point (Bräger & Schneider 1998, Rayment et al. 2011a), off the east coast (ECSI) between the Marlborough Sounds and Otago Peninsula (Dawson et al. 2004, MacKenzie & Clement 2014) and off the south coast (SCSI) between Toetoes Bay and Porpoise Bay and in Te Waewae Bay (Bejder & Dawson 2001, Dawson et al. 2004). Population densities are lower in the intervening stretches of coast, e.g., Fiordland (Bräger & Schneider 1998), in Golden Bay (Slooten et al. 2001) and along the south Otago coast (Jim Fyfe pers. comm.), suggesting a spatially discontinuous distribution.

There are clear genetic differences between Hector's dolphins in different locations, including over relatively small distances (Pichler et al. 1998, Pichler and Baker 2000, Hamner et al. 2012a, Hamner et al. 2016, 2017). Genetic differentiation at this scale is unusual among cetaceans in the absence of geographical barriers and reflects that individual Hector's dolphins are thought to have small home ranges and high philopatry (Pichler et al. 1998, Bräger et al. 2002, Rayment et al. 2009b). Genetic analysis of Hector's dolphins from the North Coast South Island (NCSI) and from Kaikōura highlight the importance of understanding connectivity between smaller local populations and larger neighbouring subpopulations (Hamner et al. 2016, Baker et al. 2017).

Bräger & Bräger (2018) found that home range sizes are likely to be population-specific and contingent on local topographic and other environmental features. For example, the deep-sea Kaikōura Canyon may constitute a substantial dispersal barrier; evidence suggests that dolphins tend not to cross the canyon, as reflected in genetic differences between dolphins north and south of the canyon (Weir and Sagnol 2015, Hamner et al. 2016, Bräger and Bräger 2018. The ECSI populations off Kaikōura and Moeraki compared with WCSI populations off Westport-Greymouth and Jackson Bay also showed significant differences in individual movement patterns, based on photo-ID observations (Bräger & Bräger 2018). Satellite tagging of three Hector's dolphins near Banks Peninsula in 2004 recorded maximum movements of 50.9 to 66.5 km over deployments lasting from four to seven months (Stone et al. 2005). Rayment et al. (2009a), using photo-ID records of 53 dolphins near Banks Peninsula, recorded maximum distances between sightings of each dolphin ranging from 9.3 km to 107.4 km for the period 1985–2006. Rare observations of Hector's dolphin movements over 400 km (Hamner et al. 2014a) are considered to be exceptional behaviour.

Genetic testing of WCNI dolphins since 2001 has identified a small number of Hector's dolphins located within the contemporary distribution of Māui dolphin as far north as the Manukau Harbour. These results confirm the occurrence of at least occasional long distance dispersal by Hector's dolphins (Hamner et al. 2012b, Baker et al. 2016b). Although some of these dolphins were observed in association with Māui dolphins, to date there is no evidence of successful interbreeding (Hamner et al. 2014b).

Presumed Māui dolphins sightings extend from Maunganui Bluff to New Plymouth (Slooten et al. 2005, Du Fresne 2010, Hamner et al. 2012a, 2012b; DOC 2020b). Research surveys since 2003 are focused in areas of highest dolphin density between Kaipara Harbour and Kawhia (Slooten et al. 2005, Du Fresne 2010, Hamner et al. 2012a, 2012b).

Historical samples from strandings and museum specimens have allowed genetic identification of Māui dolphins off the WCNI from Dargaville to Wellington (DOC 2020a, Pichler 2002). Pichler & Baker (2000) reported genetic analysis of samples of Hector's and Māui dolphins dating back to 1870 and suggest that abundance has declined and geographic range has contracted over the past 140 years. Historical strandings data also indicate that the geographical range of Māui dolphins contracted from the 1970s to the 1990s (Russell 1999), but most of these dolphins were not genetically identified and so could also have included Hector's dolphins.

There are occasional reported public sightings of Hector's and/or Māui dolphins from all around the North Island, including validated sightings (e.g., Baker 1978, Cawthorn 1988, Russell 1999, Freeman 2003, McGrath submitted). The Department of Conservation maintains a website encouraging the public to report sightings (DOC 2020b) and uses a systematic validation process whereby scientific experts contact each person reporting a sighting in the North Island[4]. The locations of both validated and un-validated sightings are shown in Figure 6.1. That even un-validated sightings tend to cluster in locations where spatial distribution models predict that the habitat is most suitable (see below) further supports their credibility. It is typically assumed that North Island sightings in locations outside the known core Māui dolphin area are indicative of transient animals from other locations rather than resident local populations; i.e., as at June 2020 there are no confirmed records of newborn calves in North Island locations outside the known Māui dolphin subpopulation area. However there remains the possibility that Māui or Hector's dolphins may expand their current distribution or disperse to recolonise suitable North Island habitats in future, e.g., in Hawke Bay or the South Taranaki Bight near Whanganui.

6.2.5 HABITAT PREFERENCE AND SPATIAL ABUNDANCE PATTERNS

Hector's and Māui dolphins typically inhabit shallow waters close to shore, including in harbours and bays and in open coastal waters (e.g., Rayment et al. 2009a, Rodda and Moore 2013, Derville et al. 2016, Bräger & Bräger 2018). There are differences in daily and seasonal distribution patterns in different locations (e.g., Dawson & Slooten 1988, Stone et al. 1995, Bräger et al. 2003, Rayment et al. 2009b, Turek et al. 2013, MacKenzie & Clement 2014, 2016). Near Banks Peninsula, Hector's dolphins are sighted most frequently close to shore, but have also been observed up to 22 nm offshore, especially in Pegasus Bay over shelf waters shallower than 50 m depth (Bräger et al. 2003; see Figure 6.5). In contrast, the WCSI Hector's dolphins generally have longer alongshore ranges, but are usually found within approximately 6 nm of shore (Bräger et al. 2003, Rayment et al. 2011a, MacKenzie & Clement 2016, Bräger & Bräger 2018). Similar to the ECSI, highest density areas are mostly within the 50 m depth contour (Figure 6.5).

Māui dolphins are most abundant in inshore waters between Manukau Harbour and Port Waikato. Most sightings are concentrated within 4 nm of the coast (Slooten et al. 2005, MPI & DOC 2012, Oremus et al. 2012); with lower numbers of sightings out to 7 nm (Du Fresne 2010, Thompson & Richard 2012) and very occasional sightings further offshore (Figure 6.5). Passive acoustic monitoring using deployed hydrophones revealed a similar pattern at the core of the Māui dolphin range near Manukau Harbour, with the majority of detections occurring within 4 nm, but occasional detections further offshore to a maximum distance of 10 nm (Nelson & Radford 2018).

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[4] The DOC sightings confirmation process is described here: https://www.doc.govt.nz/globalassets/documents/conservation/native-animals/marine-mammals/mauis-validation-system.pdf

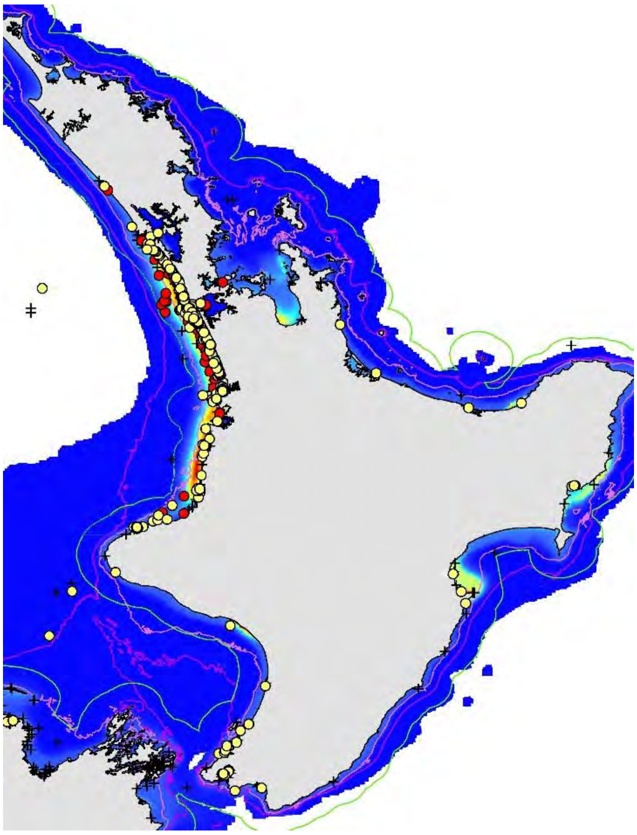

Figure 6.1: Locations of all reported public sightings of Hector's or Māui dolphins around the North Island. Yellow = validated summer sighting; Red = validated winter sighting. Black cross = un-validated sighting. Sightings locations are superimposed on outputs of a spatial habitat suitability model, in which predictions were based on water turbidity and the estimated prevalence of dolphin prey (from Roberts et al. 2019a; see section 6.2.9).

Historically, Māui dolphins have been sighted in three North Island harbours: Kaipara, Manukau, and Raglan (Slooten et al. 2005; Scali 2006); but harbour sightings are rare in recent decades (Rayment et al. 2011b, Derville et al. 2016). Passive acoustic monitoring via deployed hydrophones in these three harbours, in addition to Kawhia Harbour, revealed very occasional dolphin presence inside harbours near the harbour mouths (Rayment et al. 2011b, Wright & Treganza 2019). Distribution models fitted to public sightings data also predict very low densities inside harbours (Roberts et al. 2019a) as shown in Figure 6.3.

Numerous studies have reported an affinity for high-turbidity water and avoidance of clear water by Hector's and Māui dolphins (e.g., Abel 1971, Baker 1972, Baker 1978, Bräger & Bräger 2018, Bräger et al. 2003, Derville et al. 2016, Ferreira & Roberts 2003, Rayment et al. 2009a, Russell 1999, Rodda & Moore 2013, Weir and Sagnol 2015, Derville et al. 2016, Bräger & Bräger 2018; McGrath submitted). These observations are reflected in the outputs of quantitative habitat preference models fitted to boat-based sightings (e.g., Bräger et al. 2003, Derville et al. 2016, Miller 2015) and aerial survey sightings, as described by Roberts et al. (2019a). The dolphins' preference for turbid waters is also reported from direct behavioural observations in which dolphins following boats were typically observed to stop and turn back at the boundary between turbid and clear waters, without reference to depth or distance from shore (Russell 1999).

Dolphin distributions appear to shift further offshore during the winter, most likely associated with seasonal changes in the spatial extent of preferred turbid-water conditions and seasonal shifts in the distribution of their preferred prey (Miller 2015; Roberts et al. 2019a). Rayment et al. (2010) conducted aerial surveys of Hector's dolphins at Banks Peninsula from the coast to 15 nm offshore over three summers and winters. A significantly larger proportion of the population was sighted inside the 4 nm set net restriction zone in summer (mean = 81%; s.e. = 3.60) than in winter (mean = 44%; s.e. = 3.60). Similar seasonal differences in distribution were observed during the ECSI aerial surveys (MacKenzie & Clement 2014; Figure 6.2): in the Banks Peninsula (BP) stratum, 45% of the local population was observed inside the 4 nm set net exclusion zone in summer, compared with only 26% for the winter population. Similarly, in the Clifford Bay and Cloudy Bay (CCB) stratum, 47% of the local summer population and 14% of the local winter population were within the 4 nm set net fisheries exclusion zone (Miller 2015, Miller et al. 2013, MacKenzie & Clement 2014, Brough et al. 2019). Similar seasonal offshore movements were reported by Du Fresne & Mattlin (2009) and MacKenzie & Clement (2014).

These observations, including seasonal inshore-offshore movement patterns, are largely consistent with the predictions of Māui and Hector's dolphin spatial distribution models reflecting habitat preference functions fitted to Hector's dolphin aerial survey observations (Roberts et al. 2019a), reproduced below in Figure 6.5. That spatial predictions in the North Island are largely consistent with independent observations (i.e., public sightings), despite the preference functions having been parameterised using aerial survey data in the South Island, lends strength to the proposition that they reflect actual behavioural or habitat drivers of distribution, rather than incidental correlations. More systematic forms of model validation could include withholding a spatially contiguous portion of the data and using the remainder of the data to predict into areas in which data were withheld (e.g., using

ECSI data to predict distributions on the WCSI, and vice versa).

6.2.6 POPULATION SIZE

The population sizes of the different Hector's and Māui dolphin subpopulations have been estimated by formal surveys since the mid-1980s. Different survey methods were used through time, including: boat-based surveys (1985–2000), aerial surveys (since 2000) (section 6.2.6.2) and genetic mark recapture (since 2001) (section 6.2.6.1). Population estimates are summarised in Table 6.1. Aerial surveys (which are less sensitive to swell height) consistently produce higher population size estimates for this species compared with boat-based surveys in similar areas (MacKenzie & Clement 2014, Slooten et al. 2004). For example, the ECSI aerial survey in 2013 estimated 2–2.5 times as many dolphins within 4 nm of the coast comparedwith boat-based surveys in 1997–2000 (Dawson et al. 2004, MacKenzie & Clement 2014). Also note that the wide uncertainty around survey-based population size estimates (CV typically around 20%) hampers our ability to detect population changes, unless that change is very large (section 6.2.7). The most recent comprehensive abundance estimates for Hector's dolphins are from aerial surveys of the coastal waters (excluding harbours and enclosed bays), carried out separately for the east (ECSI), west (WCSI), and south (SCSI) coasts of the South Island (14 849 animals, CV 11%, 95% CI 11 923–18 492) (MacKenzie & Clement 2016). The most recent estimate of Māui dolphin abundance (63 dolphins aged 1+, 95% CL 57–75) is based on the 2015–2016 surveys of genetically identified individuals from the west coast North Island (WCNI) (Baker et al. 2016b). There were also two Hector's dolphins genetically identified

Table 6.1: Survey abundance estimates for Hector's and Māui dolphins by area and year. The results of the different surveys may not be directly comparable due to differences in survey methods. Studies are organised by coastal region: ECSI = East Coast South Island, NCSI = North Coast South Island, WCSI = West Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island. (Continued on next page)

Subpopulation	Survey region	Years of survey	Method	Abundance estimate (95% confidence interval)	Reference
WCNI	Kaipara Harbour to 10 nm south of Whanganui; out to 0.43 nm*	1985	Boat strip transect	134	Dawson & Slooten 1988
	Kaipara Harbour to 10 nm south of Whanganui; out to 0.43 nm*	1985	Re-analysis of Dawson & Slooten 1998	140 (46–280)	Martien et al. 1999
	Kaipara Harbour to New Plymouth; out to 800 m from shore	1998	Boat strip transect	80	Russell 1999
	Paraparaumu and North Cape; out to 10 nm	2001/02	Aerial transect	75 (48–130)	Ferreira & Roberts 2003
	Not stated	2003	Genetic capture recapture	69 (38–125)	Baker et al. 2013
	Maunganui Bluff to New Plymouth; out to 10 nm	2004	Aerial transect	111 (48–252)	Slooten et al. 2006
	Not stated	2006	Genetic capture recapture	59 (19–181)	Baker et al. 2013
	Baylys Beach to New Plymouth	2010–2011	Genetic capture recapture	55 (48–69)	Hamner et al. 2014b
	Kaipara Harbour to Mokau River, Taranaki	2015–2016	Genetic capture recapture	63 (57–75)	Baker et al. 2016b
Entire South Island	Out to 20 nm	2010–2015	Aerial line transects	14 849 (11 923–18 492)	MacKenzie & Clement 2016
	Out to 10 nm	1997–2000	Boat and aerial line transects	7270 (5303–9966)	Slooten et al. 2004; Dawson et al. 2004
	Out to 0.43 nm	1985	Boat, strip transects	3274	Dawson & Slooten 1988
WCSI	Farewell Spit to Milford Sound; out to 20 nm	2014/15	Aerial line transects	Summer: 5490 (3319–9079) Winter: 5802 (3879–8679)	MacKenzie & Clement 2016
	Farewell Spit to Milford Sound; out to 10 nm	2000–2001	Aerial line transects	5388 (3613–8034)	Slooten et al. 2004
ECSI	Kaikōura coast	2014–2015	Genetic capture recapture	480 (342–703)	Hamner et al. 2016
	Kaikōura coast	2013	Photo-ID, mark re-capture	304 (211–542)	Weir & Sagnol 2015
	Cloudy Bay	2011–2012	Genetic capture recapture	272 (236–323)	Hamner et al. 2013
	Cloudy Bay and Clifford Bay; out to at least 16 nm	2008–2009	Aerial line transects	Summer: 951 (573–1577) Winter: 315 (173–575) Spring: 188 (100–355)	Du Fresne & Mattlin 2009
	Banks Peninsula	1989–1997	Photo-ID, mark re-capture	1,119 (744–1,682)	Gormley et al. 2005
	Otago coast; out to 400m	2010–2011	Boat line transect	42 (19–92)	Turek et al. 2012
ECSI & NCSI ECSI & NCSI	Farewell Spit to Nugget Point; out to 20 nm	2012–2013	Re-analysis of Mackenzie & Clement 2014	Summer: 9728 (7001–13 517) Winter: 8208 (4888–13 785)	MacKenzie & Clement 2016

ECSI & NCSI	Farewell Spit to Nugget Point; out to 20 nm	2012–2013	Aerial line transect	Summer: 9130 (6342–13 144) Winter: 7456 (5224–10 641)	MacKenzie & Clement 2014
	Farewell Spit to Motunau; out to 20 nm	1998/99	Boat line transect	285 (137–590)	Clement et al. 2001
ECSI & SCSI	Long Point, Fiordland to Timaru; out to 20 nm	1998/99	Boat line transect	399 (279–570)	Du Fresne et al. 2001
ECSI, NCSI, & SCSI	Farewell Spit to Long Point; out to 20 nm	1997–2000	Boat line transect	1880 (1246–2843)	Dawson et al. 2004
SCSI	Long Point, Fiordland to Nugget Point; out to 20 nm	2018	Aerial line transects	332 (217–508)	MacKenzie & Clement 2019
	Long Point, Fiordland to Nugget Point; out to 20 nm	2010	Re-analysis of Clement et al. 2011	238 (113–503)	MacKenzie & Clement 2016
	Long Point, Fiordland to Nugget Point; out to 20 nm	2010	Aerial line transects	628 (301–1311)	Clement et al. 2011
	Te Waewae Bay	2005/06	Photo-ID mark-recapture	Summer: 580 (480–700) Winter: 380 (300–500)	Rodda 2014
	Te Waewae Bay	2004/05	Photo-ID mark-recapture	Summer: 403 (269–602) Autumn: 251 (183–343)	Green et al. 2007
	Porpoise Bay	1996–1997	Photo-ID mark-recapture	48 (44–55)	Bejder and Dawson 2001

* The 1985 estimates by Dawson & Slooten (1988) were adjusted upward by a factor of five to account for the assumed proportion of the population occurring within sight of the coastal transect (out to 800 m) based on the proportion of all sightings in this zone along 5 nm transects off the South Island.

during the 2015–2016 Māui dolphin surveys. The research programmes producing these estimates are described in greater detail below.

There are a few recent genetic and photo-identification mark-recapture estimates for local Hector's dolphin populations that are valuable for understanding local population dynamics and areas of conservation concern. These estimates are more accurate when dolphins have small ranges with limited offshore dispersal allowing greater chance of sampling most of the population e.g., Porpoise Bay and Kaikōura (Bejder & Dawson 2001, Weir & Sagnol 2015, Hamner et al. 2016), but are less robust when the populations range further offshore and become less accessible e.g., Cloudy Bay (Hamner et al. 2017) as highlighted by comparisons with aerial surveys with greater coverage (Du Fresne & Mattlin 2009, MacKenzie & Clement 2014, MacKenzie & Clement 2016).

Differences in the offshore extent of survey sampling effort may account for discrepancies between current aerial survey based abundance estimates and earlier population estimates from boat-based transect surveys (e.g., Dawson & Slooten 1988, Dawson et al. 2004, Slooten et al. 2004, 2006) or from photo-ID mark-recapture studies focused on particular local populations of Hector's dolphins (Gormley et al. 2005, Turek et al. 2013).

6.2.6.1 MĀUI DOLPHIN GENETIC MARK-RECAPTURE CENSUS

Beginning in 2010–11, Māui dolphin populations have been monitored with a boat-based census every 5 years, using genetic mark-recapture methods. The 2015–16 census estimated an abundance of N = 63 animals (95% CL 57–75) for the population of Māui dolphins at least one year old (Baker et al. 2016b). These estimates are comparable to, but slightly larger than the previous estimate of N = 55 (95% CL 48–69) based on comparable genotype surveys in 2010–11 (Hamner et al. 2012b). The longer time series and higher resolution mark-recapture data informed updated demographic models and improved estimation of survival rate and population trend than presented previously (Roberts et al 2019b, Cooke et al. 2018, 2019). A repeat genetic mark-recapture census using the same method is currently in progress (in 2020–21) by the University of Auckland and Oregon State University, funded jointly by DOC and Fisheries New Zealand.

6.2.6.2 HECTOR'S DOLPHIN AERIAL SURVEY PROGRAMME

Beginning in 2010, a series of aerial surveys were conducted under MPI contracts to estimate the abundance and characterise the spatial distributions of the SCSI, ECSI, and WCSI Hector's dolphin subpopulations (Clement et al. 2011, MacKenzie et al. 2012, MacKenzie & Clement 2014, 2016; MacKenzie & Clement 2019); see Figure 6.2.

The initial SCSI aerial survey programme involved two aerial surveys undertaken during March 2010 and August 2010 between Puysegur Point and Nugget Point and out to the 100 m depth contour (Clement et al. 2011). MacKenzie & Clement (2016) reanalysed the SCSI survey data from 2014 and produced an annual average population estimate for the SCSI of 238 (s.e. 94; 95% c.i. 113–503) based on revised figures for availability. In early 2018 a repeat survey successfully obtained a lower CV in the estimate of population size by adopting higher sampling intensity in the nearshore strata and in areas of high dolphin density in Te Waewae Bay. The updated SCSI population size from MacKenzie & Clement (2019) is 332 animals (95% c.i. 217–508).

The ECSI aerial survey programme involved an initial design phase (MacKenzie et al. 2012) followed by two aerial surveys conducted over summer 2012–13 and winter 2013 between Farewell Spit and Nugget Point and offshore to 20 nm (covering about 42 677 km^2; MacKenzie & Clement 2014). A total of 354 dolphin groups were sighted in the summer, along 7156 km of transect lines, and 328 dolphin groups were sighted in the winter, along 7276 km of transect lines. MacKenzie & Clement (2016) reanalysed the ECSI survey data from 2014 and produced an annual average estimate for the ECSI of 8968 animals (s.e. 1377; 95% c.i. 6649–12 096), based on revised figures for availability. Note these estimates do not include harbours and bays, which were outside the designated survey strata.

The WCSI aerial survey programme involved two separate aerial surveys in summer 2014–15 and winter 2015 (MacKenzie & Clement 2016). The population within the WCSI survey area (about 26 333 km^2 between Farewell Spit and Milford Sound) was estimated at 5490 animals (CV = 26%; 95% c.i. 3319–9079) in summer and 5802 (CV = 21%; 95% c.i. 3879–8679) in winter. These estimates were obtained by averaging the four sets of results for each season; from two different datasets using different truncation distances and two methods of estimating availability (dive cycle and circle-backs). These estimates are very similar to the previous 2000–01 WCSI estimate of 5388 Hector's dolphins by Slooten et al. (2004) (CV = 21%; 95% c.i. 3613–8034), even after accounting for differences in offshore survey areas (MacKenzie & Clement 2016).

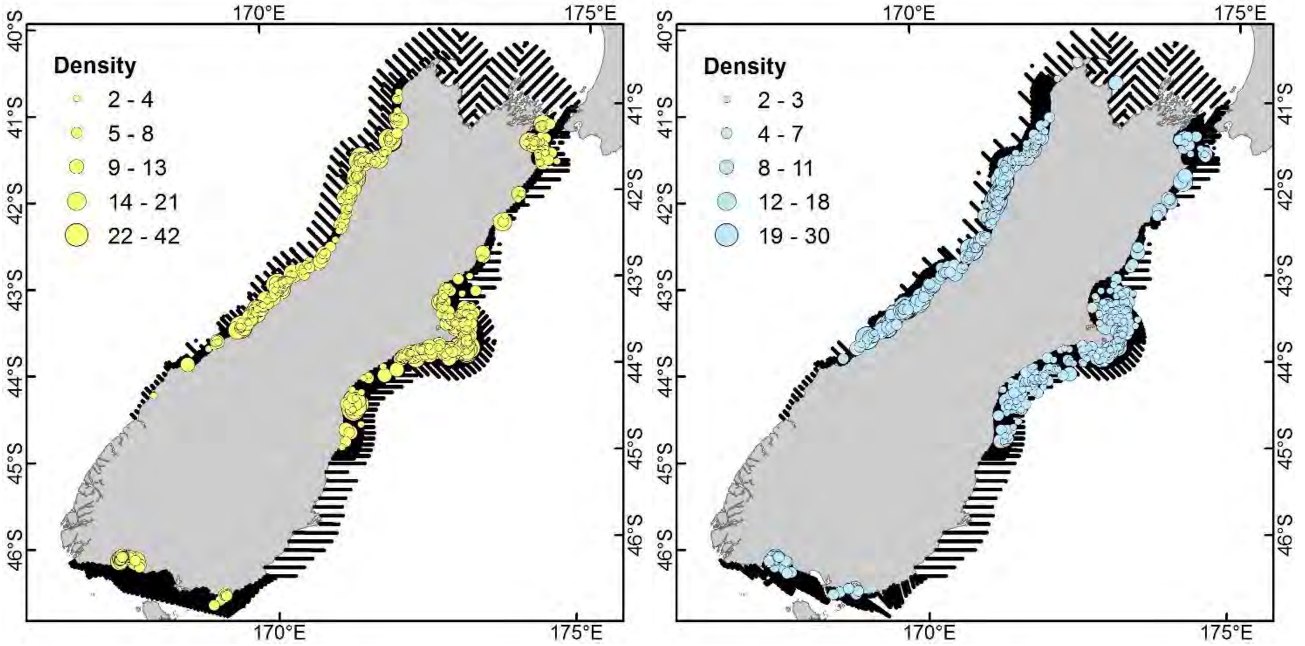

Figure 6.2: Hector's dolphin summer (left) and winter (right) sightings from the three separate abundance surveys: west coast (WCSI) completed in 2015, east and north coast (ECSI) completed in 2013, and south coast (SCSI) completed in 2010. Black lines represent the paths of aerial survey transects. Reproduced from Roberts et al. (2019a) using the outputs of MacKenzie & Clement (2016). Note that the SCSI survey was repeated in early 2018 (Mackenzie & Clement 2019).

Arising from the reanalysis of the ECSI and SCSI survey data, MacKenzie & Clement (2016) estimated the total Hector's dolphin population in coastal areas around the full South Island (excluding sounds and harbours) at 14 849 animals (CV = 11%; 95% c.i. 11 923–18 492). This estimate is approximately double the previous estimate from surveys conducted in the late 1990s–early 2000s (7300; 95% c.i. 5303–9966) (Slooten et al. 2004), with the difference primarily due to the substantial number of dolphins sighted in offshore waters at distances greater than had been extensively surveyed previously, especially in ECSI (MacKenzie & Clement 2016).

Following discussion in the International Whaling Commission (IWC) Sub-committee on Small Cetaceans, in 2015 the subcommittee agreed to an inter-sessional review of the methods used in these abundance estimates (International Whaling Commission 2016a). A formal process was agreed whereby an Inter-sessional Expert Group (IEG) reviewed the abundance methodology and estimates produced by MacKenzie & Clement (2014, 2016) (International Whaling Commission 2016b).

The IEG recognised that this study accounted for many difficulties that also affect other small cetacean abundance estimation studies using aerial surveys. It commended the ambitious and often innovative work undertaken by the authors to attempt to deal with all of those issues. After a thorough review of the survey design, analyses, and results, the IEG endorsed the abundance estimates and concluded that the estimates accurately reflected the data, were derived from appropriate data collection and analysis methods, and represented the most current abundance estimate for Hector's dolphins around the South Island (such that it would be reasonable to use them to inform a management plan). The IEG also considered this study to be a step forward in the development of survey methodology more generally (International Whaling Commission 2016b).

In 2019 the aerial survey observations were used to parameterise spatial habitat models to estimate the seasonal spatial density of Hector's and Māui dolphins, a critical input to the spatial multi-threat risk assessment used to inform the update of the TMP (Roberts et al. 2019a).

6.2.7 CHANGES IN POPULATION SIZE

Change in population size can be summarised in terms of the direction of population change (i.e., increasing or decreasing), or the annual rate of population change (λ) where $\lambda > 1$ indicates population increase, and $\lambda < 1$ indicates decline, which is used as a basis for the current domestic and international threat classification status rankings for both sub-species (Baker et al. 2019, Reeves et al. 2020).

The use of survey-based population size estimates for estimating population growth rate is hampered by changes in survey methods through time, and by the low precision of estimates (Table 6.1). A population model fitted to estimates of Māui dolphin population size estimated a slightly declining population size with reasonably high precision (λ = 0.98, 95% credible interval = 0.96–1.00) (Roberts et al. 2019b). This assessment found that Māui dolphin population change was primarily driven by female survival, which was estimated to be around 5% higher than for males.

The rate of population change can also be inferred indirectly with population simulations using prior distributions of all required demographic rates, i.e., survival and reproductive rate at age. A demographic assessment fitted to photo-ID observations of Hector's dolphins inside the Banks Peninsula Marine Mammal Sanctuary (BPMMS) found that their population trajectory is likely to be stable since the establishment of the sanctuary (λ = 1·00, 95% CI = 0·93–1·05) (Gormley et al. 2012). More precise estimates of calving interval were identified as the best way of reducing uncertainty in population growth using this method (Gormley 2009).

Population trajectory, and population status relative to historical values, have also been estimated indirectly using logistic population growth models that incorporate estimates of historical commercial fishery deaths (Martien et al. 1999, Burkhart & Slooten 2003, Slooten 2007, Slooten & Dawson 2010). The latest published analysis estimated the population size of Hector's dolphin in 2009 to be 27% of the 1970 estimate, and that the Māui dolphin was the most depleted subpopulation (Slooten & Dawson 2010). However, these assessments used population abundance estimates for the ECSI population that were later shown to underestimate actual population size by a factor of 3–5 relative to comprehensive aerial survey derived estimates of comparable areas. As a consequence these analyses have greatly over-estimated the vulnerability of dolphins to capture in commercial set nets (and, hence, historical fishery-related deaths as a proportion of total population) and will therefore estimate a lower status relative to un-impacted levels than would be obtained with comparable models using updated population estimates (J Roberts unpublished data). The assessment by Slooten & Dawson (2010) (and earlier iterations) is also inconsistent with the outputs of the most recent spatial risk model (Roberts et al. 2019a), which found that the median estimates of commercial fisheries deaths since 1992–93 would be insufficient to prevent population recovery to 90% of un-impacted levels, for both Hector's and Māui dolphins. However, current population trend and status depend also on assumptions about non-fishery threats and will be affected by assumptions regarding historical depletion, including from recreational fisheries and from commercial fishing prior to the establishment of the Quota Management System (QMS), when effort levels were higher and less regulated. For example Lallemand et al.

(2008) reported that set net fishing effort over large portions of the ECSI declined by more than 80% following the establishment of the QMS in 1986, but spatially precise effort location data are not available for this period.

Ongoing research under Fisheries New Zealand project PRO2019-11 will estimate the spatial distribution of historical set net fishing effort in the period prior to 1992–93, for which effort data were unavailable to the spatial risk model described by Roberts et al. (2019a). The historically reconstructed effort data can then be used to estimate population status for Hector's dolphin subpopulations, by applying the SEFRA modelling approach to historical fishing effort patterns from the time when monofilament nets were first employed.

6.2.8 SUBPOPULATION STRUCTURE

For purposes of the spatial risk assessment used to inform the update of the Hector's-Māui dolphin Threat Management Plan (Roberts et al. 2019a), the South Island Hector's dolphin population is divided into four subpopulations, corresponding to the east coast, south coast, west coast, and north coast, as shown in Fig 6.3. The existence of a genetically distinct NCSI population is suggested based on genetic evidence (Baker et al. 2017), but requires a higher sample size to be confirmed.

In the risk assessment by Roberts et al. (2019a), the Māui dolphin subpopulation is presumed to occupy the area from Taranaki in the south to Cape Reinga in the north. The area from Taranaki southward to the Kāpiti Coast is considered to be a potential habitat for dolphins expanding their range, and/or a transition zone for dolphins moving between the WCNI subpopulation and the South Island. The remainder of the North Island is presumed to have no current resident population, but the fisheries risk assessment can still be used to evaluate what the risk would be to any dolphins occupying preferred habitats in those areas (see below).

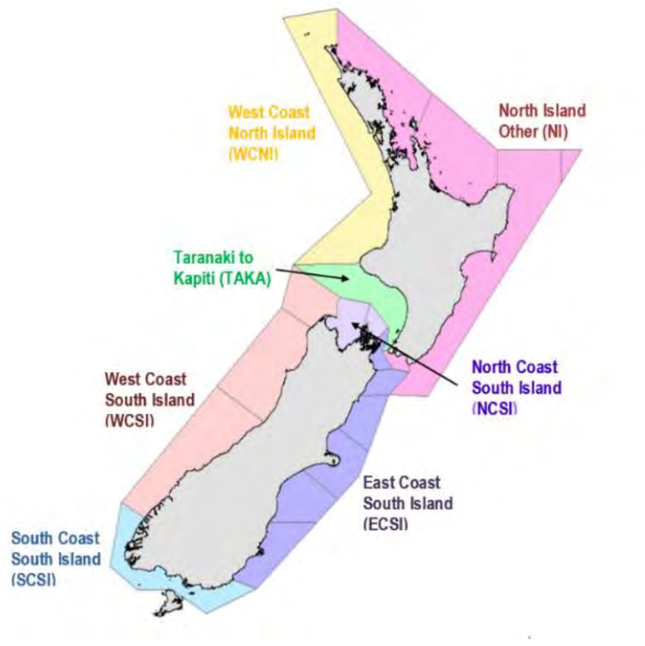

Figure 6.3: Boundaries designating Hector's and Māui subpopulations for purposes of spatial risk assessment (Robert et al. 2019a). The number of animals in each subpopulation was estimated based on aerial surveys (for Hector's dolphins) or genetic census (for Māui dolphins). Subpopulation zones with no known year round population ('Taranaki to Kāpiti' and 'other North Island') were assigned arbitrary low numbers of dolphins so that potential risk to transient or future populations could be estimated in the risk assessment.

6.2.9 SPATIAL DOLPHIN DENSITY ESTIMATION

The seasonal (summer and winter) spatial abundance of Hector's and Māui dolphins was estimated as part of the spatial risk assessment of threats to Hectors and Māui dolphins (Roberts et al. 2019a) (see section 6.4). The primary spatial abundance information for predicting the coastal abundance of Hector's and Māui dolphins came from a series of summer and winter aerial line-transect surveys for estimating the abundance and spatial distribution of Hector's dolphins, conducted around the South Island of New Zealand between 2010 and 2015 (MacKenzie & Clement 2014, 2016). Habitat models were fitted to aerial survey observations (Figure 6.2) related to candidate spatial habitat layers, including physical variables (e.g., depth or turbidity) and biotic variables (e.g., the modelled prevalence of key prey species).

The habitat model used to estimate the spatial density of Hector's and Māui dolphins included satellite derived seasonal turbidity and trawl survey derived prevalence of ahuru (*Auchenoceros punctatus*) (a key prey species, Miller et al. 2013) as predictors. The inclusion of turbidity as the primary model term is consistent with the assessments by

Bräger (1998), Torres et al. (2013), and Stephenson et al. (2020), who all found water turbidity to be a strong predictor of Hector’s and Māui dolphin presence and abundance.

The aerial survey parameterised model could not be used to estimate spatial density inside the harbours of the WCNI, which were much more turbid than the South Island regions where the model was fitted, and where physical features such as sandbars or tidal mudflats may affect dolphin distributions at scales smaller than the habitat model can predict. For these reasons, the relative spatial density of Māui dolphins in WCNI harbours was estimated using a separate habitat preference model fitted to validated public sightings data and an aerial survey of spatially resolved boat density as a proxy for spatial public sighting ‘effort’, related to locational/habitat based variables. This model also found turbidity to be the strongest predictor of sightings density and estimated a very low relative abundance inside harbours, where recreational boat density is high, but validated dolphin sightings are rare (Figure 6.4).

Roberts et al. (2019a) give a full description of the methods, data, and assumptions underlying the spatial dolphin density estimation. Section 6.5.6 below identifies particular locations in which the spatial predictions may be more uncertain, with implications for risk assessment outputs.

The final spatial dolphin density estimate was obtained by combining the habitat preference based estimate in the South Island in coastal waters of the North Island with the public sightings based estimate in WCNI harbours. Spatial abundance was rescaled for each of the subpopulation areas defined in Figure 6.3, using population size estimates from aerial surveys (for Hector’s dolphins) or genetic mark-recapture census (for Māui dolphins) as described above. The composite spatial abundance predictions for Hector’s and Māui dolphins are shown in Figure 6.4. The resulting spatial density prediction achieved a high degree of correspondence with spatial patterns of public sightings and commercial fishery observer sightings of both Hector’s and Māui dolphins (see Figure 6.5 and appendix 7 of Roberts et al. 2019a), indicating that the habitat model accurately represented the true habitat requirements of both sub-species, despite being fitted primarily to Hector’s dolphin observations.

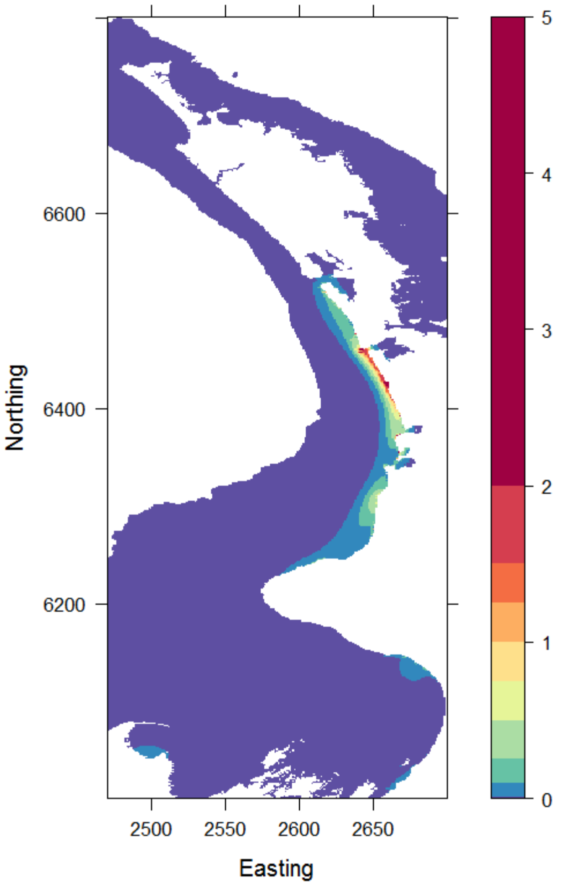

Figure 6.4: Estimated spatial density of Hector’s and Māui dolphins off the west coast of the North Island from a predictive model fitted to boat-based validated public sightings. From Roberts et al. (2019a).

6.2.10 THREATS TO HECTOR’S AND MĀUI DOLPHINS

6.2.10.1 FISHERIES BYCATCH

Fisheries bycatch, particularly in recreational and commercial set net fisheries and to a lesser extent in commercial trawls, is a known threat to Hector’s and Māui dolphins. Hector’s and Māui dolphin bycatch is thought to have increased rapidly with the widespread adoption of monofilament set nets in the 1970s and 1980s and declined thereafter (e.g., Dawson 1991, Dawson and Slooten 1993, Martien et al. 1999, Duignan et al. 2003, Currey et al. 2012, Abraham et al. 2017). Commercial and recreational set net fishing remains a threat to dolphin populations in locations where the spatial distribution of dolphins (e.g., Figure 6.5) overlaps the spatial distribution of set net and trawl fishing effort.

Observations and records of fisheries bycatch are summarised in section 6.3. These observations may provide valuable information about the nature of fisheries captures, but fisher-reported bycatch rates in isolation are not a reliable means of estimating total commercial fisheries deaths, because it is likely that not all fishers will voluntarily report all bycatch events. Government fisheries observers are deployed on a proportion of commercial fishing vessels, but historically observer coverage has been low in inshore fisheries except in locations of particular concern (such as the WCNI set net fishery since 2012, due to the urgent conservation status of Māui dolphins). Where observer coverage is low, it may not be representative of total fishing effort in space and time, so it is necessary to correct for the effects of potential coverage bias in the estimation of fisheries risk. The SEFRA method (Chapter 3) is designed to achieve this; the extent and magnitude of the fisheries risk to dolphin subpopulations has been estimated using this approach, described in section 6.4.

6.2.10.2 DISEASE

The awareness of disease as a potential serious threat to Hector's and Māui dolphins has emerged only recently, since the last update of the TMP in 2012 (e.g., see Currey et al. 2012).

There have been 5 Māui dolphin and 50 Hector's dolphin necropsies undertaken by veterinary pathologists at Massey University. Analysis of samples collected from beach-cast or entangled/bycaught Hector's and Māui dolphins revealed that disease was a major cause of death, followed by maternal separation (i.e., when a calf is separated from its mother – this is a primary cause of calf mortality), and then bycatch (Roe et al. 2013). Infectious diseases, including brucellosis, pneumonia, toxoplasmosis, and tuberculosis, were identified as the cause of death for 53% (n = 23/43) of dolphins where cause of death could be determined

The main disease of concern for Hector's and Māui dolphins is toxoplasmosis. Toxoplasmosis is a disease caused by infection with a single-celled parasite *Toxoplasma gondii*, which is capable of infecting all bird and mammal species and for which the domestic house cat (*Felis catus*), including owned, stray, and feral cats, is the only definitive host in New Zealand. It is thought that toxoplasmosis oocysts in cat faeces are transmitted to the ocean via waterways and accumulate up the marine food chain through filter feeding animals (such as shellfish or small pelagic fish that filter plankton) and then to dolphins ingesting infected prey (Massie et al. 2010).

Of the 31 non-fishery related deaths of non-calf dolphins, recorded by the Massey University SoVS Pathology Database between 2007 and 2018, nine died from toxoplasmosis of which seven (78%) were females (ECSI = 5; WCSI = 2; WCNI = 2; see also Table 6.4 below). Based on identification of toxoplasma in the tissues of bycaught and beach-cast dolphins, the majority (61%) were found to be infected (Roe et al. 2013). Factors influencing whether or not an infection causes disease (and/or becomes fatal) are poorly understood, but are related to a number of factors associated with the immune response of the host (reviewed by Roberts et al. in review). Also, toxoplasma virulence is known to be influenced by host and parasite genetics, e.g., some genetic strains of toxoplasma are more lethal than others for certain host species (for example, California sea otters (Miller et al. 2004; Kreuder et al. 2003; Conrad et al. 2005; Shapiro et al. 2019)). It may be that nutritional stress or other factors influencing immune system function can also cause dormant toxoplasma infections to become active (see below). Worldwide, toxoplasmosis is recognised as a threat to a wide range of marine and terrestrial wildlife species, especially in parts of Australia and the Pacific region (Roberts et al. in review, Barbieri et al. 2016; Work et al. 2000). Notably, these are often locations where the native fauna did not evolve in the presence of cats, as in New Zealand. Further research to understand the effects of toxoplasmosis on Māui and Hector's dolphins is being planned between agencies, led by the Department of Conservation.

Of other diseases identified in necropsy results, pneumonia was the second most common non-fishery related cause of death for non-calf dolphins (13%, n = 4/??), followed by brucellosis (6%, n =2/31, both females). Brucellosis is associated with deaths and foetal loss in mammals, and an analysis of Hector's and Māui dolphins revealed 26% (n = 7/27) tested positive for Brucella (Buckle et al. 2017). Like toxoplasmosis, dolphins can carry this disease without it causing death, but, in addition to the two female Hector's dolphin deaths from brucellosis, there was also a neonate Māui dolphin death. The form of Brucella that killed two of the dolphins had the greatest similarity to *Brucella pinnipedialis* – typically reported in seals, but it is likely that there is a Pacific form of marine Brucella not yet fully described. Since that study, in 2018, a female Māui dolphin died from septicaemia after her near-term foetus died from brucellosis and she was unable to birth the stillborn calf (Dr Wendi Roe, Massey University, unpublished data).

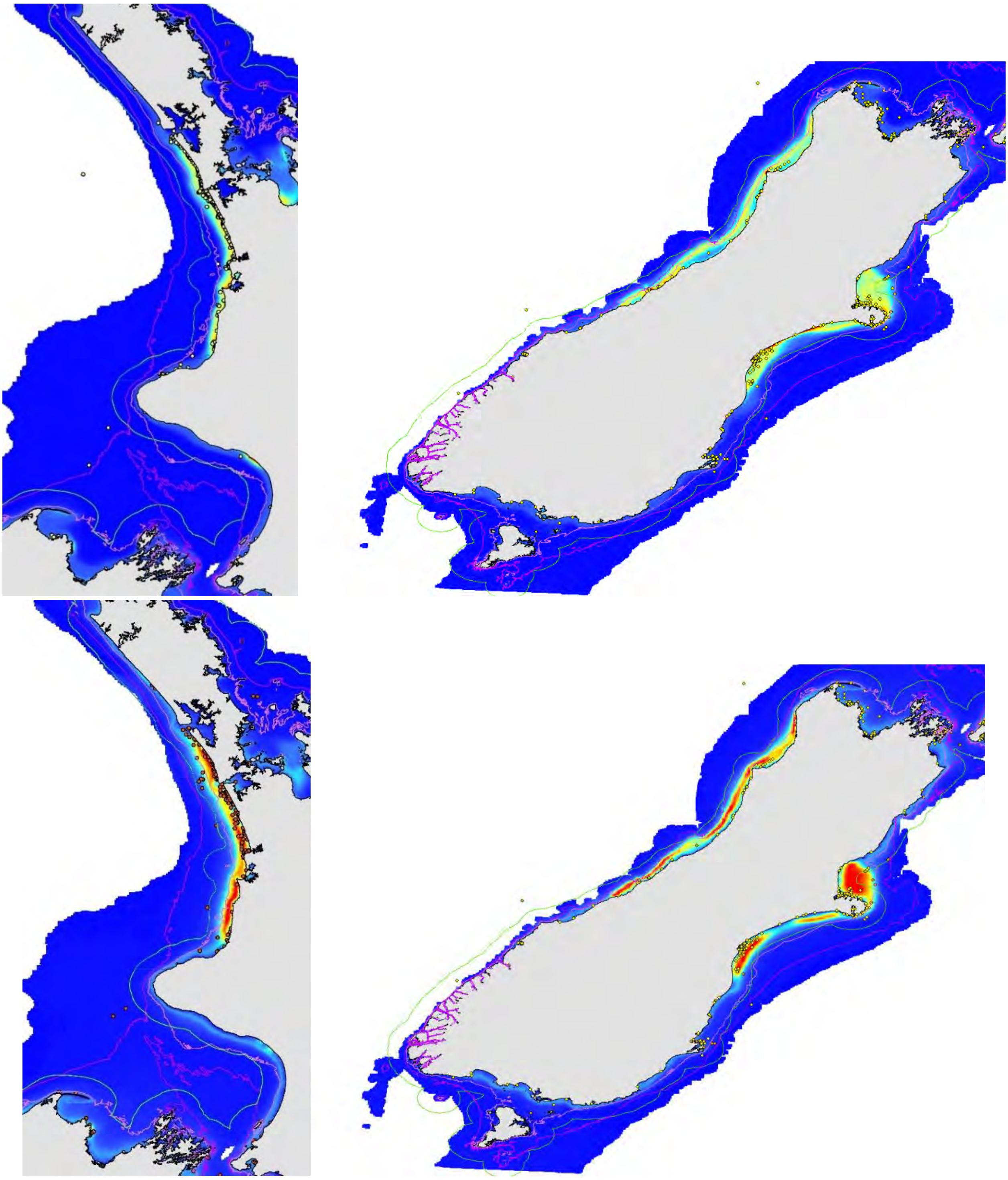

Figure 6.5: Estimated spatial density of Hector's and Māui dolphins in summer (top) and winter (bottom) used in the spatial multi-threat risk assessment by Roberts et al. (2019a). Also shown are the 50 m and 100 m depth contours (in purple), the Territorial Sea boundary (in green), and the locations of validated public sightings.[The DOC sightings confirmation process is described at https://www.doc.govt.nz/globalassets/documents/conservation/native-animals/marine-mammals/mauis-validation-system.pdf.]

'Normal' or background infection levels of these diseases in living dolphins is poorly known, as blood and/or tissue samples are required to test for *Toxoplasma gondii* and Brucella infection. Although disease is normal within the marine environment, the presence of a specific cat borne disease is of concern, as is the fact that both of these diseases appear to disproportionately affect female dolphins, and that 2 of 9 toxoplasmosis deaths and 2 of 4 Brucella-attributable deaths were of Māui rather than Hector's dolphins, despite their much smaller population size.

Reflecting the results of the spatial risk assessment by Roberts et al. (2019a) below, and the Māui dolphin population model projections of Cooke et al. (2019) below, toxoplasmosis in particular has been identified as a major priority for further research and for conservation action. The Department of Conservation has prepared a Toxoplasmosis Action Plan outlining priorities to guide this work.

6.2.10.3 TROPHIC AND/OR CLIMATIC EFFECTS

To date (June 2020), there has been no formal assessment of the potential indirect effects of fishing on Hector's and Māui dolphins, e.g., via trophic competition. All of the dolphins' main prey species (e.g., Miller et al. 2013) are either targeted or are regular bycatch of commercial fisheries, but the average size of the fish appearing as prey in dolphin stomachs is generally smaller than adult size classes that are routinely targeted or selected by fishing gear (Weir 2018). A rigorous evaluation of potential trophic effects of fishing on prey availability for dolphins would likely require spatially explicit estimates of fisheries extractions at scales relevant to individual dolphin movements and existing closed areas, considering both the size selectivity of fisheries removals and the potential for recruitment overfishing.

Climate change and/or climatic variability is likely to affect dolphins, in particular because inter-annual changes in sea temperature are likely to affect the distribution or availability of prey species or influence terrestrial run-off (Shears & Bowen 2017). It is not known how changes in spatial patterns of water turbidity or prey distributions will affect Hector's and Māui dolphins. They have the ability to disperse to other areas, and they have a varied diet so there may be shifts in habitat use and range, but the species' history of small ranges and high site fidelity may impose a behavioural limit on their ability to move, which ultimately may affect their reproductive success. Effects of climate change are likely to be greater for subpopulations with small home ranges.

Roberts et al. (2019a) estimated that for all suitable prey species, the total abundance of available prey species was many times lower in WCNI relative to suitable Hector's dolphin habitats around the South Island. Improved prey abundance modelling considering also the size distribution of the available prey may provide additional insight of the extent to which Māui dolphins may confront a shortage of suitable prey relative to Hector's dolphins. Weir (2018) notes that their income breeding strategy and high energetic demands during pregnancy may make Hector's and Māui dolphins particularly vulnerable to factors that reduce or temporarily disrupt their regular access to preferred prey, potentially affecting their reproductive success or susceptibility to disease.

6.2.10.4 OTHER ANTHROPOGENIC THREATS

Underwater noise can cause physical injury and disturbance to dolphins. Noise exposure can be estimated using underwater sound propagation modelling (e.g. McPherson et al. 2019), but the actual effects of different levels and types of sound on marine mammals are poorly understood (Forney et al. 2017, Leunissen & Dawson 2018, Lucke et al. 2019). Disturbance may be short-term and/or episodic (e.g., noise from seismic surveys, pile driving, drilling or mining, research activities, or vessel traffic), but may have a cumulative impact and/or a habitat displacement effect with consistent or repeated exposure.

Boat strikes are not thought to be a major cause of death but there has been one confirmed death, a Hector's dolphin calf in Akaroa harbour in 1999 (Stone & Yoshinaga 2000, DOC 2020a). Commercial dolphin-watch tourism may have negative effects on cetaceans (e.g., Martinez et al. 2012); these activities are regulated by the Department of Conservation.

Most marine pollutants have sub-lethal effects that may be difficult to detect. By global standards, the levels of pollutants such as DDT and PCBs in New Zealand waters are low, but their coastal habitat and preferred fish prey may make Hector's and Māui dolphins more exposed to accumulating pollutants than offshore species (Stockin et al. 2010; Jones et al. 1996, 1999) and Māui dolphins in particular may be especially vulnerable, because their spatial distribution is largely confined to turbid waters affected by freshwater river plumes that are highly contaminated (Hunt & Jones 2020).

6.2.10.5 NATURAL CAUSES OF DEATH

Hector's and Māui dolphins are vulnerable to predation by sharks and killer whales. Most predation events in New Zealand are attributed to seven-gill sharks or white sharks (Cawthorn 1988) but other large sharks may also prey upon these small dolphins (Heithaus 2006). Because these are naturally occurring events they are not managed as 'threats' but the spatial distribution of seven-gill and white shark populations is non-uniform, so understanding the level of risk from shark predation and patterns of overlap between sharks and dolphins helps us to understand cumulative threats to different dolphin subpopulations.

A major natural cause of death for Hector's and Māui dolphin calves is maternal separation (i.e., when a dependent calf is separated from its mother). As a cause of death in necropsied individuals, it is second only to disease (Roe et al. 2013); the rate at which this occurs is possibly exacerbated by extreme weather conditions (DOC & MFish 2007, MPI & DOC 2012).

6.2.11 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Threat classification is an established approach for identifying species at risk of extinction (IUCN 2013). The risk of extinction for Hector's and Māui dolphin has been assessed under two threat classification systems: the New Zealand Threat Classification System (Townsend et al. 2008) and the International Union for the Conservation of Nature (IUCN) Red List of Threatened Species (IUCN 2013).

The IUCN classifies Māui dolphin as Critically Endangered under criteria A4c,d and C2a(ii)[5] due to an ongoing or projected decline of greater than 80% over three generations, and there being fewer than 250 mature individuals remaining (Reeves et al. 2020). Hector's dolphin is classified by the IUCN as Endangered under criterion A4d[6] due to an ongoing or projected decline of greater than 50% over three generations (Reeves et al. 2020).

Under the New Zealand Threat Classification System (Baker et al. 2019), Māui dolphin is classified as Nationally Critical, the most threatened status, under criterion A(1), with the qualifier Conservation Dependent (CD)[7]. Hector's dolphin is classified as Nationally Vulnerable under criterion D(1/1), with the qualifier Conservation Dependent (CD)[8].

6.3 FISHERIES INTERACTIONS

6.3.1 DATA FROM RECOVERED CARCASSES AND FISHER-REPORTED CAPTURES

Hector's and Māui dolphins have been caught in inshore commercial and recreational set net fisheries and in inshore trawl fisheries since at least 1973 (DOC 2020a; Baker 1978). Beach-cast carcasses are frequently reported by members of the public; floating carcasses may be reported by fishers or other boaters; fishers are also required to report incidental captures. The Department of Conservation maintains a Hector's and Māui dolphin incident database in which all such deaths are recorded including the cause of death where this can be determined (DOC 2020a[9]). A summary of known, probable, or possible fisheries deaths from this database is shown in Table 6.2. Incidental fisheries mortalities have been documented throughout the species range but the greatest number of reports are from the east coast South Island.

Nineteen individual Hector's dolphins were reported caught in trawl fisheries between 1973 and 2008, with seven since 2008, in 13 separate capture incidents reported

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[5] A taxon is listed as 'Critically Endangered' if it is considered to be facing an extremely high risk of extinction in the wild. A4c,d refers to a reduction in population size (A), based on an observed, estimated, inferred, projected or suspected reduction of ≥ 80% over any 10-year or three-generation period (whichever is longer up to a maximum of 100 years (3)); with the reduction being based on a decline in area of occupancy, extent of occurrence and/or quality of habitat (c); or actual or potential levels of exploitation (d; IUCN 2010). C2a(ii) refers to a population size estimated to number fewer than 250 mature individuals (C); with a continuing decline, observed, projected, or inferred, in numbers of mature individuals (2); and a population structure (a) with at least 90% of mature individuals in one subpopulation (ii; IUCN 2013).

[6] A taxon is listed as 'Endangered' if it is considered to be facing a very high risk of extinction in the wild. A4d refers to a reduction in population size (A), based on an observed, estimated, inferred, projected or suspected reduction of ≥ 80% over any 10-year or three-generation period (whichever is longer up to a maximum of 100 years (3)); with the reduction being based on actual or potential levels of exploitation (IUCN 2013).

[7] A taxon is listed as 'Nationally Critical' under criterion A(1) when evidence indicates that there are fewer than 250 mature individuals, regardless of population trend and regardless of whether the population size is natural or unnatural (Townsend et al. 2008).

[8] A taxon is 'Nationally Vulnerable' under criterion D (1/1)when evidence indicates that the total population size is 5,000-20,000 mature individuals and there is an ongoing or predicted population decline of 30-70% over three generations, (Townsend et al. 2008).

[9] https://www.doc.govt.nz/our-work/hectors-and-maui-dolphin-incident-database/

by fishers (Table 6.2; DOC 2020a). Hector's dolphin captures in trawl nets include an individual caught in a trawl targeting red cod in Statistical Area 022 in 1997–98 (Starr & Langley 2000), the capture of three Hector's dolphins in a trawl in Cloudy Bay in 2006 (DOC & MFish 2007), and the capture of three dolphins in each of two separate inshore trawl events north of Banks Peninsula, in December 2018 and February 2019. Further investigation of the trawl gear configurations and vessel characteristics of fishing events in which dolphins were captured may prove useful to evaluate the extent to which gear design may affect dolphin catchability.

There is evidence of frequent bycatch of Hector's dolphins in set nets at Banks Peninsula extending back to at least the mid-1970s (Dawson 1991). Interviews with commercial fishers, voluntary reports by recreational fishers, and carcass retrieval indicated at least 230 deaths in set nets between 1984 and 1988 (Dawson 1991). Two hundred of these were reported by commercial fishers, who frequently supplied carcasses for dissection. A further 24 mortalities were reported by or attributed to amateur set net fishers. Six net-marked carcasses were recorded as "unknown net". The highest number of Hector's dolphin bycatch deaths reported annually by Dawson (1991) was 95 animals, in the 1985/86 season. Total deaths declined in the following two years (to 44 and 29 deaths, respectively), perhaps related to declining total effort levels coincident with the establishment of the QMS, as reported by Lallemand et al. (2008).

The DOC incident database records at least 45 Hector's or Māui dolphins were caught in commercial set nets from 1921 to 2008, and 12 since 2008. In recreational set nets, 21 confirmed deaths were recorded for 1921–2008 and 5 since 2008. Note however that a number of beach-cast carcasses were attributed to 'unknown set nets' or 'unknown nets' during these time periods; these will mostly reflect beach-cast carcasses with net marks, that is, animals that may have drowned in either recreational or commercial nets. Incidental captures have most frequently occurred in commercial set nets targeting rig (*Mustelus lenticulatus*), elephant fish (*Callorhynchus milli*), and school shark (*Galeorhinus australis*) (Dawson 1991, Baird & Bradford 2000), and in recreational nets set for flounder (*Rhomboselea sp*.) and moki (*Latridopsis ciliaris*) (Dawson 1991).

There have been four known incidents of Hector's dolphins becoming entangled in buoy lines of pots set for crayfish (*Jasus edwardsii*), all from Kaikōura (Dawson 1991; DOC & MFish 2007, DOC 2020a).

Numbers of dolphin deaths recorded in the DOC incident database are not representative of total fisheries bycatch rates. Carcasses may not be reported by fishers, may not wash ashore, may not be recovered, or may not show evidence of interaction with fishing gear (Slooten 2013). Spatial and seasonal detection bias will affect the probability that carcasses will be reported, with carcasses more likely to be reported in summer, in locations where fishing occurs closer to shore, and closer to major population centres and thoroughfares.

The information in the incident database (Table 6.2) provides only a biased indication of incidental captures. It is clear from this information, however, that incidental captures may occur in all areas where the distribution of Hector's and Māui dolphins overlaps with the distribution of fishing effort. Where overlap occurs, the rate at which dolphins are captured per unit of overlap (as a proxy for encounter rate) can be estimated using fisheries observer programmes, and potentially video monitoring (see below).

Table 6.2: Numbers of fishing-related deaths of Hector's and Māui dolphins 1921–2008 and 2008–16 by cause of death and region as listed in the DOC Incident Database (2017a). ECSI = East Coast South Island, WCSI = West Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island. See footnotes for explanation of probability categories as detailed in the database. (Continued next page)

	Cause of death	ECSI	WCSI	SCSI	NCSI	WCNI	Unknown subpopulation
From 1921 to June 2008							
Known entanglement (bycatch)[10]	Commercial set net	41	2	0	0	0	2
	Recreational set net	12	9	0	0	0	0
	Unknown set net	15	6	0	0	2	1
	Trawl net	15	4	0	0	0	0
	Commercial set net	0	0	0	0	0	0

---

[10] Animal was known (from incident report) to have been entangled and died.

	Cause of death	ECSI	WCSI	SCSI	NCSI	WCNI	Unknown subpopulation
Probable entanglement[11]	Recreational set net	0	0	0	0	0	0
	Unknown set net	1	4	0	0	0	0
	Unknown net	8	4	1	0	1	0
Possible entanglement[12]	Commercial set net	0	0	0	0	0	0
	Recreational set net	1	0	0	0	0	0
	Unknown set net	16	10	0	0	0	0
	Unknown net	16	7	1	0	2	0
From July 2008 to March 2020							
Known entanglement[13]	Commercial set net	11	0	0	0	1	0
	Recreational set net	1	1	0	0	0	0
	Trawl net	7	0	0	0	0	0
Probable entanglement[14]	Recreational set net	3	0	0	0	0	1
	Unknown set net	1	1	0	1	0	0
Possible entanglement[15]	Commercial set net	1	0	0	0	0	0

6.3.2 DATA FROM FISHERIES OBSERVERS

Fisheries observers record incidental captures of protected species including Hector's and Māui dolphins, on a proportion of commercial set net and commercial trawl fishing effort. These data are used to inform statistical models to estimate total captures across all fishing effort. Hector's and Māui dolphin captures recorded by fisheries observers are summarised in Table 6.3.

Because historical observer coverage in inshore fisheries has often been very low, simply scaling up from observed capture rate to estimate total captures yields estimates with unknown biases and very wide statistical confidence intervals. For example Baird & Bradford (2000) noted that the lack of information on the depth and position of commercial trawl effort and low observer coverage precluded any estimation of the total number of Hector's dolphins caught in trawl nets. Furthermore estimates from spatially blind models cannot be used to inform the design of spatial protection to reduce dolphin captures. For these reasons, to inform the update of the dolphin TMP, in 2019 these data were used in a spatially explicit risk assessment that estimates captures as a function of the overlap between dolphins and fishing effort, to correct for spatio-temporal bias arising from heterogeneous animal and fishing effort distributions, and non-representative fishing observer coverage.

Table 6.3: Observed commercial fishery captures of Hector's dolphin by fishing year from 1995–96 to 2016–17. All observed captures were from the east coast of the South Island. (Continued next page)

	Set net			Inshore trawl		
Fishing year	Alive	Dead	Total	Alive	Dead	Total
1995–96	0	0	0	0	0	0
1996–97	0	0	0	0	0	0
1997–98	2	6	8	0	1	1
1998–99	0	0	0	0	0	0
1999–00	0	0	0	0	0	0
2000–01	0	0	0	0	0	0
2001–02	0	0	0	0	0	0
2002–03	0	0	0	0	0	0
2003–04	0	0	0	0	0	0
2004–05	0	0	0	0	0	0

[11] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[12] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[13] Animal was known (from incident report) to have been entangled and died.
[14] As read from pathology report, or presence of net marks on body and a mention of this in incident report.
[15] As read from pathology report, or presence of net marks on body and a mention of this in incident report.

2005–06	0	0	0	0	0	0
2006–07	0	1	1	0	0	0
2007–08	0	1	1	0	0	0
2009–10	0	2	2	0	0	0
2010–11	0	0	0	0	0	0
2011–12	0	0	0	0	0	0
2012–13	1	0	1	0	0	0
2013–14	0	1	1	0	0	0
2014–15	0	0	0	0	0	0
2015–16	0	0	0	0	0	0
2016–17	0	1	1	0	0	0
2017–18	0	0	0	0	0	0
2018–19	0	0	0	0	0	0
Total	3	12	15	0	1	1

6.3.3 ELECTRONIC MONITORING

In addition to data gathered by scientific observers, electronic monitoring of inshore set net and trawl fisheries has been trialled to detect dolphin captures. In the 2012–13 year, the inshore set net fishery operating in Statistical Areas 022 and 024 was observed simultaneously by observers and electronic monitoring. During that time, at least two Hector's dolphins were captured, with one released alive. McElderry et al. (2007) describe another electronic monitoring trial that observed 89 set net events and 24 trawls off the Canterbury coast in the 2003–04 fishing year. Two Hector's dolphin captures were recorded in the set nets, reflecting a similar catch rate to previous estimates using data from observers. Observers and electronic monitoring were also deployed simultaneously in the Timaru set net fishery in 2012–13 (Archipelago Marine Research Ltd 2013) and observers were deployed again in 2013–14. One confirmed and one probable capture of Hector's dolphins were observed. These trials illustrate the potential to use electronic monitoring to increase observer coverage in inshore fisheries for purposes of managing risk to dolphins. New camera deployments are planned or underway to further develop this capability.

6.4 SPATIALLY EXPLICIT MULTI-THREAT RISK ASSESSMENT

In 2018–19 a team of New Zealand scientists collaborated to produce a comprehensive spatially explicit multi-threat risk assessment to Hector's and Māui dolphins (Roberts et al. 2019a), applying a customised adaptation of the SEFRA method described in Chapter 3 of this volume.

6.4.1 METHOD OVERVIEW

The spatial risk model was based on the SEFRA method, in which an animal's exposure to a particular threat, e.g., a fishing method, in space and time is expressed a function of the spatial overlap between the threat distribution and the animal distribution. The likelihood of impact per unit overlap, e.g., the probability of capture or death per encounter with a fishing event, can then be estimated empirically using fisheries observer data (for fishing threats) or other data indicative of cause of death (for lethal non-fishery threats). Because impacts are expressed in terms of probability of death at the scale of individual animals and individual threat events which are located in space, impacts (deaths) are additive in space and also additive across multiple threats to yield population-level risk at any spatial scale. Risk is expressed as a ratio between a threat-specific or cumulative estimate of deaths in the numerator and a PST or 'Population Sustainability Threshold' in the denominator. The PST reflects biological characteristics affecting the species' ability to sustain impact, and also a tuning factor that corresponds to a defined population outcome (a policy decision). See Chapter 3 for a fuller description of the SEFRA method.

6.4.1.1 COMMERCIAL FISHERIES RISK

Estimation of commercial fishery annual deaths and risk in the Hector's-Māui dolphin risk assessment was based on the spatial overlap of fishing events with the estimated summer/winter spatial abundance of Hector's and Māui dolphins (Figure 6.5). Two commercial fishery groups were defined: inshore set net fisheries and inshore trawl fisheries. For each method, annual deaths and risk were estimated at the sub-species (i.e., Hector's vs. Māui dolphin) and subpopulation levels (e.g., ECSI, WCSI, as shown in Figure 6.3).

This assessment included cryptic mortality and post-release survival priors specific to observer-recorded Hector's and Māui dolphin captures in commercial set net and trawl fisheries (see appendix 10 of Roberts et al. 2019a).

6.4.1.2 LETHAL NON-FISHERY THREATS

For demonstrably lethal non-commercial fishery threats (but excluding recreational fishing), e.g., toxoplasmosis, predation, and others, annual deaths were estimated using a multi-threat extension of the SEFRA approach. Briefly, this approach partitioned residual deaths (total annual deaths minus commercial fishery deaths) in accordance with the attributed primary causes of death from necropsy records from 2007 to 2018 (see Table 6.4). The necropsy sample excluded:

- known/probable/possible bycatch deaths — which comprised an unknown composition of commercial fishery and recreational fishing deaths and, for commercial fisheries, the standard SEFRA approach was a more direct means;
- calves — for which there were no estimates of annual survival for estimating total annual deaths; and
- individuals for which a 'poor' confidence rating was attributed to the diagnosed cause of death.

The resulting subset used in the risk assessment (Table 6.5) was primarily composed of dolphins that were found washed up dead on the beach (beach-cast), with a smaller number of dead dolphins found floating at sea. The extent to which this sample may be biased due to differential carcass detection rates for different causes of death was evaluated using sensitivities, and is discussed further below.

The risk model estimated posterior distributions of annual deaths for each of toxoplasmosis (the primary non-fishery cause of death, and with an indirect anthropogenic origin), predation events (considered to have a high potential for undetected mortalities, tested via sensitivities), and an 'other' group of all other non-fishery causes of death (most of which may constitute 'natural mortality'). Subpopulation-specific estimates are shown in Table 6.6 and Figure 6.10.

A "predation sensitivity" was undertaken to assess the sensitivity of risk model estimates to assuming an arbitrary ten-fold reduction in the detection probability of predation events. This has the effect of increasing annual deaths from predation and reducing the estimates of deaths from toxoplasmosis and "other" non-fishery causes of death.

Exposure to toxoplasmosis and predation threats were mapped in space, using estimates of the relative spatial density of *Toxoplasma gondii* parasite oocysts (from the combined outputs of a cat density model and a hydrological model) and of predation by broadnose sevengill sharks (*Notorynchus cepedianus*) (modelled from commercial set net fishery catch and effort records). The estimated spatial overlap of toxoplasmosis and predation mortalities with Hector's and Māui dolphins was then used to estimate subpopulation-specific annual deaths for these non-fishery causes of death.

6.4.1.3 NON-LETHAL THREATS

For non-lethal threats (potentially including lethal threats that have not yet appeared in the necropsy records) that can still be resolved spatially, an alternative approach was taken. Spatial overlap with non-lethal threats was presented in two different ways:

1. Mapping of relative overlap between spatial dolphin abundance and spatial threat intensity. This highlighted areas with a high density of Hector's and Māui dolphins and high threat intensity; and
2. Relative overlap statistic scaling for population size. This highlighted populations for which the threat intensity is high in the locations that dolphins occur (i.e., where probability of death per dolphin will be high, regardless of the relative abundance of dolphins).

6.4.1.4 RECREATIONAL FISHERIES RISK

Because fisheries observer data are not available for recreational fishing, the impact of recreational set net fishing mortality was estimated on a relative rather than an absolute scale, at the subpopulation level, using the method applied to spatial, non-lethal threats (described above). The full SEFRA approach could not be used for recreational fishing, because there was no means by which to estimate vulnerability to capture. Furthermore, necropsy records attributed to probable and possible bycatch (Table 6.4) comprised an unknown composition of commercial and recreational fishery deaths, precluding the use of the same approach as used for toxoplasmosis and other lethal non-fishery causes of death. This estimation of spatial overlap between dolphins and recreational set net fishing used seasonally resolved (summer/winter) estimates of the relative spatial intensity of recreational netting effort based on two nation-wide fisher surveys (Wynne-Jones et al. 2014, 2019). See section 6.4.4.2, below.

6.4.2 INTRINSIC POPULATION GROWTH (RMAX)

The spatial risk model developed for Hector's and Māui dolphins required a prior distribution of intrinsic population growth rate ($r^{\max}$), the maximum growth rate that will occur at small population size when resources are replete. The $r^{\max}$ affects the Population Sustainability Threshold (PST), reflecting the ability of the species to sustain and recover from impact while meeting a defined population objective.

The risk assessment used individual size at age and maturity stage information to estimate an $r^{\max}$ prior for Hector's dolphins (Edwards et al. 2018). This analysis followed the approach of Dillingham et al. (2016), implemented by Moore (2015), which uses an allometric invariant between optimal generation time (the average age of a breeder during optimal growth) and $r^{\max}$ observed across a wide range of vertebrate species. See Edwards et al. (2018) for a detailed description of methods and sensitivity runs.

The assessment by Edwards et al. (2018) was updated with supplementary ageing and maturity information. This produced a Monte Carlo distribution of $r^{\max}$ with median of 0.050 and 95% credible interval of 0.029–0.071. This empirical estimate replaced a previous base case $r^{\max}$ of 0.018 assumed by the most recent Māui dolphin multi-threat assessment (Currey et al. 2012); the previous estimate was based on an applied maximum longevity of 20 years (Slooten & Ladd 1991), which is now known to be an underestimate for this species (e.g., Gormley 2009).

The revised estimate of $r^{\max}$ for Hector's dolphin is now consistent with age at first reproduction, given the relationship observed across other mammalian species (Figure 6.6), and is at the low end of the cloud of values defined by other cetacean species (in grey) indicating that at this value of $r^{\max}$ Hector's dolphins are still among the slowest-reproducing cetaceans relative to their other life history traits, but within plausible bounds.

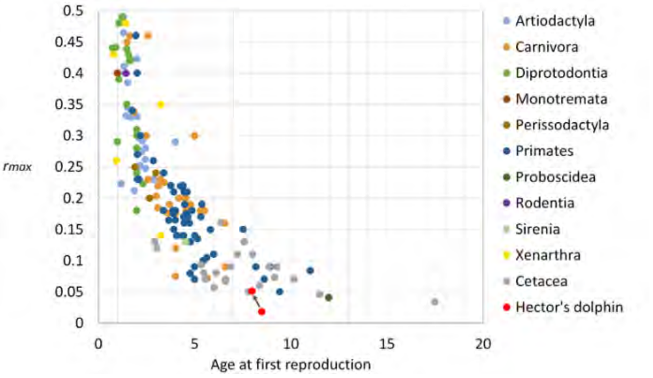

Figure 6.6: Comparative plot of r^{max} against age at first reproduction for a variety of mammalian orders (Duncan et al. 2007). For Hector's dolphin, both the previous and updated values are shown (red points). The updated value of r^{max} derived here for Hector's dolphin is now consistent with that expected from other mammals, given estimated age at first reproduction. From Edwards et al. (2018).

At very small population sizes (relevant to Māui dolphin), Allee effects may adversely affect realised population growth despite ample resources (and increase the probability of extinction). The mechanisms from which Allee effects arise all impact on individual survival and reproduction and include an array of demographic, genetic, social, and potentially anthropogenic mechanisms. Population simulations were completed using an individual-based model, accounting for demographic stochasticity and inbreeding depression. This produced a mean $r^{\max}$ of approximately 0.045 for a starting population of 50 dolphins (under the latest estimate for Māui dolphins of 63 individuals). The prior used for assessing the population risk of threats to Māui dolphins was adjusted accordingly.

Note that this assessment did not account for catastrophic events or disruption to social systems that might occur at small population size and that would further reduce $r^{\max}$.

6.4.3 COMMERCIAL FISHERY OVERLAP, DEATHS, AND RISK

6.4.3.1 SPATIAL FISHING EFFORT DISTRIBUTIONS

Spatial patterns of inshore set net and inshore trawl fishing effort are shown in Figure 6.7. The risk assessment uses a 3-year average to indicate patterns of 'current' fishing effort (i.e., 2014/15–2016/17 fishing years at the time that the risk assessment was run).

In South Island, inshore trawl effort is highest off the ECSI near Timaru. Other locations with concentrated trawl fishing effort also appear in the NCSI in Golden Bay and Tasman Bay and in the SCSI in Te Waewae Bay. Set net fishing effort is most concentrated around Kaikōura Canyon.

For the North Island, set net effort that overlaps the Māui dolphin distribution is most concentrated inside WCNI harbours and near New Plymouth. Inshore trawl effort is highest offshore from Raglan and Kawhia harbours, beyond the limits of the existing 2 nm closure.

6.4.3.2 ESTIMATED VS ACTUAL PATTERNS OF OBSERVED CAPTURES

The risk assessment model estimates dolphin captures in space as a function of spatial overlap between dolphins and fishing effort. All observed commercial fishery captures from 1995/96 to 2016/17 were in ECSI set nets, located in areas of high observed overlap near Kaikōura and the Canterbury Bight, which corresponded well with model predictions (Figure 6.8). Good agreement between predicted vs. observed patterns of observed captures indicates that the estimated spatial density of dolphins was a good approximation to the true density, and that vulnerability to capture was relatively constant across inshore set nets operating in different areas. A similar comparison for trawl captures is not meaningful because there has only been a single observed inshore trawl capture.

6.4.3.3 ESTIMATED CAPTURES, DEATHS AND RISK

Commercial set net fisheries were estimated to kill considerably more Hector's and Māui dolphins than inshore trawl fisheries (Table 6.6), despite considerably higher effort levels and also higher overlap (because existing trawl fishery closures are not as large as set net fishery closures). This reflects that dolphin catchability is substantially higher in set nets than in trawls: the model estimates that a dolphin is roughly 20–30x more likely to be killed in a set net than in a trawl in any given location.

The ECSI subpopulation had the greatest estimated number of annual deaths from both commercial set nets (38.9 individuals per annum, 95% CI = 18.6–88.3) and inshore trawls (3.0 individuals per annum, 95% CI = 0.1–15.6) for the period 2014/15 to 2016/17 (Table 6.6). For the WCSI, the estimated annual deaths are low from both commercial set nets (0.3 individuals per annum, 95% CI = 0.2–0.7) and inshore trawls (1.8 individuals per annum, 95% CI = 0.1–9.4), reflecting low effort.

The estimate of Māui dolphin commercial fisheries deaths was 0.10 deaths per annum (95% CI = 0.0–0.25) in commercial set nets, and 0.02 deaths per annum (95% CI = 0–0.05) in inshore trawl fisheries. No Māui dolphin captures were observed in either fishery from 1995/96 to 2016/17 (or subsequently, to June 2020), with very high observer coverage since 2013 (reflecting Ministerial directives arising from the previous update of the Māui dolphin TMP; MPI & DOC 2012).

For commercial set net fisheries, the median value of the estimated risk ratio was below 1 for all subpopulations. This result suggests that the best estimate of annual mortalities in commercial fisheries did not exceed the PST_{90} between 2014/15 and 2016/17, suggesting that recent commercial fishery mortality levels for set nets in isolation are most likely not sufficiently high to suppress the equilibrium population below 90% of carrying capacity (with over 50% certainty). However the upper 95th percentile of the fisheries deaths estimate did exceed the PST_{90} for all subpopulations except the WCSI, suggesting that it is possible given current uncertainty that fisheries impacts may exceed this level. Risk assessment outputs are typically communicated with reference to both the median and the 90% or 95% confidence level. The level of certainty that decision-makers require that the population objective will be achieved is a policy decision.

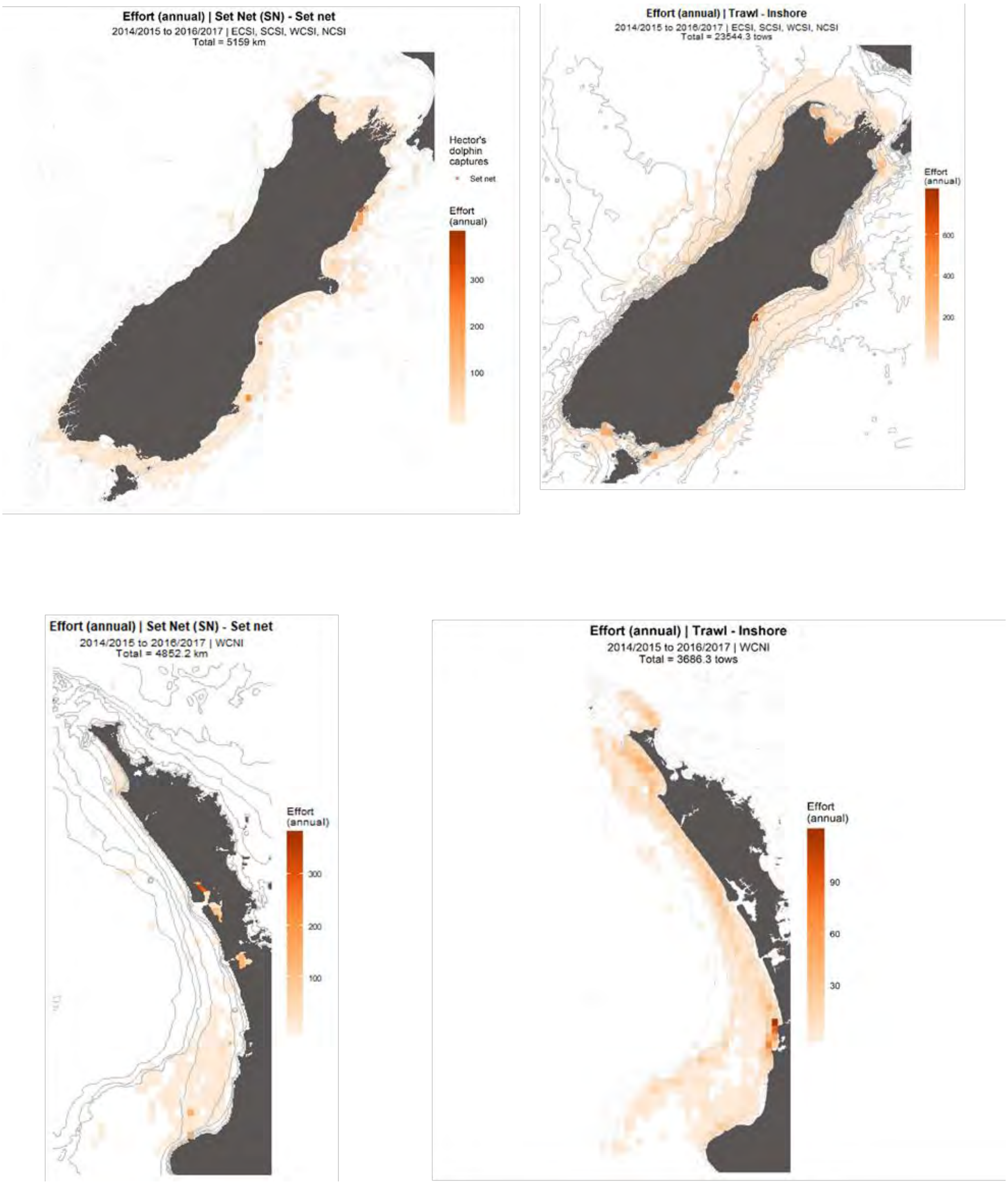

Figure 6.7: Spatial patterns of commercial fishing effort density (2014/15–2016/17) for inshore set net and inshore trawl fisheries.

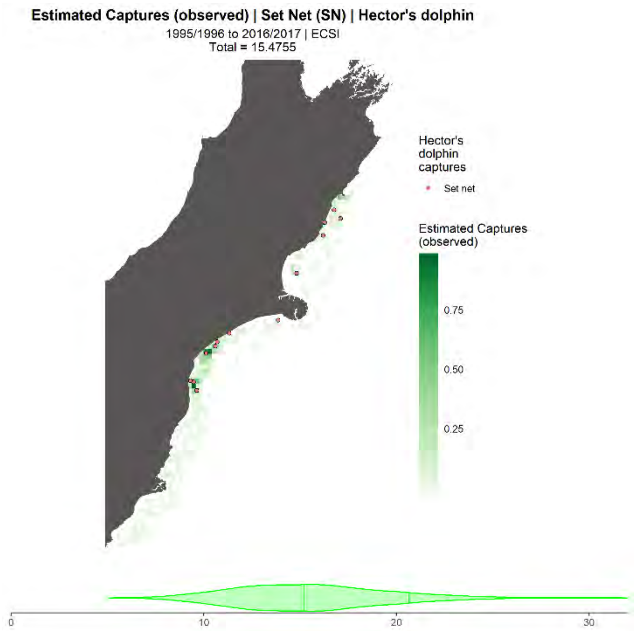

Figure 6.8: Predicted and actual observed Hector's dolphin captures in commercial set nets in the ECSI subpopulation from 1995/96 to 2016/17. The expected pattern of observed captures (as a function of observed spatial overlap) is shown in green; actual observed captures are superimposed in red. The posterior distribution of estimated observed captures is displayed as the violin along the bottom with the median and upper 90% quantile indicated by vertical lines within the violin.

6.4.3.4 ADJUSTING FOR MULTIPLE-CAPTURE TRAWL EVENTS

Fisheries observers have only recorded a single Hector's dolphin capture in inshore trawl fisheries. In contrast, there have been 13 inshore trawl events in which Hector's dolphin captures were reported by fishers. Following two fisher-reported multiple capture events in early 2019 in which three individuals were reported captured on a single trawl, the AEWG judged that observed captures (Table 6.2) were indicative of the likelihood of a capture event, but that the number of animals captured per event may be more accurately reflected in the frequency of multiple-capture events reported by fishers. In 13 such events, six captured between 2 and 4 individuals; in total 25 animals were captured (i.e., 1.92 individuals per positive capture event). Assuming that on average two individuals are captured per inshore trawl capture event, this would lead to a doubling of the estimated annual deaths and risk from the risk model (values shown in Table 6.6). These double trawl risk values are shown as a sensitivity in appendix 17 of Roberts et al. (2019a); on the advice of the AEWG these are the trawl risk numbers that were used to inform the update of the Hector's and Māui dolphin TMP.

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Table 6.4: Diagnosed primary cause of death of non-calf Hector's and Māui dolphins by population, from necropsy information for WHAT TIME Period?. Intermediate and full confidence rated diagnoses from Roberts et al. (2019a). ECSI = East Coast South Island, WCSI = West Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island.

	Hector's dolphin				Māui dolphin	
Cause of death	ECSI	WCSI	SCSI	WCNI	WCNI	Total
Brucellosis	0	1	0	0	1	2
Deformity	0	0	1	0	0	1
Disease (other)	4	2	0	1	0	7
Miscellaneous	2	2	0	0	1	5
Pneumonia	3	1	0	0	0	4
Predation	0	0	1	0	1	2
Toxoplasmosis	5	2	0	0	2	9
Tuberculosis	1	0	0	0	0	1
Known bycatch	5	0	0	0	0	5
Probable bycatch	4	2	0	0	0	6
Possible bycatch	0	1	0	0	0	1
Unknown/Open	9	3	0	0	0	12
Total	33	14	2	1	5	55

Table 6.5: Necropsy observations used in the estimation of risk from non-fishery threats in the spatial risk model of Roberts et al. (2019a). ECSI = East Coast South Island, WCSI = West Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island.

	Hector's dolphin				Māui dolphin	
Cause of death	ECSI	WCSI	SCSI	WCNI	WCNI	Total
Toxoplasmosis	5	2	0	0	2	9
Predation	0	0	1	0	1	2
Other	10	6	1	1	2	20
Total	15	8	2	1	5	31

Table 6.6: Spatial risk model estimates of annual deaths by threat and subpopulation based on data from 2014/15 to 2016/17. This model run assumed that an average of two individuals were killed per inshore trawl capture event. The median and 95% credible intervals are shown.

Cause of death	Subpopulation	50.0%	2.5%	97.5%
Set net	MĀUI	0.10	0.00	0.30
Set net	NI	0.07	0.04	0.17
Set net	TAKA	0.06	0.03	0.13
Set net	NCSI	0.65	0.31	1.47
Set net	WCSI	0.32	0.15	0.74
Set net	ECSI	38.86	18.57	88.25
Set net	SCSI	0.80	0.38	1.81
Inshore trawl	MĀUI	0.00	0.00	0.10
Inshore trawl	NI	0.00	0.00	0.04
Inshore trawl	TAKA	0.00	0.00	0.00
Inshore trawl	NCSI	0.20	0.00	1.08
Inshore trawl	WCSI	3.68	0.16	18.80
Inshore trawl	ECSI	6.08	0.28	31.12
Inshore trawl	SCSI	0.22	0.00	1.12
Toxoplasmosis	MĀUI	1.90	0.96	3.27
Toxoplasmosis	NI	0.25	0.09	0.58
Toxoplasmosis	TAKA	0.40	0.15	0.93
Toxoplasmosis	NCSI	1.10	0.40	2.54
Toxoplasmosis	WCSI	187.03	67.86	432.09
Toxoplasmosis	ECSI	115.06	41.75	265.81
Toxoplasmosis	SCSI	5.05	1.83	11.67
Predation	MĀUI	0.53	0.11	1.42
Predation	NI	0.00	0.00	0.00
Predation	TAKA	0.03	0.01	0.11
Predation	NCSI	0.77	0.16	2.63
Predation	WCSI	62.64	12.72	214.41
Predation	ECSI	17.64	3.58	60.37
Predation	SCSI	2.63	0.53	9.00
Other	MĀUI	4.06	2.65	5.99
Other	NI	0.42	0.17	0.88
Other	TAKA	0.56	0.23	1.16
Other	NCSI	9.06	3.69	18.78
Other	WCSI	232.05	94.49	480.99
Other	ECSI	411.79	167.67	853.54
Other	SCSI	14.05	5.72	29.13

6.4.3.5 CHANGE IN FISHERIES RISK OVER TIME

Figure 6.9 illustrates temporal trends in commercial set net fishing effort and spatial overlap per unit effort, and resulting estimates of fisheries deaths and risk over time at the subpopulation scale (Māui dolphins and Hector’s dolphins). For Hector’s dolphins, estimated trawl risk has declined steadily since 1992/93 due to reduced effort, but in the commercial set net fishery an overall reduction in effort has been offset by a general increase in overlap per unit effort resulting in little change in risk through time (Figure 6.9). This result was counterintuitive given the establishment of spatial set net closures in 2008, but the spatial risk model suggests that in places where Hector’s dolphins are abundant, their distribution extends well beyond these closures into areas where set net fishing is ongoing. Nonetheless the median estimate of risk ratio was below 1 in all years since 1992/93, indicating that (in the absence of other anthropogenic mortality) the estimated annual mortalities across the last 25 years are unlikely to have been sufficient to prevent population recovery to or stabilisation at levels above at least 90% of carrying capacity.

For Māui dolphins, the estimated annual set net deaths and risk ratios have declined steadily since 1992/93, due to decreasing total effort and reduced spatial overlap per unit effort, reflecting previous spatial fishery closures. Trawl fishery estimates show similar trends.

6.4.3.6 SPATIAL PATTERNS OF FISHERIES RISK

Spatial distributions of model estimated fisheries deaths are shown in Figure 6.10. The spatial patterns of captures, deaths, and risk are the same because all are proportional to spatial overlap.

In the South Island the spatial risk model predicts that set net deaths will occur mostly in Pegasus Bay north of Banks Peninsula where there are large numbers of dolphins in locations further offshore than the existing fisheries closures, and also in Kaikōura, where dolphin numbers are moderate to low, but fishing effort is highly concentrated. The model predicts that trawl fishery deaths will be highest near Timaru, where dolphins are abundant and fishing effort is high.

In the North Island the spatial risk model predicts that set net captures are most likely in low dolphin density locations, because set net fishing has already been eliminated from areas with high dolphin densities. Captures are predicted to be most likely in WCNI harbours, due to very high fishing effort (note however that the structural assumptions underlying this prediction are untested). The model also predicts captures in low dolphin-density locations near New Plymouth, for similar reasons (but in this instance model predictions of low but non-zero dolphin presence are supported by sightings data). The model predicts that trawl captures are most likely to occur near Raglan and Kawhia harbours, where the estimated dolphin distribution extends beyond the existing 2 nm trawl closure.

Using the risk spatial risk model described by Roberts et al. (2019a) and a customised query interface (‘Risk Atlas’; D Webber unpublished), in 2019 spatial risk estimate outputs analogous to Figures 6.9 and 6.10 were used at the smaller scales of subpopulations and local populations, and within user-defined boundaries simulating the effects of various spatial fishery closures, to evaluate alternate fisheries risk reduction options under an updated Threat Management Plan. These options are currently (June 2020) under consideration.

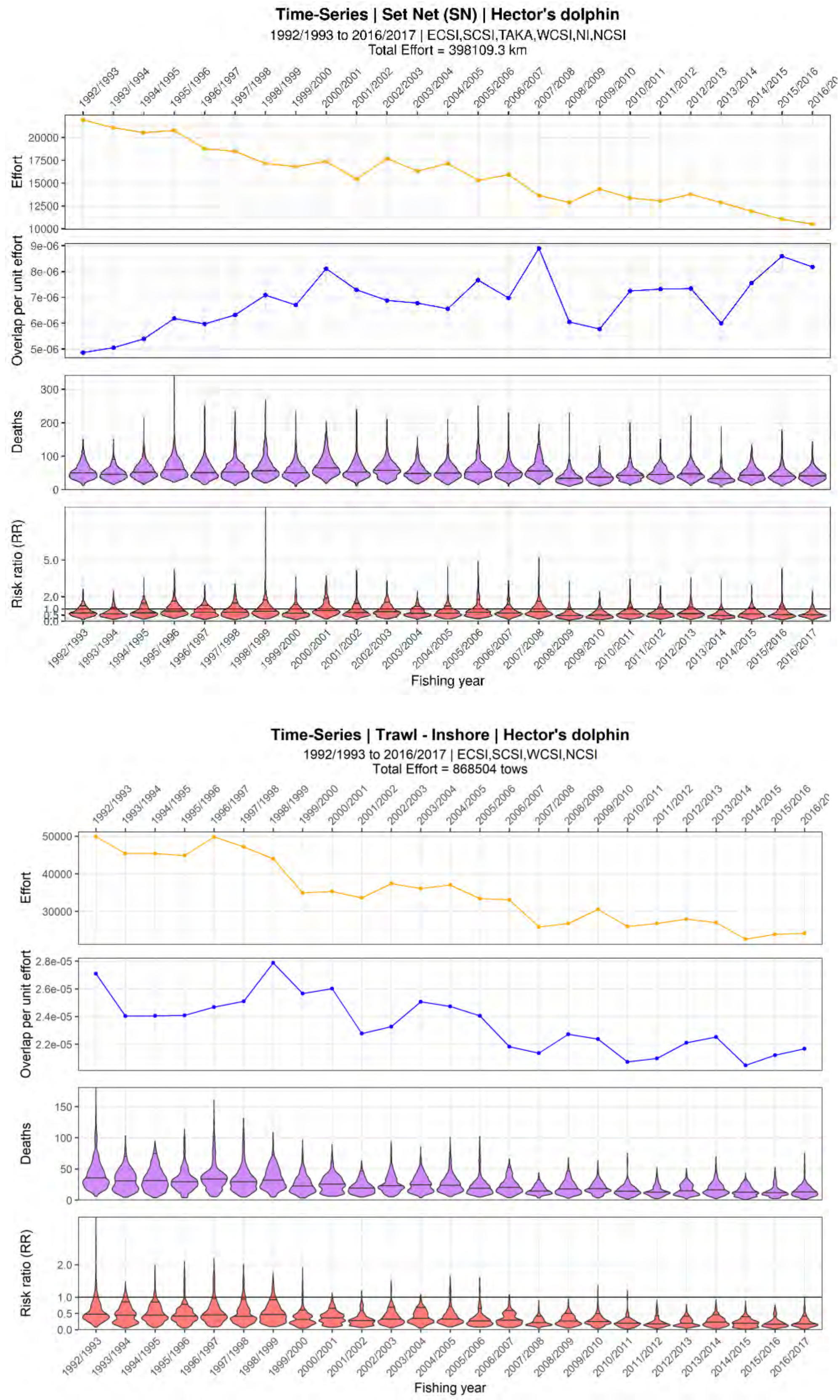
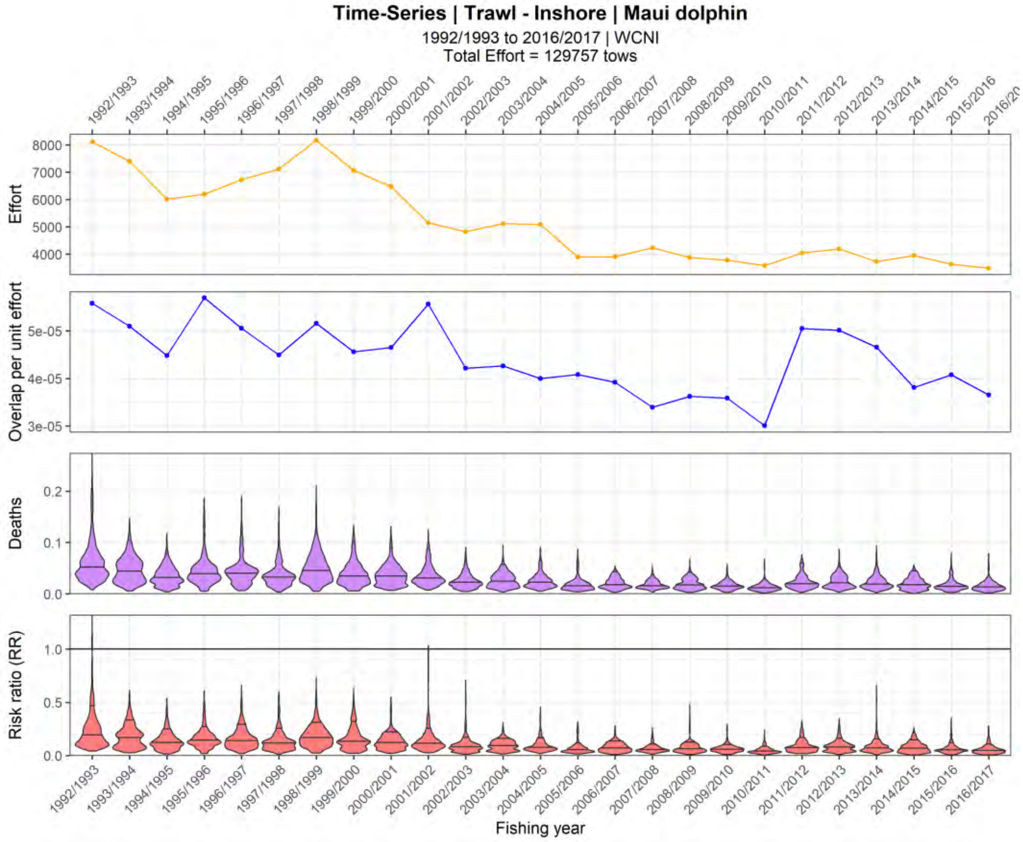


Figure 6.9: Change in estimated fisheries deaths and risk to Hector's dolphins over time arising from changes in effort and spatial overlap, for set net (upper) and inshore trawl (lower). Note that trawl deaths depicted here reflect the x2 sensitivity whereby on average two animals are killed per capture event. Risk score outputs in this model run applied a calibration coefficient (ϕ) of 0.2. [Continued over the page]



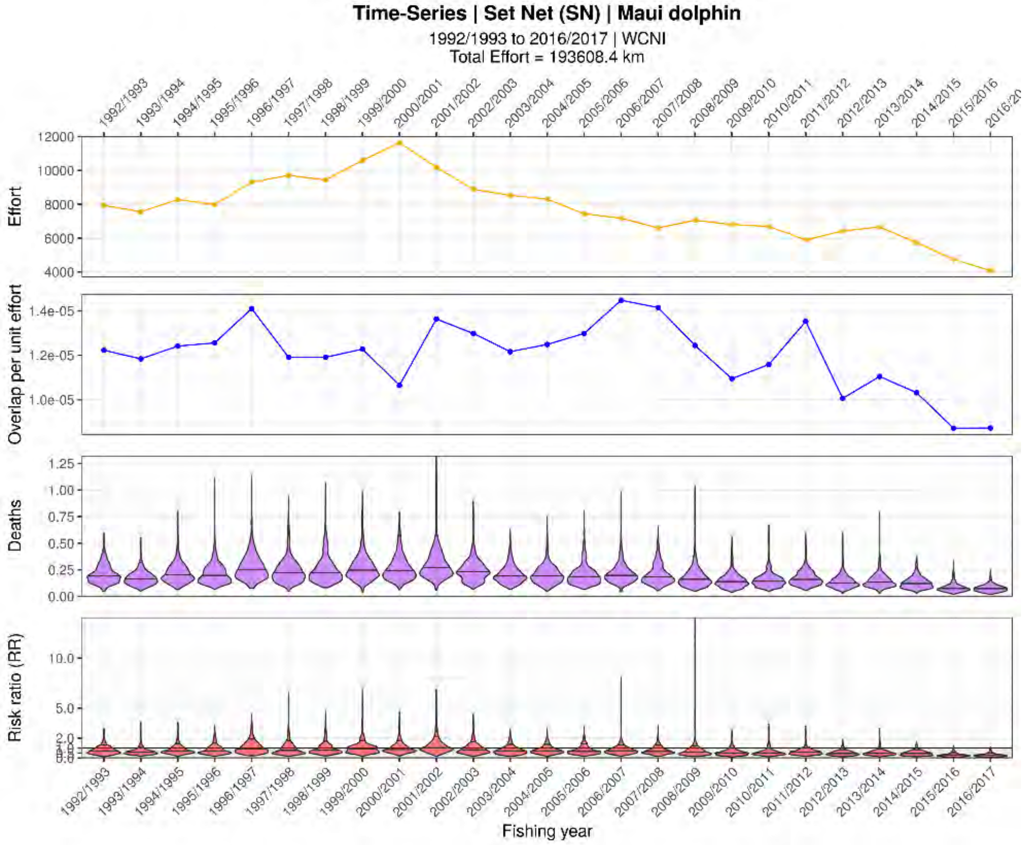


Figure 6.9: Change in estimated fisheries deaths and risk to Māui dolphins over time arising from changes in effort and spatial overlap, for set net (upper) and inshore trawl (lower). Note that trawl deaths depicted here reflect the x2 sensitivity whereby on average two animals are killed per capture event. Risk score outputs in this model run applied a calibration coefficient (ϕ) of 0.2.

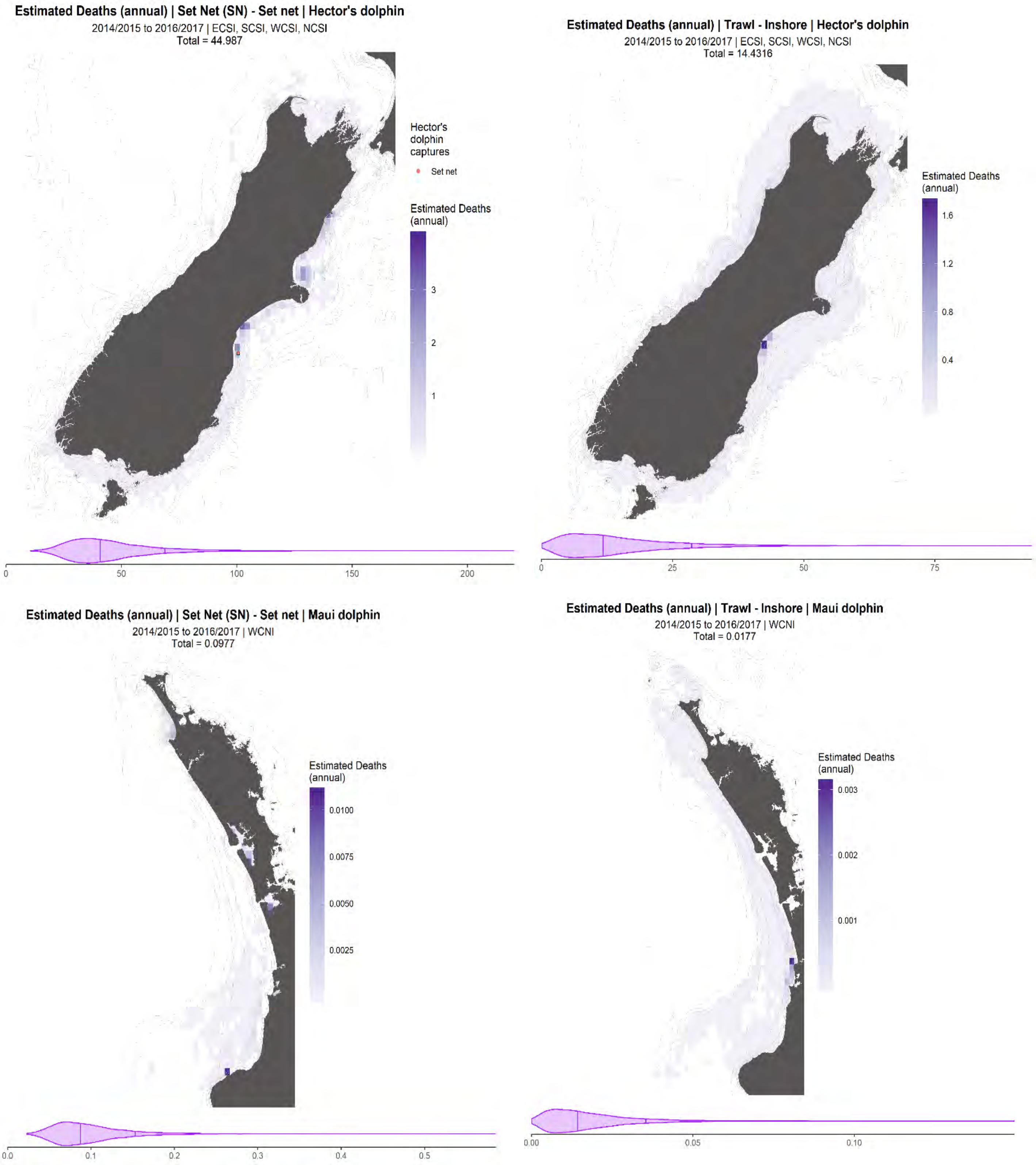


Figure 6.10: Spatial patterns of estimated annual fisheries deaths (hence also capture and risk) for Hector's dolphins (upper) and Māui dolphins (lower) from 2014/15 to 2016/17. Violin plots below each map depict the model posterior estimates of deaths; median and 90th percentile estimates are marked by vertical lines. Note that trawl deaths depicted here reflect the x2 sensitivity whereby on average two animals are killed per capture event.

6.4.3.7 FACTORS AFFECTING STATISTICAL UNCERTAINTY IN FISHERIES RISK

The wide model posteriors of commercial set net fishery deaths and risk ratios estimates primarily stem from vague priors with respect to the cryptic mortality multiplier (reflecting bodies that may drop out of the net without being observable). In contrast, the posterior for set net catchability was well estimated despite an uninformed prior, indicating that existing set net observer data are sufficient to inform relatively precise total captures estimates. For this reason increased observer coverage would lead to a relatively minor improvement in the precision of set net risk ratio estimates.

In contrast, in trawl fisheries cryptic mortality has far less influence, but trawl catchability is less well estimated, reflecting that there has only been a single observed trawl fishery capture. In locations where dolphins are abundant, increased observer coverage or electronic monitoring of trawl fishing effort can be expected to yield improved precision in the estimation of trawl fisheries risk.

Note that because the estimation of dolphin catchability in the spatial risk assessment uses only the data from observed fishing events, for which geographical locations are recorded by the fisheries observer, this part of estimation of fisheries risk is not subject to potential bias arising from non-representative fisheries observer coverage or imprecise position reporting. However where the catchability estimate is then applied to estimate total deaths across the fishery including in un-observed fishing events, these estimates will be affected by the accuracy with which fishing effort locations are recorded.

In the spatial risk assessment by Roberts et al. (2019a), all fishing effort locations representative of ‘current’ commercial fisheries risk (in Figure 6.7) are recorded precisely (with the exception of small vessel set net fisheries operating inside WCNI harbours, for which locations were estimated using data from a cellphone tracking app; see Roberts et al. 2019a). Hence the estimates of current fisheries risk (shown in Figures 6.10 and 6.13, and Table 6.6) will not be affected by bias arising from spatial effort reporting imprecision. However, inshore trawl fishing effort locations have only been recorded precisely since the introduction of new reporting forms in 2006 and 2007; fishing effort in earlier years was reported to fisheries statistical areas, and the actual position estimated subsequently. For this reason the year-specific estimates from the 1990s and mid-2000s (as in the annual time series in Figure 6.9, below) may be subject to greater uncertainty than is shown.

6.4.3.8 RISK ASSESSMENT OUTPUTS COMPARED WITH PRE-EXISTING ESTIMATES OF COMMERCIAL FISHERIES CAPTURES

Prior to 2012, the only observer programme with sufficient coverage to yield a robust estimate of the rate of incidental capture of Hector’s dolphins in inshore commercial set nets (Baird & Bradford 2000) was an observer programme in Statistical Areas 018, 020, and 022 (FMA 3) on the east coast of the South Island in the 1997/98 fishing year, which observed 214 inshore set net events, targeting shark species and elephant fish. Eight Hector’s dolphins were caught in five sets, of which two were released alive. Capture rates were most precise in Area 022, where six of the catches were reported, following observer coverage of 39% (Baird & Bradford 2000). Capture rate was estimated at 0.064 dolphins per set (CV = 43%) in Area 022 and 0.037 dolphins per set (CV = 39%) in Areas 020 and 022 combined (Baird & Bradford 2000). A total of 16 dolphins (CV = 43%) were estimated to have been captured in Area 022, and 18 dolphins (CV = 38%)[16] dolphins captured in Areas 020 and 022 combined (noting these are captures not deaths, i.e. not including cryptic mortality) (Baird & Bradford 2000). These estimates are from Statistical Areas containing the Banks Peninsula Marine Mammal Sanctuary (BPMMS), which at that time effectively prohibited commercial set netting between Sumner Head and the Rakaia River out to 4 nm from the coast (Dawson & Slooten 1993).

Using Risk Atlas, the risk assessment outputs by Roberts et al. (2019a) can be queried within user-defined boundaries, including estimates for previous fishing years. An extract performed to replicate the boundary conditions of Baird &

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[16] This was reported as either 16 or 18 dolphins in the cited reference, but has been confirmed as 18 dolphins by correspondence with the author (S. Baird, pers. comm.).

Bradford (2000), i.e., estimating set net risk within Statistical Areas 020 and 022 in the 1997/98 fishing year – yielded a mean estimate of 16.4 set net captures (or 30.7 deaths (90% c.i. 14.1–54.8) including cryptic mortality). This is very similar to the Baird and Bradford (2000) estimate of 18 captures (CV = 38%), which did not include cryptic mortality.

Slooten & Davies (2012) used the observed set net data from 2009/10 to estimate total captures on the ECSI of 23 dolphins (CV = 0.21). This was the first published capture estimate since extensive protection measures to mitigate Hector's dolphin risk were introduced in 2008.

An extract from the model of Roberts et al. (2019a), performed to replicate the boundary conditions of Slooten & Davies (2012), yielded a mean estimate of 23.0 set net captures (or 43.0 deaths (90% c.i. 23.9–74.3) including cryptic mortality). This is very similar to the Slooten & Davies (2012) estimate of 23 captures (CV = 0.21), which did not include cryptic mortality.

To inform the update of the Māui dolphin TMP in 2012, an expert panel identified 23 threats potentially relevant to Māui dolphins. For those threats that could be mapped spatially, the spatial distribution of the resulting impact was estimated by calculating the spatial overlap between the threat and the spatial density of the dolphins. In the absence of any empirical means of estimating the number of dolphin deaths attributable to each threat, these were estimated qualitatively via structured expert elicitation (Currey et al. 2012). The expert panel workshop judged that, in decreasing order of magnitude, the most important threats to Māui dolphins were: commercial set nets (2.3 deaths per year); commercial trawls (1.1 deaths per year); recreational set nets (0.9 deaths per year); mining and oil activities (0.1 equivalent deaths per year); vessel traffic (0.07 deaths or equivalent deaths per year); pollution (0.05 deaths or equivalent deaths per year); and disease < 0.01 deaths or equivalent deaths per year).

Notably, the expert panel estimates do not resemble the model estimate fitted to necropsy data (Roberts et al. 2019a), e.g., as in Table 6.6. The most notable omission is that disease was not recognised as a significant threat among the assembled experts by Currey et al. (2012). It was only subsequently that Roe et al. (2013) published the result that 2 of 3 Māui dolphins necropsied and tested in the period 2007–2011 had died as a result of *Toxoplasma gondii* infection, suggesting that the expert panel may have underestimated mortality from this source. Currey et al. (2012) reported that toxoplasmosis was not among the threats considered by the expert panel because this information was not available at the time of the risk assessment workshop. Instead the assembled experts estimated that 95% percent of annual deaths were likely to be attributable to commercial or recreational fisheries.

6.4.4 NON-COMMERCIAL FISHERY THREATS

6.4.4.1 TOXOPLASMOSIS, PREDATION, AND OTHER LETHAL NON-FISHERY THREATS

Updated necropsy observations used to inform the estimation of non-fishery deaths are given in Table 6. and 6.5. Toxoplasmosis was the primary cause of death other than entanglement in fisheries gear that was considered to have an anthropogenic origin. Toxoplasmosis was first identified as a major non-fishery threat for both Hector's and Māui dolphins towards the end of the 2012 TMP risk assessment process (Roe et al. 2013). Prior information suggests that toxoplasmosis mortalities affect both Hector's and Māui dolphins, and toxoplasmosis mortalities are geographically widespread around the South Island (Roe et al. 2013). The spatial risk assessment estimated the summer and winter relative densities of *T. gondii* oocytes in coastal waters (Figure 6.11) as a function of the estimated distribution of cats around New Zealand and rainfall or run-off using an existing hydrology model (see appendix 9 of Roberts et al. 2019a). Spatial overlap between toxoplasma exposure densities and the spatial abundance of Hector's and Māui dolphins were then used to estimate relative risk levels to different subpopulations in the spatial risk model.

Estimates of cause of death arising from the necropsy observations fitted to the spatial risk model are shown in

Table 6.. For the purposes of model fitting, the total for each cause of death were used (across all sub-areas). Of the 31 non-calf dolphins for which a primary cause of death could be determined and that were not attributed to bycatch mortality, 9 deaths were attributed to toxoplasmosis, 2 were attributed to predation, and the remainder were attributed to "other" non-fishery causes of death (mostly constituting 'natural mortality'). The sample size of observations by sub-area were small, with the bulk of sample coming from the ECSI (15) or WCSI (8), with a

similar composition of causes of death in these two areas. Model estimated annual deaths from toxoplasmosis were greater than those from commercial fisheries for all subpopulations (Table 6.6 and Figure 6.10). This was the case for the model run assuming an equal detection probability of non-fishery causes of death and also for the predation sensitivity model run, which assumed a 10-fold reduction in the detection probability of predation deaths (hence 10x more predation deaths occurring). Note however that because exposure to both toxoplasmosis and shark predation vary substantially in different locations, the relative importance of different threats or natural causes of death at the scale of particular subpopulations may be more sensitive to assumptions about predation detectability than are conclusions at the scale of the entire species. The highest number of annual toxoplasmosis deaths was estimated for the WCSI population (187.0 individuals per annum, 95% CI = 67.9–432.1); this estimate was lower for the predation sensitivity (106.8 individuals per annum, 95% CI = 32.7–284.4) under which predation was responsible for a corresponding increased proportion of non-fishery deaths.

Figure 6.11: Estimated relative coastal water density of *Toxoplasma gondii* oocysts in winter (red = high density, blue = low density). From Roberts et al. (2019a).

For the WCNI where Māui dolphins occur (labelled “MĀUI” in Table 6.6), the estimated annual deaths from toxoplasmosis (1.9 individuals per annum, 95% CI = 1.0–3.0) were much higher than from either commercial set nets (0.10 individuals per annum, 95% CI = 0.0–0.25) or the inshore trawl fishery (0.02 individuals per annum, 95% CI = 0.0–0.05). The relatively high toxoplasmosis risk ratio for Māui dolphins reflects very high estimates of *T. gondii* oocyst densities around the North Island, particularly in winter months, in the area south of Manukau Harbour (Figure 6.11). The Waikato River in particular drains a very large catchment with high domestic and feral cat densities. The core of the Māui dolphin range is largely confined to the area of freshwater influence from this and other WCNI rivers (Hunt & Jones 2020).

Note that uncertainty arising from low sample size in the necropsy data is reflected in the wide confidence intervals around estimates of toxoplasmosis deaths (although for Māui dolphins, even the lower bound of this statistical uncertainty exceeds the upper bound of the estimated commercial fishery deaths). Furthermore the comparability of estimates of commercial fishery deaths and non-fishery deaths, e.g., from toxoplasmosis, will potentially be affected by biases associated with using a necropsy sample primarily obtained from beach-cast individuals. Potential sources of bias include threat-specific differences in the timing and location of death which may affect the relative likelihood of carcass recovery for necropsy. To illustrate, deaths that occur in summer are more likely to be reported by beachgoers, so may be over-represented in necropsy results. All nine confirmed toxoplasmosis mortalities to date were recovered in the period from September to November (Roe et al. 2013; DOC 2020a).

6.4.4.2 RECREATIONAL FISHERIES RISK

The relative spatial threat posed by recreational set netting to Hector’s and Māui dolphin subpopulations was estimated based on relative spatial overlap. The locations of reported recreational set net fishing events were compiled from the results of two nationwide recreational fishing panel surveys (Wynne-Jones et al. 2014, 2019) and manually assigned to named locations by Fisheries New Zealand fisheries managers familiar with the operation of these fisheries. Note that survey answers did not distinguish between recreational set net, dragnet and throw-net fishing effort; of these only set nets are thought to pose a risk to dolphins. During data compilation, obvious throw-net and drag-net effort was excluded, but it is likely that the conflation of these three effort types nonetheless creates additional noise within the data.

Locations around the North Island with the highest recreational set net effort included the Hauraki Gulf and the area from the Kāpiti Coast northward to Wanganui. High recreational effort locations around the South Island included Golden Bay and Tasman Bay, and to a much lesser extent Banks Peninsula and the coastline adjacent to Invercargill. The spatial distribution of recreational effort used in the risk assessment is shown in figure A12 of Roberts et al. (2019a).

For all subpopulations, recreational fisheries overlap was estimated to be much higher in the summer period (Table 6.7). When scaled for population size, the highest overlap (and hence the highest estimated encounter rate per dolphin with recreational netting events) was estimated for the 'Taranaki to Kāpiti' (TAKA) sub-area. Despite the absence of evidence of a resident dolphin population here, these results indicate that any dolphin transiting this area is more than three times as likely to encounter recreational set net effort relative to dolphins in any established subpopulation. The second highest relative risk was estimated for the NCSI population (Golden Bay and Tasman Bay); the lowest levels were estimated for the large Hector's dolphin populations of the ECSI and WCSI (Table 6.7), reflecting historical fishery closures in 2008 that substantially reduced opportunities for recreational set netting. The recreational set netting that remains in these areas is confined to harbours and subject to seasonal restrictions.

Potential risks to dolphins arising from illegal set net fishing was also discussed. Spatial patterns of illegal set net behaviour effort are inherently difficult to estimate, but it is likely that where recreational set netting is uniformly banned over large areas then illegal fishing behaviour will also diminish, because uniform regulations become easier to enforce.

6.4.4.3 NON-LETHAL OR HABITAT-DISRUPTIVE THREATS

With respect to aquaculture facilities, regions of relatively high spatial overlap were limited to a small number of farms off the ECSI (Pegasus Bay and Cloudy Bay) and NCSI (Golden Bay and Tasman Bay). When scaling for population size, the highest overlap (i.e., highest likelihood of encounter per dolphin) was estimated for the NCSI subpopulation, However, the spatial extent of this overlap was small (Table 6.7).

Spatial oil spill risk is relatively high on the north coast of the North Island, in Cook Strait, and off northern Banks Peninsula. Of these locations, only northern Banks Peninsula has a high estimated density of Hector's dolphins, and so this location has the greatest threat to dolphins in terms of number of dolphins that might be affected (if oil spill events are consistent with the estimated spatial threat intensity). When scaling for population size, the ECSI had the greatest overlap with oil spill risk (Table 6.7).

The spatial cumulative underwater noise from vessel traffic (using AIS data) and selected oil and gas seismic surveys were estimated for the region to the west of the North Island from noise modelling by McPherson et al. (2019) (Figure 6.12). This modelling was supplemented by a review of the potential impacts of petroleum and mineral exploration and production on Hector's and Māui dolphins by Lucke et al. (2019), which illustrated the spatial distribution of seismic surveys around New Zealand since 1960. The most intensive historical activity was to the west of the North Island, including survey activity prior to 2010 in regions that would overlap with regions of moderately high Hector's and Māui dolphins (although not since 2010).

Lucke et al. (2019) concluded that seismic surveys and offshore pile driving pose the greatest risk for auditory impairment, but that the risk from lower frequencies primarily emitted by seismic testing is lower if the probable frequency-specific sensitivity of Hector's dolphin is considered (i.e., because these dolphins have a high-frequency auditory and vocalisation range; note however that cetaceans often respond to sounds outside the frequency range of their own vocalisations). Behavioural reactions (i.e., spatial avoidance) were considered the most probable responses to the assessed noise sources and expected sound exposure levels, but scientifically-robust data are lacking for assessing the behavioural responses of Hector's and Māui dolphins to sound. Elsewhere Leunissen & Dawson (2018) and Leunissen et al. (2019) document behavioural responses to pile driving / construction noise in the inshore environment, showing that spatial displacement can occur.

Table 6.7: Relative overlap between threats and Hector's/Māui dolphins by threat, subpopulation and season. Rescaled as a proportion of the maximum value for a respective threat across all subpopulations and both seasons. ECSI = East Coast South Island, SCSI = South Coast South Island, WCNI = West Coast North Island, TAKA = South Taranaki and Kāpiti Coast, WCSI = West Coast South Island, NI = All other coasts of the North Island, NCSI = North Coast South Island.

Subpopulation	Toxoplasmosis	Predation	Recreational netting	Oil spill risk	Aquaculture
Summer					
ECSI	0.15	0.16	0.04	0.95	0.12
SCSI	0.19	0.69	0.21	0.36	0.02
WCNI	0.20	0.23	0.16	0.08	0.00
TAKA	0.24	0.20	1.00	0.30	0.00
WCSI	0.48	1.00	0.02	0.01	0.00
NI	0.14	0.01	0.41	0.77	0.20
NCSI	0.06	0.29	0.29	0.19	1.00
Winter					
ECSI	0.26	0.15	0.01	1.00	0.17
SCSI	0.34	0.68	0.02	0.38	0.01
WCNI	1.00	0.20	0.02	0.10	0.00
TAKA	0.82	0.22	0.09	0.34	0.00
WCSI	0.71	0.98	0.01	0.01	0.00
NI	0.73	0.01	0.05	0.67	0.20
NCSI	0.12	0.33	0.05	0.18	0.80

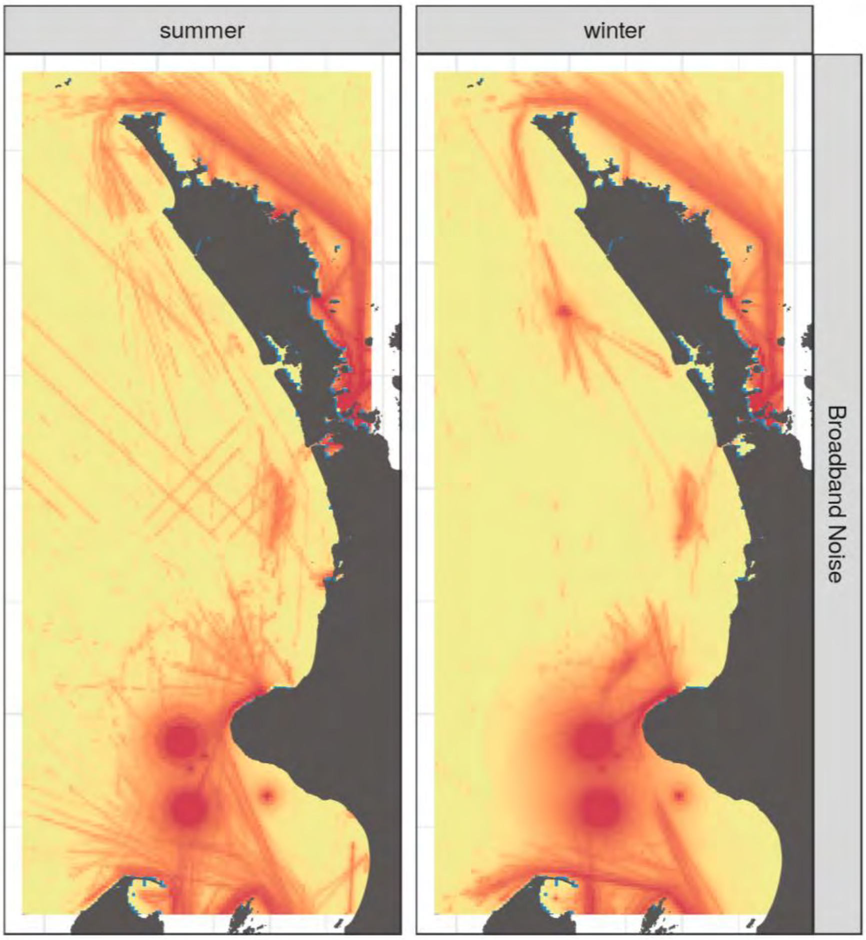

Figure 6.12: Estimated cumulative high-frequency noise in summer (left) and winter (right). Spatial estimates from noise modelling by McPherson et al. (2019).

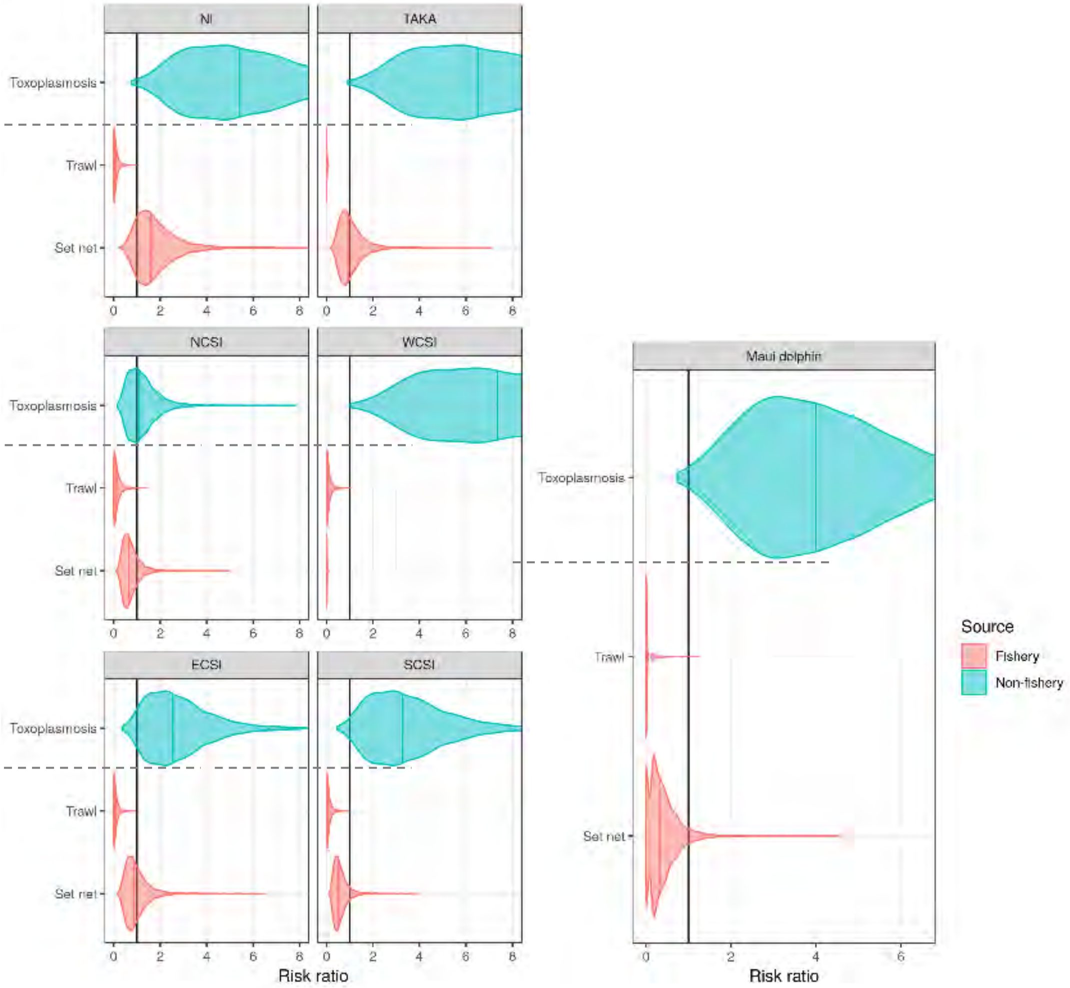

Figure 6.13: Annual commercial fishery (set net and inshore trawl) and toxoplasmosis risk ratios for Hector's dolphins by subpopulation area, under the base case (equal detection probability) scenario for non-fishery causes of death. The median and 97.5% quantile are indicated as vertical lines within each density. Dashed lines delineate threats for which differing methods were used to estimate annual risk ratio (above the line = based on proportions in the necropsied sample; below the line = using fisheries observer data). Reproduced from Roberts et al. (2019a).

6.5 HECTOR'S AND MĀUI DOLPHIN DEMOGRAPHIC POPULATION MODELS

In 2018–19 Fisheries New Zealand commissioned two separate demographic population assessments for Māui dolphins, to inform the revised TMP for Hector's and Māui dolphins.

6.5.1 POPULATION-BASED MODELS

A Bayesian population model described by Roberts et al. (2019b) was developed for Māui dolphins using the SeaBird demographic software (e.g., Roberts & Doonan 2016) integrating information from genetic "mark-recapture" observations and a population size time series, also from genetic biopsy (e.g., Baker et al. 2016b). Model runs made alternative assumptions of historical threat-specific mortality from direct fishery interactions (set net and trawl) and toxoplasmosis, as estimated by the spatial risk assessment of Roberts et al. (2019a). Models estimated a higher median annual non-calf (1+) survival probability for females (~0.89) than for males (~0.83), consistent with other assessments using the same data (e.g., Cooke et al. 2019).

Because current annual commercial fishery deaths are estimated to be low relative to other causes of death, model projections estimated only a minor effect of alleviating estimated trawl and set net mortalities, even when the upper 95% credible interval estimate of annual deaths (from the spatial risk assessment) was assumed. Where adult survival was estimated in a single time block (i.e., assuming constant 'background' mortality with only the fisheries deaths changing), this effect was insufficient to stabilise or reverse a declining population trend. In contrast, model runs alleviating toxoplasmosis deaths and commercial fishery deaths simultaneously produced increasing or stable population trends, depending on the detection probability of toxoplasmosis deaths relative to predation events (Figure 6.14)

6.5.2 INDIVIDUAL-BASED MODELS

Cooke et al. (2018) fitted an individual-based model to a time series of genetic mark-recapture data (Baker et al. 2013, 2016b) to estimate survival and other demographic rates. Cooke et al. (2019) modified this model to incorporate (from the spatial risk assessment by Roberts et al. 2019a) priors for biological parameters and for empirical estimates of time-varying annual commercial fisheries exploitation rates. Cooke et al. (2019) then used this model to simulate forward population trajectories for Māui dolphins under four sets of structural assumptions, as follows:

a) commercial fisheries deaths estimated by the spatial risk assessment are accurate in an absolute sense, there are no other anthropogenic deaths. These model runs did not fit the observed population trend.

b) commercial fisheries deaths estimated by the spatial risk assessment are accurate as a relative index only; there are no other anthropogenic deaths. These model runs fit the observed population trend but required that the catchability of Māui dolphins per encounter with fishing effort is roughly 10–20x higher than was estimated empirically for Hector's dolphins. Such a dramatic difference in inherent characteristics between subpopulations is considered implausible.

c) Commercial fisheries deaths estimated by the spatial risk assessment are accurate, and there are other (unspecified) anthropogenic causes of death that are constant over time. These runs estimate that 2.9–4.3 excess deaths per year are required to best fit the observed population trend.

d) commercial fisheries deaths estimated by the spatial risk assessment are accurate, and toxoplasmosis is also present, at levels estimated in the model based on WCNI necropsy results. These runs suggest that toxoplasmosis is responsible for 2.8–4.1 deaths per year, comparable to all excess deaths required to best fit the observed trend under group c) above.

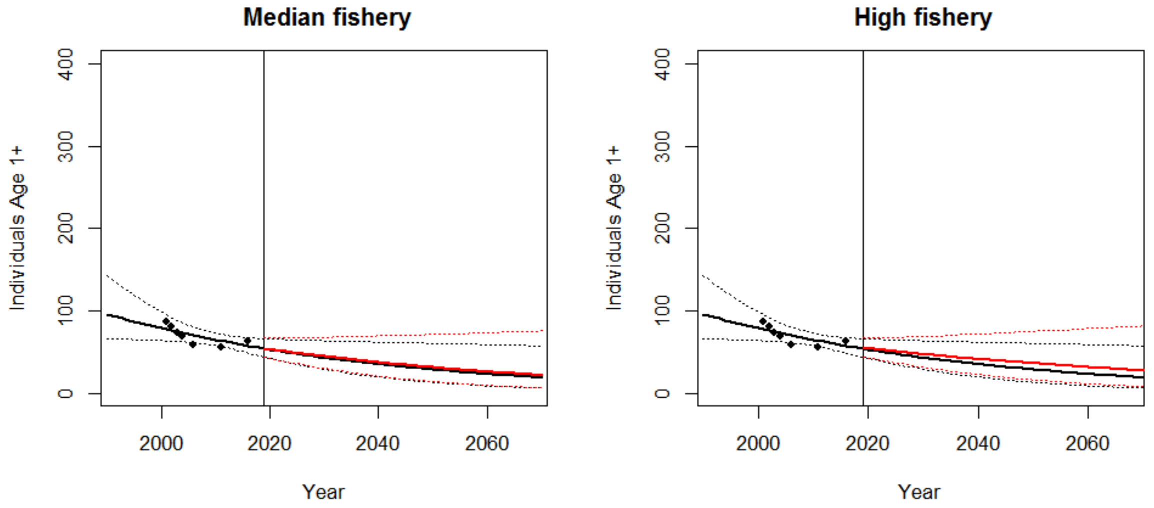

Figure 6.14: Māui dolphin population projections with alternative assumptions with respect to alleviating future threat-specific annual deaths (from Roberts et al. 2019b). Current year (2019) is highlighted by vertical line. To the left of this line: black lines are the median and 95% CI of MCMC estimates of non–calf population trajectory; and census estimates are indicated by black points. To the right of the vertical line: black lines are the projected population trajectory for a model continuing recent demographic rates. Projections *alleviating* threat-specific mortality are shown in red: median estimates of annual set net and trawl deaths (top left); the upper 95% estimates of annual set net and trawl deaths (top right); the median fishery deaths and toxoplasmosis, assuming equal detection probability of non–fishery causes of death (bottom right); and the median fishery deaths toxoplasmosis, assuming 10–fold decrease in detection probability of predation mortality (bottom left). For all trajectories, the three lines represent median and 95% CI of MCMC estimates. (Continued next page)

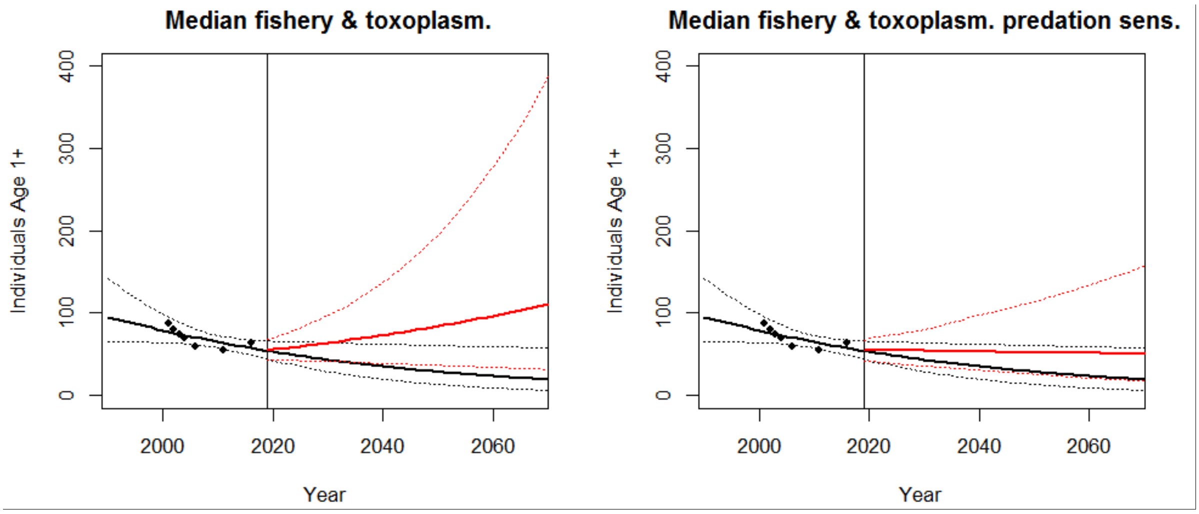

Figure 6.14: Māui dolphin population projections with alternative assumptions with respect to alleviating future threat-specific annual deaths (from Roberts et al. 2019b). Current year (2019) is highlighted by vertical line. To the left of this line: black lines are the median and 95% CI of MCMC estimates of non–calf population trajectory; and census estimates are indicated by black points. To the right of the vertical line: black lines are the projected population trajectory for a model continuing recent demographic rates. Projections *alleviating* threat-specific mortality are shown in red: median estimates of annual set net and trawl deaths (top left); the upper 95% estimates of annual set net and trawl deaths (top right); the median fishery deaths and toxoplasmosis, assuming equal detection probability of non–fishery causes of death (bottom right); and the median fishery deaths toxoplasmosis, assuming 10–fold decrease in detection probability of predation mortality (bottom left). For all trajectories, the three lines represent median and 95% CI of MCMC estimates. (Continued)

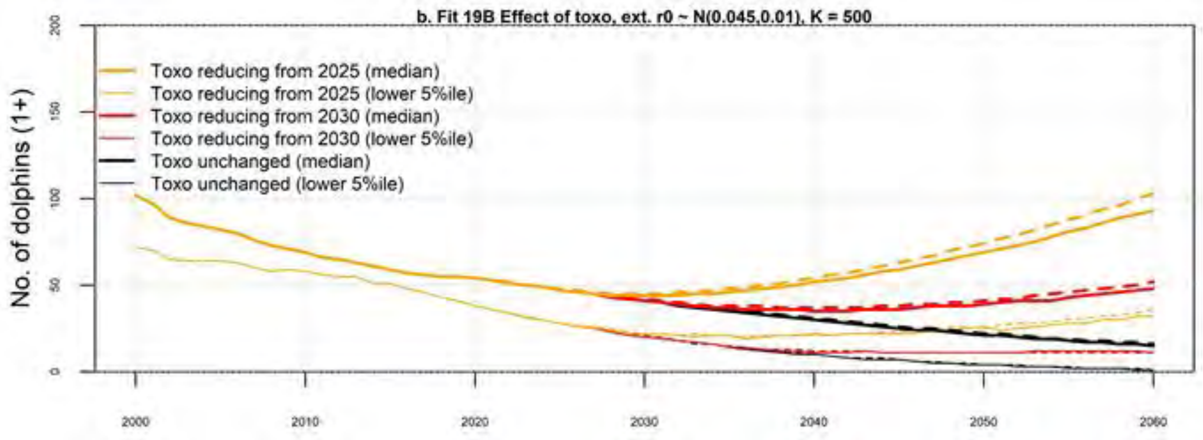

Figure 6.15: Māui dolphin population projections from Cooke et al. (2019) simulating effects of alleviating toxoplasmosis on different time frames (median and lower fifth percentile; solid lines) and eliminating commercial fisheries risk (dashed lines). This figure illustrates a model run under assumption set 'd' described in section 6.5.2, under which fisheries deaths are as estimated by Roberts et al. (2019a) and toxoplasmosis risk levels are estimated independently using North Island necropsy results.

On this basis, the model described by Cooke et al. (2019) suggests that one or more non-commercial fisheries threats may collectively be responsible for 3–4 Māui dolphin deaths per year. Assuming that toxoplasmosis is the sole non-commercial-fisheries threat, population projections under assumption set d) estimated that action to reduce the impact of toxoplasmosis would need to be successful within 5–10 years if Māui dolphins are to avoid the risk of extinction (Figure 6.15). In these simulations, toxoplasmosis risk is assumed to be constant in time beginning in 2000 and continuing until such time as risk reduction efforts begin to take effect, in either 2025 or 2030. The black line assumes no reduction in toxoplasmosis risk; the red and orange lines assume that toxoplasmosis risk is reduced by half each decade, beginning either in 2030 or 2025, respectively. The relative effect of also eliminating fisheries risk as estimated by Roberts et al. (2019a) is also shown.

Note that all of these sets of model runs assumed that excess deaths from all causes except commercial fisheries are constant over time. However recreational fishing risk may have been substantial prior to 2008 but mostly eliminated thereafter, raising the possibility that at least a portion of those 3–4 excess deaths attributed here to toxoplasmosis were actually attributable to recreational fishing and have already been eliminated. Such a hopeful interpretation is consistent with the results of Roberts et al. (2019b) in which adult survival was estimated to have improved in the period after 2008. However this result is highly uncertain, effectively fitting to a single data point from the genetic census of Baker et al. (2016b) suggesting a slight population increase between 2010 and 2015 (but with overlapping confidence intervals). Results of a repeat genetic census in 2020–21 will help to resolve these uncertainties.

6.5.3 CHARACTERISATION OF DOLPHIN DEATHS RECORDED IN DOC INCIDENTS DATABASE

Further characterisation of beach-cast dolphins in the DOC Hector's and Māui dolphin incidents database was undertaken to search for patterns potentially indicative of the nature and magnitude of lethal impacts on the dolphins, and of potential biases arising from use of these data to infer cause of death (Roberts in prep). This analysis found evidence for strong seasonality in the proportion of recovered carcasses attributed to different causes of death. Causes of death showing strong seasonality included entanglement mortalities in fishing gear (primarily in summer), disease mortality (late winter/early spring), neonate deaths (summer), and other mortality categories (variable across seasons; Roberts, unpublished data).

Other patterns were suggestive of seasonal threats that differentially affected dolphins by sex. Most significantly, among non-calf beach-cast carcasses in late winter and early spring months (August to October), and for which the sex was determined, 29 of 35 carcasses were female (Figure 6.16). The corresponding proportion of females (0.83) was significantly different from 0.50 ($p < 0.001$, 2-sided). This is coincident with the period in which all known toxoplasmosis mortalities have occurred to date (September-November). The female bias in mortalities during early spring was evident from 2000 to 2020, but not in the preceding period (Figure 6.17), potentially suggesting a change over time in the impact of whatever threat is responsible for this pattern. Increased carcass recovery rates in winter and early spring, and increased resourcing for necropsy programmes, may help to identify and understand the full range of threats affecting Hector's and Māui dolphins.

There was only tentative, non-significant evidence for changes in the demographic composition of fishery entanglement mortalities with the implementation of different fishery area restrictions around New Zealand. Fewer females were confirmed amongst incidental mortalities since the implementation of wide-ranging fishing restrictions in 2008–09, although a large component of the mortalities during this period that were in the database were not necropsied and sexed. Further analysis of stored tissue samples to sex and, if possible, age these animals may reveal important information.

The study by Roberts (in prep) also developed a rudimentary proxy for seasonal carcass detection probability, based on seasonal public beachgoer count data. This suggested that carcass detection probability is likely to be highly seasonal, such that threats that primarily kill dolphins outside the summer period (e.g., diseases) will be under-represented in the beach-cast sample of dolphins relative to those that kill dolphins in summer (e.g., bycatch and the separation of neonates from their mothers). This conclusion highlights the importance in particular of improving data collection to better understand whatever threats are responsible for the preponderance of female deaths in late winter/early spring. If seasonal detection bias results in these carcasses being under-represented in the beach-cast and necropsied sample, this may produce a negative bias in the associated threat-specific risk estimates.

6.5.4 PREVIOUS DEMOGRAPHIC POPULATION MODELS

A number of modelling exercises have aimed to assess the effect of various proposed management approaches on the future population trajectory of Hector's and Māui dolphins. (Martien et al. 1999, Burkhart & Slooten 2003, Slooten 2007, Slooten & Dawson 2010, Davies & Gilbert 2003, Davies et al. (2008).

The various models share some necessary similarities given the available information:

- each assumes a particular form of population model and uses this to project dolphin numbers forward and backward from a single population estimate;

- none of the models used the most recent aerial survey derived estimates of abundance and distribution in SCSI and ECSI;
- none of these models used spatially explicit estimates of overlap with fisheries to estimate encounter rate and capture rate per encounter, instead a single estimate of dolphin capture rate from the ECSI was applied to historical fishing effort levels to estimate fishing-related dolphin mortalities in other subpopulations.

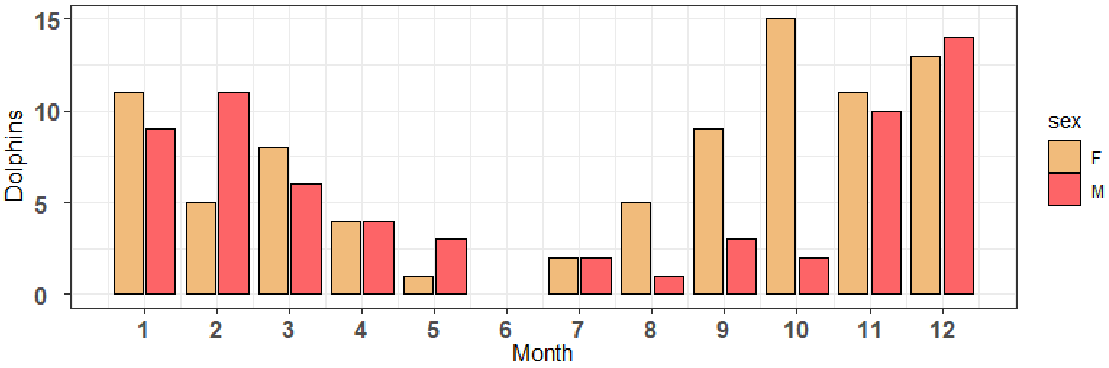

Figure 6.16: Sex composition of adult Hector's or Māui dolphins found beach-cast (dead on shore) since 1984/85, by month of reporting. This plot excludes juveniles and individuals for which the sex was not determined, and a small number of records prior to 1984/85.

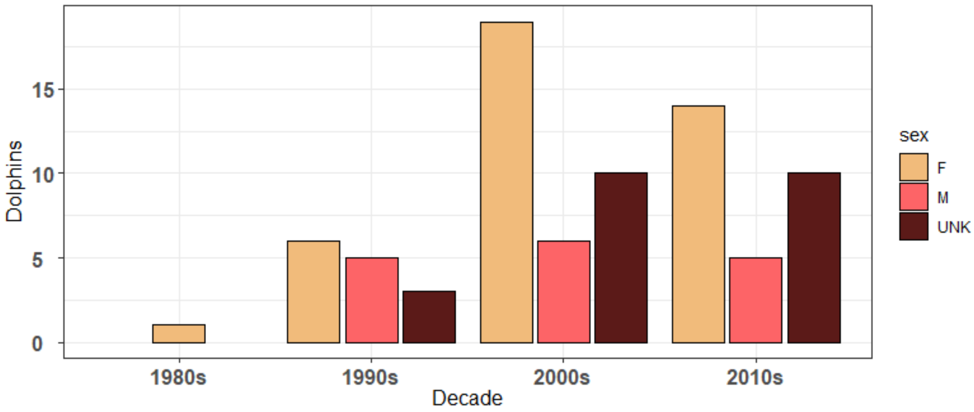

Figure 6.17: Sex composition of Hector's or Māui dolphins found beach-cast (dead on shore) in late winter and spring (August to November) by decade. This plot excludes a small number of records prior to 1984/85. Reproduced from Roberts (2020).

In multiple rounds of review and discussion, the AEWG identified that both forward and rearward population projections using many of these models were reliant on informative priors or strong structural assumptions, and may be sensitive to input parameters that derive from earlier analyses that may become dated, in particular if input parameterisation was a function of population size, or dependent on assumptions about population productivity (r^{max}). The AEWG noted that model estimates were likely to be more reliable for local dolphin populations near Banks Peninsula where most of the data was collected, but the population size or spatial scale to which model outputs could be safely applied was not always clear. The AEWG concluded that if the outputs of these analyses were to be communicated to managers, it should be with appropriate caveats.

6.5.5 BANKS PENINSULA DEMOGRAPHIC MARK-RECAPTURE

Demographic models and analyses based on an ongoing mark-recapture database of dolphins sighted near Banks Peninsula have produced a number of potentially valuable publications to understand population dynamics of Hector's dolphins (e.g., Cameron et al. 1999, Du Fresne 2005; Gormley et al. 2012). In particular the conclusions of Gormley et al. (2012) are frequently cited to infer fisheries mortality rates before and after the establishment of a spatial fisheries closure, but the spatial scale over which the apparent demographic signals are indicative of actual population dynamics has been unclear.

In 2020 MacKenzie & Roberts (in prep) analysed mark-recapture data given in the appendix of Du Fresne (2005), collected around Banks Peninsula from 1985 to 2002. The purpose of the analysis was to evaluate the sensitivity of the findings of Gormley et al. (2012) to alternative model structures. Gormley et al. (2012) analysed data for the period 1986–2006, sourced from the same database as Du Fresne (2005), and reported evidence of increased survival from 1990 onwards, which was interpreted as evidence for the beneficial effects of the Banks Peninsula Marine Mammal Sanctuary. The data used by Gormley et al. (2012) were unavailable for the reanalysis, hence the data from Du Frense (2004) were used for the period 1986–2002. Although the datasets are similar they are not identical for the overlapping time periods.

MacKenzie & Roberts (in prep) found that survival estimates could be sensitive to the model fit to the data, and that the structure used by Gormley et al. (2012) had greater support on the basis of leave-one-out information criterion (LOOIC; Gelman et al. 2014). Importantly, the survival estimates obtained by MacKenzie & Roberts (in prep) were substantially higher than those by Gormley et al. (2012) pre-1990, and slightly lower than Gormley's for post-1990, suggesting an overall decline in survival or occupancy from 1990 onwards. This is at odds with the results of Gormley et al. (2012), (Figure 6.18), but in keeping with the results of Cameron et al. (1999) who also estimated higher pre-1990 survival than Gormley et al. (2012).

The reversal of the apparent change in adult survival relative to the assessment by Gormley et al. (2012) was because different subsets of the mark-recapture data were included in these different analyses over time. It appears that the dataset available to the analysis by MacKenzie & Roberts (in prep) included some dolphin re-sights that had been used by Du Fresne (2005) but were reclassified or otherwise excluded from later mark-recapture analyses by Gormley et al. (2012). Researchers at Otago University report that data selection for inclusion in different analyses has changed over time to more robustly estimate adult survival rates for different subsets of the population. However the nature and consequences of these changes have not always been well documented.

MacKenzie & Roberts (in prep) noted that clarification is required on the following points to resolve conflicting estimates of how Hector's dolphin survival may have changed with the establishment of the BPMMS. Until that time, the authors judged, and the AEWG concurred, that any of the pre-1990 survival estimates from various sources should be regarded as provisional and not yet verified for use to inform management or policy decisions.

The following clarifications were requested:

- details on the spatial distribution of survey effort and dolphin sightings, and how that has changed over time.
- details on the number of individuals that have been reclassified or otherwise excluded, and which sightings are affected.
- sensitivity of the estimated time series of survival estimates to temporal changes in the spatial distribution of survey effort.
- sensitivity of any results to the spatial and temporal sub-setting of the full dataset for analyses.

The long-term mark-recapture data collected near Banks Peninsula are currently (June 2020) being analysed by researchers at Otago University. When these analyses are complete they will provide a significantly longer time series than has been available for any previous analysis estimating Hector's dolphin adult survival rates.

6.5.6 KEY AREAS OF UNCERTAINTY IN THE SPATIAL MULTI-THREAT RISK ASSESSMENT

6.5.6.1 SPATIAL UNCERTAINTY

The following are identified as areas where the outputs of the spatial multi-threat risk assessment may be more uncertain than elsewhere.

- Low dolphin density areas of Māui dolphin habitat. The spatial estimates of dolphin density are most accurate in locations with more dolphins and become less reliable (in a proportional sense not an absolute sense) in locations with very low dolphin densities. For this reason, fisheries risk estimates may be more uncertain in the following locations:

- inside west coast North Island harbours – the models estimate that dolphins enter the harbours very infrequently, but it’s possible that these estimates are wrong in either direction, including the possibility that dolphins never penetrate the interior of these harbours;
- the northern and southern extreme of the Māui dolphin distribution – the southern extreme is verified by sightings data; the northern extreme is unverified;
- The extreme offshore distribution of the Māui dolphin distribution – the habitat model is well-specified and verified by actual data out to around 10 to 12 nautical miles offshore, but at further ranges it predicts a uniformly low “background” density that never drops to zero even at very far distances offshore. It is likely that the model is overestimating the density of dolphins at distances far offshore.
- The Cape Egmont to Wellington “transition” zone. Dolphin density estimates are assigned arbitrarily to illustrate what risk dolphins would face if they were resident in this area. It appears instead that dolphin sightings in this area are of transient or dispersing dolphins; actual densities are unknown.
- • North coast South Island. The estimated spatial distribution here is more uncertain than the east and west coast South Island subpopulation distributions, due to very few aerial survey observations.
- • South coast South Island. The estimated spatial distribution here is more uncertain than the east and west coast South Island subpopulation distributions, due to the absence of a key data layer in the spatial model (representing availability of dolphin prey).

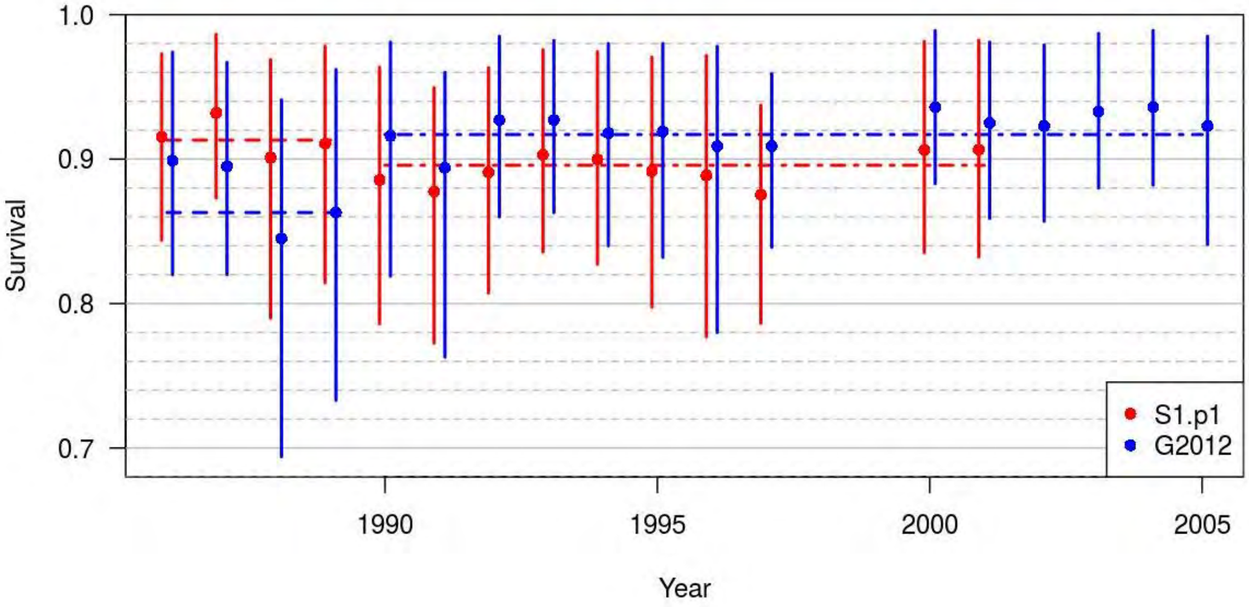

Figure 6.18: Comparison of estimated survival probabilities (mean and 95% c.i.) obtained from the data used by MacKenzie & Roberts (in prep; S1.p1, in red) to those reported by Gormley et al. (2012; G2012, in blue). The Banks Peninsula Marine Mammal Sanctuary was established in 1989; horizontal lines indicate the posterior mean of the mean survival probability from each assessment for the defined pre- and post-sanctuary periods.

6.5.6.2 POPULATION UNCERTAINTY

At the scale of whole subpopulations, the west, east, and south coast South Island subpopulations and Māui dolphin populations are estimated using consistent methods for which estimates of statistical uncertainty are thought to be accurate and unbiased. However:

- North coast South Island population size. Very little is known about how many dolphins there are in the north coast South Island subpopulation. Uncertainty about population size means that estimates of total deaths are also uncertain but estimates of risk (that is, probability of death per individual animal) are unaffected.

Local population sizes in other locations with low population density may be poorly estimated by aerial surveys.

6.5.6.3 FISHERIES RISK UNCERTAINTY

Set net catchability (probability of capture per encounter) is well estimated. Trawl catchability is estimated with less precision but is known to be much lower than for set nets in an absolute sense. However:

- Cryptic mortalities (unobservable captures) are uncertain for set net fisheries.

- The number of animals dying per trawl event is not well estimated. Evidence suggests that trawl captures may arise from social interactions that sometimes involve more than one dolphin. The model run currently used to inform the TMP assumes that each trawl capture event kills two dolphins on average, but this multiplier is uncertain.

- Fishery groups. All set net fisheries are treated as a single group, and all inshore trawl fisheries are treated as a single group, for purposes of estimating catchability. If some fishers use different gear, or have consistently different behaviour in ways that affect dolphin catchability, then this contrast will not be reflected in local risk estimates. However, unless fisheries observer coverage is also biased, the result will be increased uncertainty but not systematic bias.

- Harbour set nets. The risk assessment model treats WCNI harbour set nets as if they have the same probability of catching a dolphin as do set nets in offshore areas. This approximation may not be valid, so the estimate of risk in harbours is more uncertain than reflected numerically.

- Recreational fisheries risk is not estimated quantitatively. In areas where recreational fishing is still permitted, this could be a substantial but unquantified cause of death. In areas where recreational fisheries previously occurred, but were subsequently eliminated, this could cause a major historical change in threat level that the model is unable to estimate.

6.5.6.4 NON-FISHERIES RISK UNCERTAINTY

The use of beach-cast dolphin carcasses to estimate rates of death relies on assumptions about the rate that carcasses are recovered for necropsy and may be biased. As a consequence:

- Toxoplasmosis death estimates are more uncertain than represented in the numerical estimates and could be biased either high or low due to factors affecting carcass detection rate.

- A possible sex bias in toxoplasmosis deaths (if more females are dying) may have important implications; if the sex bias is real, then toxoplasmosis risk is higher than estimated by Roberts et al. (2019a).

Brucellosis is grouped under 'other' causes of death in Roberts et al. (2019a) hence effectively treated as a component of natural mortality. If brucellosis risk is related to anthropogenic causes or affects some subpopulations in particular, then this treatment may underestimate the importance of this threat.

Non-lethal habitat disruptive threats cannot be quantified:

- Seismic risks from underwater sound are estimated in the risk assessment in a relative sense only, and only for Māui dolphins. Although the level of sound the dolphins experience has been estimated quantitatively, it is unknown how this level of sound may affect dolphins.

6.6 MANAGING FISHERIES AND NON-FISHERIES RISK

The following section describes management tools and measures already in place to manage fisheries and non-fisheries risks to Māui and Hector's dolphins (as of June 2020). These measures do not reflect new management under a revised TMP in 2020; options for such measures are currently being considered by Ministers.

To reduce fisheries risk to Hector's and Māui dolphins, restrictions on commercial and recreational set net, driftnet and trawl fisheries have been established under both the Marine Mammals Protection Act 1978 and Fisheries Act 1996. The first protected area designated for this purpose was with the establishment of a sanctuary at Banks Peninsula in 1988. In 2007, the Hector's and Māui dolphin Threat Management Plan (TMP) was developed by DOC and the former Ministry of Fisheries and included restrictions under the Fisheries Act. Subsequently four additional Marine Mammal Sanctuaries were established in 2008; note however that spatially these later MMS designations were wholly contained within the boundaries of existing fisheries closures (below) so had the effect of managing non-fishery risks rather than further reducing fisheries risk.

6.6.1 MARINE MAMMAL SANCTUARIES

1) The *Bank's Peninsula Marine Mammal Sanctuary* was established in 1988, the first Hector's dolphin sanctuary. This region on the east coast of the South Island is a dolphin hotspot and was subject to high levels of bycatch from recreational and commercial set nets at least from the 1970s, until at least as recently as the mid-1980s (Dawson 1991, Dawson & Slooten 1993). The sanctuary was extended in 2008 and now covers 389 km of coastline, extending from the southern end of the Rakaia River mouth to the northern end of the Waipara River mouth and out to 12 nm (22.2 km) offshore.

2) The *Clifford and Cloudy Bay Marine Mammal Sanctuary* was established in 2008 and covers an area of 338 km of coastline from Cape Campbell to a point 12 nm (22.2 km) offshore in a direct line to Tory Channel, northeast South Island. This region is an area with relatively high numbers of Hector's dolphins observed over 20 km offshore (Du Fresne & Mattlin 2009, MacKenzie & Clement 2014, Hamner et al. 2017).

3) The *Catlins Coast Marine Mammal Sanctuary* was established in 2008, covering an area with small, genetically isolated populations of Hector's dolphins from Porpoise Bay and Toetoes Bay, southeast South Island (Bejder & Dawson 2001, Hamner et al. 2012, MacKenzie & Clement 2018). The sanctuary covers 161 km of coastline extending from Three Brother's Point offshore 5 nm (9.3 km) to a point 6.9 nm (12.8 km) offshore from Bushy Point Beacon.

4) The *Te Waewae Bay Marine Mammal Sanctuary* was established in 2008, covering 113 km of Southland coastline. The boundaries encompass Te Waewae Bay from Pahia Point to Sand Hill Point into shore. This sanctuary covers the main habitat and most of the population of the genetically distinct SCSI population of Hector's dolphins (Hamner et al. 2012, Rodda & Moore 2013, MacKenzie & Clement 2019).

5) The *West Coast North Island Marine Mammal Sanctuary* was established in 2008 and is the largest, covering 2164 km of coastline from Maunganui Bluff, Northland to Oakura, Taranaki. In 2013, there was an amendment under the Marine Mammals Protection Act 1988 to the activities restricted within a portion of the sanctuary due to a new abundance estimate and bycatch event off Taranaki. This is the most complex of the protected areas; under the Fisheries Act 1996, commercial and recreational set net, driftnet, and trawl restrictions are in place with variations by location. Protection also includes an intensive fisheries observer programme for the set net fisheries in the southern Taranaki region, and trawl fisheries between Maunganui Bluff and Pariokariwa Point.

6.6.2 SPATIAL CLOSURES UNDER THE FISHERIES ACT

In 2008, an extensive package of spatial closures was implemented under the Fisheries Act to reduce fisheries risk to dolphins, largely superseding the existing discrete closures under the Marine Mammal Protection Act. In the North Island these restrictions were extended further in 2012 and 2013. Current spatial fisheries closures are depicted in Figures 6.19–6.22 and described below.

On the WCNI, the set net restrictions on were extended to 7 nm offshore between Maunganui Bluff and Pariokariwa Point (including the entrances to the Kaipara, Manukau, and Raglan harbours and the entrance to the Waikato River). Trawling was prohibited to 2 nm offshore between Maunganui Bluff and Pariokariwa Point, and to 4 nm offshore between Manukau Harbour and Port Waikato. In 2012, the set net restrictions on the WCNI were extended further south, banning commercial and recreational set netting to 2 nm offshore from Pariokariwa Point to Hawera. New requirements were also implemented requiring an MPI observer on any commercial set net vessel operating within 7 nm of shore. In 2013, the set net restrictions were extended again, banning commercial and recreational set netting between 2 and 7 nm from Pariokariwa Point to the Waiwhakaiho River mouth.

On the ECSI and SCSI, most set netting was prohibited within 4 nm of the coast, and trawling within 2 nm offshore was limited only to trawl vessels employing nets with a low headline height (generally targeting flatfish) on the presumption that dolphin catchability using low headline height gear is likely to be lower[17].

On the WCSI, recreational set netting was banned within 2 nm of the coast and commercial set netting was subject to a seasonal restriction (Figure 6.20).

6.6.3 OPTIONS FOR FISHERIES RISK MITIGATION

To date most management of fisheries risk to Hector’s and Māui dolphins has emphasized spatial management, to reduce the overlap between dolphins and fishing effort distributions. Other options for gear modifications aimed at reducing cetacean captures in set net fisheries include changing the way that fishing gear is deployed to reduce the risk of entanglement (e.g., Hembree & Harwood 1987) or adding acoustic alarms (pingers) to alert dolphins to the presence of the gear (Dawson et al. 2013b). Some ECSI set net fishermen use pingers under a voluntary Code of Practice (Southeast Finfish Management Company 2000). The effectiveness of pingers has been demonstrated in some experimental trials for other small cetaceans (e.g., Kraus et al. 1997, Trippel et al. 1999, Bordino et al. 2002; see review in Dawson et al. 2013b); however their utility is not universal, for example because cetaceans can become habituated to their presence (Cox et al. 2001) or because they are not always properly deployed (Cox et al. 2007, Dawson et al. 2013b).

To address fisheries risk in trawl fisheries, Fisheries New Zealand is progressing new research (under project SEA2019-27) to investigate options for deployed hydrophone arrays that can detect the vocalisations of the dolphins during fishing operations, to better understand interactions between dolphins and fishing gear. Outcomes of this work will be used to inform further investigations of potential trawl risk mitigation systems, e.g., to alert fishers in real time to the presence of dolphins near the gear.

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[17] Detailed descriptions of the restrictions can be found at: Ministry for Primary Industries. Protecting Hector’s and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

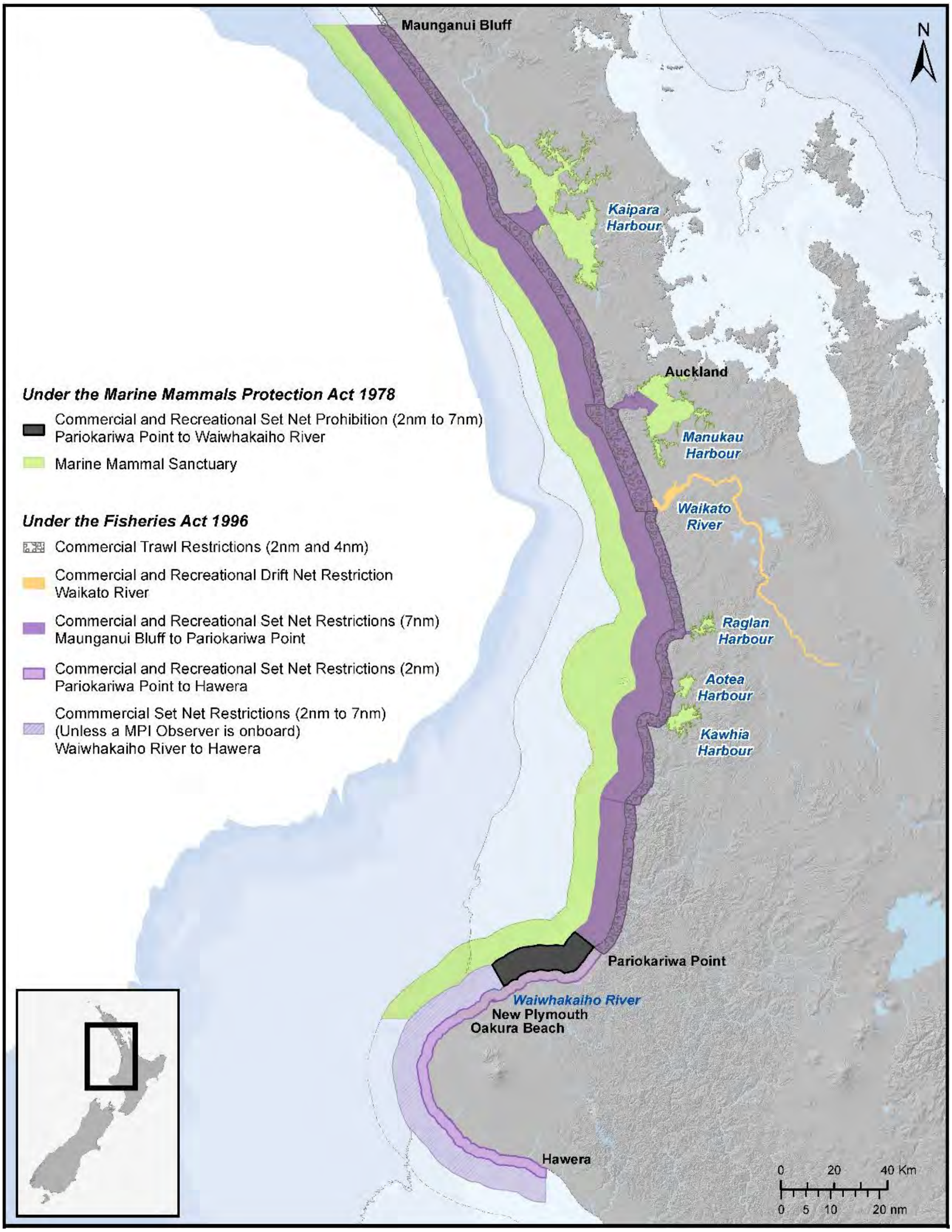

Figure 6.19: Summary of restrictions on commercial and amateur set netting on the WCNI. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector’s and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

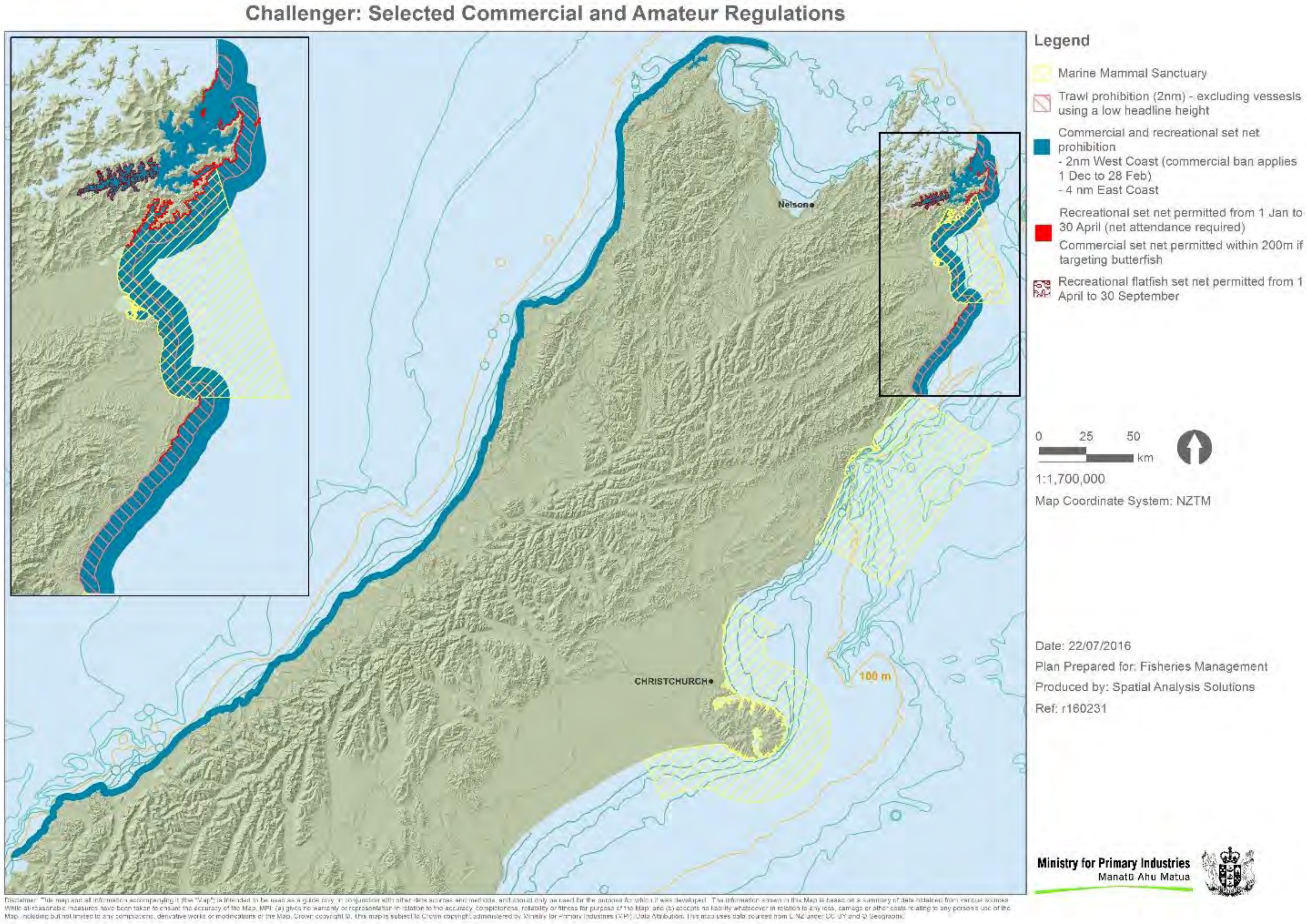

Figure 6.20: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the Challenger area (west coast) and north-east coast of the South Island. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

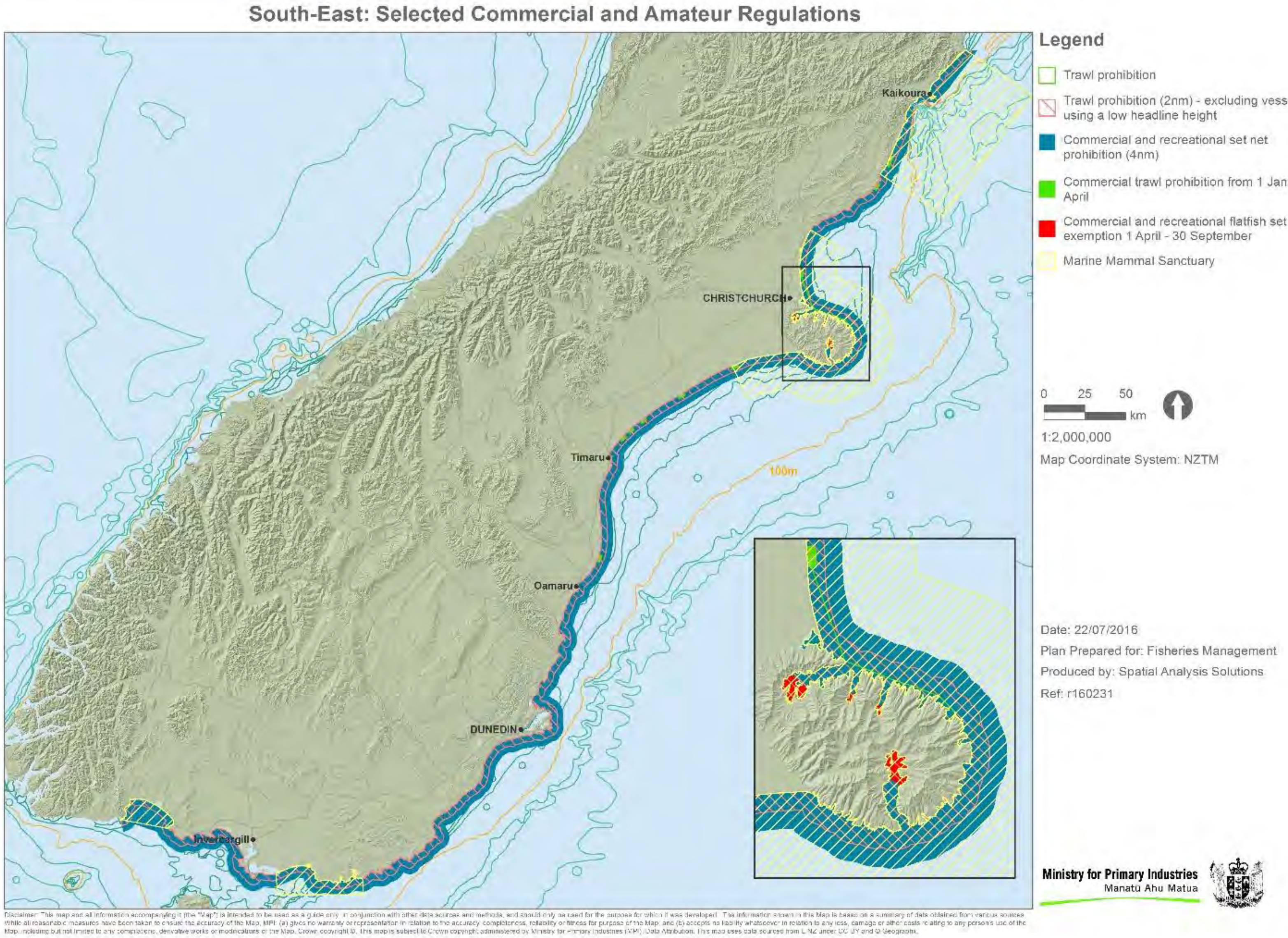

Figure 6.21: Summary of restrictions on commercial and amateur set netting, and commercial trawling in the south-east of New Zealand. For a full description of the restrictions, see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

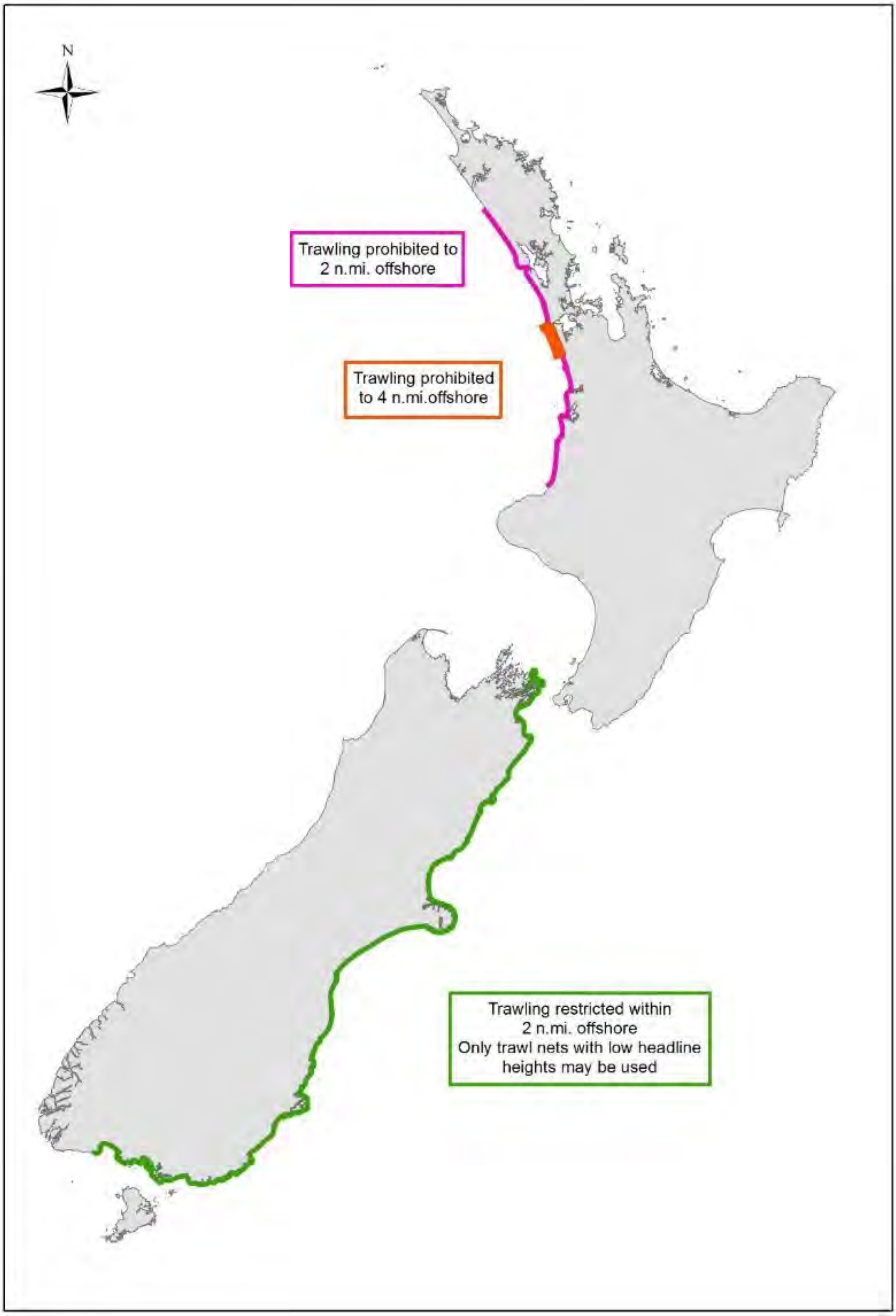

Figure 6.22: Summary of restrictions on trawling. For a full description of the restrictions see: Ministry for Primary Industries. Protecting Hector's and Māui dolphins. Retrieved from https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/protecting-hectors-and-maui-dolphins.

6.7 INDICATORS AND TRENDS

Population size	**Māui dolphins**: 55 (95% c.i.: 48–69) in 2010–11[18] 63 (95% c.i.: 57–75) in 2015–16[19] **ECSI Hector's dolphins**: Annual median estimate: 8968 (s.e.: 1377; 95% c.i.: 6649–12 096) Seasonal estimate: 9728 (CV: 17%; 95% c.i.: 7001–13 517) in summer 2012–13 and 8208 (CV 27%; 95% c.i.: 4888–13 785) in winter 2013 (out to 20 nm)[20] **WCSI Hector's dolphins**: Annual estimate: 5388 (CV = 21%; 95% c.i.: 3613–8034) in 2000–01 (out to 4 nm)[21] Annual median estimate: 5642 (s.e.: 936; 95% c.i.: 4085–7792) Seasonal estimate : 5490 (CV: 26%; 95% c.i.: 3319–9079) in summer and 5802 (CV: 21%; 95% c.i.: 3879–8679) in winter (out to 20 nm) [20] **SCSI Hector's dolphins**: Annual median estimate: (95% c.i. = 217-508) in 2018[20] Seasonal estimates: 177 (CV: 37%; 95% c.i.: 88–358) in March 2011; 299 (CV: 47%; 95% c.i.: 125–714) in August 2011[20]
Population trend	**Māui dolphins**: Declining over longer time period although some evidence of possible stabilisation from 2010/11 to 2015/16. **ECSI Hector's dolphins**: Unknown. Inconsistent evidence from abundance estimates, risk analyses and demographic estimates of population growth rates. **SCSI Hector's dolphins**: Unknown. Population size estimated but too uncertain to discern trend **WCSI Hector's dolphins:** Unknown; Population estimated but too uncertain to discern trend **NCSI Hector's dolphins**: Unknown; population size unknown
Threat status	**Māui dolphins**: NZ: Nationally Critical, Criterion A(1), Conservation Dependent in 2013[22] IUCN: Critically Endangered, Criteria A4c,d and C2a(ii) in 2013[23] **Hector's dolphins**: NZ: Nationally Vulnerable, Criterion D(1/1), Conservation Dependent in 2018[23] IUCN: Endangered, Criterion A4d in 2013[23]
Number of fisheries deaths (includes cryptic deaths)	Hector's dolphin set net: 44 (21–80) Hector's dolphin trawl: 14 (1–43) Māui dolphin set net: 0.10 (0–0.25) Māui dolphin trawl: 0.02 (0–0.05)
Trends in interactions	Hector's dolphin set net: stable Hector's dolphin trawl: decreasing Māui dolphin set net: decreasing Māui dolphin trawl decreasing [see Figure 6.9]

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Chapter 7: Common dolphin (*Delphinus delphi delphi*) - Technical Summary

Common dolphin
(*Delphinus delphi delphi*)

Not threatened (Baker et al. 2019)

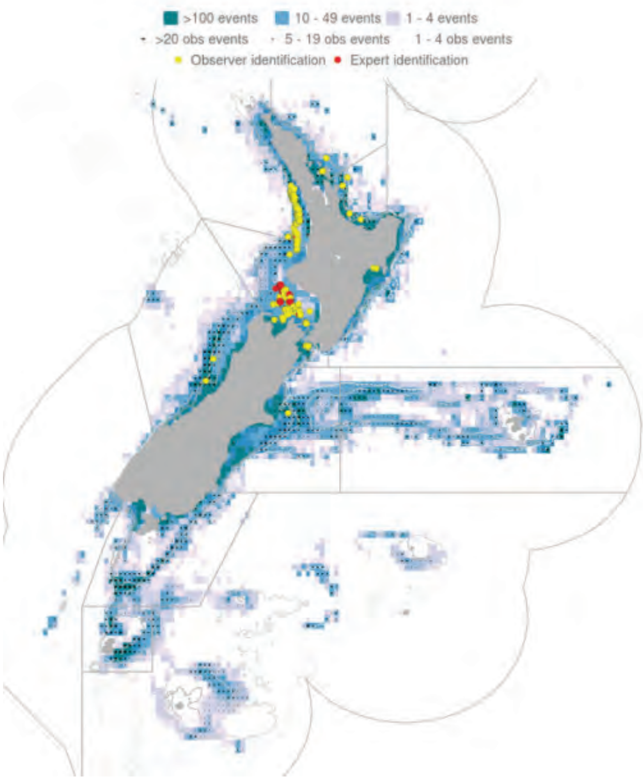
1. THE ISSUE IN BRIEF

- The common dolphin (*Delphinus delphi delphi*) is a marine mammal species with a worldwide distribution
- Common dolphins are abundant and generally not threatened worldwide. They are locally threatened in some areas (e.g., the Mediterranean Sea)
- Like other marine mammals, common dolphins are protected under the Marine Mammals Protection Act 1978 and the Fisheries Act 1996
- Tourism and fisheries are considered two of the greatest potential threats to common dolphin in NZ waters

2. POTENTIAL THREATS

Potential threats to common dolphins include stranding, diseases, natural predation, toxins, habitat loss, ship strikes, tourism-related disturbance, fishing mortality, and negative trophic interactions with fisheries

3. INCIDENTAL CAPTURES - LOCATION



- Between 2002–03 and 2017–18, there were 214 observed incidental captures of common dolphins in trawl fisheries, two in surface longline fisheries, and six in set net fisheries

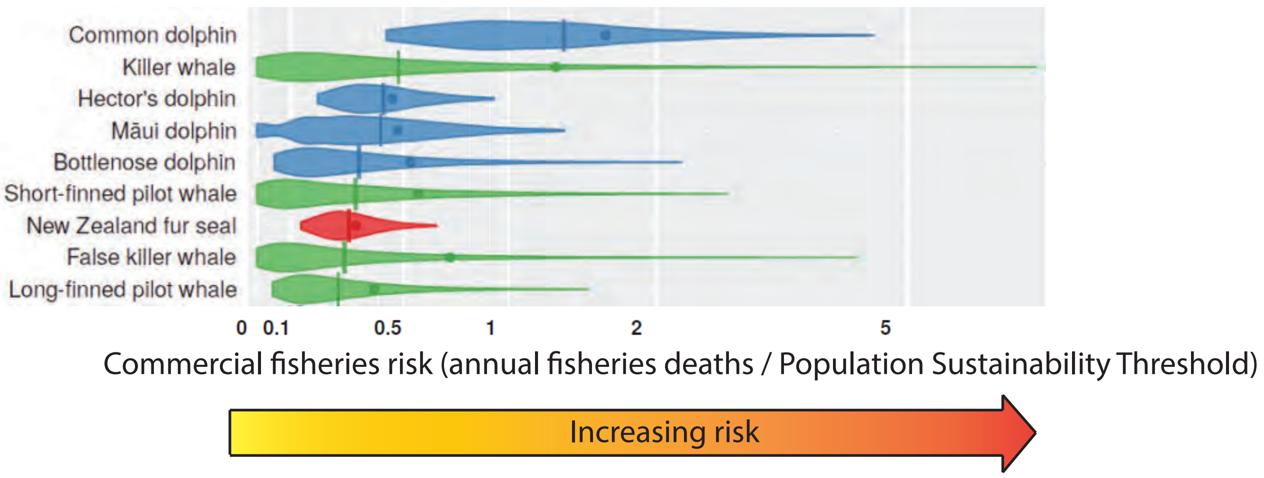
- The main trawl fisheries contributing to incidental captures of common dolphins are jack mackerel, as well as inshore and midwater (mainly hoki) trawls. The jack mackerel midwater trawl fishery contributed the most, and captures mostly resulted in fatal events

- The most common location for incidental captures has been the west coast of the North Island, with a hotspot in the Taranaki Bight

- Observations are limited due to the complexity of the inshore trawl fishery therefore we are uncertain about capture levels

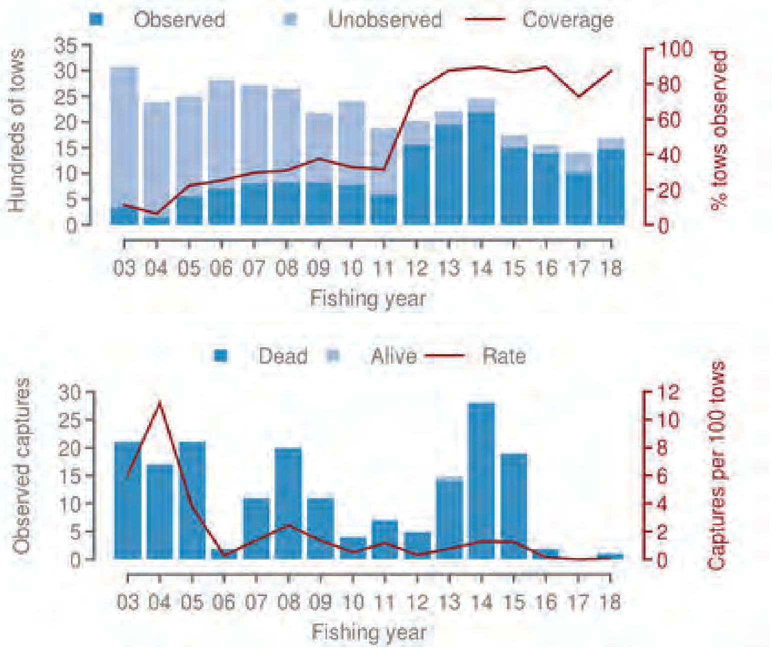
Map of common dolphin captures in NZ trawl fisheries between 2002 and 2018. Yellow and red dots indicate common dolphin capture events, reported by observers and experts, respectively. Blue shades represent the trawl fishing effort

4. RISK ASSESSMENT



The 2016 multi-species marine mammal risk assessment (MMRA) estimated that commercial fisheries risk to common dolphins may be high, but is highly uncertain, reflecting biological unknowns (uncertain population size and population structure) and low observer coverage in inshore trawl fisheries. The New Zealand threat status for common dolphins is 'not threatened' (Baker et al. 2019). Improved population assessment will reduce this uncertainty and inform a more reliable assessment of commercial fisheries risk to common dolphins.

5. INCIDENTAL CAPTURES - JACK MACKEREL TRAWL FISHERIES



The observer coverage on board of jack mackerel trawl vessels has been steadily increasing since 2003, reaching nearly full coverage in recent years

Captures of common dolphin have been decreasing since 2003–04, due to mitigation measures in fishing operations, and have been negligible in the last 3 years

Fishing effort (above) and observed captures (below) of common dolphin in NZ trawl fisheries

6. ONGOING RESEARCH

- Improved estimates on population size and structure of common dolphin in NZ to inform risk assessment at smaller spatial scales
- Improved models of common dolphin spatial distribution, based on environmental and sighting data

7 COMMON DOLPHIN (*DELPHINUS DELPHIS DELPHIS*)

Status of chapter	This chapter has not been updated for AEBAR 2021.
Scope of chapter	This chapter briefly describes: the biology of short-beaked common dolphins (*Delphinus delphis delphis*); the nature and extent of potential interactions with fisheries; management of fisheries interactions; means of estimating fisheries impacts and population level risk; and remaining sources of uncertainty, to guide future work.
Area	The New Zealand EEZ and Territorial Sea.
Focal localities	Areas where significant fisheries interactions are known to have occurred include waters off the west coast of the North Island (including Taranaki Bight) and to a lesser extent Cook Strait.
Other than the Key issues	Improved means of estimating incidental captures and risk in poorly observed inshore fisheries; improved understanding of population size and structure; improved understanding of common dolphin spatio-temporal distributions affecting interaction rates with fishing effort.
Emerging issues	Improved ability to assess risk and apply risk management solutions on a regional subpopulation basis, or at finer spatial and temporal scales
MPI research (current)	PRO2013-01 *Estimation of Seabird and Marine Mammal Captures*; PRO2014-01 *Improving information on the distribution of seabirds and marine mammals*; PRO2017-08A *Research into the demographic parameters for at-risk marine mammals as identified by the risk assessment (common dolphins).*
NZ government research (current)	DOC Marine Conservation Services Programme (CSP): INT2015-01 *To understand the nature and extent of protected species interactions with New Zealand commercial fishing activities*; INT2015-03 *To determine which marine mammal, turtle and protected fish species are captured in fisheries and their mode of capture*.
Other research	Massey University: Skull morphometrics, growth and reproductive biology, diet and nutritional ecology, fine-scale distribution and abundance, and mother-offspring dynamics of common dolphins in New Zealand. Auckland University: Impacts of tourism on dolphin behaviour examining and the effectiveness of permit changes to the dolphins' responses to swimmers and boats.
Related chapters/issues	Chapter 3: Spatially Explicit Fisheries Risk Assessment (SEFRA); See also the JMA chapter, page 557, of the Fisheries Assessment Plenary Volume 2 (MPI 2017)

7.1 CONTEXT

Short-beaked common dolphins (*Delphinus delphis delphis*) were first described by Linnaeus in 1758 and have a worldwide distribution. In New Zealand waters, this species is protected under the Marine Mammal Protection Act (MMPA) of 1978 and the Fisheries Act (FA) of 1996. All marine mammals are protected under the s.2 (1) of the FA. The ministers for the Department of Conservation (DOC) and the Ministry for Primary Industries (MPI) can jointly approve a population management plan (PMP) for one or more species under s.14F of the Wildlife Act or s.3E of the MMPA. This PMP can include a maximum allowable level of fishing-related mortality of the species in New Zealand waters and recommendations to the Minister of Fisheries on 1) measures to mitigate fishing-related mortality and 2) the standard of information to be collected on fishing-related mortality. Currently, a PMP does not exist for common dolphins.

MPI manages fishing-related mortalities of common dolphins under s.15 (2) of the FA *'to avoid, remedy, or mitigate the effect of fishing-related mortality of any protected species and such measures may include setting a limit on fishing-related mortality.'* The 2005 Conservation General Policy administered by DOC specifies that *'protected marine species should be managed for their long-term viability and recovery throughout their natural range'*. The management of fisheries interactions with common dolphins aligns with the 2030 objective 6 to *'manage impacts of fishing and aquaculture'* and Strategic Action 6.2 to *'set and monitor environmental standards, including for threatened and protected species and seabed impacts'*.

Under the National Deepwater Plan, Objective 2.5 is most relevant to the management of common dolphins in New

Zealand waters: *'manage deepwater and middle-depth fisheries to avoid or minimise adverse effects on the long-term viability of endangered, threatened, and protected species'* (Ministry for Primary Industries 2012). The National Deepwater Plan contains information for fisheries to assess and manage marine mammal interactions with the deepwater fishing activity including a Marine Mammal Operating Procedure (MMOP), which outlines specific mitigation practices and proper handling of incidental marine mammal captures (Ministry for Primary Industries 2012).

Management Objective 7 of the National Fisheries Plan for Highly Migratory Species (HMS) is to 'implement an ecosystem approach to fisheries management, taking into account associated and dependent species' (Ministry of Fisheries 2010). The goals under this objective are as follows:

1. Avoid, remedy, or mitigate the adverse effects of fishing on associated and dependent species, including through maintaining food chain relationships.
2. Minimise unwanted bycatch and maximise survival of incidental catches of protected species in HMS fisheries using a risk management approach.
3. Increase the level and quality of information available on the capture of protected species.

The Draft National Fisheries Plan for Inshore Finfish states that the objectives of all groups is *'to minimise the adverse impact of fishing actives on the aquatic environment, including on biological diversity'* (Ministry of Fisheries 2011).

7.2 BIOLOGY

7.2.1 TAXONOMY

Within the Delphindae family, common dolphin are a member of the subfamily Delphininae (Perrin 1989). Based on genetic and morphological differences, there are two currently recognised species of common dolphins, the short-beaked (*Delphinus delphis*) and the long-beaked (*D. capensis*) (Rosel et al. 1994, Heyning & Perrin 1994). There are two subspecies of the short-beaked common dolphin (*D. d. Delphis* and *D. d. ponticus*), which is found only in the black sea) and two subspecies of long-beaked common dolphin (*D. c. capensis* and a nominal subspecies recognized as *D. c. tropicalis*; Jefferson & Waerebeek 2002). Genetic and morphometric differences between common dolphin populations in the South Pacific and those from other parts of the world have cast uncertainty as to the taxonomic identity of the New Zealand population of common dolphins (Bell et al. 2002, Stockin 2008, Stockin & Visser 2005). Skull morphometry values from Australia and New Zealand common dolphins fall between those reported for short- and long-beaked common dolphins. However, initial evidence suggests that the species in New Zealand waters is a larger form of the short-beaked common dolphin found elsewhere (Jordan et al. 2015, Jordan 2012, Bell et al. 2002). For the remainder of this chapter, 'common dolphin' will refer to the short-beaked species – *D. d. delphis*.

7.2.2 DISTRIBUTION

Common dolphins are found worldwide in tropical, subtropical, and temperate waters of the Pacific and Atlantic oceans (Hammond et al. 2008, Evans 1994) (Figure 7.1). This species also occurs in confined seas such as the Sea of Okhotsk and Sea of Japan as well as in small subpopulations in places such as the Mediterranean and Black Seas (Hammond et al. 2008). New Zealand waters represent the southern-most limit of common dolphins. Common dolphins are found around both the North and South Island (Brager & Schnieder 1998, Gaskin 1968, Berkenbusch et al. 2013, Constantine & Baker 1997) (Figures 1.2 and 1.3). However, Gaskin (1968) suggests that the distribution of common dolphins in New Zealand waters is constrained to warmer waters (greater than ca. 14°C) and is limited by the subtropical East Cape Current in the north and the subtropical convergence in the south.

Common dolphins are frequently observed along the northern and eastern coast of the North Island in the Bay of Islands, Hauraki Gulf, Mercury Bay, and in small groups, outside Wellington Harbour (Gaskin 1968, Constantine & Baker 1997, Neumann & Orams 2005, O'Callaghan & Baker 2002). Similar to other populations, common dolphins in New Zealand waters exhibit inshore and offshore daily and seasonal movements (Meynier et al. 2008, Neumann 2001c, Stockin 2008). The seasonal distribution of common dolphins is largely determined by the behaviour of their prey. Common dolphins are known to forage on small schooling fish that are strongly linked to sea surface temperature (SST). As a result, both common dolphins and their prey are found close to shore in the spring and summer when SST is high and further offshore in the autumn when SST drops (Neumann 2001, Stockin 2008, Neumann 2001). This species is also known to adjust their seasonal movements to take advantage of warmer water during a La Niña event (Neumann 2001).

Common dolphins are encountered in single and large multi-species groups with both seabirds and other marine mammals (hundreds to thousands) and found in waters both nearshore and thousands of kilometres offshore, in pelagic waters (Evans 1994). In New Zealand waters, they are known to form large aggregations with approximately 10 seabird and seven cetacean species. Of the seabird species, common dolphins are most often associated with the Australasian gannet (*Morus serrator*). Associations with other cetaceans include: bottlenose dolphins (*Tursiops truncatus*), striped dolphins (*Stenella coeruleoaba*), Hector's dolphin (*Cephalorhynchus hectori hectori*), Dusky dolphin (*Lagenorhynchus obscurus*), Minke whale (*Balaenoptera acutorostrata*), Sei whale (*Balaenoptera borealis*), and Bryde's whales (*Balaenoptera brydei*) (Stockin 2009).

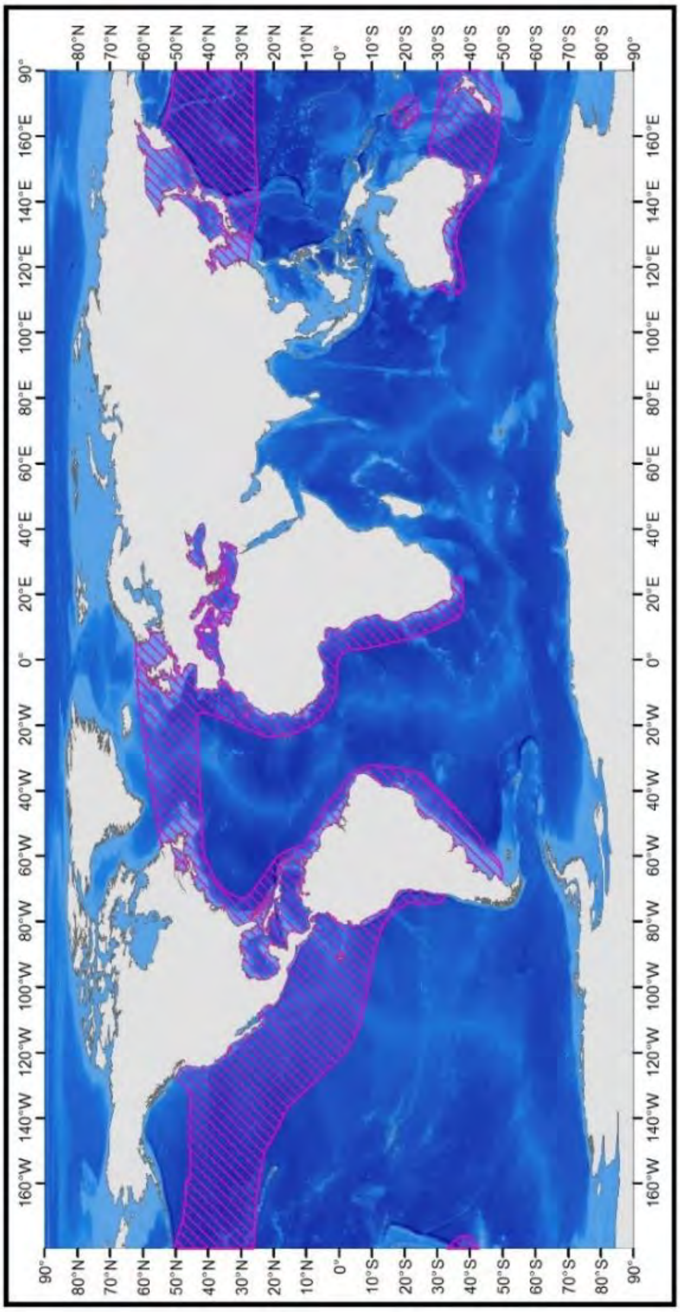


Figure 7.1: Worldwide distribution of short-beaked common dolphins (*Delphinus delphis delphis*) provided by the International Union for the Conservation of Nature (IUCN) (Hammond et al. 2008). Magenta hatched areas indicate range.

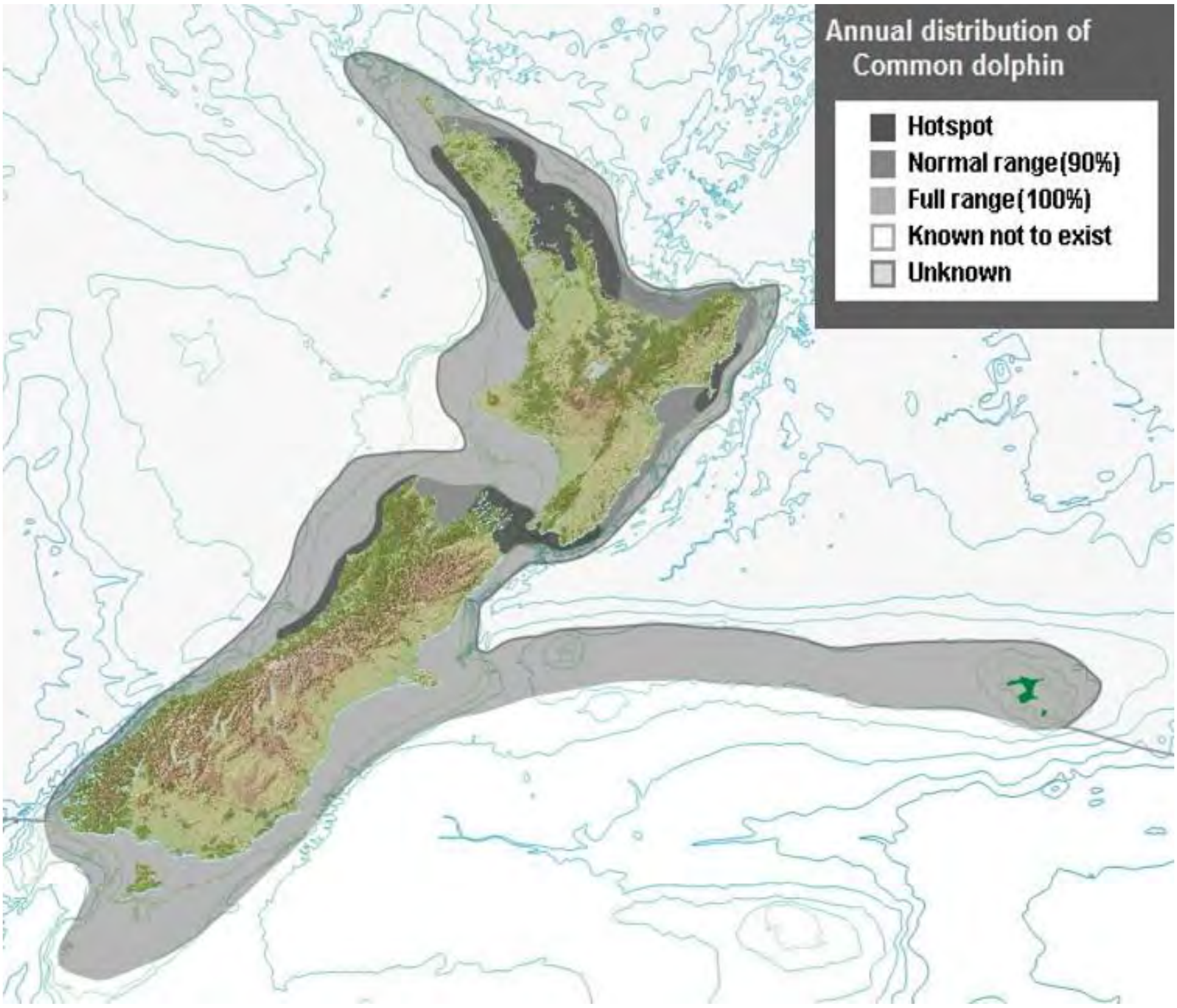


Figure 7.2: Distribution of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters (from www.nabis.govt.nz).

7.2.3 FORAGING ECOLOGY

The diet of common dolphins has primarily been assessed from the stomach contents of stranded and incidentally captured animals. Studies on common dolphins worldwide have documented the primary prey items as small schooling epipelagic and mesopelagic fish such as mackerel, sardines, and anchovies, as well as squid (Hammond et al. 2008, Young & Cockcroft 1994, Silva 1999, Bearzi et al. 2003, Pusineri et al. 2007, Overholtz 1991, Morizur et al. 1999). While there is abundant information on the diet of common dolphins for many populations, there is relatively little information for common dolphins in New Zealand waters.

Although research has specifically identified the Hauraki Gulf as an area extensively used for feeding, common dolphins forage in waters all around New Zealand (Stockin et al. 2009a). In one study, common dolphins off the east coast of the North Island were observed foraging on schools of jack mackerel (*Trachurus novaezelandiae*), schools of juvenile kahawai (*Arripis trutta),* yellow-eyed mullet (*Aldrichetta forsteri*), flying fish (*Cypselurus lineatus*), and, on one occasion, a school of parore (*Girella tricuspidata*), and garfish (*Hyporamphus ihi*) (Neumann & Orams 2003). The prevalent prey species from the stomach contents of animals stranded around the New Zealand coastline (n=27) and animals incidentally captured in the jack mackerel fishery off the west coast of the North Island (n=10) included arrow squid (*Nototodarus* sp.), anchovy (*Engraulis australis*), jack mackerel (*Trachurus* spp.) (Meynier et al. 2008). In another study, pilchard (*Sardinops neopilchardus*), and garfish (*Hyporhamphus ihi*) were the predominant prey items found in the stomachs of nine New Zealand common dolphin carcasses (n=9) classified as 'entanglement' (Stockin et al. 2009b).

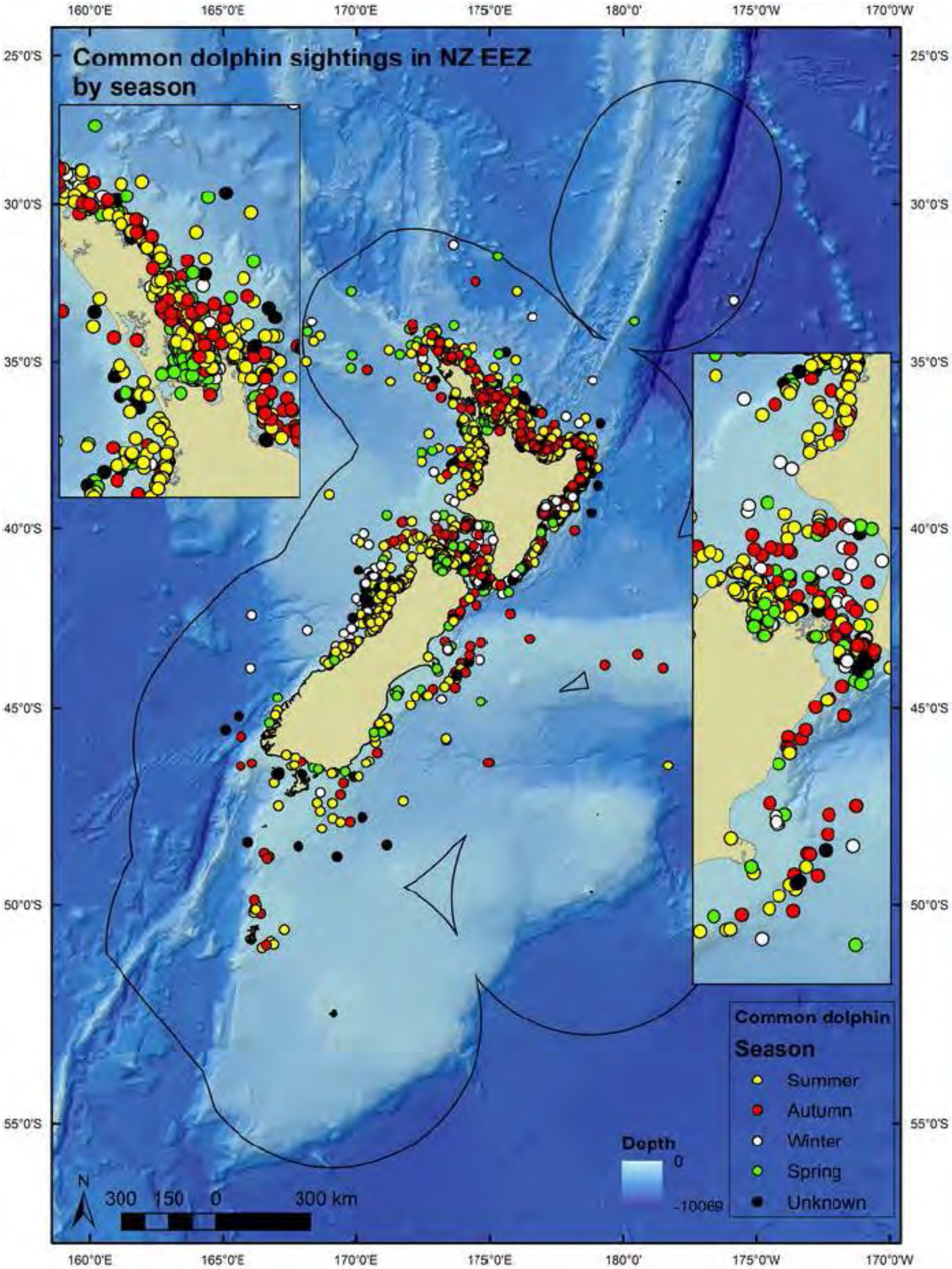


Figure 7.3: Systematic and opportunistic sightings of short-beaked common dolphins (*Delphinus delphis delphis*) in New Zealand waters between 1970 and 2013. Data sources include Department of Conservation (DOC), Cawthorn (2009), opportunistic at-sea sightings (NIWA), and the Centralised Observer Database (COD). (Sightings are indicative of the distribution only). Figure from Berkenbusch et al. (2013).

The similarity in prey items found in the stomachs of coastal and offshore animals provides further support that common dolphins in New Zealand make daily excursions between nearshore and offshore environments (Meynier et al. 2008). In addition, many of the prey species (e.g., squid) found in the stomachs of common dolphins are found in the deep scattering layer, which migrates towards the surface at night (Hammond et al. 2008, Neumann & Orams 2003). Neumann & Orams (2003) cite personal communication with S. Morrison in which common dolphin were sighted by crew members of squid boats during nocturnal fishing in Mercury Bay suggesting that time of day may provide important foraging opportunities for this species. The ability of common dolphins to feed on small schooling fish in shallow coastal waters during the day and on prey in the deep scattering layer in pelagic waters at night may indicate foraging plasticity (Neumann 2001). Acoustic research in New Zealand waters showed that during the day the mesopelagic layer occupied waters deeper than 200 m, then rapidly ascended to close to the surface after sunset; throughout the night, this layer dispersed downwards but remained in depths of less than 200 m until dawn when the it descended to day depths (McClatchie & Dunford 2003, O'Driscoll et al. 2009). O'Driscoll et al. (2013) found that schools of jack mackerel ascended and dispersed at night and were seen in depths of 10–30 m before dawn.

To exploit a large range of prey species, common dolphins exhibit a variety of foraging strategies. In New Zealand waters, both individual and coordinated feeding strategies have been documented (Neumann & Orams 2003). Individual foraging strategies include four types of behaviour: high-speed pursuit (traveling at high velocity in a zig-zag erratic fashion), fish-whacking (fish whacked with tail-fluke) and kerplunking (rapid tail-fluke movement in shallow water) (Neumann & Orams 2003, Constantine & Baker 1997). Furthermore, coordinated foraging strategies include: wall formation (driving fish into shallower water), carouseling (herding fish against the water surface), and bubble-blowing (startling herded fish).

Common dolphins are often observed foraging in association with other species (Neumann & Orams 2003). Rather than initiating feeding as a multi-species group, research indicates that birds and cetaceans may alert one another to prey by their presence and behaviour (Neumann & Orams 2003).

7.2.4 REPRODUCTIVE BIOLOGY

Despite their global distribution, relatively little information exists on the reproductive biology of common dolphins. Most of the existing information comes from studies on common dolphin populations in the North Pacific, Eastern Tropical Pacific, or North Atlantic and may or may not be applicable to the population of animals in New Zealand waters. Male and female age of sexual maturity for common dolphins in the North Pacific is 10.5 years for males and 8.0 years for females with lengths ranging 179–182 cm and 170.7–172.8 cm, respectively (Ferrero & Walker 1995). In the North Atlantic population, males reach sexual maturity at 9.5 years and 213 cm and females at 8.3 years and 200 cm (Westgate & Read 2007). A later age of sexual maturity for males may be the result of delayed breeding until the testes are large enough to compete with other males (Westgate & Read 2007).

Testes weight of sexually mature males ranges from 273.2 to 1190 g (Ferrero & Walker 1995). Male common dolphins in the North Atlantic exhibit seasonal changes in testes size with largest testes occurring in mid-July and smallest in October (Westgate & Read 2007). The peak in testes size corresponds with the timing of ovulation, conception and parturition and changes five-fold between maximum expansion and retraction. In mature males, testes comprised 2.2–3% of their total body mass (Westgate & Read 2007). Results from Westgate & Read (2007) suggest common dolphins in the North Atlantic engage in sperm competition as evidences by the seasonal change in testes size. The slight sexual dimorphism between sexes, in addition to seasonal changes in testes size, indicate that males compete for access to oestrous females and that females likely mate with many males (Westgate & Read 2007). Given that many common dolphins in temperate environments exhibit reproductive seasonality, it is likely that the New Zealand population of animals also exhibits a peak in reproduction that may correspond to seasonally abundant prey or optimal water temperatures.

Although gestation time for common dolphins in New Zealand waters is unknown, the length of gestation for this species is about 11 months for the North Pacific population, 11–12 months for the North Atlantic population, and 11 months for the Black Sea population (Westgate & Read 2007, Ferrero & Walker 1995, Gaskin 1972). Like all odontecetes, common dolphins give birth to a single calf, though one occurrence of a twin birth was reported off the coast of Spain (Gonzalez et al. 1999). At parturition, Westgate & Read (2007) estimated the length of neonate common dolphins in the North Atlantic at 93.2 cm. Neonates nurse for approximately six months and begin foraging at three to six months of age (Brophy et al. 2009). Common dolphins in the North Atlantic were found to have a minimum inter-birth interval of two years (Westgate & Read 2007).

In New Zealand waters, calves are seen year-round in the Hauraki Gulf, however, peak numbers are recorded in late spring and early summer months of December and January (Stockin et al. 2008). Common dolphins are considered a social species, showing non-random associations with other individuals. Sexual segregation in which animals divide into 'bachelor' (adult males), and 'nursery' (adult females and calves) groups has been observed in common dolphins in New Zealand waters (Neumann 2001, Neumann et al. 2002, Viricel et al. 2008). Mixed-sex groups also occur though they are usually associated with mating activities. The lack of stability in group composition is known as a 'fission-fusion' society in which group composition changes almost daily (Connor et al. 2000, Neumann 2001).

7.2.5 POPULATION BIOLOGY

The abundance of common dolphins is estimated at 4 000 000 worldwide with population estimates existing for many regions: 370 000 in the western US; 3 000 000 in the Eastern Tropical Pacific; 30 000 off the eastern US; 96 000 in the Black Sea; 60 000 on the eastern Atlantic continental shelf; 14 700 in the Alboran Sea; 75 000 in the Celtic Sea Shelf; and 19 400 in the western Mediterranean Sea (Jefferson et al. 2011, Hammond et al. 2008).

Although there is currently no abundance estimate for common dolphins in New Zealand waters, they are considered the most abundant and widespread cetacean recorded in the Hauraki Gulf, an important foraging and nursery area, in the summer (O'Callaghan & Baker 2002). Unlike common dolphins in other areas of New Zealand waters, in the Hauraki Gulf, this species exhibits high site fidelity (Stockin et al. 2008, 2014).

The maximum age of short-beaked common dolphins in western North Atlantic teeth was estimated at over 30 years using teeth samples from 204 bycaught and stranded animals (Westgate & Read 2007). Similarly, growth layers of teeth collected from 206 common dolphins in New Zealand waters that were stranded or bycaught in the midwater trawl fishery for jack mackerel (*T. novaezelandiae*) estimated maximum age at over 20 years and 29 years for males and females, respectively (Stockin et al. 2011, Murphy et al. 2014). Seven common dolphins incidentally caught by New Zealand fisheries and returned for autopsy were aged between 4 and 11 years (based on dentinel growth layers) (Duignan et al. 2003, 2004, Duignan & Jones 2005).

Microsatellite analyses of nearshore and offshore New Zealand common dolphins suggest that these animals have recently diverged (Stockin et al. 2014). In addition, the presence of high genetic variation at the southern limit of their distribution suggests that the overall population in New Zealand waters may be expanding and that there are fine-scale population level differences (Stockin et al. 2014).

Common dolphin populations are subject to many natural and anthropogenic threats that include but are not limited to: stranding, disease, predation, toxins, habitat loss, vessel-strike, recreational and commercial fishing and tourism-based activities. The cumulative impact of these threats on common dolphin populations has not been assessed. Drivers of common dolphin mortality include seasonal environmental variation, commercial fisheries interactions, habitat degradation, high-intensity acoustic disturbance, and disease (Murphy et al. 2013).

The Mediterranean Sea population of common dolphins was greatly reduced due to five main factors: 1) habitat loss, 2) prey depletion, 3) incidental captures by fisheries, and 4) immuno-suppression caused by chemical contamination, and 5) environmental fluctuations (Bearzi et al. 2003). In addition, at least 840 000 animals were removed from the Black Sea by hunters between 1946 and 1983, after which the population further declined due to disease and overfishing of prey species (Hammond et al. 2008).

In the absence of a population estimate for common dolphins in New Zealand waters, the impact of natural and human-induced effects cannot be accurately determined. Two of the main known threats to common dolphins in New Zealand waters are incidental capture by fisheries and tourism-related impacts (Thompson et al.

2013, Neumann & Orams 2005, Meissner et al. 2015, Constantine & Baker 1997, Stockin 2009). Fisheries-related threats are discussed in detail in the Sections 7.3 and 7.4.

7.2.6 CONSERVATION BIOLOGY AND THREAT CLASSIFICATION

Common dolphins are currently listed as a species of least concern under the International Union for the Conservation of Nature (IUCN) Red List of Threatened species with the exception of the Mediterranean subpopulation, which is listed as 'endangered' (Hammond et al. 2008).

In 2010, the conservation status of New Zealand marine mammals was reassessed using the 2008 version of the New Zealand Threat Classification system (Baker 2010). Based on several levels of criteria, common dolphins were classified as 'not threatened' with the qualifiers that the information was considered 'data poor', that the species was 'secure overseas', and that some subpopulations were 'threatened overseas'.

7.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Interactions between cetaceans and fisheries occur worldwide. Cetaceans have been incidentally captured by numerous types of fishing gear including trawl nets, purse seine nets, and static nets such as driftnets or gillnets (Reeves et al. 2005). Hall et al. (2000) state that cetaceans are at greater risk of capture by midwater trawls, which are towed faster than bottom trawls and usually target fish and squid. As a result, cetaceans may be captured when foraging in areas where fisheries using such gear also operate.

Due to their high global abundance, interactions between common dolphins and fisheries are not unusual. The highest rates of interactions are associated with fisheries that use trawl, purse seine, and drift nets. Outside New Zealand, perhaps the most well-known interaction occurs in the Eastern Tropical Pacific where common dolphins are found in association with yellowfin tuna (*Thunnus albacares*). In the 1960s, about 350 000 common dolphins were estimated to have been taken by this purse seine fishery (Joseph 1994). However, due to mitigation measures introduced in the 1970s, the rate of dolphin captures has been greatly reduced and is no longer a conservation concern (Reeves 2003).

In addition to the Eastern Tropical Pacific, interactions between common dolphins and fisheries are known to occur in the north and south Atlantic and Pacific oceans. Common dolphins were the most commonly caught cetacean in the US shark and swordfish gillnet fishery with an estimated mortality of 861 dolphins between 1996 and 2002 (Carretta et al. 2005). In the UK and the French pelagic trawl fishery for bass, ca. 800 common dolphins were taken annually (Hammond et al. 2008). In addition, the pelagic pair-trawl fishery off southwest England captured approximately 200 common dolphins per annum, with most animals being captured at night (de Boer et al. 2012). Male dolphins were at a greater risk of capture in pair-trawls offshore whereas females and calves were more vulnerable to gillnets close to shore (de Boer et al. 2012). Other areas where interactions between common dolphins and fisheries are known to occur include:

- The North Sea, predominantly in gillnets (Reijnders & Lankester 1990).
- Off the coast of Africa, predominantly in gillnet and purse seine fisheries (Maigret 1994, Jefferson et al. 1997).
- Off the south coast of Australia, mostly in gillnets or anti-shark netting (Kemper et al. 2005).
- Off the coast of Portugal, where 59% of 124 bycaught common dolphins were bycaught in primarily gill and seine nets between 1975 and 1998 and where fisheries interactions were responsible for up to 44% of strandings (Silva & Sequeira 2003).
- The Mediterranean Sea, where dolphins have a moderate (6–30% of sightings) or strong (35–50% of sightings) association with foreign purse seine tuna fishing, dolphin fish fishing activities, and illegal drift nets for swordfish offshore (Vella 2005, Tudela et al. 2005, Bearzi et al. 2008).
- The Black Sea, in pelagic trawl nets (Hammond et al. 2008, Reeves & Notarbartolo di Sciara 2006).

The Mediterranean Sea subpopulation of common dolphins has been declining since the 1960s and has been subjected to the effects of illegal drift-netting and other anthropogenic impacts (Reeves 2003, Forcada & Hammond 1998, Piroddi et al. 2011). It is believed that overfishing in the Mediterranean Sea has outcompeted common dolphins for prey (Bearzi et al. 2003). Bearzi et al. (2008) found that 10 active purse seine vessels were responsible for removing 33% of the biomass and suggested that they had the largest impact on dolphin prey species.

To reduce mortality from incidental captures, many countries have put implemented monitoring programmes to mitigate direct fisheries impacts to common dolphins. For example, after the creation of the US Marine Mammal Protection Act in 1972, observer coverage in the purse seine fishery was increased to 100% to ensure compliance. The European Union has also introduced legislation to establish observer programmes for most fisheries (Hammond et al. 2008). Other measures to reduce unwanted bycatch include: modification of fishing gear and methods (acoustic deterrents), input and output controls (limiting fishing effort or capacity), compensatory mitigation (investing in conservation projects), establishment of Marine Protected Areas (MPAs), fleet communication (reporting real-time observation of unpredictable bycatch hotspots), industry self-policing (peer pressure from within the industry), handling and release practices (backing down and hand rescue procedures to release dolphins), and changing gear (using alternative fishing methods that results in lower bycatch) (Gilman & Lundin 2009).

7.4 STATE OF KNOWLEDGE IN NEW ZEALAND

Common dolphins and fisheries in New Zealand waters often target the same fish species in the same areas. Early reports to the International Whaling Commission suggested that during June 1979 and April 1992, common dolphins were captured in trawl nets, crayfish pots, and purse seine nets (see Berkenbusch et al. 2013). Scientific observer data show that the primary fishery in New Zealand waters that is responsible for common dolphin mortality has been the midwater trawl fishery for jack mackerel species. Evidence from the early 1990s, after the establishment of the government observer programme, indicated that single and multiple captures of common dolphins occurred in the trawl nets of foreign-chartered trawlers targeting jack mackerel species off the west coast of New Zealand, in Quota Management Areas 7, 8 and 9 (61 animals between 1989–90 and 1992–93; see Baird 1994). This fishery operated offshore in the north and south Taranaki Bight waters, mainly in the summer months of November to April. During these years, observers reported a change in this fleet from the use of bottom trawls with headline heights of 5.2–9.8 m to midwater trawls with headline heights of 20–45 m (MPI unpublished observer data). The midwater trawls could be towed near the bottom during the day and in the water column at night and thus follow the movement of the jack mackerel schools. Alternatively, both gear types were used, alternating according to time of day.

Midwater nets were towed for 4–6 hrs and nets hauled between 2330 and 0615 h were responsible for almost all the dolphin captures, particularly in south Taranaki Bight in 70–130 m depths (Baird 1994). These mortalities resulted in the development of voluntary Codes of Practice (COPs) by the company operating the vessels, which aims to outline best practices to remedy, mitigate, or avoid incidental captures (Rowe 2007) (see Baird 1994, Appendix 9). The COPs addressed several aspects of the fishing operation thought to increase the likelihood of capture, mainly: the practice of undertaking a U-turn with the trawl doors up but the net in the water near the surface; the timing of setting; and the vessel lighting during night fishing activities. In addition, the codes may include recommendation for gear modifications and voluntary area closures (Rowe 2007). The government response led to increased observer coverage and provision for the necropsy of captured animals. MPI observer data shows that 10 common dolphins have been autopsied since 1994 (see also Duignan et al. 2003, 2004, Duignan & Jones 2005). However, capture incidents continued to occur until this fleet of vessels ceased fishing in New Zealand waters in the mid–late 1990s (Baird 1996).

Subsequently, midwater trawling for jack mackerels has remained the main method and target fishery responsible for common dolphin captures (based on observer data) (see Abraham & Thompson 2011). However, since the late 1990s, the observed common dolphin captures have been almost entirely from a different fleet of large foreign-charted trawlers operating mainly off the west coast of the North Island during summer months (Thompson et al. 2013a).

These vessels use midwater nets with headline heights of 30–60 m in depths of less than 200 m. The largest capture event in this fishery caught nine dolphins in one tow (Thompson et al. 2013a). Observer coverage between 1995–96 and 2010–11 was at least 20% for most fishing years but fluctuated considerably between 7 and 70% (Thompson et al. 2013a). The vessels are required to follow Operational Procedures for mitigating incidental captures of marine mammals as agreed by quota owners (see Section 7.4.2 for a fuller explanation).

Headline depth of trawl nets (distance from the headline to the surface) was found to be an important factor in explaining common dolphin captures in this fishery (Thompson et al. 2013). The majority of dolphin captures occurred when headline depth was between 10 and 40 m; however, 50% of observed capture events and 54% of common dolphins captured in large vessel mackerel fishery occurred on the 10% of the observed trawls that had a headline depth shallower than 30 m (Figure 7.4) (Thompson et al. 2010). Thompson et al. (2013, 2010) estimated that an increase of 21 m in headline depth may reduce the number of common dolphin captures by half. Longer tows caught more dolphins, as did tows in darkness, and tows conducted in the waters off the north Taranaki Bight. Of all shallow trawl tows (headline depths shallower than 40 m), 69% occurred at night when the fish migrate to the surface (Thompson et al. 2013). Common dolphins are known to follow diel migrating prey, which likely explains higher captures rates in shallow waters at night. Table 7.1 shows common dolphin captures in the jack mackerel fishery from 1989–90 to 1994–95. Most common dolphin captures occurred when conducting midwater trawls at night. The number of captures between 1995–96 and 2001–02 fishing years ranged between zero and 31 animals (Thompson et al. 2013). Captures have also been reported occasionally from observed trawl fisheries that targeted other middle depth species such as barracouta, hoki and arrow squid, as well as trawl nets targeting inshore species such as trevally and tarakihi (MPI unpublished data). The distributions of the fishing effort and observed captures for 2002–03 to 2015–16 are shown for all trawl fisheries (Figure 7.5) and for jack mackerel fisheries (Figure 7.6). During this time period there were 150 observed captures of common dolphins in trawl fisheries, 134 of which occurred in the jack mackerel fishery (see Section 7.4.1, Tables 7.2 and Table 7.3).

There were no observed common dolphin captures by the following New Zealand fisheries between 2002–03 and 2015–16: trawl (all except jack mackerel, hoki, middle depth and inshore); surface longline (southern bluefin, albacore and swordfish); bottom longline (ling, snapper); set net (flatfish and mullet); and purse seine (mackerel and skipjack tuna). There was a single common dolphin observed caught in the bigeye surface-longline fishery, in 2014–15. It should be noted that the proportion of the commercial effort covered by observers is highest in deepwater trawl fisheries, with relatively small amounts of effort observed for inshore trawl fisheries and fisheries using other types of fishing gear (see Abraham & Thompson 2011). Between 1995–96 and 2011–12 fishing years, observer effort in the middle-depth, inshore, and flatfish trawl fisheries was 3.4%, 0.5% and 0.3%, respectively (Berkenbusch et al. 2013).

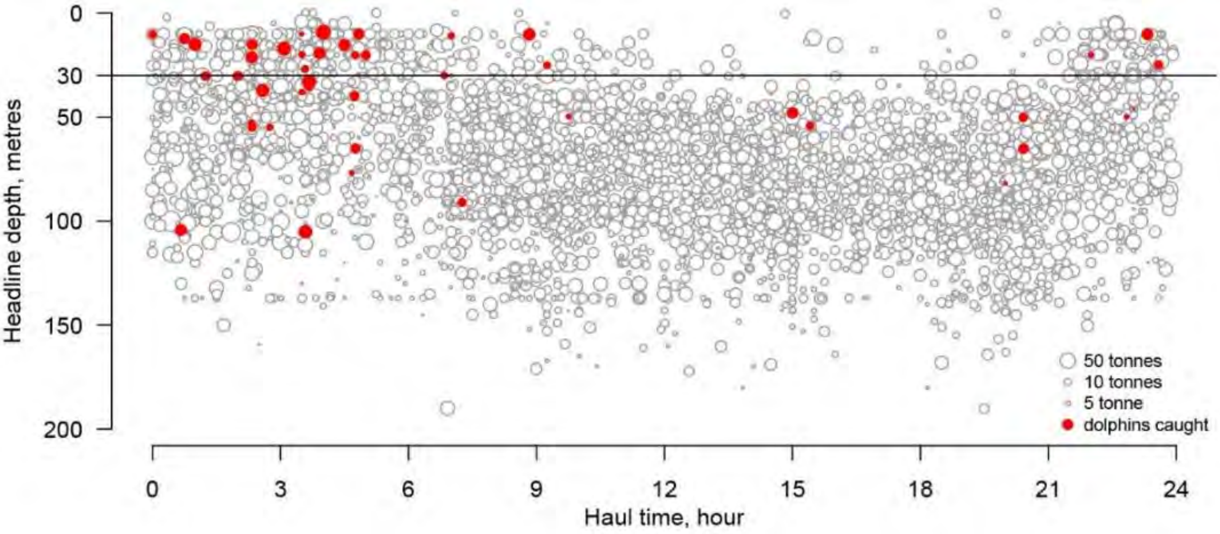


Figure 7.4: Headline depth versus the haul time for observed trawls in the large-vessel jack mackerel fishery. The catch weight is indicated by the size of the circles. Tows where an observed common dolphin capture event occurred are filled (from Thompson et al. 2010).

Table 7.1: Total and observed numbers of tows, observed number of dolphin mortalities and the number of events (tows) that incidentally caught dolphins in the jack mackerel fishery around the North (NT) and South (ST) Taranaki Bights by gear type (MW: midwater and BT: Bottom Tow), and time of day (D: Day and N: Night) for fishing years 1989–90 to 1994–95. Red bold numbers indicate that the species was confirmed as common dolphin (*Delphinus delphis delphis*). Table reproduced from Baird (1994, 1996). [Continued on next page]

Fishing year	Region	Gear	Time of day	Effort		Observed captures	
				Fishing tows	% observed	Mortality	Events
1989–90	NT	BT	D	1191	48	0	0
	NT	MW	D	41	0	0	0
	NT	BT	N	173	6	0	0
	NT	MW	N	28	1	0	0
	ST	BT	D	1418	139	0	0
	ST	MW	D	15	6	0	0
	ST	BT	N	186	6	0	0
	ST	MW	N	105	90	**23**	10
1990–91	NT	BT	D	603	2	0	0
	NT	MW	D	53	0	0	0
	NT	MT	N	72	0	0	0
	NT	MW	N	63	0	0	0
	ST	BT	D	676	47	0	0
	ST	MW	D	147	110	0	0
	ST	BT	N	84	12	0	0
	ST	MW	N	146	73	0	0
1991–92	NT	BT	D	1523	101	0	0
	NT	MW	D	361	4	0	0
	NT	BT	N	279	36	**2**	2
	NT	MW	N	500	3	**5**	3
	ST	BT	D	618	74	**1**	1
	ST	MW	D	151	3	0	0
	ST	BT	N	95	7	**5**	1
	ST	MW	N	146	15	16	5
1992–93	NT	BT	D	1759	135	0	0
	NT	MW	D	21	3	0	0
	NT	BT	N	438	22	0	0
	NT	MW	N	156	16	0	0
	ST	BT	D	588	112	0	0
	ST	MW	D	51	0	0	0
1992–93	ST	BT	N	48	6	0	0
	ST	MW	N	305	28	**9**	3
1993–94	NT	BT	D	1494	78	0	0
	NT	BT	D	219	19	0	0
	NT	MT	N	309	13	0	0
	NT	MW	N	300	28	0	0
	ST	BT	D	645	155	0	0
	ST	MW	D	120	20	0	0
	ST	BT	N	35	14	0	0
	ST	MW	N	279	71	8	5
1994–95	NT	BT	D	391	17	0	0
	NT	MW	D	399	80	0	0
	NT	BT	N	93	9	0	0
	NT	MW	N	258	74	0	0
	ST	BT	D	198	41	0	0
	ST	MW	D	228	73	6	3
	ST	BT	N	27	13	0	0
	ST	MW	N	147	74	15	3

Table 7.2: Fishing and observed effort (number of tows), the number and rate of observed captures, and estimated mean from statistical models of common dolphin (*Delphinus delphis delphis*) captures by all trawl fisheries by fishing year in the New Zealand EEZ (see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). For each fishing year, the table gives the total number of fishing tows, the percentage of tows that were observed; the number of observed captures (both dead and alive); the capture rate (captures per hundred tows); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Thompson et al. 2010 and 2013).

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% observed	Number	Rate	Mean	95% c.i.
2002–03	130 119	5.3	21	0.31	271	146–440
2003–04	120 819	5.4	17	0.26	239	129–396
2004–05	120 430	6.4	22	0.29	221	123–367
2005–06	109 944	6.0	4	0.06	125	52–242
2006–07	103 314	7.7	11	0.14	178	87–315
2007–08	89 531	10.1	20	0.22	143	71–250
2008–09	87 549	11.2	20	0.20	135	64–248
2009–10	92 893	9.7	4	0.04	137	55–266
2010–11	86 078	8.7	9	0.12	155	75–274
2011–12	84 418	11.1	5	0.05	108	41–210
2012–13	83 837	14.8	17	0.14	116	52–218
2013–14	85 110	15.6	30	0.23	118	61–208
2014–15	78 765	17.2	21	0.15	104	50–190
2015–16	78 029	16.6	7	0.05	3	2–7
2016–17	78 173	17.6	1	0.01	1	0–5
2017–18	74 243	20.1	1	0.01	0	0–4
2018–19	70 924	19.6	0	0.00	–	–
2019–20	65 994	23.6	0	0.00	–	–

Table 7.3: Fishing and observed effort (number of tows) and the number, rate, and estimated mean for common dolphin (*Delphinus delphis delphis*) captures by jack mackerel fisheries by fishing year in the New Zealand EEZ (see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). For each fishing year, the table gives the total number of trawl tows, the number of tows observed and the percentage of tows that were observed; the number of observed captures (both dead and alive); the capture rate (captures per hundred tows); and the mean number of estimated total captures (with 95% confidence interval). For more information on the methods used to prepare the data, see Thompson et al. 2010 and 2013.

Fishing year	Effort		Observed captures		Estimated captures	
	Fishing tows	% Observed	Number	Rate	Mean	95% c.i.
2002–03	3 067	11.3	21	6.07	141	60–259
2003–04	2 383	6.4	17	11.18	99	45–181
2004–05	2 509	22.2	21	3.76	85	46–139
2005–06	2 809	25.2	2	0.28	12	2–33
2006–07	2 711	29.6	11	1.37	55	23–102
2007–08	2 652	30.8	20	2.44	42	24–70
2008–09	2 169	37.5	11	1.35	23	11–43
2009–10	2 406	32.7	4	0.51	17	4–42
2010–11	1 882	31.5	7	1.18	53	18–108
2011–12	2 032	76.2	5	0.32	7	5–13
2012–13	2 213	87.7	15	0.77	16	15–20
2013–14	2 447	89.4	28	1.28	29	28–35
2014–15	1 750	86.4	19	1.26	21	19–28
2015–16	1 544	89.6	2	0.14	3	2–7
2016–17	1 407	72.8	0	0	1	0–5
2017–18	1 688	87.3	0	0	0	0–4
2018–19	1 627	78.5	0	0	–	–
2019–20	1 747	77.4	0	0	–	–

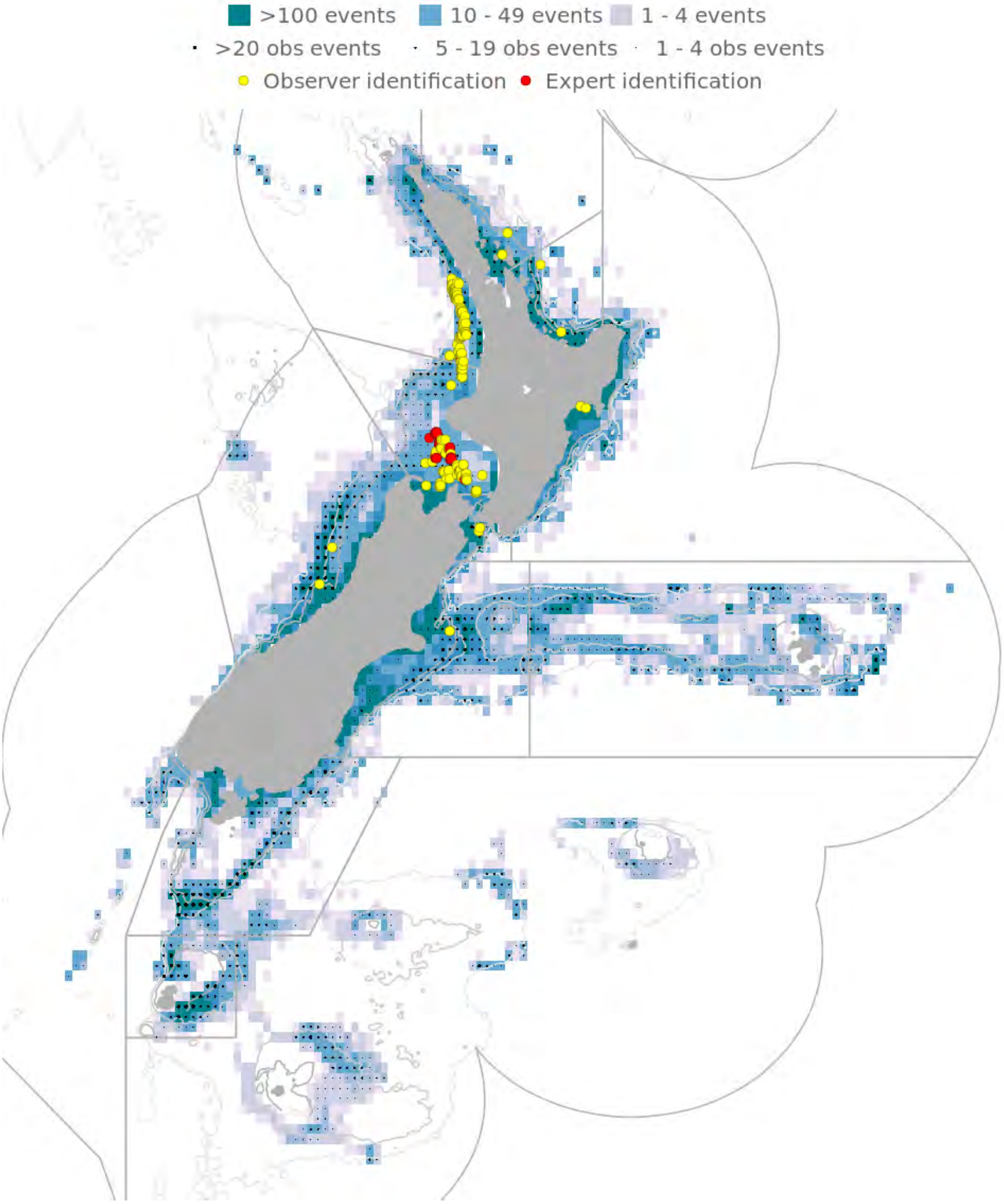


Figure 7.5: Distribution of all trawl fishing effort and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2019–20 (for more information see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Fishing effort is mapped into 0.2-degree cells, coloured to represent the number of tows. Observed fishing events are indicated by black dots, and observed capture events are indicated by red dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell.

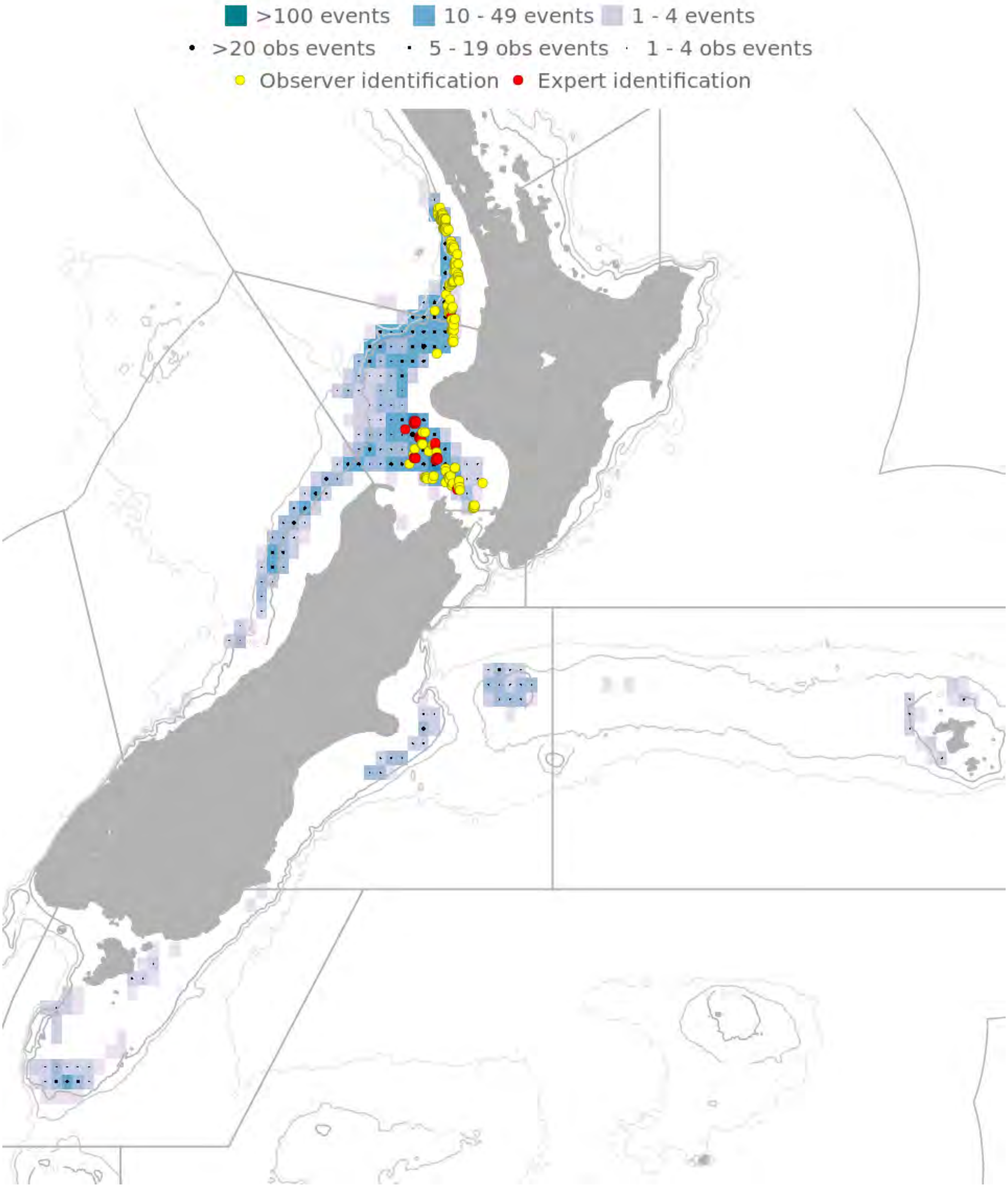


Figure 7.6: Distribution of trawl fishing effort for jack mackerel and observed common dolphin (*Delphinus delphis delphis*) captures, 2002–03 to 2019–20 (for more information see MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Fishing effort is mapped into 0.2-degree cells, coloured to represent the number of tows. Observed fishing events are indicated by black dots, and observed captures are indicated by red and yellow dots. Fishing effort is shown for all tows with latitude and longitude data, where three or more vessels fished within a cell.

7.4.1 QUANTIFYING FISHERIES INTERACTIONS

Bayesian models have been applied to fishing effort and observer data collected from trawl fisheries to estimate the number of common dolphin captures within New Zealand's EEZ (Abraham & Thompson 2011) (Figure 7.7). Note that while there were a small number of live captures, most capture events resulted in dolphin mortality. A separate two-step Bayesian hurdle model was developed by Thompson et al. (2010) to estimate the number of captures by the jack mackerel trawl fishery off the west coast of the North Island (Figure 7.8). The first part of the model estimated the presence of a capture event and the second part estimated how many capture events occurred if a capture event was estimated to have been present. Because no captures were recorded from smaller vessels, this analysis only included data from vessels over 90 m in length (Thompson et al. 2010). However, observer coverage of these vessels was limited to 0–0.5% for the years analysed (Thompson et al. 2010). Model-based capture estimates have been created for fishing years since 1995–96 (Thompson et al. 2013) and updated estimates to 2015–16 are presented in Table 7.2.

During the 2002–03 and 2015–16 fishing seasons, less than 3% of the total trawl effort (number of tows) occurred in the jack mackerel fishery, yet 90% of the 206 common dolphin captures recorded by observers occurred in this fishery (Tables 7.2 and 7.3).

7.4.2 MANAGING FISHERIES INTERACTIONS

Because little is known about the population of common dolphins in New Zealand, the level of fisheries impact and population level risk cannot be estimated with certainty. Given the large numbers of common dolphins worldwide, it is unlikely that the interaction between common dolphins and fisheries will have an adverse effect at the scale of the global population. However, there is still debate regarding the taxonomy of common dolphins found in New Zealand waters and whether a unique subpopulation inhabits New Zealand's EEZ. New research is currently underway to investigate population size and structure of common dolphins, to enable assessment of fisheries impacts and risk at the scale of regional subpopulations (if any).

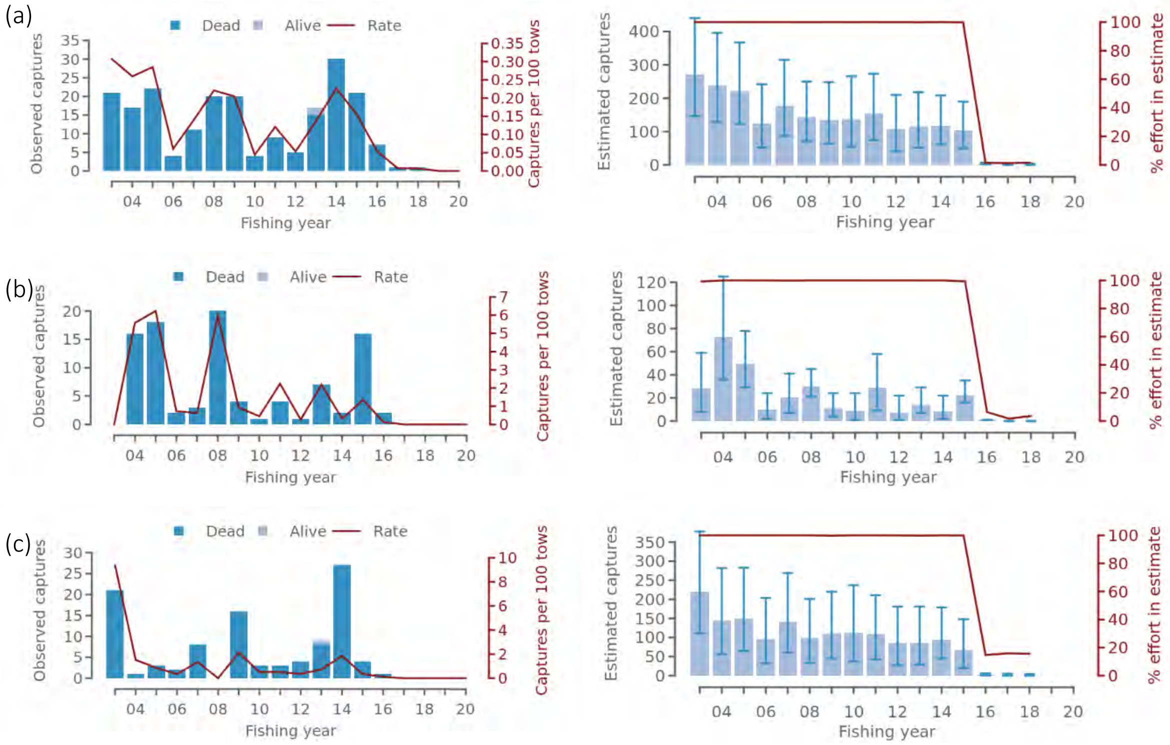


Figure 7.7: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in all trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2019–20, inclusive of three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Thompson el al. 2010 and 2013.

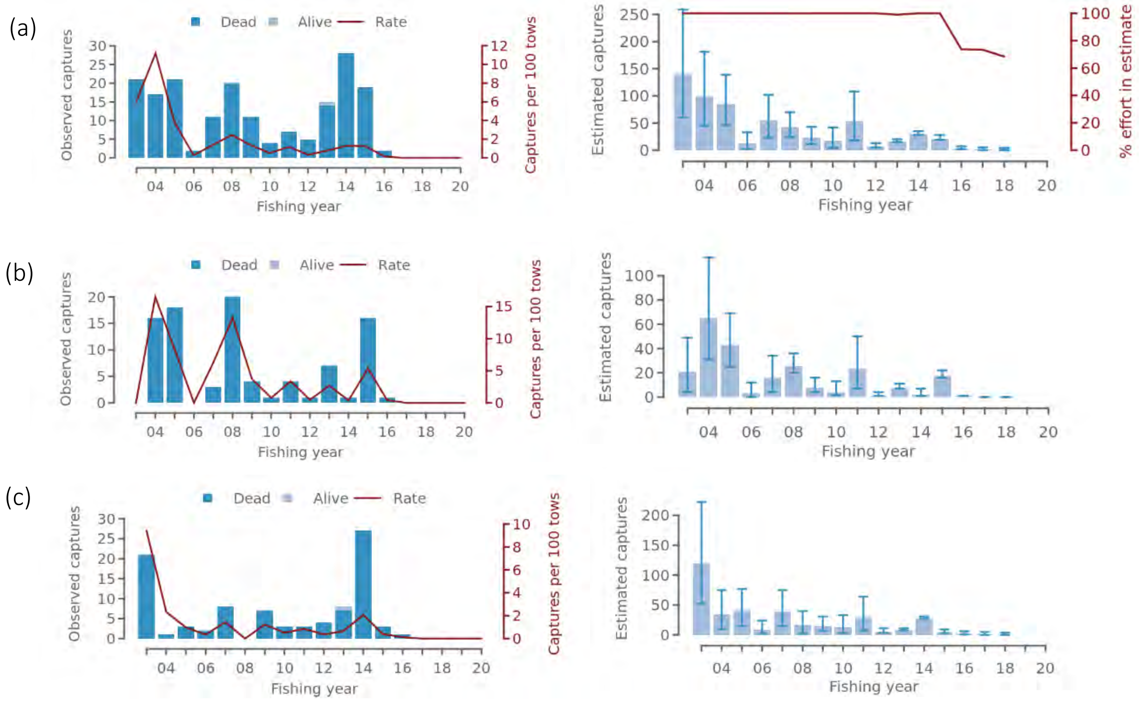


Figure 7.8: Observed captures of common dolphins (*Delphinus delphis delphis*) (dead and alive) in the jack mackerel trawl fisheries, the capture rate (captures per hundred tows) and the mean number of estimated total captures (with 95% confidence interval) by fishing years from 2002–03 to 2019–20 for three regions: (a) New Zealand's EEZ; (b) West coast of North Island; and (c) the Taranaki region (MPI data analysis at https://protectedspeciescaptures.nz/PSCv6/, data version v11). Percentage effort included in the estimation is shown when it was less than 100%. For more information on the methods used to prepare the data, see Thompson el al. 2010 and 2013.

MPI monitors interactions between fishing vessels and marine mammals primarily via the observer programme. In addition, MPI and the deepwater quota owners and trawl operators have developed a Marine Mammal Operating Procedure (MMOP) that specifies how skippers of trawlers greater than 28 m in length are expected to provide reports to the government of all marine mammal interactions, and specifies what fishers should do reduce capture rates and fisheries risk. Observer reviews provide information that contributes to managing interaction of the deepwater fleet. Specific risk management actions are identified for implementation in all JMA trawl fisheries, and there are additional requirements north of latitude 40° 30' S where most interactions occur.

Vessel practices required under the MMOP include: refraining from deploying fishing gear when dolphins are present; assigning an officer on watch and deck to report all sightings; ensuring trawl gear is closed during turns, by keeping doors at or above surface; using acoustic dissuasive devices attached to net on night-time tows for jack mackerel species; and (in the northern area) refraining from deploying trawl gear between 0230 and 0430 h. Additionally, under the MMOP all vessel officers are briefed annually on the risk factors regarding common dolphin captures especially area, depth and temporal factors. The full requirements can be seen at http://deepwatergroup.org/wp-content/uploads/2016/11/Marine-Mammals-Operational-Procedures-2016-17.pdf.

Vessels are required to report any captures to all vessels in the vicinity (by VHF radio) and must also notify the DeepWater Group (DWG) within a 24 hour period, and record captures in the ship's log, any time a common dolphin is caught (see Annual Review Report for Deepwater Fisheries, http://www.mpi.govt.nz/document-vault/4090, for more information).

7.4.3 MODELLING POPULATION-LEVEL IMPACTS OF FISHERIES INTERACTIONS

Because common dolphins are abundant and widespread, fisheries interactions are not considered a threat to the population at a global scale. However, small subpopulations of common dolphins such as the

Mediterranean Sea population have been significantly impacted by fishing.

The number of common dolphins captured in deepwater trawl fisheries is known with high certainty, due to high levels of observer coverage. Capture rates in poorly observed inshore fisheries are far less certain. Regardless of captures, the level of fisheries risk to common dolphins is estimated very poorly, in large part due to unknown population structure, such that there is no clear understanding of what size population these impacts should be considered against. New MPI research (PRO2017-08A) is underway applying genetic analyses to better understand common dolphin population structure and population size for potentially impacted populations, to improve estimates of fisheries risk. Other research is also in progress to estimate spatial distributions for New Zealand cetacean species (PRO2014-01), including common dolphins. Outputs from this work will inform spatially explicit estimates of encounter rate and capture rate in fisheries, which can then be applied to estimate population level risk at any spatial scale, applying the SEFRA method (Chapter 3, and below).

Total estimated captures per year varied between 0.15 (95% c.i.: 0.00–1.74) and 6.27 (95% c.i.: 2.49–12.27) captures per 100 tows over this 16-year period between the 1995–96 and 2011–12 fishing years (Thompson et al. 2013, Berkenbusch et al. 2013, Abraham & Berkenbusch 2017). The majority of observed common dolphin bycatch events in New Zealand waters have been in trawl fisheries targeting jack mackerel (*Trachurus declivis, T. murphyi* and *T. novaezelandiae*) on the west coast of the North Island.

7.4.4 MULTI-SPECIES MARINE MAMMAL RISK ASSESSMENT

In 2017, a New Zealand Marine Mammal Risk Assessment (MMRA) was completed (Abraham et al. 2017) applying a modification of the SEFRA method described in Chapter 3. Outputs of the MMRA suggest that common dolphins are the species potentially most at risk from New Zealand commercial fisheries. Fisheries risk to common dolphins is attributed primarily to pelagic trawl fisheries, for which historically observed captures are sufficient to estimate vulnerability and risk with some confidence, and also to inshore trawl and set-net fisheries, for which species vulnerability (hence total captures) is very poorly estimated (due to very low levels of historical observer coverage). Furthermore, as previously noted, estimates of biological population size are highly uncertain due to unknown population structure. As a consequence, cumulative fisheries risk for common dolphins remains highly uncertain, with an estimated risk score that may be less than half the Population Sustainability Threshold (PST) or may exceed the PST by a factor of two (Figure 7.9). (Note that the particular definition of PST used in the multi-species MMRA represents a number of anthropogenic deaths that would allow population recovery to, or stabilisation at, 50% of K with 90% certainty. Other species-specific risk assessments may adopt other population reference outcomes in the definition of PST, reflecting policy choices.)

Estimated fishery-related deaths for common dolphins in each fishery group, as estimated in the MMRA, are shown in Figure 7.10.

In 2017 an independent expert review of the SEFRA method and its implementations, including the (at that time unpublished) MMRA, made recommendations to improve this and future implementations of the MMRA (Lonergan et al. 2017). Of particular relevance to common dolphins, the review cautioned against uncritical use of Delphi-derived spatial species distribution layers as inputs. Research is currently in progress to estimate common dolphin distributions empirically on a finer spatial scale, using habitat suitability models informed by sightings data (PRO2014-01). When outputs of this work is available, it is expected that these will be combined with improved population estimates (from PRO2017-08A) in an updated marine mammal risk assessment.

7.4.5 SOURCES OF UNCERTAINTY

While there is an abundance of knowledge on common dolphins worldwide, relatively little is known about this species in New Zealand waters. The latest research suggests that common dolphins in New Zealand waters are a larger form of the short-beaked common dolphin found elsewhere; however further work is needed to verify this conclusion, which is based on a study with small sample size (Jordan 2012). As identified above, there is considerable uncertainty regarding population size and/or subpopulation structure of common dolphins around New Zealand. MPI project PRO2017-08A will address this uncertainty.

Due to historically low levels of observer coverage incidental captures of common dolphins by inshore fisheries are only poorly estimated. Improved observer coverage or monitoring by other means may help to address this uncertainty. Where captures are observed, improved understanding of factors affecting capture rates in different parts of the fishing event (i.e., setting, towing, or hauling) may be useful to inform management strategies or mitigation options to reduce captures.

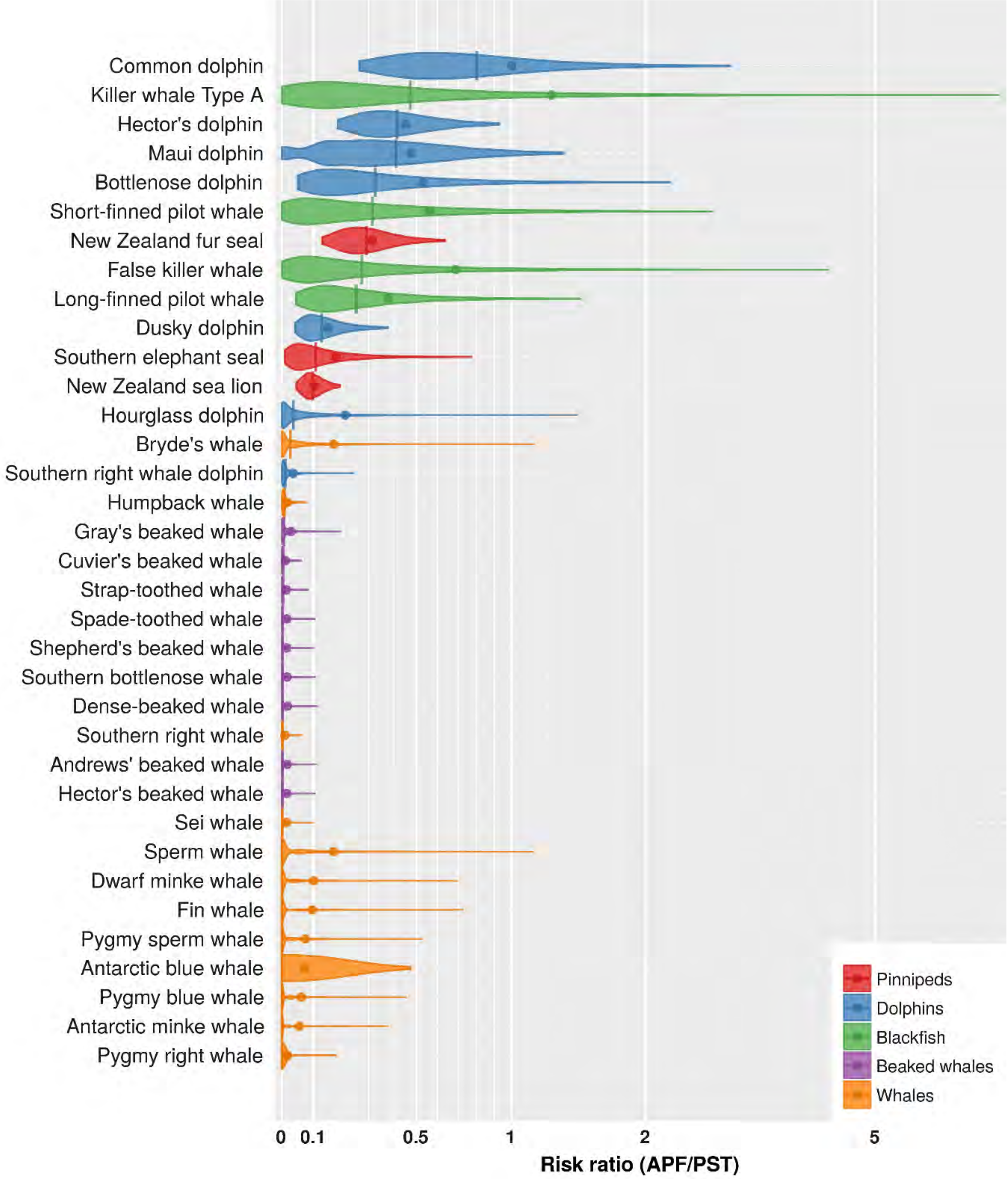


Figure 7.9: Cumulative fishery risk across all fishery groups as estimated by the 2016 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017). Species groups are colour coded.

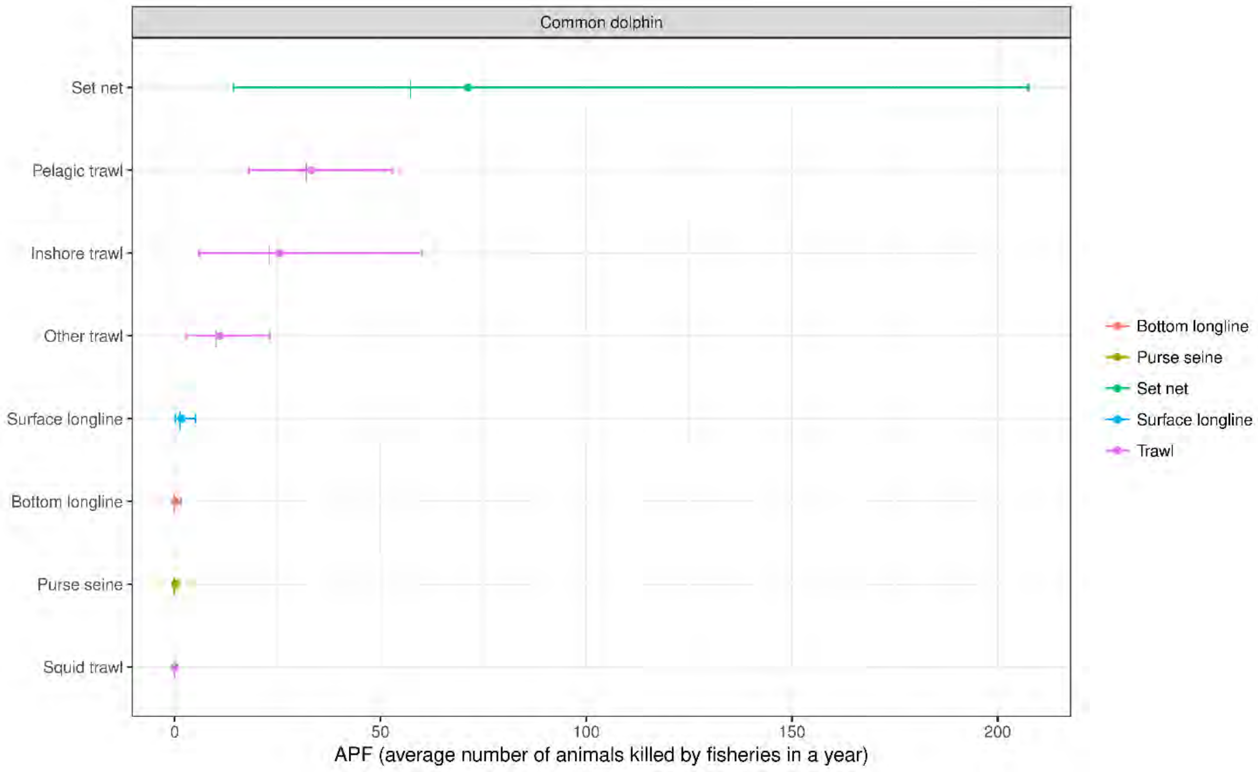
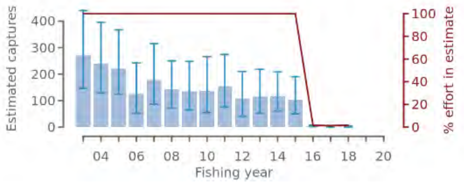


Figure 7.10: Annual fishery-related deaths of common dolphins in each fishery group, as estimated by the 2017 New Zealand Marine Mammal Risk Assessment (NZMMRA; Abraham et al. 2017).

7.5 INDICATORS AND TRENDS

Population size	Unknown in New Zealand EEZ, but approximately 4 000 000 worldwide.[1]
Population trend	Unknown.
Threat status	New Zealand: Not Threatened; Data Poor, and Secure Overseas in 2013.[2] IUCN: Least Concern, in 2008.[3]
Number of interactions[4]	104 estimated captures (95% c.i.: 50–189) in modelled trawl fisheries in 2014–15[4] 1 observed captures in trawl fisheries in 2017–18[4] 21 estimated captures (95% c.i.: 19–28) in the jack mackerel trawl fisheries in 2014–15[4] 0 observed captures in the jack mackerel trawl fisheries in 2017–18[4] 0 observed captures in the jack mackerel trawl fisheries in 2016–17[4] 142.7 estimated annual potential fatalities (APF) (95% c.i.: 70.7–285.1) [5]
Trends in interactions[4]	Trawl fisheries:  

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[1] Hammond et al. (2008).

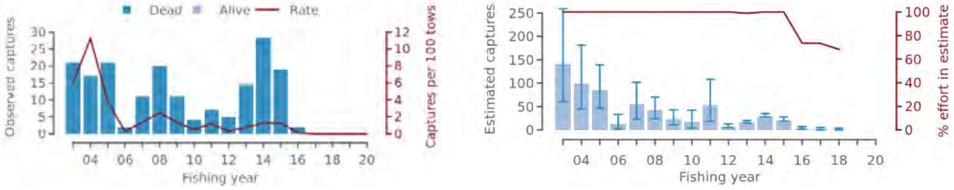
[2] Baker et al. (2016).

[3] Hammond et al. (2008).

[4] For more information, see: https://protectedspeciescaptures.nz/PSCv6/

[5] Abraham & Berkenbusch 2017.

Jack mackerel trawl fishery:



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Chapter 8: Seabirds - Technical Summary

Seabird accidental catch

Albatrosses, petrels, shearwaters

1. THE ISSUE IN BRIEF

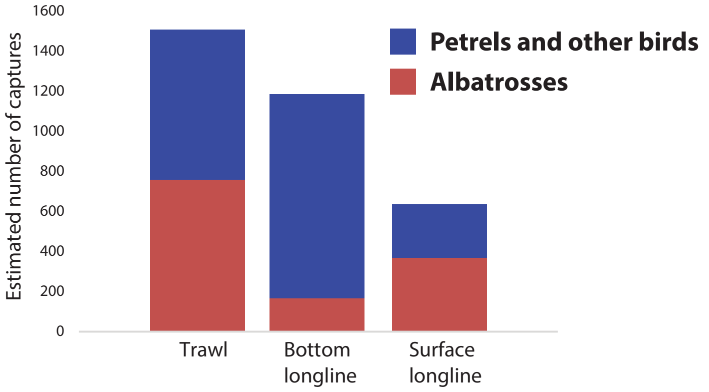
- New Zealand has 145 seabird species in its waters, with 95 species breeding here, of which over a third are endemic
- Seabirds are attracted to fishing activities and can get entangled in wires or nets, suffer damage by flying into trawl warps, get hooked by longlines, confused by lights, or drowned by the gear.
- Many species are classified as endangered or threatened. Fisheries New Zealand, industry, and DOC work together to reduce risk of accidental captures and mortality

2. NEW ZEALAND'S NATIONAL PLAN OF ACTION FOR SEABIRDS (NPOA-S)

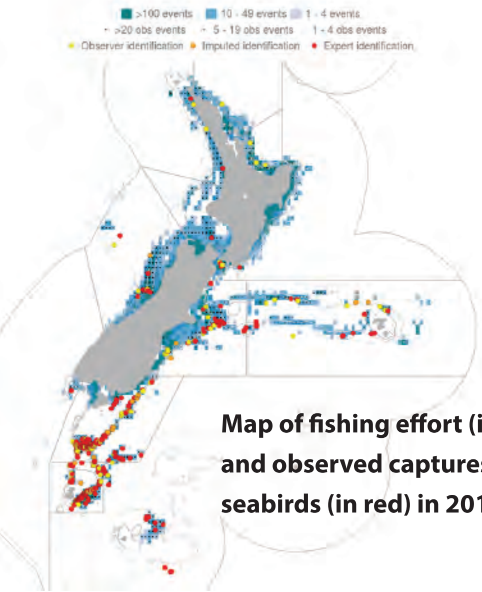
- The 2020 NPOA-S describes objectives for the prevention, monitoring, and management of incidental seabird capture in New Zealand fisheries and identifies nine action points outlined in this chapter
- The NPOA-S is administered by DOC and FNZ, and is implemented through a range of national fisheries planning processes and industry-led initiatives

3. INCIDENTAL CAPTURES OF SEABIRDS

- Seabirds are incidentally captured by a range of fishing activities, including:
  - inshore trawl (snapper, tarakihi)
  - deepwater trawl (hoki, squid, scampi)
  - bottom longline (ling, snapper)
  - surface longline (southern bluefin tuna, bigeye tuna, swordfish)



Estimated captures of seabirds by fishing gear type in 2017–18



Map of fishing effort (in blue) and observed captures of seabirds (in red) in 2017–18

- Estimated captures of seabirds in fisheries are based on statistical modelling of observed captures across several years. A total of 3328 seabird captures were estimated in 2017–18

- Estimates of seabird deaths and population risk are modelled from observer data, distribution maps and include cryptic mortality (unobservable deaths). A total of 12 900 seabird fatalities are estimated per year

4. RISK ASSESSMENT

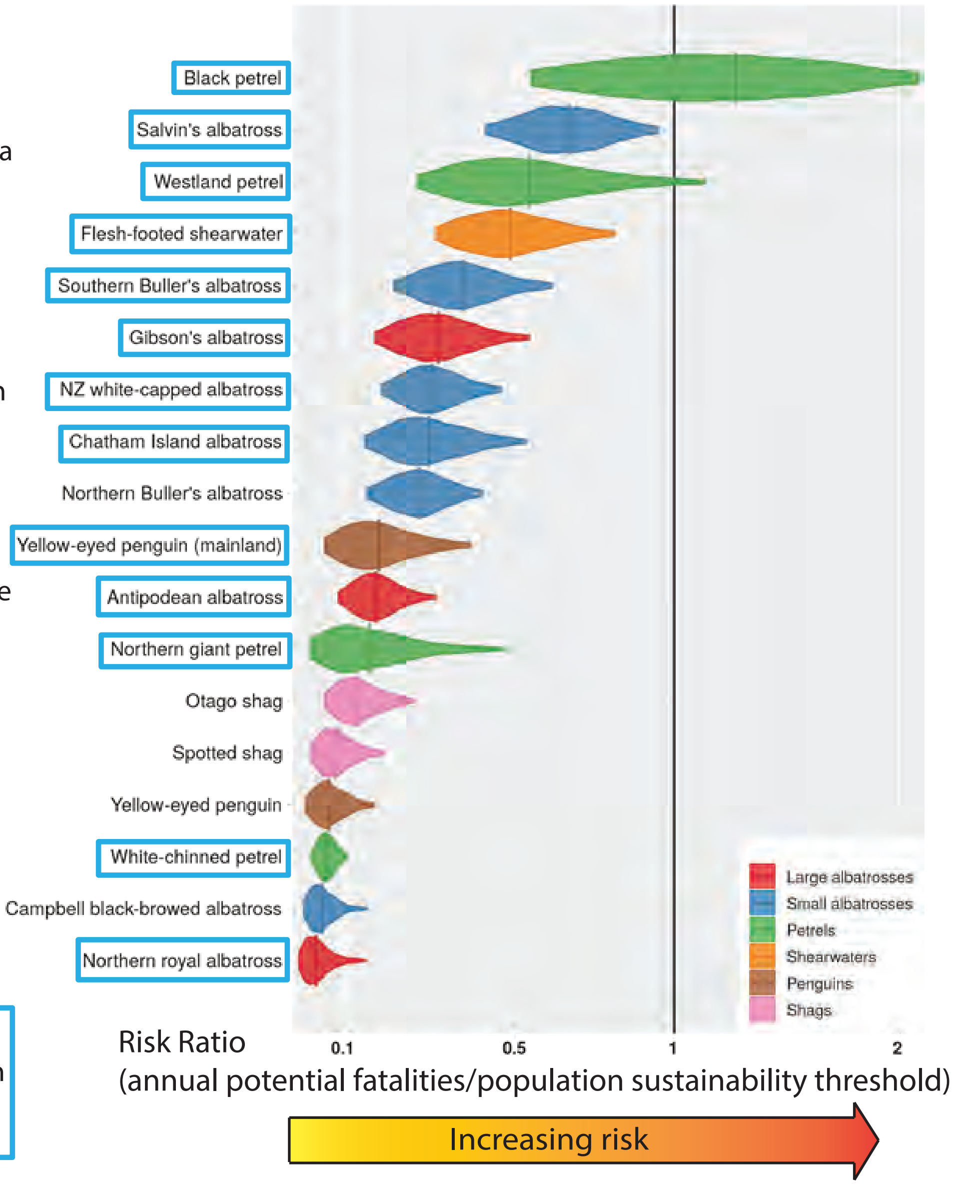
- The assessment uses the SEFRA methodology (see Chapter 3) to estimate the risk to multiple seabird species from multiple fisheries in New Zealand

- The risk ratio is an estimate of annual fisheries-related deaths relative to the population sustainability threshold (PST). It includes cryptic mortality and is specific to a desired population outcome (e.g., a population recovery target)

- Risk values above 1 indicate a substantial risk that the desired population outcome will not be achieved

- Black petrel is the only species at very high risk (mean risk ratio > 1), and there are 5 other species at high risk (0.25 > mean risk < 1)

- Work is in progress to include recreational fisheries and fisheries outside the New Zealand EEZ



Additional species-specific demographic and distribution modelling is carried out for these species

5. ONGOING RESEARCH

- Scientifically robust species-specific demographic and distribution studies have been completed for a large number of the species assessed. The risk assessment is a useful tool to prioritise research on populations of the species which have not yet been assessed
- Population trends require ongoing monitoring and interpretation to keep track of natural population fluctuations and anthropogenic pressures
- There remains some uncertainty on the role of fishing-related mortality in driving population trends. Determining the sources of uncertainty in risk assessments, and addressing them, remains critical
- Mitigation measures to decrease bird mortality and injury continue to be monitored and refined

8 NEW ZEALAND SEABIRDS

Status of chapter	This chapter has been partially updated for AEBAR 2021.
Scope of chapter	This chapter focuses on estimates of captures and risk assessments conducted for seabirds that breed in New Zealand waters. Also included are descriptions of the nature of fishing interactions, the management context and approach, trends in key indicators, and major sources of uncertainty. It includes details only on species that have been the focus of MPI research; demographic studies on individual seabird species (10 taxa) and 5 taxa for which quantitative population modelling has been conducted.
Area	New Zealand EEZ and Territorial Sea (TS) (noting that many seabirds are highly migratory and spend prolonged periods outside the New Zealand EEZ; on the high seas these effects are considered by CCSBT, WCPFC, CCAMLR, SPRFMO, etc., and New Zealand capture estimates are reported to those bodies).
Focal localities	Interactions with fisheries occur in many parts of the EEZ and TS as well as on the high seas and in the EEZs of other nations.
Key issues	Quantitative and semi-quantitative risk assessments can be improved through better estimates of: incidental captures in fisheries that are poorly observed or unobserved; species identity, especially of birds released alive; cryptic mortality rates; survival of birds released alive; improved understanding of seabird distributions; and the ability of seabird populations to sustain given levels of fisheries mortality, especially given fisheries interactions and captures outside the New Zealand EEZ and in non-commercial fisheries. Improved assessment of total fisheries impacts (i.e., including non-commercial and out-of-zone) and fisheries impacts is required, in the context of other factors influencing seabird survival and reproduction and other anthropogenic effects. Consolidation of population modelling and risk assessment results are key challenges. Black petrel remains a key focus of risk assessment research, with Antipodean albatross and yellow-eyed penguin also key species in terms of understanding the drivers of population trajectory and of demographic parameters.
Emerging issues	Potential new fishery monitoring techniques. New fisher mitigation technology. Global observer programmes.
Fisheries New Zealand research (current)	PRO2021-02 *Estimation of warp capture cryptic mortality multipliers with seabird corpse catcher device*, PRO2021-03 *Antipodean albatross multi-threat* risk *assessment*, PRO2021-04 *Comparing results of protected species captures interactions using different data collection methods*, PRO2021-06 *Identification of seabird capture 'hotspots' in the Regional Fisheries Management Organisation CCSBT*, PRO2021-07 *Review, cataloguing and continuation of footage collected from the 2020/21 Black Petrel Electronic Monitoring Project*, PSB2020-01 *Continued population monitoring of black petrel*, PSB2020-04 *Spatial Distribution Modelling for Hoiho*, PSB2020-05 *Grooming and preparation of the hoiho database*, PSB2020-06 *Characterisation of all fishing activity that overlaps with hoiho including fish bycatch*, PSB2020-07 *Factors affecting protected species captures in domestic longline fisheries*, PSB2020-*08 Desktop update of estimation of seabird cryptic mortality in trawls, via warp and net captures in the New Zealand domestic fleet using standard mitigation*, PSB2020-10 *Review and continuation of footage collection from the 2020-21 Black Petrel Electronic Monitoring project*, PSB2019-01 *Estimation of total seabird captures using standardised estimation methods,* PSB2019-08 *Feasibility trial of the Underwater Baitsetter*, PSB2019-09 *Opportunistic Aerial survey of white-capped albatross on the Auckland Islands*, PRO2017-01A *Demographic parameters of black petrels,* PRO2017-05A *Population modelling of black petrels*, PRO2017-06 *Characterisation of yellow-eyed penguin/fishery interactions*, PRO2017-19 *Capture rate of black petrels and flesh-footed shearwaters*.

New Zealand government research (current)	DOC Conservation Services Programme (CSP) projects: INT2018-01 *Observing commercial fisheries*, INT2016-02 *Identification of seabirds captured in New Zealand fisheries,* INT2017-02 *Supporting the utility of electronic monitoring to identify protected species interacting with commercial fisheries,* POP2017-03 *Salvin's albatross: Bounty Islands population*, POP2017-04 *Seabird population research: Auckland Islands 2017-2020,* POP2017-*06 Indirect effects on seabird in north-east North Island region,* POP2018-02 *Hoiho population and tracking,* POP2018-04 *Flesh-footed shearwater population monitoring,* POP2018-05 *Westland petrel population estimate,* MIT2017-01 *Protected species liaison,* MIT2017-02 *Characterisation and development of offal management for small vessels*, MIT2018-01 *Protected species engagement project,* MIT2018-02 *Haul mitigation for small longline vessels*, MIT2018-03 *Setting mitigation for small longline vessels.*
Related chapters/issues	National Plan of Action (2020) Reducing the incidental mortality of seabirds in fisheries (Fisheries New Zealand 2020). https://www.mpi.govt.nz/dmsdocument/40652-National-Plan-Of-Action-Seabirds-2020-Report.

8.1 CONTEXT

Seabird names and taxonomy in this document generally follow that adopted by the Ornithological Society of New Zealand (OSNZ 2010) except where a different classification has been agreed by the parties to the Agreement for the Conservation of Albatrosses and Petrels (ACAP) or the New Zealand Threat Classification Scheme (NZTCS) has classified multiple taxa within a single OSNZ species. There are probably more than 10 000 bird species worldwide, but fewer than 400 are classified as seabirds (being specialised marine foragers). All but seven seabird taxa in New Zealand are absolutely protected under s3 of the Wildlife Act 1953, meaning that it is an offence to hunt or kill them. Southern black-backed gull, *Larus dominicanus dominicanus*, is the only species that is not protected. Black shag, *Phalacrocorax carbo novaehollandiae*, and subantarctic skua, *Catharacta antarctica lonnbergi*, are partially protected; sooty shearwater, *Puffinus griseus*, and grey-faced petrel, *Pterodroma macroptera gouldi*, may be hunted; and little shag, *Phalacrocorax melanoleucos brevirostris*, and pied shag, *Phalacrocorax varius varius* may be killed subject to Minister's notification. Of the 99 seabird taxa that breed in New Zealand, 30 are considered threatened (Table 8.1). For albatrosses and petrels, a key threat is injury or death in fishing operations, although the Wildlife Act provides defences if the accidental or incidental death or injury took place in the course of fishing pursuant to a permit, licence, authority, or approval issued, granted, or given under the Fisheries Act 1996, as long as the interaction is reported. Commercial fishers are required to complete a Non-Fish and Protected Species Catch Return (NFPSCR, s11E of the Fisheries (Reporting) Regulations 2001).

The Minister of Conservation may approve a Population Management Plan (PMP) for one or more species under s14F of the Wildlife Act and a PMP can include a maximum allowable level of fishing-related mortality for a species (MALFiRM). Such a limit would apply to New Zealand fisheries waters and would be for the purpose of enabling a threatened species to achieve a non-threatened status as soon as reasonably practicable, and in any event within a period not exceeding 20 years, or, in the case of non-threatened species, neither cause a net reduction in the size of the population nor seriously threaten the reproductive capacity of the species (s14G). No PMPs are in place for seabirds but, in the absence of a PMP, the Minister of Oceans and Fisheries may, after consultation with the Minister of Conservation, take such measures as they consider necessary to avoid, remedy, or mitigate the effect of fishing-related mortality on any protected species (s15(2) of the Fisheries Act 1996).

Relevant, high-level guidance from the 2005 statement of General Policy, amendments to Chapters 2 and 4, June 2007, technical amendment July 2019 under the Conservation Act 1987 and Wildlife Act 1953 includes the following stated policies:

> 4.4 (f) Marine protected species should be managed for their long-term viability and recovery throughout their natural range.
>
> 4.4 (g) Where unprotected marine species are identified as threatened, consideration will be given to amending the Wildlife Act 1953 schedules to declare such species absolutely protected.
>
> 4.4 (j) Human interactions with marine mammals and other marine protected species should be managed to avoid or minimise adverse effects on populations and individuals.

4.4 (l) The Department should work with other agencies and interests to protect marine species.

New Zealand is a signatory to a number of international conventions and agreements to provide for the management of threats to seabirds, including:

- the United Nations Convention on the Law of the Sea (UNCLOS);
- the United Nations Fish Stocks Agreement (insofar as it relates to the conservation of non-target, associated, and dependent species);
- the Convention on Biological Diversity (CBD);
- the Convention on Migratory Species (CMS);
- the Food and Agriculture Organisation's (FAO) International Plan of Action for Reducing the Incidental Catch of Seabirds in Longline Fisheries (IPOA);
- the FAO Code of Conduct for Responsible Fisheries and the interpretive Best Practice Technical Guidelines;
- the Agreement on the Conservation of Albatrosses and Petrels (ACAP);
- Western & Central Pacific Fisheries Commission (WCPFC);
- Convention on the Conservation and Management of High Seas Fishery Resources in the South Pacific Ocean (SPRFMO);
- Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR).

The ACAP agreement requires that parties achieve and maintain a favourable conservation status for selected albatross and petrel taxa. Under the IPOA-seabirds, New Zealand developed a National Plan of Action (NPOA) to reduce the incidental catch of seabirds in New Zealand fisheries in 2004 (MFish & DOC 2004) and recently revised NPOA-seabirds (MPI 2020) https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/managing-our-impact-on-marine-life/seabirds/). The scopes of the 2004,2013 and 2020 NPOA are broader than the original IPOA to facilitate a coordinated and long-term approach to reducing the impact of fishing activity on seabirds. The 2020 NPOA covers all New Zealand fisheries and has a long-term objective that *'New Zealand strives for no fishing-related seabird captures'* There are high-level subsidiary objectives related to practical aspects, avoiding bycatch, healthy seabird populations, research and information, and international enagement. Implementation is largely through Fisheries New Zealand fisheries plans (see below). More detail is included in section 8.4.1. *Quantifying fisheries interactions*.

Strictly speaking, birds considered 'captured' in this context are those that have been brought on board a fishing vessel having been impaled on a hook, entangled by a line, trapped in a net, snagged on a trawl warp, or otherwise retrieved by some part of the fishing gear.

All National Fisheries Plans except that for freshwater fisheries are relevant to the management of fishing-related mortality of seabirds.

Under the National Fisheries Plan for Deepwater and Middle-depth Fisheries (Fisheries New Zealand 2019a) and the draft National Inshore Finfish Fisheries Plan (Fisheries New Zealand 2019), the objective most relevant for management of seabirds is Management Objective 8 and Objective 11 respectively: *Manage deepwater and middle-depth fisheries (or inshore fisheries) to avoid, remedy or mitigate the adverse effects of these fisheries on the long-term viability of endangered, threatened and protected species populations.*

Management Objective 5 of the National Fisheries Plan for Highly Migratory Species (HMS) 2019 (Fisheries New Zealand 2019b) is to '*Implement an ecosystem approach to fisheries management, taking into account associated and dependent species*'.

The Environment Objective for the draft National Fisheries Plan for Inshore Shellfish (Ministry of Fisheries 2011b), to *'Minimise adverse effects of fishing on the aquatic environment, including on biological diversity'*. The draft National Fisheries Plan for Freshwater (Ministry of Fisheries 2011c) has the same objective but is unlikely to be relevant to management of fishing-related mortality of seabirds.

Table 8.1: List of New Zealand seabird taxa, excluding occasional visitors and vagrants, according to the Ornithological Society of New Zealand (OSNZ 2010) unless otherwise indicated (all taxa under the New Zealand Threat Classification System are listed, ACAP taxonomy generally takes precedence). International Union for the Conservation of Nature (IUCN) and New Zealand (DOC) classifications are shown (http://www.iucnredlist.org and Robertson et al. 2017 at http://www.doc.govt.nz/documents/science-and-technical/nztcs19entire.pdf). [Continued on following pages]

Common name	Scientific name	DOC category	IUCN category
Albatrosses			
Antipodean albatross	*Diomedea antipodensis antipodensis*	Threatened: Nationally Critical	Endangered#
Gibson's albatross	*Diomedea antipodensis gibsonii*	Threatened: Nationally Critical	Endangered#
Southern royal albatross	*Diomedea epomophora*	At Risk: Naturally Uncommon	Vulnerable
Wandering albatross	*Diomedea exulans*	Non-Resident Native: Migrant	Vulnerable
Northern royal albatross	*Diomedea sanfordi*	At Risk: Naturally Uncommon	Endangered
Light mantled sooty albatross	*Phoebetria palpebrata*	At Risk: Declining	Near Threatened
Southern Buller's albatross	*Thalassarche bulleri bulleri*	At Risk: Naturally Uncommon	Near Threatened#
Northern Buller's albatross	*Thalassarche bulleri platei*	At Risk: Naturally Uncommon	Near Threatened#
Indian yellow-nosed albatross	*Thalassarche carteri*	Non-Resident Native: Coloniser	Endangered
Grey-headed albatross	*Thalassarche chrysostoma*	Threatened: Nationally Vulnerable	Endangered
Chatham albatross	*Thalassarche eremita*	At Risk: Naturally Uncommon	Vulnerable
Campbell albatross	*Thalassarche impavida*	Threatened: Nationally Vulnerable	Vulnerable
Black-browed albatross	*Thalassarche melanophris*	Non-Resident Native: Coloniser	Least Concern
Salvin's albatross	*Thalassarche salvini*	Threatened: Nationally Critical	Vulnerable
White-capped albatross*	*Thalassarche steadi**	At Risk: Declining	Near Threatened
Shearwaters			
North Island little shearwater	*Puffinus assimilis haurakiensis*	At Risk: Recovering	Least Concern#
Kermadec little shearwater	*Puffinus assimilis kermadecensis*	At Risk: Relict	Least Concern#
Buller's shearwater	*Puffinus bulleri*	At Risk: Naturally Uncommon	Vulnerable
Flesh-footed shearwater	*Puffinus carneipes*	Threatened: Nationally Vulnerable	Near Threatened
Subantarctic little shearwater	*Puffinus elegans*	At Risk: Naturally Uncommon	Least Concern
Fluttering shearwater	*Puffinus gavia*	At Risk: Relict	Least Concern
Sooty shearwater	*Puffinus griseus*	At Risk: Declining	Near Threatened
Hutton's shearwater	*Puffinus huttoni*	Threatened: Nationally Vulnerable	Endangered
Wedge-tailed shearwater	*Puffinus pacificus*	At Risk: Relict	Least Concern
Short-tailed shearwater	*Puffinus tenuirostris*	Non-Resident Native: Migrant	Least Concern
Petrels and prions			
Southern Cape petrel	*Daption capense capense*	Non-Resident Native: Migrant	Least Concern#
Snares Cape petrel	*Daption capense australe*	At Risk: Naturally Uncommon	Least Concern#
White-bellied storm petrel	*Fregetta grallaria grallaria*	Threatened: Nationally Endangered	Least Concern

Common name	Scientific name	DOC category	IUCN category
New Zealand storm petrel	*Fregetta maoriana*	Threatened: Nationally Vulnerable	Critically Endangered
Black-bellied storm petrel	*Fregetta tropica*	Not Threatened	Least Concern
Antarctic fulmar	*Fulmarus glacialoides*	Non-Resident Native: Migrant	Least Concern
Grey-backed storm petrel	*Garrodia nereis*	At Risk: Relict	Least Concern
Blue petrel	*Halobaena caerulea*	Non-Resident Native: Migrant	Least Concern
Kerguelen petrel	*Lugensa brevirostris*	Non-Resident Native: Migrant	Least Concern
Southern giant petrel	*Macronectes giganteus*	Non-Resident Native: Migrant	Least Concern
Northern giant petrel	*Macronectes halli*	At Risk: Recovering	Least Concern
Wilson's storm petrel	*Oceanites oceanicus*	Non-Resident Native: Migrant	Least Concern
Fulmar prion	*Pachyptila crassirostris crassirostris*	At Risk: Naturally Uncommon	Least Concern#
Lesser fulmar prion	*Pachyptila crassirostris flemingi*	At Risk: Naturally Uncommon	Least Concern#
Chatham fulmar prion	*Pachyptila crassirostris pyramidalis*	At Risk: Naturally Uncommon	Least Concern#
Slender-billed prion	*Pachyptila belcheri*	Non-Resident Native: Migrant	Least Concern
Antarctic prion	*Pachyptila desolata*	At Risk: Naturally Uncommon	Least Concern
Salvin's prion	*Pachyptila salvini*	Non-Resident Native: Migrant	Least Concern
Fairy prion	*Pachyptila turtur*	At Risk: Relict	Least Concern
Broad-billed prion	*Pachyptila vittata*	At Risk: Relict	Least Concern
South Georgian diving petrel	*Pelecanoides georgicus*†	Threatened: Nationally Critical	Least Concern
Southern diving petrel	*Pelecanoides urinatrix chathamensis*	At Risk: Relict	Least Concern#
Subantarctic diving petrel	*Pelecanoides urinatrix exsul*	Not Threatened	Least Concern#
Northern diving petrel	*Pelecanoides urinatrix urinatrix*	At Risk: Relict	Least Concern#
Kermadec storm petrel	*Pelagodroma albiclunis*	Threatened: Nationally Critical	–
New Zealand white-faced storm petrel	*Pelagodroma marina maoriana*	At Risk: Relict	Least Concern#
White-chinned petrel	*Procellaria aequinoctialis*	Not Threatened	Vulnerable
Grey petrel	*Procellaria cinerea*	At Risk: Naturally Uncommon	Near Threatened
Black petrel	*Procellaria parkinsoni*	Threatened: Nationally	Vulnerable
Westland petrel	*Procellaria westlandica*	At Risk: Naturally Uncommon	Endangered
Chatham petrel	*Pterodroma axillaris*	Threatened: Nationally Vulnerable	Vulnerable
White-necked petrel	*Pterodroma cervicalis*	At Risk: Relict	Vulnerable
Cook's petrel	*Pterodroma cookii*	At Risk: Relict	Vulnerable
Mottled petrel	*Pterodroma inexpectata*	At Risk: Relict	Near Threatened
White-headed petrel	*Pterodroma lessonii*	Not Threatened	Least Concern
Grey-faced petrel	*Pterodroma macroptera gouldi*	Not Threatened	Least Concern
Chatham Island tāiko	*Pterodroma magentae*	Threatened: Nationally Critical	Critically Endangered
Soft-plumaged petrel	*Pterodroma mollis*	At Risk: Naturally Uncommon	Least Concern

Common name	Scientific name	DOC category	IUCN category
Kermadec petrel	*Pterodroma neglecta* “summer”†	Threatened: Nationally Endangered	Least Concern#
Kermadec petrel	*Pterodroma neglecta* “winter”†	At Risk: Naturally Uncommon	Least Concern#
Black-winged petrel	*Pterodroma nigripennis*	Not Threatened	Least Concern
Pycroft’s petrel	*Pterodroma pycrofti*	At Risk: Recovering	Vulnerable
Penguins			
Eastern rockhopper penguin	*Eudyptes filholi*	Threatened: Nationally Vulnerable	Vulnerable#
Fiordland crested penguin	*Eudyptes pachyrhynchus*	Threatened: Nationally Vulnerable	Vulnerable
Snares crested penguin	*Eudyptes robustus*	At Risk: Naturally Uncommon	Vulnerable
Erect-crested penguin	*Eudyptes sclateri*	At Risk: Declining	Endangered
White-flippered blue penguin**	*Eudyptula minor albosignata***	At Risk: Declining	Least Concern#
Chatham Island blue penguin**	*Eudyptula minor chathamensis***	At Risk: Naturally Uncommon	Least Concern#
Northern blue penguin**	*Eudyptula minor iredalei***	At Risk: Declining	Least Concern#
Southern blue penguin**	*Eudyptula minor minor***	At Risk: Declining	Least Concern#
Australian little penguin	*Eudyptula novaehollandiae*	At Risk: Recovering	–
Yellow-eyed penguin	*Megadyptes antipodes*	Threatened: Nationally Endangered	Endangered
Shags			
Campbell shag	*Leococarbo campbelli*	At Risk: Naturally Uncommon	Vulnerable
King shag	*Leucocarbo carunculatus*	Threatened: Nationally Endangered	Vulnerable
Otago shag	*Leucocarbo chalconotus*	At Risk: Recovering	Vulnerable
Auckland Island shag	*Leucocarbo colensoi*	Threatened: Nationally Vulnerable	Vulnerable
Chatham Island shag	*Leucocarbo onslowi*	Threatened: Nationally Critical	Critically Endangered
Bounty Island shag	*Leucocarbo ranfurlyi*	At Risk: Naturally Uncommon	Vulnerable
Foveaux shag	*Leucocarbo stewarti*	Threatened: Nationally Vulnerable	–
Black shag	*Phalacrocorax carbo novaehollandiae*	At Risk: Naturally Uncommon	Least Concern#
Little shag	*Phalacrocorax melanoleucos brevirostris*	Not Threatened	Least Concern#
Little black shag	*Phalacrocorax sulcirostris*	At Risk: Naturally Uncommon	Least Concern
Pied shag	*Phalacrocorax varius varius*	At Risk: Recovering	Least Concern#
Pitt Island shag	*Phalacrocorax featherstoni*	Threatened: Nationally Critical	Endangered
Blue shag	*Stictocarbo punctatus oliveri*	At Risk: Naturally Uncommon	Least Concern#
Spotted shag	*Stictocarbo punctatus punctatus*	Not Threatened	Least Concern#
Gulls and terns			

Common name	Scientific name	DOC category	IUCN category
White-capped noddy	*Anous minutus minutus*	At Risk: Naturally Uncommon	Least Concern#
Common noddy	*Anous stolidus pileatus*	Non-Resident Native: Coloniser	Least Concern#
Black-fronted tern	*Childonias albostriatus*	Threatened: Nationally Endangered	Endangered
White-winged black tern	*Childonias leucopterus*	Non-Resident Native: Migrant	Least Concern
White tern	*Gygis alba candida*	Threatened: Nationally Critical	Least Concern#
Caspian tern	*Hydroprogne caspia*	Threatened: Nationally Vulnerable	Least Concern
Black-billed gull	*Larus bulleri*	Threatened: Nationally Critical	Endangered
Southern black-backed gull	*Larus dominicanus dominicanus*	Not Threatened	Least Concern#
Red-billed gull	*Larus novaehollandiae scopulinus*	At Risk: Declining	Least Concern#
Sooty tern	*Onychoprion fuscatus serratus*	At Risk: Recovering	Least Concern#
Grey ternlet	*Procelsterna cerulea albivittata*	At Risk: Naturally Uncommon	Least Concern#
New Zealand fairy tern	*Sternula nereis davisae*	Threatened: Nationally Critical	Vulnerable#
Arctic tern	*Sterna paradisaea*	Non-Resident Native: Migrant	Least Concern
Southern white-fronted tern***	*Sterna striata aucklandorna****	Threatened: Nationally Vulnerable	Near Threatened#
White-fronted tern***	*Sterna striata striata****	At Risk: Declining	Near Threatened#
New Zealand Antarctic tern	*Sterna vittata bethunei*	At Risk: Recovering	Least Concern#
Eastern little tern	*Sternula albifrons sinensis*	Non-Resident Native: Migrant	Least Concern#
Skuas			
Brown skua	*Catharacta antarctica lonnbergi*	At Risk: Naturally Uncommon	Least Concern#
South Polar skua	*Catharacta maccormicki*	Non-Resident Native: Migrant	Least Concern
Pomarine skua	*Stercorarius pomarinus*	Non-Resident Native: Migrant	Least Concern
Long-tailed skua	*Stercorarius longicaudus*	Non-Resident Native: Migrant	Least Concern
Arctic skua	*Stercorarius parasiticus*	Non-Resident Native: Migrant	Least Concern
Gannets and tropicbird			
Australasian gannet	*Morus serrator*	Not Threatened	Least Concern
Red-tailed tropicbird	*Phaethon rubricauda*	At Risk: Recovering	Least Concern
Masked booby	*Sula dactylatra tasmani*	Threatened: Nationally Endangered	Least Concern#

* OSNZ (2010) classify New Zealand white-capped albatross as a subspecies *Thalassarche cauta steadi*. Full species status is used here following ACAP.

** OSNZ (2010) classify a single species, little penguin *Eudyptula minor*. Multiple taxa are included here to reflect classification in the New Zealand Threat Classification Scheme.

*** OSNZ (2010) classify a single species, white-fronted tern *Sterna striata*. Multiple taxa are included here to reflect classification in the New Zealand Threat Classification Scheme.

# Indicates that the International Union for the Conservation of Nature (IUCN) classification is based on a broader definition of the species than listed in this table.

† Taxonomically Indeterminate in the New Zealand Threat Classification Scheme (NZTCS).

8.2 BIOLOGY

Taylor (2000) provided an excellent summary of the characteristics, ecology, and life history traits of seabirds, defined for the purpose of this document by the list given in Table 8.1 which is further summarised here. Table 8.1 includes all seabird taxa listed by Robertson et al. (2017) – the most recent iteration of the NZTCS as applied to birds, with the exception of those seabird taxa classified as ‘vagrant’. For both the New Zealand Threat Classification System and the IUCN ‘Red List’ system, seabird taxa with relatively small populations, and/or which have declining populations, and/or which occupy small spatial extents tend to be categorised with a higher conservation status. The New Zealand system includes additional categories not found in the Red List classifications. Specifically, the ‘threatened’ category is sub-divided into ‘nationally critical’, ‘nationally endangered’, and ‘nationally vulnerable’ categories (analogous to the Red List’s ‘critically endangered’, ‘endangered’, and ‘vulnerable’ categories). Additionally, the New Zealand system has four ‘at risk’ categories: ‘declining’, ‘recovering’, ‘relict’, and ‘naturally uncommon’. The New Zealand system also recognises ‘migrant’, ‘vagrant’, and ‘coloniser’ categories, in addition to a ‘not threatened’ category, which is similar to the Red List’s ‘near threatened’ category.

All seabirds spend part of their lifecycle feeding over the open sea. They have webbed feet, water-resistant feathering to enable them to fully immerse in salt water, and powerful wings or flippers. All have bills with sharp hooks, points, or filters, which enable them to catch fish, cephalopods, crustaceans, and plankton. Seabirds can drink saltwater and have physiological adaptations to remove excess salt.

Most seabird taxa are relatively long-lived; most live to 20 years and 30–40 years is typical for the oldest individuals. A few groups, notably albatrosses, can live for 50–60 years. Most taxa have relatively late sexual maturity. Red-billed gull *Larus novaehollandiae scopulinus* and blue penguin *Eudyptula minor* have been recorded nesting as yearlings, and diving petrels *Pelecanoides urinatrix* and yellow-eyed penguin *Megadyptes antipodes* can begin as 2-year-olds, but most seabirds start nesting only at age 3–6 years, and some albatross and petrel taxa delay nesting until 8–15 years old. In these late developers, individuals first return to colonies at 2–6 years old. Most seabirds, and especially albatrosses and some petrels, usually return to the breeding colony where they were reared, or nest close-by. Seabirds also have a tendency to mate for long periods with the same partner, and albatross pairs almost always remain together unless one partner dies.

The number of eggs laid varies among families. Albatrosses and petrels lay only one egg per year (sometimes nesting every other year) and do not lay again that year if it is lost. Other taxa such as gannets lay one egg but can replace it if the egg is lost. Most penguins lay two eggs but some raise only one chick and eject the second egg; replacement laying is uncommon. Blue penguins, gulls, and terns lay 1–3 eggs and can lay up to three clutches in a year if eggs are damaged or lost. Shags lay 2–5 eggs, can replace clutches, and have several breeding seasons in a year. Incubation in albatrosses and petrels lasts 40–75 days and chick rearing 50–280 days. In gulls and terns, incubation is completed in 20–25 days and chicks fledge in 20–40 days. In general, the lower the potential reproductive output of a taxon, the higher the adult survival rates and longevity.

Some seabirds such as shags, blue penguin, and yellow-eyed penguin live their lives and forage relatively close to where they breed, but many, including most albatrosses and petrels, spend large parts of their lives in international waters or in the waters of other nations far from their breeding locations. They can travel great distances across oceans during foraging flights and migratory journeys.

8.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

Fishing-related mortality of seabirds has been recognised as a serious, worldwide issue for only about 30 years (Bartle 1991, Brothers 1991, Brothers et al. 1999, Croxall 2008, Dias et al. 2019) and the Food and Agriculture Organisation of the United Nations (FAO) released its International Plan of Action for reducing incidental catch of seabirds in longline fisheries (IPOA-seabirds) in 1999 (FAO 1999). The IPOA-seabirds called on countries with (longline) fisheries that interact with seabirds to assess their fisheries to determine if a problem exists and, if so, to develop national plans (NPOA-seabirds) to reduce the incidental seabird catch in their fisheries. Lewison et al. (2004) noted that, in spite of the recognition of the problem, few comprehensive assessments of the effects of fishing-related mortality had been conducted in the decade or so after the problem was recognised. They reasoned that: many vulnerable species live in pelagic habitats, making surveys logistically complex and expensive; capture data are sparse; and understanding of the potential for affected populations to sustain additional mortality is poor. Soykan et al. (2008) identified similar questions in a Theme Section published in *Endangered Species Research*, including: Where is bycatch most prevalent? Which species are taken as bycatch? Which fisheries and gear types result in the highest bycatch of marine megafauna? What are the population-level effects on bycatch species? How can bycatch be reduced?

There has been substantial progress on these questions since 2004. Dias et al. (2019) reviewed the threats for all 359 seabird taxa and concluded that: seabirds are more threatened than other comparable groups of birds; their status has deteriorated faster over recent decades, with invasive alien species (46% of species), bycatch in fisheries (27.9%), hunting/trapping (27%), climate change/severe weather (26.7%), and disturbance (20.3%) being the top threats, and invasive alien species and bycatch in fisheries being the most immediate and potentially manageable threat to many albatrosses, petrels, and penguins. They listed the principal threats while at sea as being posed by commercial fisheries (through mortality associated with fishing gear and to a lesser extent overfishing) and pollution, and those on land as being invasive alien species, hunting and trapping, climate change/severe weather, and disturbance. Impacts of pollution, overfishing, problematic native species, energy production and mining, residential and commercial development, light pollution, diseases, agriculture, transportation and service corridors, aquaculture, logging and wood harvesting, and geological events were listed as threats for some taxa, with between 0.3% and 18.7% of species affected by these threats.

Dias et al. (2019) and Croxall et al. 2012 categorised responses to the major threats at sea as:

- using long-term demographic studies of relevant seabird species, linked to observational and recovery data to identify the cause of population declines (e.g., Croxall et al. 1998, Tuck et al. 2004, Poncet et al. 2006);
- risk assessments, based on spatio-temporal overlap between seabird species susceptible to bycatch and effort data for fisheries likely to catch them (e.g., Waugh et al. 2008b, Filippi et al. 2010, Tuck et al. 2011);
- implementation of operational and technical measures, such as discard management, bird-scaring lines, and hook shielding devices depending on the characteristics of the fishery;
- ensuring practical implementation of the measures and compliance with regulations;
- management of secondary impacts of fishing, through the effective implementation of ecosystem-based management;
- effective research programmes to ensure that current management measures are effective and that problems are not underestimated.

Seabirds are ranked by the IUCN as the world's most threatened bird grouping (Croxall et al. 2012, Dias et al. 2019). Globally they face a number of threats to their long-term viability, both at their breeding sites and while foraging at sea. Work at the global level on reducing threats at breeding sites is a major focus of ACAP, for which DOC is the lead New Zealand agency. However, a key threat to seabirds at sea, especially albatrosses penguins and petrels, is incidental capture and death in fisheries (Croxall et al. 2012), which in New Zealand is managed by Fisheries New Zealand.

Some seabirds do not range far from their breeding or roosting sites and incidental captures of these taxa can be managed by a single jurisdiction. Conversely, conservation of highly migratory taxa such as albatrosses and petrels cannot be achieved by one country acting independently of other nations that share the same populations. Because of this, in recent years, countries that share populations of threatened seabirds have sought to take action on an international level (e.g., ACAP) to complement policy and actions taken within their own jurisdictions.

The ICES Working Group on Seabird Ecology agreed (WGSE 2011) that the three most important indirect effects of fisheries on seabird populations were: the harvesting of seabird food; discards as food subsidies; and modification of marine habitats by dredges and trawls. Many seabird prey species are fished commercially (e.g., Furness 2003) or can be impacted indirectly by fishing of larger predators. These relationships are complex and poorly understood but WGSE (2011) agreed that impacts on populations of seabirds were inevitable. Fishery discards and offal have the potential to benefit seabird species, especially those that ordinarily scavenge (Furness et al. 1992, Wagner &

Boersma 2011). However, discarding can also modify the way in which birds forage for food (e.g., Bartumeus et al. 2010, Louzao et al. 2011), sometimes with farther-reaching behavioural consequences with negative as well as positive effects (including the 'junk food hypothesis', e.g., Romano et al. 2006, Grémillet et al. 2008). Louzao et al. (2011) stated that discards can affect movement patterns (Arcos & Oro 1996), improve reproductive performance (Oro et al. 1997, 1999) and increase survival (Oro & Furness 2002, Oro et al. 2004). Benefits for scavengers and kleptoparasitic taxa (those that obtain food by stealing from other animals) feeding on discards can also have consequent negative impacts on other species, especially diving species, that share breeding sites or are subject to displacement (Wagner & Boersma 2011). Dredging and bottom trawling both affect benthic habitat and fauna (see Rice 2006 and the benthic effects chapter in this document) and WGSE (2011) agreed that this probably affects some seabird populations, although little work has been done in this area.

8.4 STATE OF KNOWLEDGE IN NEW ZEALAND

Before the arrival of humans, the absence of terrestrial mammalian predators in New Zealand made it a relatively safe breeding place for seabirds and large numbers of a wide variety of taxa bred here, including substantial numbers on the main North and South islands. Today, New Zealand's extensive coastline, numerous inshore and offshore islands (many of them predator free) and surrounding seas and oceans continue to make it an important foraging and breeding ground for about 168 seabird taxa, second only to the USA (GA Taylor, Department of Conservation, personal communication). Roughly 99 of these taxa breed in New Zealand (Figure 8.1 and Figure 8.2, Table 8.2), including the greatest number of albatrosses (14), petrels (32), shags (13), and penguins (9) of any area in the world (Miskelly et al. 2008). More than a third are endemic (i.e., breed nowhere else in the world), giving New Zealand by far the largest number of endemic seabird taxa in the world.

Some seabirds use New Zealand waters but do not breed here. Some visit here occasionally to feed (e.g., wandering albatross *Diomedea exulans* and southern giant petrel *Macronectes giganteus*), whereas others are frequent visitors (e.g., short-tailed shearwater *Puffinus tenuirostris* and Wilson's storm petrel *Oceanites oceanicus*), sometimes for extended durations (e.g., Arctic skua *Stercorarius parasiticus*).

Taylor (2000) listed a wide range of threats to New Zealand seabird taxa including introduced mammals, avian predators (e.g., weka), disease, fire, weeds, loss of nesting habitat, competition for nest sites, coastal development, human disturbance, commercial and cultural harvesting, volcanic eruptions, pollution, plastics and marine debris, oil spills and exploration, heavy metals or chemical contaminants, global sea temperature changes, marine biotoxins, and fisheries interactions. Seabirds are caught in commercial trawl, longline, set net, and, occasionally, other fisheries (e.g., annual assessments by SJ Baird from 1994 to 2005, Baird & Smith 2008, Waugh et al. 2008a, 2008b, Abraham et al. 2010b, 2016, Abraham & Richard 2019) as well as in non-commercial fisheries (Abraham et al. 2010a). New Zealand released its first National Plan of Action to reduce the incidental catch of seabirds (NPOA-seabirds) in 2004; this was revised in 2013 and most recently in 2020. The 2004 plan stated that there was, at that time, limited information about the level of incidental catch and population characteristics of different seabird taxa, and that this made quantifying the overall impact of fishing difficult. This situation had improved by the time 2020 NPOA-seabirds was published with several iterations of the seabird risk assessment having been published, nevertheless, that document seeks to ensure, among other things, that the development of new mitigation measures, new observation and monitoring methods, and relevant research are encouraged and resourced. Seabird taxa caught in New Zealand fisheries range in IUCN threat ranking from critically endangered (e.g., Chatham Island shag *Leucocarbo onslowi*) to least concern (e.g., common diving petrel) (e.g., Vié et al. 2009).

Different taxa and populations face different threats from fishing operations depending on their biological characteristics and foraging behaviours. Biological traits such as diving ability, agility, size, sense of smell, eyesight, and diet, and foraging factors such as the season and areas they forage, their aggressiveness, and the boldness (or shyness) they display in their attraction to fishing activity can all affect their susceptibility to capture, injury, or death from fishing operations. Some fishing methods pose particular threats to some guilds or types of seabirds. For example, penguins are particularly vulnerable to set net operations and large albatrosses appear to be vulnerable to most forms of longlining. The nature and extent of interactions differs spatially, temporally, seasonally, and diurnally between sectors and fisheries and between fleets and vessels within fisheries. In 2017–18 the taxa most frequently observed caught in New Zealand commercial fisheries in descending order were: white-chinned petrel *Procellaria aequinoctialis* (243), New Zealand white-capped albatross *Thalassarche steadi* (160), southern Buller's albatross *Thalassarche bulleri bulleri* (63), sooty shearwater (56), Salvin's albatross *Thalassarche salvini* (35), flesh-footed shearwater *Puffinus carneipes* (16), black petrel *Procellaria parkinsoni* (12), Westland petrel *Procellaria westlandica* (10), and grey petrel *Procellaria cinerea* (9). It should be noted that some of these totals included birds not technically 'captured' (see definition in section 8.1) but which were deck strikes.

The management of fisheries to ensure the long-term viability of seabird populations requires an understanding of the risks posed by fishing and other anthropogenic drivers. Several studies have already estimated the number of seabirds caught annually within the New Zealand Exclusive Economic Zone (EEZ) in a range of fisheries (e.g., Baird & Smith 2008, Waugh et al. 2008a, 2008b, Abraham et al. 2010b, 2016, Abraham & Richard 2019). Seabirds that breed in New Zealand die as a result of interactions with commercial or recreational fishing operations in waters under New Zealand jurisdiction, through interactions with New Zealand vessels or other nations' vessels on the High Seas, and through interactions with commercial, recreational, or artisanal fishing operations in waters under the jurisdiction of other states.

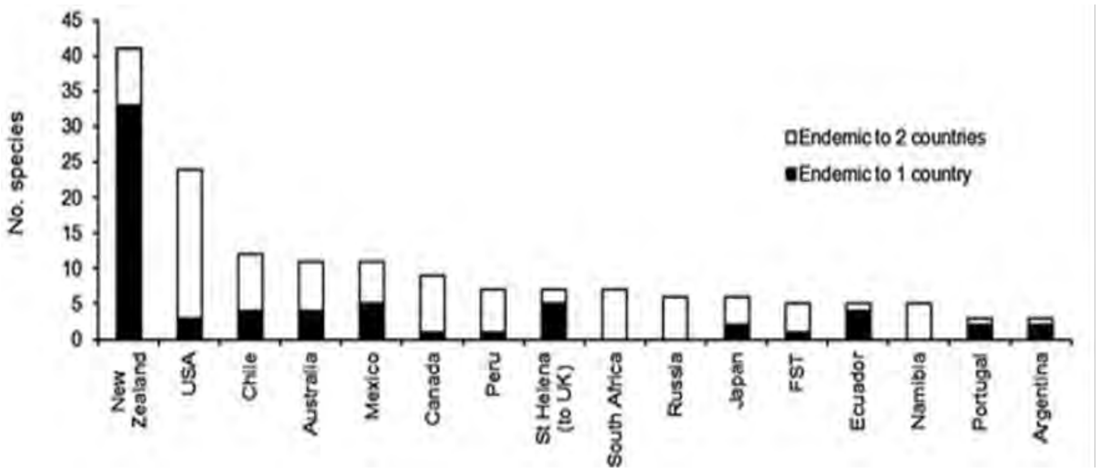


Figure 8.1: (from Croxall et al. 2012). Number of endemic breeding seabird taxa by country.

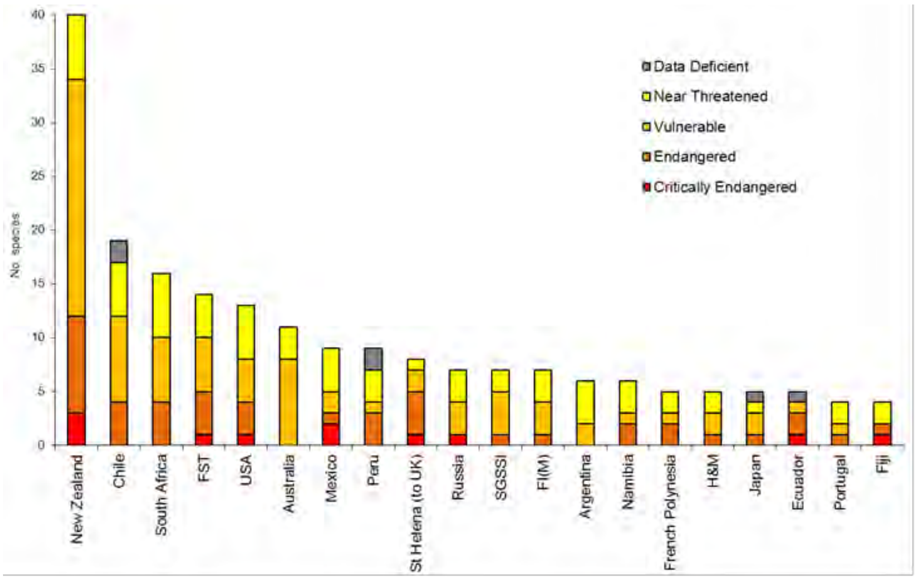


Figure 8.2: (from Croxall et al. 2012, supplementary material). The number of breeding and resident seabird species by country in each IUCN category (excluding Least Concern). FST, French Southern Territories; SGSSI, South Georgia and South Sandwich Islands; FI(M), Falkland Islands (Malvinas); H&M, Heard Island and McDonald Islands.

Table 8.2: (from Taylor 2000). Number of species (spp.) and taxa of seabirds of different families in New Zealand and worldwide in 2000. Additional taxa may have been recorded since. '–' No species from family in this group.

Family	Common name	World breeding		NZ breeding		NZ visitors, vagrants	
		N spp.	N taxa	N spp.	N taxa	N spp.	N taxa
Spheniscidae	Penguins	17	26	6	10	8	10
Gaviidae	Divers, loons	4	6	–	–	–	–
Podicipedidae	Grebes	10	20	2	2	–	–
Diomedeidae	Albatrosses	24	24	13	13	7	7
Procellariidae	Petrels, shearwaters	70	109	28	31	20	23
Hydrobatidae	Storm petrels	20	36	4	5	2	3
Pelecanoididae	Diving petrels	4	9	2	4	–	–
Phaethontidae	Tropicbirds	3	12	1	1	1	1
Pelecanidae	Pelicans	7	12	–	–	1	1
Sulidae	Gannets	9	19	2	2	1	1
Phalacrocoracidae	Shags	39	57	12	13	–	–
Fregatidae	Frigatebirds	5	11	–	–	2	2
Anatidae	Marine ducks	18	27	–	–	–	–
Scolopacidae	Phalaropes	2	2	–	–	2	2
Chionididae	Sheathbills	2	5	–	–	–	–
Stercorariidae	Skuas	7	10	1	1	4	4
Laridae	Gulls	51	78	3	3	–	–
Sternidae	Terns, noddies	43	121	10	11	8	8
Rynchopidae	Skimmers	2	4	–	–	–	–
Alcidae	Auks, puffins	22	45	–	–	–	–
Total		359	633	84	96	56	62

To evaluate whether the viability of seabird populations is jeopardised by incidental mortality from commercial fishing, the number of annual fatalities needs to be compared with the capacity of the populations to replace those losses; this depends on the size and productivity of each population. Sufficient data to build fully quantitative population models to assess risks and explore the likely results of different management approaches are available for only very few taxa (e.g., Fletcher et al. 2008, Francis et al. 2008, Francis & Bell 2010, Dillingham & Fletcher 2011). For this reason, seabird risk assessments are generally made using only expert knowledge (Level 1) or when available, semi-quantitative using a mix of expert knowledge and data (Level 2) (Hobday et al. 2007). Rowe (2013) described a Level 1 seabird risk assessment and Baird et al. (2006, updated by Baird & Gilbert 2010) described a semi-quantitative assessment for seabird taxa for which reasonable numbers of observed captures were available. These assessments could not be used directly to quantify risk for all seabird taxa and fisheries. More comprehensive and quantitative Level 2 risk assessments have since been conducted and are described in more detail in section 8.4.2.3 *Seabird Spatially Explicit Fisheries Risk Assessment*. Furthermore, observer programmes overseas are enabling quantitative Level 2 risk assessments to be carried out over larger spatial scales. For example, current work supported by Fisheries New Zealand aims to develop a Level 2 risk assessment for a selection of New Zealand seabirds across the entire southern hemisphere.

8.4.1 QUANTIFYING FISHERIES INTERACTIONS

Onboard captures recorded by observers represent the most reliable source of information for monitoring trends in total captures and capture rates, but these data have three main deficiencies with respect to estimating total fatalities, especially to species level. First, some captured seabirds are released alive (28% in trawl fisheries on average between 2002–03 and 2017–18, 27% in surface longline fisheries, and 27% in bottom longline fisheries), meaning that, all else being equal, estimates of captures may result in the overestimation of total fatalities, depending on the survival rate of those released. There is a trend in the percentage of albatross observed caught on trawl vessels that were released alive, with a general increase from 2009–10; this trend is less apparent for all birds or in other methods (Table 8.3). Second, identifications by observers are not completely reliable and sometimes use generic codes rather than species codes. From 2002–03 to 2015–16, 68% of all observed seabird captures have either been returned for necropsy or photographs taken for confirmation of identification. As a result of the expert review, 22.4% of the species identifications made by observers were changed (Thompson et al. unpublished data). Third, not all birds killed or mortally wounded by fishing gear are recovered on a fishing vessel. Some birds caught on longline hooks fall off before being recovered, and birds that collide with trawl warps may be dragged under the water and drowned or injured to the extent that they are unable to fly or feed. Excluding this 'cryptic' mortality means that, all else being equal, estimates of captures will underestimate total fatalities, and the extent of underestimation will vary among taxa and fisheries. These deficiencies do not greatly affect the suitability of estimates of captures and capture rates for monitoring purposes, but they have necessitated the development of alternative methods for assessing risk and population consequences.

Information with which to characterise seabird interactions with fisheries comes from a variety of sources. Some is opportunistically collected, whilst other information collection is targeted at specifically describing the nature and extent of seabird captures in fisheries. This section is focused on the targeted information collection.

Many New Zealand commercial fisheries have Fisheries New Zealand observer coverage, some of which is funded by DOC's CSP programme (e.g., Rowe 2009, 2010, Ramm 2011, 2012). Observers collect independent data on the number of captures of seabirds, the number of fishing events observed, and at-sea identification of the seabirds for these fisheries. Commercial fishers are legally required to provide effort data allowing estimation of the total number of fishing events in a fishery. In combination these data have been used for many years to assess the nature and extent of seabird captures in fisheries (e.g., Baird 1994, 1995, 1996, 1997, Baird et al. 1998, Baird 1999, Baird et al. 1999, Baird 2000, 2001a, 2001b, 2003, 2004a–c, Ayers et al. 2004, Baird & Griggs 2004, Baird 2005, Abraham & Thompson 2009a, Thompson & Abraham 2009, Abraham & Thompson 2010, Abraham et al. 2010b, 2011a, 2011b). In this context, 'captures' include all seabirds observed by an observer to be brought onboard a fishing vessel, whether reported as live or dead, but exclude non-fishing-related events (e.g., birds striking the superstructure and landing on deck) and decomposed carcasses. Specimens and photographs (especially for birds released alive) are also collected allowing verification of at-sea identifications (from carcasses, tissue samples, or photographs) and description of biological characters (sex, age, condition, etc., available only from carcasses).

Table 8.3: Percentage of observed captures that were released alive (https://protectedspeciescaptures.nz/PSCv6/. Data version v11.)

Fishing year	All birds			Albatross spp. only		
	Trawl	SLL	BLL	Trawl	SLL	BLL
2002–03	25	18	9	8	27	11
2003–04	9	30	32	4	31	80
2004–05	18	41	33	10	48	-
2005–06	19	38	49	7	40	43
2006–07	19	22	12	11	24	0
2007–08	20	38	10	18	38	30
2008–09	27	26	37	19	34	50
2009–10	37	30	57	30	32	-
2010–11	31	45	45	38	51	100
2011–12	25	16	70	23	18	88
2012–13	39	26	0	34	27	0
2013–14	40	25	15	27	26	10
2014–15	52	42	15	46	47	100
2015–16	30	10	8	28	9	20
2016–17	27	26	31	38	32	0

2017–18	36	9	12	44	11	0
2018–19	33	16	17	30	15	0
2019–20	32	35	19	32	11	90

In some fisheries, observer data are temporally and spatially well stratified, whereas, in others, data are only available from a spatially select part of the fishery, or a limited part of the year. Where sufficient observer data are available, estimates of total seabird captures in the fishery are calculated. The methods currently used in estimating seabird captures in New Zealand fisheries are described by Abraham & Richard (2019). In this context, captures include all seabirds recovered on a fishing vessel except birds that land on the deck or collide with a vessel's superstructure, decomposing animals, records of tissue fragments, and birds caught during trips carried out under special permit (e.g., for trials of mitigation methods). See section 8.1 for a definition of 'seabird capture'. Observer coverage has been variable in that some fisheries and areas have had much higher coverage than others. This complicates estimation of the total number of seabirds captured, especially when estimates include more than one fishery, because the distribution of birds and captures is also heterogeneous (Figure 8.3).

Fisher-reported captures (on NFPSCR forms available since 1 October 2008) have not been used to estimate total captures because the reported capture rates vary by fishery and the species identification can be less certain.

Abraham et al. (2016) made model-based estimates of captures in New Zealand trawl and longline fisheries for the following taxa or groups: sooty shearwater, white-chinned petrel, black petrel, grey petrel, white-capped albatross, Salvin's albatross, southern Buller's albatross, other albatrosses, and all other birds. These individual species were chosen because they are the most frequently caught in trawl and longline fisheries. Captures of other albatrosses are mostly Gibson's *Diomedea antipodensis gibsonii* or Antipodean albatrosses or Campbell Island albatross *Thalassarche impavida*. The 'other birds' category includes many taxa but grey, black, grey-faced *Pterodroma macroptera gouldi*, and Cape petrels (both subspecies but mostly southern Cape petrels, *Daption capense capense*), flesh-footed shearwater, and spotted shag *Stictocarbo punctatus punctatus* are relatively common observed captures (the latter based on few observations that included 31 captures in one event). Estimated captures up to and including the 2019–20 year are shown in Table 8.4 to Table 8.12.

Observed captures of seabirds in trawl fisheries were most common off both coasts of the South Island, along the Chatham Rise, on the fringes of the Stewart-Snares shelf, and around the Auckland Islands (Figure 8.4). This largely reflects the distribution of the major commercial fisheries for squid, hoki, and middle-depth species, which have tended to have relatively high observer coverage. White-capped, Salvin's, and southern Buller's albatrosses have been the most frequently observed captured species of albatrosses, and white-chinned petrel and sooty shearwater have been the other species most frequently observed (Table 8.13). About 38% of observed captures were albatrosses.

Observed captures of seabirds in surface longline fisheries were most common off the south-west coast of the South Island and the north-east coast of the North Island (Figure 8.5), again largely reflecting the distribution of the major commercial fisheries (for southern bluefin and other tunas). The foreign charter fleet targeting tuna has historically had much higher observer coverage than the domestic fleet. Southern Buller's and white-capped albatrosses have been the most frequently observed captured species of albatrosses, and grey, white-chinned, and black petrels have been the other species most frequently observed (Table 8.14). About 80% of observed captures were albatrosses.

Observed captures of seabirds in bottom longline fisheries were most common off the south coast of the South Island, along the Chatham Rise, scattered throughout the subantarctic, and off the north-east coast of the North Island, especially around the Hauraki Gulf (Figure 8.6). This distribution largely reflects the distribution of the ling and snapper longline fisheries that have received most observer coverage; other bottom longline fisheries have had much less coverage. Salvin's and Chatham albatrosses have been the most frequently observed captured of the albatross species, and white-chinned petrel, flesh-footed shearwater, grey petrel, sooty shearwater, and black petrel have been the other species most frequently observed

(Table 8.15). Only about 14% of observed captures were albatrosses.

Model-based estimates of captures can be compared across trawl and longline fisheries (Figure 8.7). Summed across all bird taxa, trawl, surface longline, and bottom longline fisheries account for 41%, 17%, and 42% of estimated captures, respectively, but there are substantial differences in these proportions among seabird taxa. A high proportion (81% between 2002–03 and 2016–17) of estimated total captures of white-capped albatross are from trawl fisheries with most of the remainder estimated from the surface longline fisheries. The trawl fishery also accounts for 92% of the estimated captures of sooty shearwaters, with most of the remainder taken by bottom longlines. The proportion of estimated captures by trawl fisheries reduces to 12% for all other albatrosses (i.e., not including white-capped, Salvin's, and Buller's (both southern Buller's and northern Buller's) combined, with 44% and 43% taken in surface and bottom longline fisheries, respectively. Bottom longline and trawl fisheries account for 29% and 66%, of the estimated total captures of white-chinned petrels respectively.

Over the 2002–03 to 2017–18 period, there appear to have been downward trends (across all fisheries) in the estimated captures of albatross and non-albatross taxa other than white-chinned petrel (Figure 8.7). Estimated captures of white-chinned petrel appear to have fluctuated without much trend, although there is some evidence for an increasing trend for white-chinned petrel, especially in trawl fisheries, although with large annual variations.

Because fishing effort often changes with time, estimates of total captures may not be the only index required for comprehensive monitoring. The number of captures is clearly more biologically relevant for birds, but capture rates by fishery may be more useful measures to assess fishery performance and the effectiveness of mitigation approaches. Dividing modelled catch estimates by the number of tows or hooks set in a particular fishery in each year provides capture rate indices by fishery. These are typically reported as the number of birds captured per 100 trawl tows or per 1000 longline hooks (Figures 8.8 to 8.14).

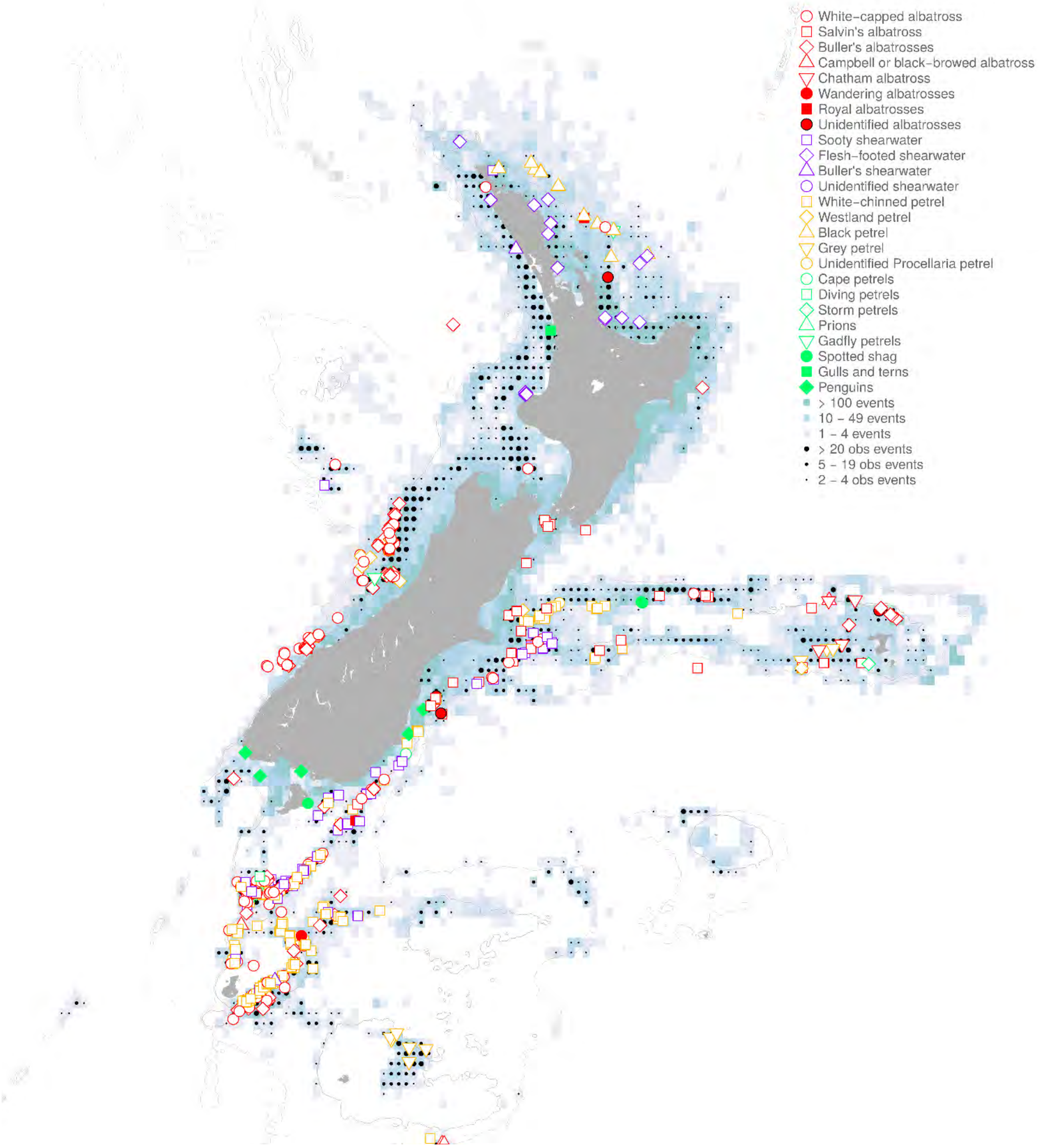


Figure 8.3: All observed seabird captures in trawl, surface longline, bottom longline, set net, and purse seine fishing within New Zealand region, between October 2017 and September 2018. The colour within each 0.2 degree cell indicates the number of fishing events (tows and sets, darker colours indicate more fishing) and the black dots indicate the number of observed fishing events (larger dots indicate more observations). The coloured symbols indicate the location of observed seabird captures, randomly jittered by 0.2 degrees. The 500 m and 100 m depth contours are shown. Data version v2019001.

Table 8.4: Summary of model-estimated total captures of all seabirds combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. [Protected species bycatch (protectedspeciescaptures.nz)](protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	2.65	1 128	952–1338	50 302	0.3	0.70	581	430–763
2008–09	35 926	22.4	5.10	1 478	1269–1719	51 623	3.2	3.34	636	488–808
2009–10	36 770	22.3	3.14	1 124	958–1331	56 123	1.4	1.93	668	505–860
2010–11	35 525	18.9	5.48	1 491	1278–1743	50 553	1.4	0.27	620	466–797
2011–12	34 124	25.9	2.72	1 018	873–1181	50 294	1.0	1.33	590	447–767
2012–13	33 305	36.6	5.82	1 314	1198–1444	50 532	0.4	0.89	623	466–809
2013–14	34 616	33.4	4.17	1 111	995–1245	50 494	3.4	0.99	644	485–828
2014–15	34 742	33.1	5.27	1 333	1194–1496	44 023	4.7	0.83	542	408–697
2015–16	34 302	32.0	4.01	1 018	911–1137	43 727	4.6	1.35	541	408–696
2016–17	33 884	30.9	4.06	1 042	925–1173	44 289	7.3	0.34	531	395–686
2017–18	34 556	36.9	3.89	1 026	922–1146	39 687	5.5	0.27	497	368–649
2018–19	33 561	37.2	4.45	1 088	987–1203	37 363	3.8	0.63	504	375–653
2019–20	32 837	41.0	4.21	1 073	974–1189	33 157	6.5	0.61	430	320–560
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.01	324	172–614	27 370 157	1.8	0.04	1 493	1 131–2 021
2008–09	12 861 501	24.8	0.00	296	117–672	24 572 503	3.6	0.04	1 406	1 067–1 892
2009–10	13 607 740	12.6	0.01	319	149–647	26 846 371	2.7	0.09	1 436	1 095–1 911
2010–11	12 914 717	11.8	0.01	341	166–663	27 983 729	1.1	0.01	1 575	1 199–2 109
2011–12	11 560 277	17.5	0.00	194	81–404	26 313 426	0.3	0.07	1 412	1 062–1 920
2012–13	8 240 515	3.3	0.00	206	108–394	24 270 564	1.9	0.01	1 270	965–1 722
2013–14	16 448 081	11.7	0.02	570	325–1 003	24 416 374	4.1	0.06	1 194	938–1 532
2014–15	14 076 799	2.5	0.03	414	216–827	25 284 349	2.1	0.03	1 113	866–1 456
2015–16	18 603 012	10.8	0.04	525	325–865	24 885 644	2.5	0.04	1 054	809–1 410
2016–17	22 163 805	17.6	0.00	479	248–908	24 398 951	4.5	0.04	1 036	798–1 364
2017–18	16 481 655	31.4	0.00	184	99–343	23 696 351	3.0	0.02	1 002	761–1 341
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.03	–	–
2019–20	17 601 499	17.6	0.02	–	–	22 950 633	5.8	0.03	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.08	42	28–72	1 677 154	8.1	0.10	403	317–508
2008–09	809 230	97.2	0.05	44	42–52	2 306 403	6.5	0.10	511	403–647
2009–10	478 558	100	0.12	56	56–56	2 516 706	7.3	0.43	648	531–793
2010–11	503 370	100	0.06	29	29–29	2 684 809	6.4	0.11	620	497–778
2011–12	551 440	100.6	0.06	33	33–33	2 548 437	6.8	0.18	691	565–844
2012–13	487 520	100	0.01	5	5–5	2 389 412	3.1	0.30	661	544–802
2013–14	653 330	100	0.02	16	16–16	1 897 484	6.8	0.15	544	445–668
2014–15	622 300	99.4	0.04	22	22–23	1 791 086	6.0	0.15	488	395–597
2015–16	54 450	43.6	1.14	44	30–76	2 304 091	13.0	0.35	672	563–799

2016–17	0					2 094 236	16.5	0.16	546	451–655
2017–18	0					2 292 381	12.9	0.33	636	536–758
2018–19	0					2 056 736	9.4	0.35	–	–
2019–20	0					2 000 759	9.8	0.12	–	–

Table 8.5: Summary of model-estimated total captures of white-capped albatross combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.51	200	146–264	50 302	0.3	0.00	144	78–232
2008–09	35 926	22.4	0.98	283	218–358	51 623	3.2	0.60	153	90–238
2009–10	36 770	22.3	0.46	180	130–240	56 123	1.4	0.39	173	99–272
2010–11	35 525	18.9	0.64	211	153–281	50 553	1.4	0.14	163	90–256
2011–12	34 124	25.9	0.71	224	170–291	50 294	1.0	1.14	160	91–248
2012–13	33 305	36.6	1.10	229	195–272	50 532	0.4	0.44	174	98–274
2013–14	34 616	33.4	0.67	171	137–215	50 494	3.4	0.18	171	98–270
2014–15	34 742	33.1	0.67	162	130–203	44 023	4.7	0.00	143	80–226
2015–16	34 302	32.0	0.97	203	169–247	43 727	4.6	0.15	150	86–235
2016–17	33 884	30.9	0.79	180	146–225	44 289	7.3	0.00	148	81–233
2017–18	34 556	36.9	0.83	186	155–224	39 687	5.5	0.05	130	72–207
2018–19	33 561	37.2	0.99	207	176–243	37 363	3.8	0.00	132	73–212
2019–20	32 837	41.0	0.94	217	183–260	33 157	6.5	0.14	113	63–178
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0	1	0–5	27 370 157	1.8	0.00	22	5–50
2008–09	12 861 501	24.8	0	1	0–4	24 572 503	3.6	0.00	23	5–51
2009–10	13 607 740	12.6	0	1	0–4	26 846 371	2.7	0.00	20	4–46
2010–11	12 914 717	11.8	0	1	0–5	27 983 729	1.1	0.00	29	7–67
2011–12	11 560 277	17.5	0	1	0–3	26 313 426	0.3	0.00	26	7–57
2012–13	8 240 515	3.3	0	2	0–6	24 270 564	1.9	0.00	20	5–48
2013–14	16 448 081	11.7	0	3	0–8	24 416 374	4.1	0.00	21	5–48
2014–15	14 076 799	2.5	0	2	0–7	25 284 349	2.1	0.00	19	5–42
2015–16	18 603 012	10.8	0	2	0–7	24 885 644	2.5	0.00	19	4–44
2016–17	22 163 805	17.6	0	2	0–6	24 398 951	4.5	0.02	23	7–48
2017–18	16 481 655	31.4	0	1	0–4	23 696 351	3.0	0.00	18	4–41
2018–19	18 007 521	11.5	0	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0	–	–	22 950 633	5.8	0.00	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.01	6	3–14	1 677 154	8.1	0.00	34	15– 63
2008–09	809 230	97.2	0.00	2	2–3	2 306 403	6.5	0.01	44	22– 76
2009–10	478 558	100	0.02	11	11–11	2 516 706	7.3	0.10	72	48–106
2010–11	503 370	100	0.01	4	4–4	2 684 809	6.4	0.00	53	27– 89
2011–12	551 440	100.6	0.01	6	6–6	2 548 437	6.8	0.01	148	86–229

2012–13	487 520	100	0.00	2	2–2	2 389 412	3.1	0.14	140	86–216
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.05	115	69–179
2014–15	622 300	99.4	0.00	3	3–3	1 791 086	6.0	0.04	105	62–164
2015–16	54 450	43.6	0.46	15	11–29	2 304 091	13.0	0.10	140	93–207
2016–17	0					2 094 236	16.5	0.05	125	80–190
2017–18	0					2 292 381	12.9	0.18	132	97–180
2018–19	0					2 056 736	9.4	0.11	–	–
2019–20	0					2 000 759	9.8	0.01	–	–

Table 8.6: Summary of model-estimated total captures of Salvin's albatross combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.09	121	72–180	50 302	0.3	0.70	111	53–190
2008–09	35 926	22.4	0.35	163	114–225	51 623	3.2	0.60	125	65–204
2009–10	36 770	22.3	0.44	176	126–237	56 123	1.4	1.03	137	71–227
2010–11	35 525	18.9	0.33	173	121–235	50 553	1.4	0.00	126	61–215
2011–12	34 124	25.9	0.27	152	106–207	50 294	1.0	0.00	116	56–197
2012–13	33 305	36.6	0.43	179	134–231	50 532	0.4	0.00	128	62–220
2013–14	34 616	33.4	0.44	192	145–248	50 494	3.4	0.00	142	70–239
2014–15	34 742	33.1	0.38	207	153–274	44 023	4.7	0.00	112	55–190
2015–16	34 302	32.0	0.29	160	115–214	43 727	4.6	0.00	97	46–167
2016–17	33 884	30.9	0.24	142	99–195	44 289	7.3	0.00	107	52–184
2017–18	34 556	36.9	0.28	161	116–215	39 687	5.5	0.00	118	56–203
2018–19	33 561	37.2	0.16	134	92–184	37 363	3.8	0.00	123	58–212
2019–20	32 837	41.0	0.36	159	120–205	33 157	6.5	0.37	110	56–182
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.00	21	2–72	27 370 157	1.8	0.00	55	16–139
2008–09	12 861 501	24.8	0.00	29	3–95	24 572 503	3.6	0.00	54	17–121
2009–10	13 607 740	12.6	0.00	32	1–153	26 846 371	2.7	0.00	54	18–123
2010–11	12 914 717	11.8	0.00	21	4–59	27 983 729	1.1	0.00	65	22–147
2011–12	11 560 277	17.5	0.00	24	1–108	26 313 426	0.3	0.00	66	21–150
2012–13	8 240 515	3.3	0.00	27	6–79	24 270 564	1.9	0.00	59	20–135
2013–14	16 448 081	11.7	0.00	66	23–160	24 416 374	4.1	0.00	54	18–117
2014–15	14 076 799	2.5	0.00	25	6–58	25 284 349	2.1	0.00	49	16–109
2015–16	18 603 012	10.8	0.00	48	21–97	24 885 644	2.5	0.00	46	14–100
2016–17	22 163 805	17.6	0.00	40	10–105	24 398 951	4.5	0.00	36	12–77
2017–18	16 481 655	31.4	0.00	10	2–25	23 696 351	3.0	0.00	45	14–101
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.00	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0	0	0–1	1 677 154	8.1	0.01	10	4–21

2008–09	809 230	97.2	0	2	2–2	2 306 403	6.5	0.01	12	4–26
2009–10	478 558	100	0	0	0–0	2 516 706	7.3	0.01	14	4–30
2010–11	503 370	100	0	0	0–0	2 684 809	6.4	0.00	9	2–21
2011–12	551 440	100.6	0	0	0–0	2 548 437	6.8	0.01	8	2–17
2012–13	487 520	100	0	0	0–0	2 389 412	3.1	0.00	8	1–17
2013–14	653 330	100	0	0	0–0	1 897 484	6.8	0.00	6	1–13
2014–15	622 300	99.4	0	0	0–0	1 791 086	6.0	0.00	4	0–11
2015–16	54 450	43.6	0	0	0–1	2 304 091	13.0	0.00	6	1–13
2016–17	0					2 094 236	16.5	0.00	4	0–9
2017–18	0					2 292 381	12.9	0.00	9	2–20
2018–19	0					2 056 736	9.4	0.00	–	–
2019–20	0					2 000 759	9.8	0.00	–	–

Table 8.7: Summary of model-estimated total captures of southern Buller's albatross combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl*						Inshore trawl*				
2007–08	39 220	22.7	0.19	101	69–143	50 302	0.3	0.00	23	1–72
2008–09	35 926	22.4	0.25	93	63–129	51 623	3.2	0.00	23	1–70
2009–10	36 770	22.3	0.15	86	55–123	56 123	1.4	0.00	25	1–78
2010–11	35 525	18.9	0.30	93	65–128	50 553	1.4	0.00	21	1–64
2011–12	34 124	25.9	0.41	128	95–170	50 294	1.0	0.00	21	1–66
2012–13	33 305	36.6	0.49	117	94–149	50 532	0.4	0.00	21	0–65
2013–14	34 616	33.4	0.31	99	73–134	50 494	3.4	0.00	21	1–65
2014–15	34 742	33.1	0.28	103	77–136	44 023	4.7	0.00	16	0–50
2015–16	34 302	32.0	0.51	125	98–159	43 727	4.6	0.00	19	0–59
2016–17	33 884	30.9	0.24	85	62–116	44 289	7.3	0.00	19	0–57
2017–18	34 556	36.9	0.36	101	77–134	39 687	5.5	0.00	15	0–46
2018–19	33 561	37.2	0.44	–	–	37 363	3.8	0.00	–	–
2019–20	32 837	41.0	0.17	–	–	33 157	6.5	0.00	–	–
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.00	14	5–32	27 370 157	1.8	0.00	48	13–115
2008–09	12 861 501	24.8	0.00	4	0–11	24 572 503	3.6	0.00	27	7–65
2009–10	13 607 740	12.6	0.00	8	1–22	26 846 371	2.7	0.00	31	8–76
2010–11	12 914 717	11.8	0.00	8	1–23	27 983 729	1.1	0.00	37	9–94
2011–12	11 560 277	17.5	0.00	3	0–11	26 313 426	0.3	0.04	38	1197
2012–13	8 240 515	3.3	0.00	5	0–14	24 270 564	1.9	0.00	22	5–54
2013–14	16 448 081	11.7	0.00	9	2–23	24 416 374	4.1	0.00	22	5–53
2014–15	14 076 799	2.5	0.00	7	1–19	25 284 349	2.1	0.00	18	4–45
2015–16	18 603 012	10.8	0.00	10	3–24	24 885 644	2.5	0.00	17	4–42
2016–17	22 163 805	17.6	0.00	9	1–27	24 398 951	4.5	0.00	17	4–42
2017–18	16 481 655	31.4	0.00	4	0–13	23 696 351	3.0	0.00	16	3–39

2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.00	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.06	30	18–58	1 677 154	8.1	0.03	56	33– 86
2008–09	809 230	97.2	0.04	29	28–38	2 306 403	6.5	0.01	75	44–120
2009–10	478 558	100	0.09	41	41–41	2 516 706	7.3	0.15	113	80–154
2010–11	503 370	100	0.05	23	23–23	2 684 809	6.4	0.02	88	55–131
2011–12	551 440	100.6	0.05	27	27–27	2 548 437	6.8	0.02	126	82–183
2012–13	487 520	100	0.00	2	2–2	2 389 412	3.1	0.11	108	73–153
2013–14	653 330	100	0.02	15	15–15	1 897 484	6.8	0.06	94	62–135
2014–15	622 300	99.4	0.03	18	18–19	1 791 086	6.0	0.03	78	49–116
2015–16	54 450	43.6	0.59	19	14–40	2 304 091	13.0	0.14	135	101–179
2016–17	0					2 094 236	16.5	0.04	103	70–149
2017–18	0					2 292 381	12.9	0.06	96	66–135
2018–19	0					2 056 736	9.4	0.09	–	–
2019–20	0					2 000 759	9.8	0.00	–	–

Table 8.8: Summary of model-estimated total captures of white-chinned petrel combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.65	215	140–316	50 302	0.3	0.00	16	1–47
2008–09	35 926	22.4	1.31	311	219–429	51 623	3.2	0.06	14	2–40
2009–10	36 770	22.3	0.88	250	166–360	56 123	1.4	0.00	17	1–50
2010–11	35 525	18.9	1.87	373	265–514	50 553	1.4	0.00	17	1–51
2011–12	34 124	25.9	0.71	204	136–294	50 294	1.0	0.00	14	1–41
2012–13	33 305	36.6	2.36	404	353–473	50 532	0.4	0.00	17	1–49
2013–14	34 616	33.4	1.31	246	202–304	50 494	3.4	0.00	18	1–51
2014–15	34 742	33.1	2.58	450	385–534	44 023	4.7	0.00	16	1–47
2015–16	34 302	32.0	1.47	240	203–288	43 727	4.6	0.00	15	1–45
2016–17	33 884	30.9	1.37	259	207–325	44 289	7.3	0.00	13	1–40
2017–18	34 556	36.9	1.72	302	262–352	39 687	5.5	0.00	13	1–39
2018–19	33 561	37.2	1.48	252	219–296	37 363	3.8	0.00	12	1–37
2019–20	32 837	41.0	2.04	356	319–406	33 157	6.5	0.00	12	1–37
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.00	194	60–474	27 370 157	1.8	0.01	344	121–772
2008–09	12 861 501	24.8	0.00	180	27–533	24 572 503	3.6	0.00	325	113–727
2009–10	13 607 740	12.6	0.00	171	32–475	26 846 371	2.7	0.00	308	109–691
2010–11	12 914 717	11.8	0.01	208	70–488	27 983 729	1.1	0.00	352	125–800
2011–12	11 560 277	17.5	0.00	95	12–285	26 313 426	0.3	0.00	332	108–814
2012–13	8 240 515	3.3	0.00	100	20–283	24 270 564	1.9	0.00	298	99–702
2013–14	16 448 081	11.7	0.02	391	165–816	24 416 374	4.1	0.00	223	81–499

2014–15	14 076 799	2.5	0.03	291	104–680	25 284 349	2.1	0.00	202	68–470
2015–16	18 603 012	10.8	0.04	385	202–718	24 885 644	2.5	0.01	240	85–546
2016–17	22 163 805	17.6	0.00	343	129–766	24 398 951	4.5	0.02	226	89–495
2017–18	16 481 655	31.4	0.00	125	48–279	23 696 351	3.0	0.00	212	69–495
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.02	–	–	22 950 633	5.8	0.01	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.01	5	4–11	1 677 154	8.1	0.00	15	4–32
2008–09	809 230	97.2	0.00	2	2–3	2 306 403	6.5	0.01	18	6–40
2009–10	478 558	100	0.00	2	2–2	2 516 706	7.3	0.01	23	8–49
2010–11	503 370	100	0.00	2	2–2	2 684 809	6.4	0.02	24	10–46
2011–12	551 440	100.6	0.00	0	0–0	2 548 437	6.8	0.02	24	10–47
2012–13	487 520	100	0.00	1	1–1	2 389 412	3.1	0.00	27	9–59
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.00	20	6–42
2014–15	622 300	99.4	0.00	0	0–0	1 791 086	6.0	0.02	25	8–59
2015–16	54 450	43.6	0.00	1	0–10	2 304 091	13.0	0.00	32	10–78
2016–17	0					2 094 236	16.5	0.00	19	5–45
2017–18	0					2 292 381	12.9	0.01	29	11–61
2018–19	0					2 056 736	9.4	0.03	–	–
2019–20	0					2 000 759	9.8	0.01	–	–

Table 8.9: Summary of model-estimated total captures of sooty shearwaters combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.91	348	231–505	50 302	0.3	0.00	55	11–141
2008–09	35 926	22.4	1.90	494	351–686	51 623	3.2	0.06	55	13–135
2009–10	36 770	22.3	0.61	266	168–399	56 123	1.4	0.00	62	14–152
2010–11	35 525	18.9	1.70	468	323–659	50 553	1.4	0.00	61	14–148
2011–12	34 124	25.9	0.36	192	112–297	50 294	1.0	0.00	54	11–133
2012–13	33 305	36.6	1.13	256	196–340	50 532	0.4	0.00	59	13–149
2013–14	34 616	33.4	1.11	274	206–367	50 494	3.4	0.00	63	13–153
2014–15	34 742	33.1	1.15	302	222–410	44 023	4.7	0.05	50	11–123
2015–16	34 302	32.0	0.57	172	116–254	43 727	4.6	0.00	49	10–127
2016–17	33 884	30.9	1.26	265	201–356	44 289	7.3	0.09	53	13–129
2017–18	34 556	36.9	0.42	148	96–232	39 687	5.5	0.00	47	10–116
2018–19	33 561	37.2	0.79	214	154–298	37 363	3.8	0.07	47	11–116
2019–20	32 837	41.0	0.51	169	116–247	33 157	6.5	0.00	42	8–104
Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.00	21	8– 46	27 370 157	1.8	0.00	10	0–38
2008–09	12 861 501	24.8	0.00	21	0– 74	24 572 503	3.6	0.00	11	0–36
2009–10	13 607 740	12.6	0.00	46	14–118	26 846 371	2.7	0.00	10	0–35

2010–11	12 914 717	11.8	0.00	19	0– 66	27 983 729	1.1	0.00	12	0–41
2011–12	11 560 277	17.5	0.00	16	0– 55	26 313 426	0.3	0.00	10	0–33
2012–13	8 240 515	3.3	0.00	16	0– 56	24 270 564	1.9	0.00	9	0–29
2013–14	16 448 081	11.7	0.00	21	3– 59	24 416 374	4.1	0.00	6	0–22
2014–15	14 076 799	2.5	0.00	13	0– 46	25 284 349	2.1	0.00	6	0–21
2015–16	18 603 012	10.8	0.00	8	0– 26	24 885 644	2.5	0.00	7	0–23
2016–17	22 163 805	17.6	0.00	11	0– 40	24 398 951	4.5	0.00	6	0–21
2017–18	16 481 655	31.4	0.00	2	0– 10	23 696 351	3.0	0.00	8	1–25
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.00	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.00	0	0–1	1 677 154	8.1	0.00	2	0– 9
2008–09	809 230	97.2	0.00	0	0–0	2 306 403	6.5	0.00	3	0–11
2009–10	478 558	100	0.00	0	0–0	2 516 706	7.3	0.00	3	0–11
2010–11	503 370	100	0.00	0	0–0	2 684 809	6.4	0.00	3	0–12
2011–12	551 440	100.6	0.00	0	0–0	2 548 437	6.8	0.00	2	0– 9
2012–13	487 520	100	0.00	0	0–0	2 389 412	3.1	0.00	3	0– 9
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.00	2	0– 8
2014–15	622 300	99.4	0.00	0	0–0	1 791 086	6.0	0.00	2	0– 7
2015–16	54 450	43.6	0.00	0	0–1	2 304 091	13.0	0.00	3	0– 9
2016–17	0					2 094 236	16.5	0.00	2	0– 8
2017–18	0					2 292 381	12.9	0.00	3	0–10
2018–19	0					2 056 736	9.4	0.00		
2019–20	0					2 000 759	9.8	0.00		

Table 8.10: Summary of model-estimated total captures of black petrels combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.01	4	1–17	50 302	0.3	0.00	32	5–85
2008–09	35 926	22.4	0.01	5	1–22	51 623	3.2	0.00	33	6–88
2009–10	36 770	22.3	0.00	4	0–21	56 123	1.4	0.00	32	5–85
2010–11	35 525	18.9	0.00	3	0–18	50 553	1.4	0.00	30	6–80
2011–12	34 124	25.9	0.00	4	0–19	50 294	1.0	0.19	30	5–79
2012–13	33 305	36.6	0.00	4	0–19	50 532	0.4	0.00	26	3–76
2013–14	34 616	33.4	0.00	3	0–18	50 494	3.4	0.29	29	8–74
2014–15	34 742	33.1	0.00	3	0–16	44 023	4.7	0.10	23	4–61
2015–16	34 302	32.0	0.00	4	0–20	43 727	4.6	0.65	35	15–76
2016–17	33 884	30.9	0.00	3	0–17	44 289	7.3	0.18	27	9–69
2017–18	34 556	36.9	0.00	3	0–15	39 687	5.5	0.00	20	2–59
2018–19	33 561	37.2	0.00	3	0–18	37 363	3.8	0.07	21	3–59
2019–20	32 837	41.0	0.01	3	1–15	33 157	6.5	0.00	16	1–49

Large BLL*						Small BLL*				
2007–08	14 123 096	21.8	0.00	1	0– 4	27 370 157	1.8	0.01	314	170–526
2008–09	12 861 501	24.8	0.00	2	0– 7	24 572 503	3.6	0.01	286	160–476
2009–10	13 607 740	12.6	0.00	2	0– 7	26 846 371	2.7	0.06	328	196–522
2010–11	12 914 717	11.8	0.00	2	0– 8	27 983 729	1.1	0.01	315	173–531
2011–12	11 560 277	17.5	0.00	3	0–10	26 313 426	0.3	0.00	259	143–440
2012–13	8 240 515	3.3	0.00	1	0– 4	24 270 564	1.9	0.00	226	126–375
2013–14	16 448 081	11.7	0.00	1	0– 6	24 416 374	4.1	0.01	222	125–366
2014–15	14 076 799	2.5	0.00	1	0– 4	25 284 349	2.1	0.00	216	117–362
2015–16	18 603 012	10.8	0.00	1	0– 4	24 885 644	2.5	0.00	170	93–291
2016–17	22 163 805	17.6	0.00	1	0– 4	24 398 951	4.5	0.01	190	110–310
2017–18	16 481 655	31.4	0.00	0	0– 2	23 696 351	3.0	0.00	151	83–262
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.00	–	–
Large SLL*						Small SLL*				
2007–08	568 285	50.2	0.00	0	0–0	1 677 154	8.1	0.01	44	26– 69
2008–09	809 230	97.2	0.00	0	0–0	2 306 403	6.5	0.01	52	30– 80
2009–10	478 558	100	0.00	0	0–0	2 516 706	7.3	0.03	55	34– 84
2010–11	503 370	100	0.00	0	0–0	2 684 809	6.4	0.01	80	48–120
2011–12	551 440	100.6	0.00	0	0–0	2 548 437	6.8	0.01	69	40–105
2012–13	487 520	100	0.00	0	0–0	2 389 412	3.1	0.00	60	35– 91
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.00	49	28– 77
2014–15	622 300	99.4	0.00	0	0–0	1 791 086	6.0	0.00	41	22– 69
2015–16	54 450	43.6	0.00	0	0–2	2 304 091	13.0	0.02	50	31– 76
2016–17	0					2 094 236	16.5	0.03	50	31– 73
2017–18	0					2 292 381	12.9	0.03	59	36– 91
2018–19	0					2 056 736	9.4	0.03	–	–
2019–20	0					2 000 759	9.8	0.03	–	–

Table 8.11: Summary of model-estimated total captures of grey petrels combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.02	11	3–26	50 302	0.3	0.00	4	0–19
2008–09	35 926	22.4	0.00	10	0–26	51 623	3.2	0.00	4	0–19
2009–10	36 770	22.3	0.11	21	11–39	56 123	1.4	0.00	5	0–23
2010–11	35 525	18.9	0.10	19	9–36	50 553	1.4	0.00	4	0–20
2011–12	34 124	25.9	0.01	5	1–16	50 294	1.0	0.00	4	0–19
2012–13	33 305	36.6	0.07	11	9–19	50 532	0.4	0.00	4	0–18
2013–14	34 616	33.4	0.10	13	11–21	50 494	3.4	0.00	4	0–18
2014–15	34 742	33.1	0.04	7	5–14	44 023	4.7	0.00	3	0–14
2015–16	34 302	32.0	0.03	6	3–14	43 727	4.6	0.00	3	0–15

2016–17	33 884	30.9	0.04	7	4–14	44 289	7.3	0.00	3	0–14
2017–18	34 556	36.9	0.05	9	6–17	39 687	5.5	0.00	3	0–15
2018–19	33 561	37.2	0.22	31	27–41	37 363	3.8	0.00	3	0–13
2019–20	32 837	41.0	0.03	7	4–16	33 157	6.5	0.00	2	0–12
Large BLL						Small BLL				
2007–08	14 123 096	21.8	0.00	36	8–103	27 370 157	1.8	0.00	112	28–286
2008–09	12 861 501	24.8	0.00	27	4– 86	24 572 503	3.6	0.00	97	24–255
2009–10	13 607 740	12.6	0.00	23	1– 91	26 846 371	2.7	0.00	122	29–321
2010–11	12 914 717	11.8	0.00	47	1–232	27 983 729	1.1	0.00	109	26–284
2011–12	11 560 277	17.5	0.00	10	0– 47	26 313 426	0.3	0.00	97	24–251
2012–13	8 240 515	3.3	0.00	12	1– 41	24 270 564	1.9	0.00	86	20–224
2013–14	16 448 081	11.7	0.00	13	1– 47	24 416 374	4.1	0.00	98	25–246
2014–15	14 076 799	2.5	0.00	26	0–132	25 284 349	2.1	0.01	99	28–248
2015–16	18 603 012	10.8	0.00	20	0– 92	24 885 644	2.5	0.00	85	20–211
2016–17	22 163 805	17.6	0.00	20	0– 88	24 398 951	4.5	0.00	86	21–221
2017–18	16 481 655	31.4	0.00	12	4– 31	23 696 351	3.0	0.00	95	24–247
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.00	–	–
Large SLL						Small SLL				
2007–08	568 285	50.2	0.00	0	0–1	1 677 154	8.1	0.01	12	5–24
2008–09	809 230	97.2	0.01	5	5–5	2 306 403	6.5	0.01	13	5–26
2009–10	478 558	100	0.00	0	0–0	2 516 706	7.3	0.01	18	7–33
2010–11	503 370	100	0.00	0	0–0	2 684 809	6.4	0.00	13	4–27
2011–12	551 440	100.6	0.00	0	0–0	2 548 437	6.8	0.01	12	5–23
2012–13	487 520	100	0.00	0	0–0	2 389 412	3.1	0.00	15	5–27
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.01	15	6–27
2014–15	622 300	99.4	0.00	0	0–0	1 791 086	6.0	0.00	14	5–26
2015–16	54 450	43.6	0.00	0	0–1	2 304 091	13.0	0.00	16	6–30
2016–17	0					2 094 236	16.5	0.00	16	7–30
2017–18	0					2 292 381	12.9	0.00	19	8–35
2018–19	0					2 056 736	9.4	0.00	–	–
2019–20	0					2 000 759	9.8	0.00	–	–

Table 8.12: Summary of model-estimated total captures of flesh-footed shearwaters combined by 1 October to 30 September fishing year in inshore trawl fisheries (including flatfish target fisheries) and deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, mackerel, scampi, and middle depth target fisheries) (effort in tows); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL, effort in hooks); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL, effort in hooks) between 2007–08 and 2019–20. Observed and modelled rates are per 100 trawl tows or 1000 longline hooks. % obs, percentage of effort observed; capture rate, observed capture rate. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication. (Continued on next page)

Fishing year	Fishing effort		Capture	Modelled estimates		Fishing effort		Capture	Modelled estimates	
	All effort	% obs	rate	Mean	95% c.i.	All effort	% obs	Rate	Mean	95% c.i.
DW trawl						Inshore trawl				
2007–08	39 220	22.7	0.07	29	11–62	50 302	0.3	0.00	69	24–144
2008–09	35 926	22.4	0.04	24	7–52	51 623	3.2	0.00	69	25–141
2009–10	36 770	22.3	0.01	29	8–64	56 123	1.4	0.13	73	28–150
2010–11	35 525	18.9	0.22	41	21–74	50 553	1.4	0.00	66	24–132
2011–12	34 124	25.9	0.00	19	4–45	50 294	1.0	0.00	65	24–134

2012–13	33 305	36.6	0.00	25	6–56	50 532	0.4	0.00	64	23–131
2013–14	34 616	33.4	0.02	24	7–52	50 494	3.4	0.41	67	28–129
2014–15	34 742	33.1	0.01	21	5–49	44 023	4.7	0.34	60	25–120
2015–16	34 302	32.0	0.01	24	6–54	43 727	4.6	0.05	53	18–113
2016–17	33 884	30.9	0.01	24	7–54	44 289	7.3	0.00	48	16–104
2017–18	34 556	36.9	0.01	28	8–60	39 687	5.5	0.14	49	17–103
2018–19	33 561	37.2	0.00	26	6–61	37 363	3.8	0.00	43	13–94
2019–20	32 837	41.0	0.00	18	3–44	33 157	6.5	0.05	37	12–79
Large BLL						Small BLL				
2007–08	14 123 096	21.8	0.00	0	0– 3	27 370 157	1.8	0.00	271	187–376
2008–09	12 861 501	24.8	0.00	2	0– 7	24 572 503	3.6	0.02	287	203–393
2009–10	13 607 740	12.6	0.00	1	0– 4	26 846 371	2.7	0.02	273	190–374
2010–11	12 914 717	11.8	0.00	1	0– 3	27 983 729	1.1	0.00	304	210–426
2011–12	11 560 277	17.5	0.00	1	0– 4	26 313 426	0.3	0.00	272	187–380
2012–13	8 240 515	3.3	0.00	7	1–14	24 270 564	1.9	0.00	276	191–384
2013–14	16 448 081	11.7	0.00	6	1–13	24 416 374	4.1	0.03	272	193–367
2014–15	14 076 799	2.5	0.00	5	1–11	25 284 349	2.1	0.02	248	173–341
2015–16	18 603 012	10.8	0.00	4	0– 9	24 885 644	2.5	0.02	229	159–315
2016–17	22 163 805	17.6	0.00	4	1–10	24 398 951	4.5	0.00	224	154–314
2017–18	16 481 655	31.4	0.00	0	0– 3	23 696 351	3.0	0.02	223	156–311
2018–19	18 007 521	11.5	0.00	–	–	23 354 322	3.0	0.00	–	–
2019–20	17 601 499	17.6	0.00	–	–	22 950 633	5.8	0.01	–	–
Large SLL						Small SLL				
2007–08	568 285	50.2	0.00	0	0–0	1 677 154	8.1	0.01	118	60–205
2008–09	809 230	97.2	0.00	0	0–0	2 306 403	6.5	0.00	155	79–274
2009–10	478 558	100	0.00	0	0–0	2 516 706	7.3	0.00	156	80–277
2010–11	503 370	100	0.00	0	0–0	2 684 809	6.4	0.01	181	96–318
2011–12	551 440	100.6	0.00	0	0–0	2 548 437	6.8	0.00	138	69–246
2012–13	487 520	100	0.00	0	0–0	2 389 412	3.1	0.00	134	71–233
2013–14	653 330	100	0.00	0	0–0	1 897 484	6.8	0.00	112	58–200
2014–15	622 300	99.4	0.00	0	0–0	1 791 086	6.0	0.01	77	39–141
2015–16	54 450	43.6	0.00	2	0–12	2 304 091	13.0	0.00	104	54–182
2016–17	0					2 094 236	16.5	0.00	82	41–147
2017–18	0					2 292 381	12.9	0.01	111	59–193
2018–19	0					2 056 736	9.4	0.01	–	–
2019–20	0					2 000 759	9.8	0.03	–	–

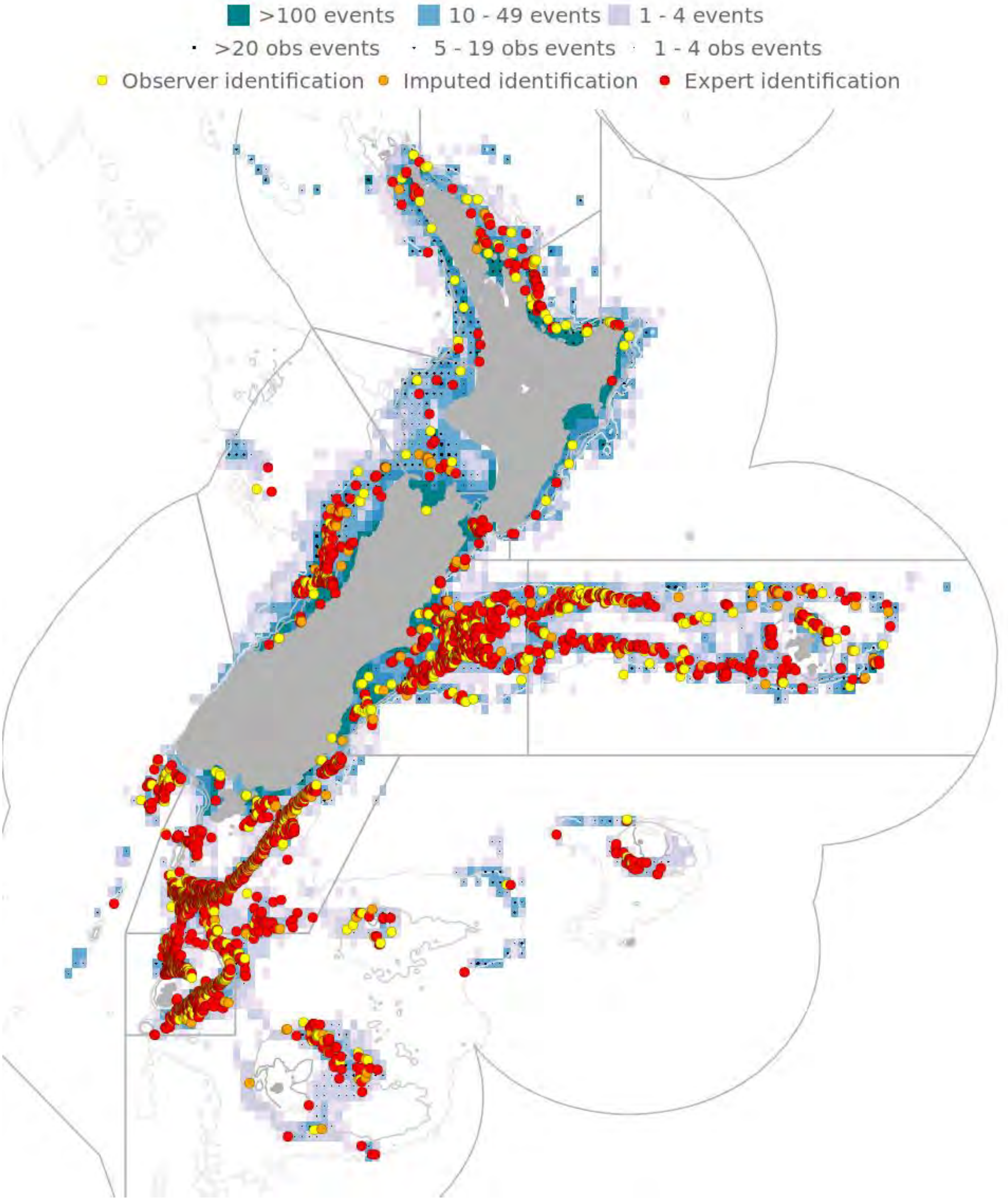


Figure 8.4: Map of trawl fishing effort and all observed seabird captures in trawls, October 2002 to September 2020. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell. https://protectedspeciescaptures.nz/PSCv6/h

Table 8.13: Summary of seabirds observed captured in trawl fisheries 2002–03 to 2019–20. Declared target species are: SQU, arrow squid; HOK+, hoki, hake, ling; MID, other middle-depth species – silver, white, and common warehou, barracouta, alfonsinos, redbait, rubyfish; SCI, scampi; ORH+, orange roughy and oreos; SBW, southern blue whiting; JMA, jack mackerels; INS, inshore species for which one or more captures have been observed – elephantfish, gemfish, red gurnard, tarakihi, red cod, spiny dogfish, John dory, snapper; FLA, flatfishes. https://protectedspeciescaptures.nz/PSCv6/. (Continued on next page)

Species name or group	Declared target species									
	SQU	HOK+	MID	SCI	ORH+	SBW	JMA	INS	FLA	ALL
New Zealand white-capped albatross	1 221	138	169	41	6	2	19	31	4	**1 631**
Salvin's albatross	35	254	110	48	8	17	3	28	0	**503**
Southern Buller's albatross	206	163	73	28	4	1	6	0	0	**481**
Southern royal albatross	15	3	2	0	2	2	0	0	0	**24**
Albatrosses	9	5	1	3	1	0	0	1	0	**20**
Campbell black-browed albatross	1	12	1	1	0	4	0	0	0	**19**
Chatham Island albatross	1	3	3	1	8	0	0	0	0	**16**
Smaller albatrosses	0	2	1	2	0	1	0	0	0	**6**
Black-browed albatross	1	1	2	0	0	0	0	0	0	**4**
Royal albatrosses	3	0	0	0	0	0	0	0	0	**3**
Buller's albatross	1	1	0	0	0	0	0	0	0	**2**
Great albatrosses	0	2	0	0	0	0	0	0	0	**2**
Northern royal albatross	0	1	0	0	1	0	0	0	0	**2**
Wandering albatrosses	0	1	1	0	0	0	0	0	0	**2**
Black-browed albatrosses	0	1	0	0	0	0	0	0	0	**1**
Gibson's albatross	0	0	0	0	1	0	0	0	0	**1**
Light-mantled sooty albatross	1	0	0	0	0	0	0	0	0	**1**
Northern Buller's albatross	0	1	0	0	0	0	0	0	0	**1**
All Albatrosses	**1 494**	**588**	**363**	**124**	**31**	**27**	**28**	**60**	**4**	**2 719**
White-chinned petrel	1 797	210	171	76	3	0	38	1	0	**2 296**
Sooty shearwater	1 045	331	261	39	2	0	13	6	0	**1 697**
Grey petrel	2	2	0	1	1	88	0	0	0	**94**
Flesh-footed shearwater	0	4	1	37	0	0	0	21	0	**63**
Cape petrel	1	30	0	3	8	5	1	0	0	**48**
Common diving petrel	13	8	8	3	3	1	2	5	0	**43**
Westland petrel	0	31	3	1	0	0	1	5	0	**41**
Spotted shag	0	0	0	2	0	0	0	0	32	**34**
Black petrel	1	0	0	1	1	0	0	29	0	**32**
Fairy prion	3	14	1	0	0	1	7	0	0	**26**
Grey-backed storm petrel	5	2	3	2	0	6	2	1	0	**21**
Snares Cape petrel	1	8	0	0	0	7	0	1	0	**17**
Antarctic prion	15	1	0	0	0	0	0	0	0	**16**
New Zealand white-faced storm petrel	1	0	0	0	2	0	4	4	0	**11**
Fulmar prion	0	0	0	0	0	0	10	0	0	**10**
Northern giant petrel	0	6	2	1	1	0	0	0	0	**10**
Cape petrels	2	2	0	0	0	0	2	1	0	**7**
Broad-billed prion	0	0	6	0	0	0	0	0	0	**6**
Grey-faced petrel	0	0	0	0	0	0	0	6	0	**6**
Black-bellied storm petrel	1	2	2	0	0	0	0	0	0	**5**
Fulmars, petrels, prions and shearwaters	2	1	0	2	0	0	0	0	0	**5**
Mottled petrel	2	1	1	0	0	0	0	1	0	**5**

Table 8.13 [Continued]:

Species name or group	Declared target species									
	SQU	HOK+	MID	SCI	JMA	INS	SBW	ORH+	FLA	ALL
Petrels, prions, and shearwaters	0	1	1	0	0	2	0	1	0	**5**
Prions	2	2	0	0	0	0	0	1	0	**5**
Shearwaters	0	0	0	0	0	0	0	4	0	**4**
Buller's shearwater	0	0	0	0	0	0	0	3	0	**3**
Large seabirds	0	1	0	0	0	1	1	0	0	**3**
Southern giant petrel	0	3	0	0	0	0	0	0	0	**3**
Storm petrels	0	2	1	0	0	0	0	0	0	**3**
Procellaria petrels	2	0	0	0	0	0	0	0	0	**2**
Seabirds	1	1	0	0	0	0	0	0	0	**2**
Short-tailed shearwater	0	0	2	0	0	0	0	0	0	**2**
Small seabirds	0	1	0	0	0	0	0	1	0	**2**
Southern black-backed gull	0	0	0	0	0	0	0	1	1	**2**
Wilson's storm petrel	0	0	0	0	0	0	0	2	0	**2**
Australasian gannet	0	0	0	0	0	0	1	0	0	**1**
Cook's Petrel	0	0	0	0	0	0	1	0	0	**1**
Double-banded plover	0	1	0	0	0	0	0	0	0	**1**
Gadfly petrels	0	1	0	0	0	0	0	0	0	**1**
Mid-sized petrels & shearwaters	1	0	0	0	0	0	0	0	0	**1**
White-headed petrel	1	0	0	0	0	0	0	0	0	**1**
All other birds	**2 898**	**666**	**463**	**168**	**21**	**111**	**83**	**94**	**33**	**4 537**
Grand total	**4 392**	**1 254**	**826**	**292**	**52**	**138**	**111**	**154**	**37**	**7 256**

For white-capped albatross, total estimated captures have fluctuated between 407 in 2007-08 to 567 in 2012-13. Together, deepwater, midwater, and inshore trawl fisheries account for 67% of all estimated captures of white-capped albatross in the 2017–18 fishing year the last year estimates were available for all fisheries.

For Salvin's albatross, captures rates have fluctuated without trend in all fisheries except small surface longline vessels where they showed a decrease between 2007–08 and 2016–17. Total estimated captures for all fisheries have fluctuated between 318 in 2007-08 and 460 in 2013-14. Together, trawl fisheries account for 81% of all estimated captures of Salvin's albatross in the 2017–18 fishing year the last year estimates were available for all fisheries.

For southern Buller's albatross, total estimated captures have fluctuated between 232 in 2017-18 to 343 in 2011-12 (Figure 8.10). Together small surface longline fisheries account for 41% of all estimated captures of southern Buller's albatross in the 2017–18 fishing year the last year estimates were available for all fisheries.

For white-chinned petrel, total estimated captures have fluctuated between 669 in 2011-12 to 984 in 2014-15 (Figure 8.11). Together, deepwater trawl fisheries account for 44% of all estimated captures of white-chinned petrel in these years the last year estimates were available for all fisheries.

For black petrels, total estimated captures have fluctuated between 233 in 2017-18 to 430 in 2010-11. Together, small bottom longline fisheries account for 62% of all estimated captures of black petrels in the 2017–18 fishing year the last year estimates were available for all fisheries.

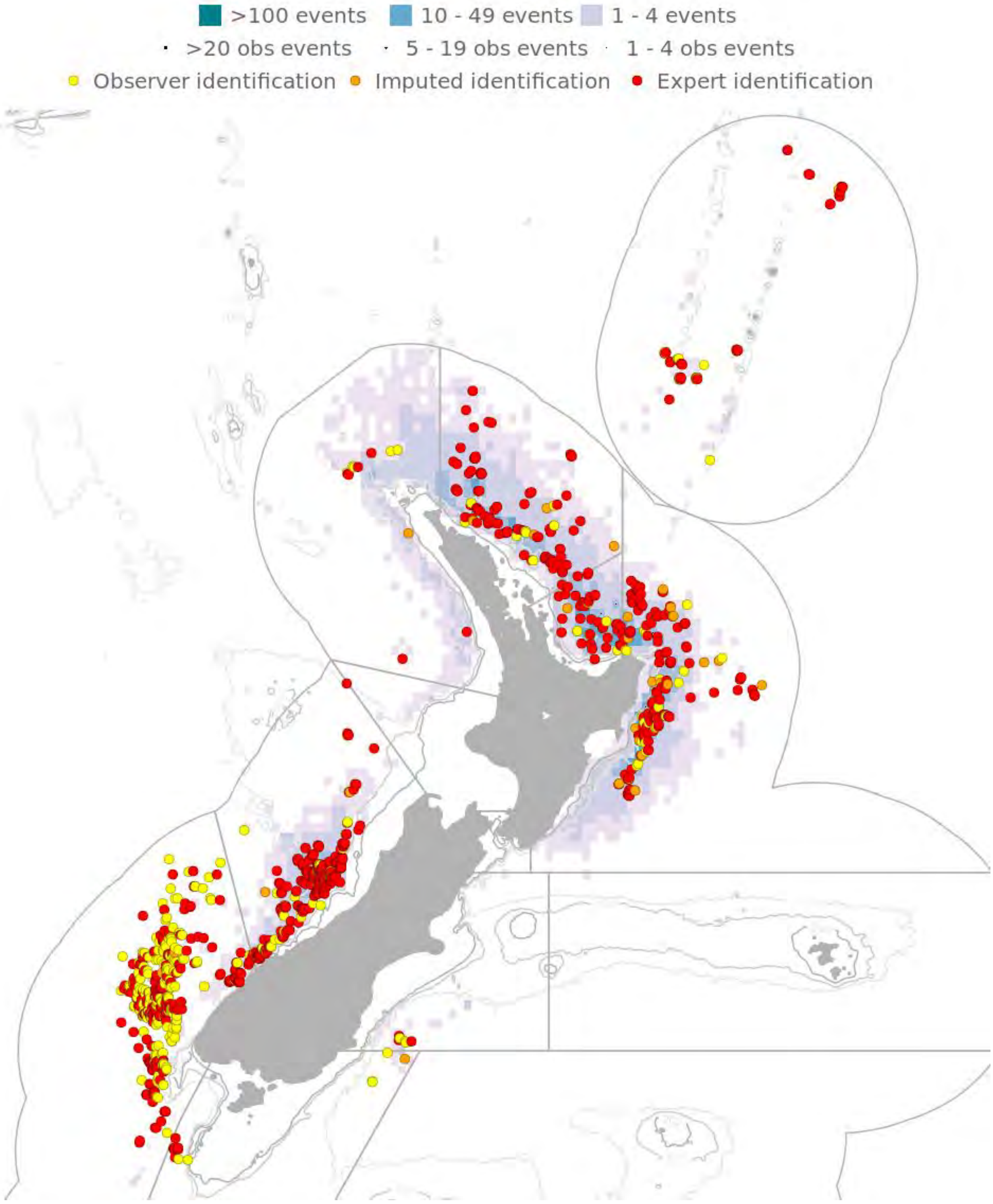


Figure 8.5: Map of surface longline fishing effort and all observed seabird captures by surface longlines, October 2002 to September 2020. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell (here, 90.4% of effort is displayed). https://protectedspeciescaptures.nz/PSCv6/.

Table 8.14: Summary of seabirds observed captured in surface longline fisheries October 2002 to September 2020. Declared target species are: SBT, southern bluefin tuna; BIG, bigeye tuna; SWO, broadbill swordfish; ALB, albacore tuna. https://protectedspeciescaptures.nz/PSCv6/. Data version v11.

Species name or group	Declared target species				
	SBT	SWO	BIG	ALB	ALL
Southern Buller's albatross	480	1	13	9	503
New Zealand white-capped albatross	249	5	4	0	258
Campbell black-browed albatross	27	3	3	17	50
Gibson's albatross	13	12	11	7	43
Antipodean albatross	6	15	18	3	42
Albatrosses	1	33	2	0	36
Southern royal albatross	13	0	3	0	16
Antipodean and Gibson's albatrosses	0	6	2	0	8
Salvin's albatross	5	0	3	0	8
Wandering albatross	2	2	2	0	6
Black-browed albatross	5	0	0	0	5
Black-browed albatrosses	1	2	2	0	5
Wandering albatrosses	5	0	0	0	5
Great albatrosses	2	0	0	0	2
Northern royal albatross	0	0	2	0	2
Grey-headed albatross	1	0	0	0	1
Light-mantled sooty albatross	1	0	0	0	1
Northern Buller's albatross	1	0	0	0	1
Smaller albatrosses	1	0	0	0	1
All albatrosses	813	79	65	36	993
Black petrel	0	5	43	2	50
Grey petrel	42	2	0	5	49
White-chinned petrel	32	7	6	4	49
Westland petrel	46	1	0	1	48
Flesh-footed shearwater	1	2	21	0	24
Grey-faced petrel	3	2	4	15	24
Sooty shearwater	3	1	0	7	11
Cape petrels	2	0	0	0	2
Fulmars, petrels, prions and shearwaters	0	1	0	1	2
Large seabirds	2	0	0	0	2
Southern giant petrel	2	0	0	0	2
White-headed petrel	0	0	0	2	2
Gadfly petrels	0	0	1	0	1
All other birds	133	21	75	37	266
Grand total	946	100	140	73	1 259

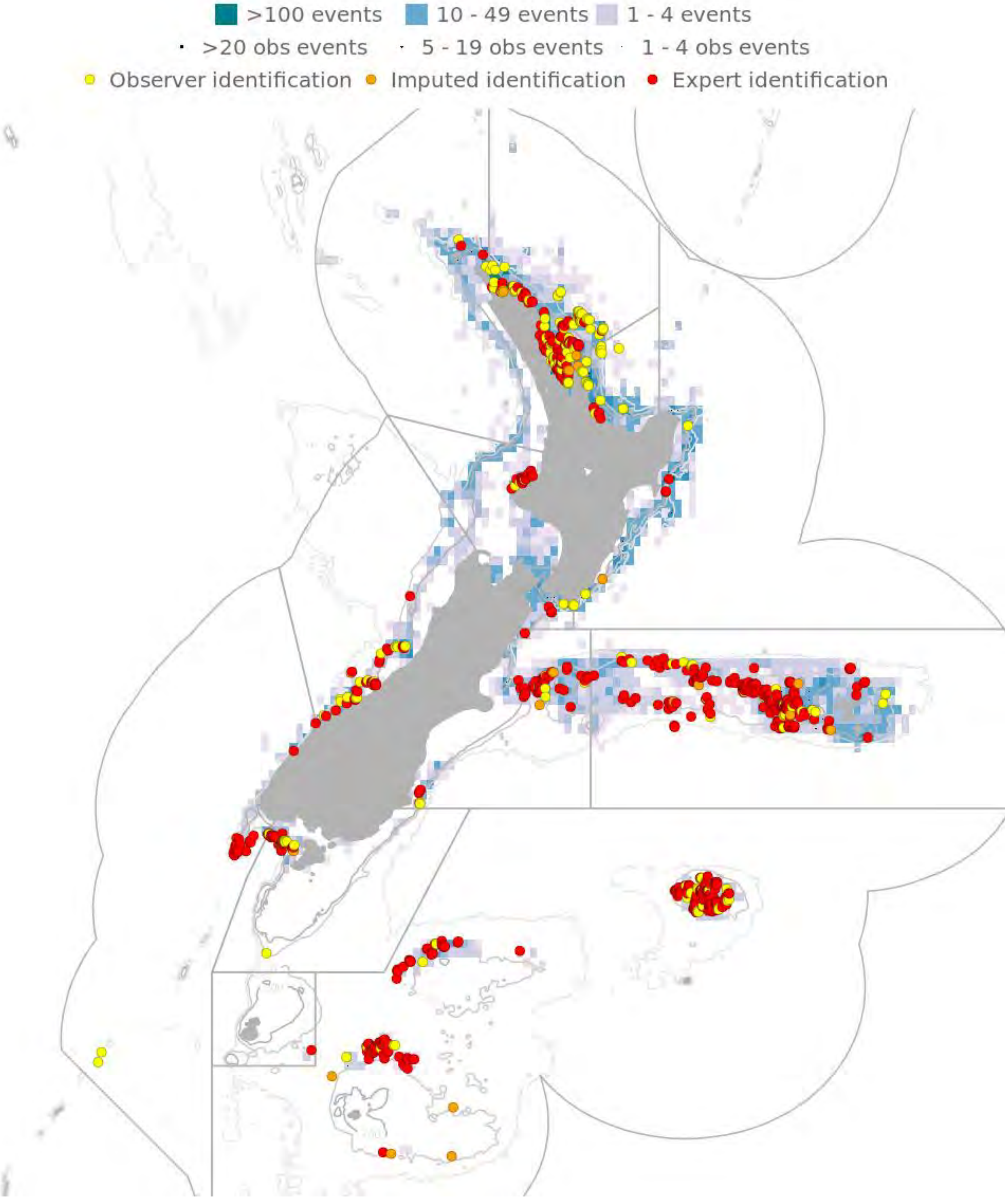


Figure 8.6: Map of bottom longline fishing effort and all observed seabird captures by bottom longline, October 2002 to September 2020. Fishing effort is mapped into 0.2-degree cells, with the colour of each cell being related to the amount of effort (events). Observed fishing events are indicated by black dots, and observed captures are indicated by dots, red for those identified to species by experts, orange for those with identification imputed, and yellow for those using observer identification. Fishing is shown only if the effort could be assigned a latitude and longitude, and if there were three or more vessels fishing within a cell (here, 96.9% of effort is displayed). https://protectedspeciescaptures.nz/PSCv6/. Data version v11.

Table 8.15: Summary of seabirds observed captured in bottom longline fisheries, October 2002 to September 2020. Declared target species are: LIN, ling; SNA, snapper; BNS, bluenose; HPB, hāpuku or bass; Other – red gurnard, kahawai, toothfish, ribaldo, school shark, and tarakihi. https://protectedspeciescaptures.nz/PSCv6/Data version v11.

Species name or group	Declared target species					
	LIN	SNA	BNS	HPB	Other	ALL
Salvin's albatross	64	0	0	0	3	**67**
Chatham Island albatross	21	0	0	0	0	**21**
New Zealand white-capped albatross	14	0	0	0	0	**14**
Southern Buller's albatross	10	0	3	0	0	**13**
Southern royal albatross	6	0	0	0	0	**6**
Campbell black-browed albatross	0	0	2	1	1	**4**
Albatrosses	3	0	0	0	0	**3**
Wandering albatrosses	2	0	1	0	0	**3**
Black-browed albatrosses	2	0	0	0	0	**2**
Black-browed albatross	1	0	0	0	0	**1**
Indian Ocean yellow-nosed albatross	1	0	0	0	0	**1**
All albatrosses	**124**	**0**	**6**	**1**	**4**	**135**
White-chinned petrel	437	0	2	2	20	**461**
Flesh-footed shearwater	0	94	1	3	23	**121**
Black petrel	0	40	50	4	3	**97**
Grey petrel	74	2	0	0	0	**76**
Sooty shearwater	73	2	0	1	0	**76**
Cape petrel	18	1	0	0	0	**19**
Westland petrel	16	0	0	0	2	**18**
Buller's shearwater	0	14	0	0	0	**14**
Fulmars, petrels, prions and shearwaters	1	9	0	0	0	**10**
Fluttering shearwater	0	6	0	0	2	**8**
Southern black-backed gull	0	5	0	0	3	**8**
Common diving petrel	7	0	0	0	0	**7**
Northern giant petrel	4	1	0	0	2	**7**
Grey-faced petrel	0	0	0	6	0	**6**
Prions	6	0	0	0	0	**6**
Cape petrels	4	0	0	0	0	**4**
New Zealand white-faced storm petrel	1	2	1	0	0	**4**
Small seabirds	4	0	0	0	0	**4**
Red-billed gull	0	3	0	0	0	**3**
Australasian gannet	0	2	0	0	0	**2**
Little penguin	0	2	0	0	0	**2**
Pied shag	0	2	0	0	0	**2**
Seagulls	2	0	0	0	0	**2**
Snares Cape petrel	2	0	0	0	0	**2**
Storm petrels	2	0	0	0	0	**2**
Crested penguins	1	0	0	0	0	**1**
Giant petrels	1	0	0	0	0	**1**
Grey-backed storm petrel	1	0	0	0	0	**1**
Petrels, prions, and shearwaters	0	1	0	0	0	**1**
Short-tailed shearwater	1	0	0	0	0	**1**
All other birds	**655**	**186**	**54**	**16**	**55**	**966**
Grand total	**779**	**186**	**60**	**17**	**59**	**1 101**

Figure 8.7: Model-based estimates of captures of the most numerous seabird taxa observed captured in inshore trawl fisheries (including flatfish target fisheries); deepwater (DW) trawl fisheries (including squid, hoki, orange roughy, oreo, southern blue whiting, jack mackerels, scampi, and middle-depth target fisheries); large (> 28 m) and small (< 28 m) surface longline fisheries (SLL); and large (> 28 m) and small (< 28 m) bottom longline fisheries (BLL) between 2002–03 and 2017–18. For confidence limits see Tables 8.5–8.12. Note that this level of aggregation conceals any different trends between areas. Data version v2019001. This figure has not been updated for the 2021–22 edition due to the unavailability of the data at time of publication but will be updated in the next edition.

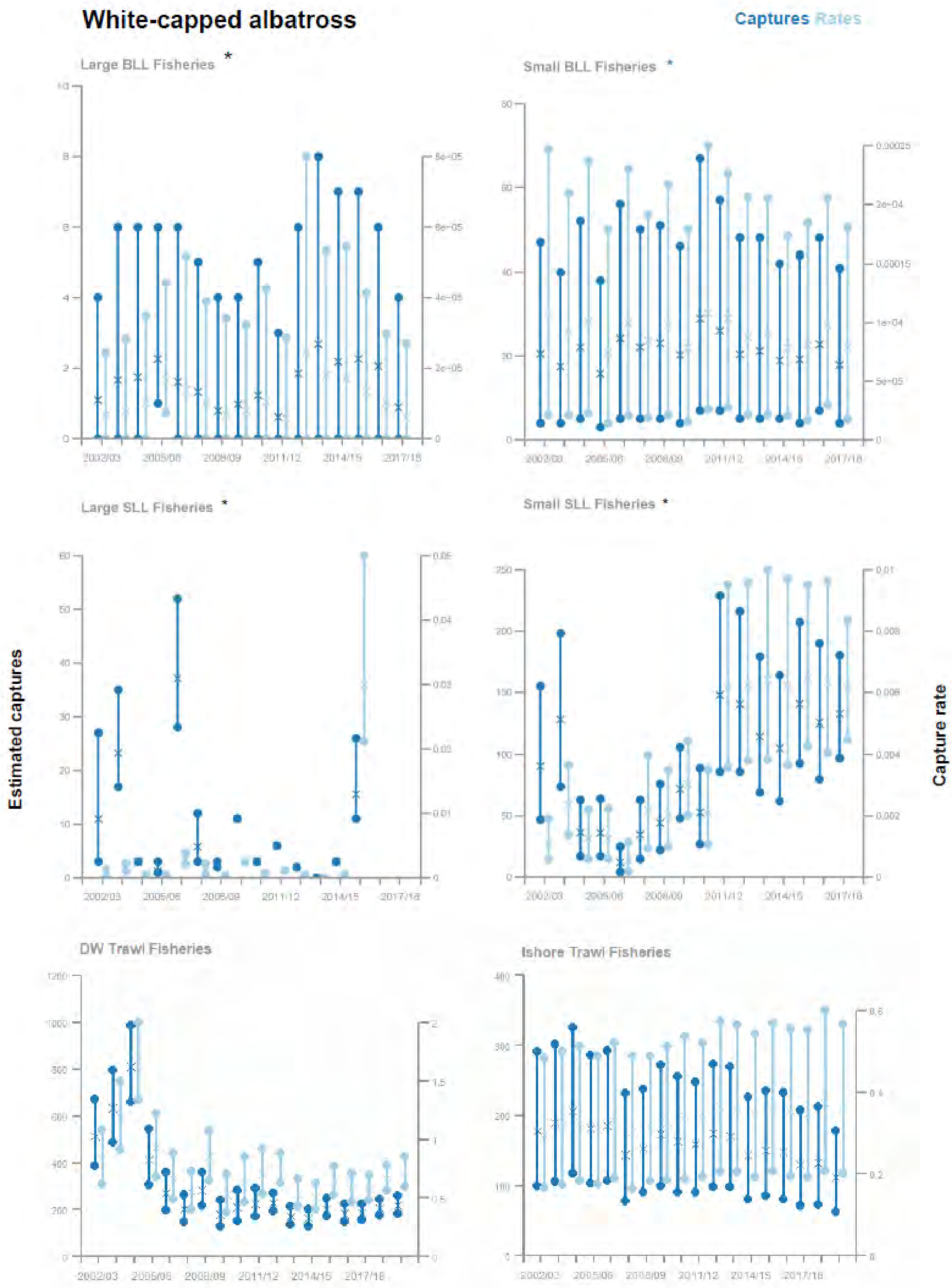


Figure 8.8: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of white-capped albatross in fisheries between 2002–03 and 2019–20. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

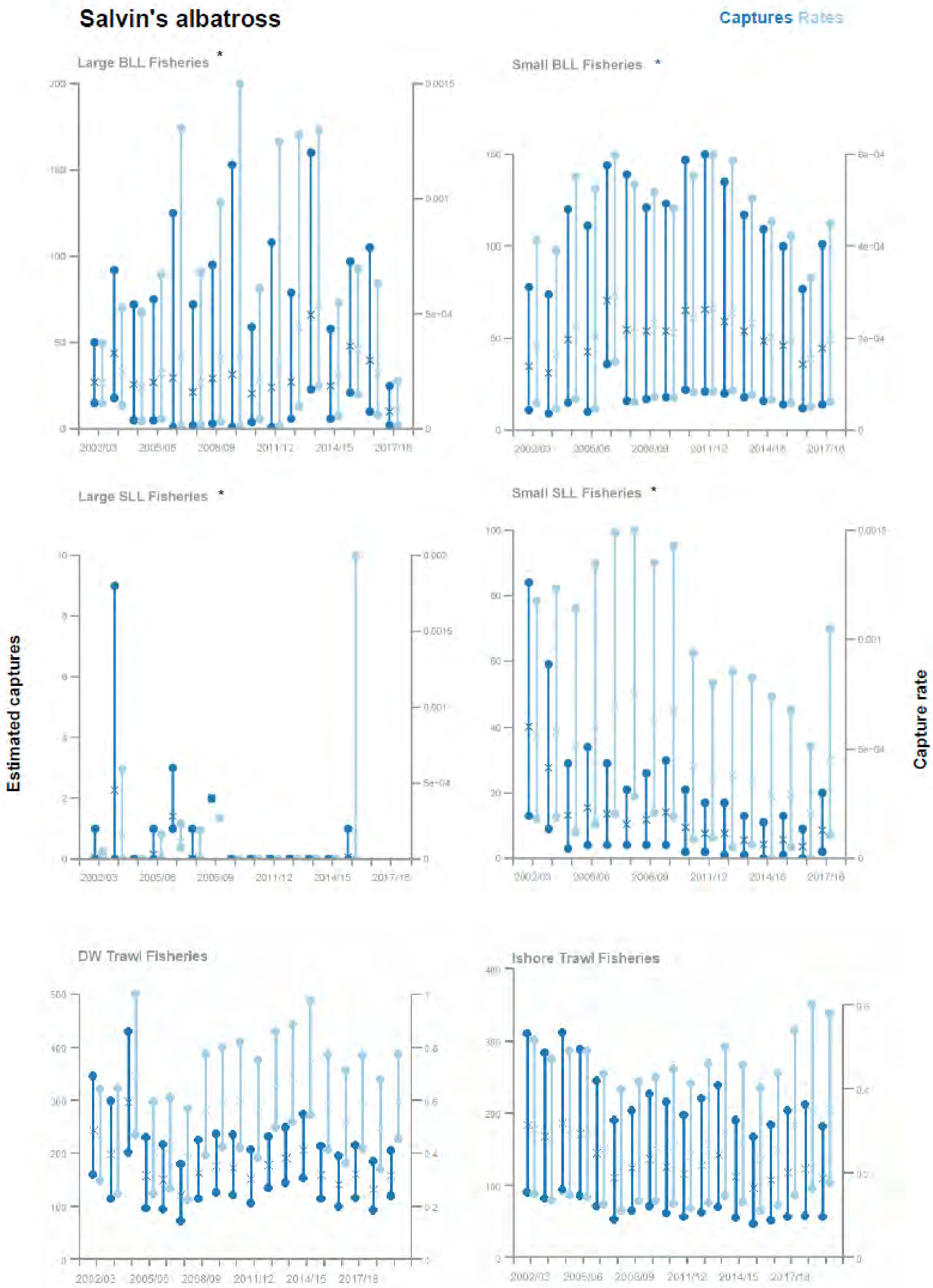


Figure 8.9: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of Salvin's albatross in fisheries between 2002–03 and 2019–20. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

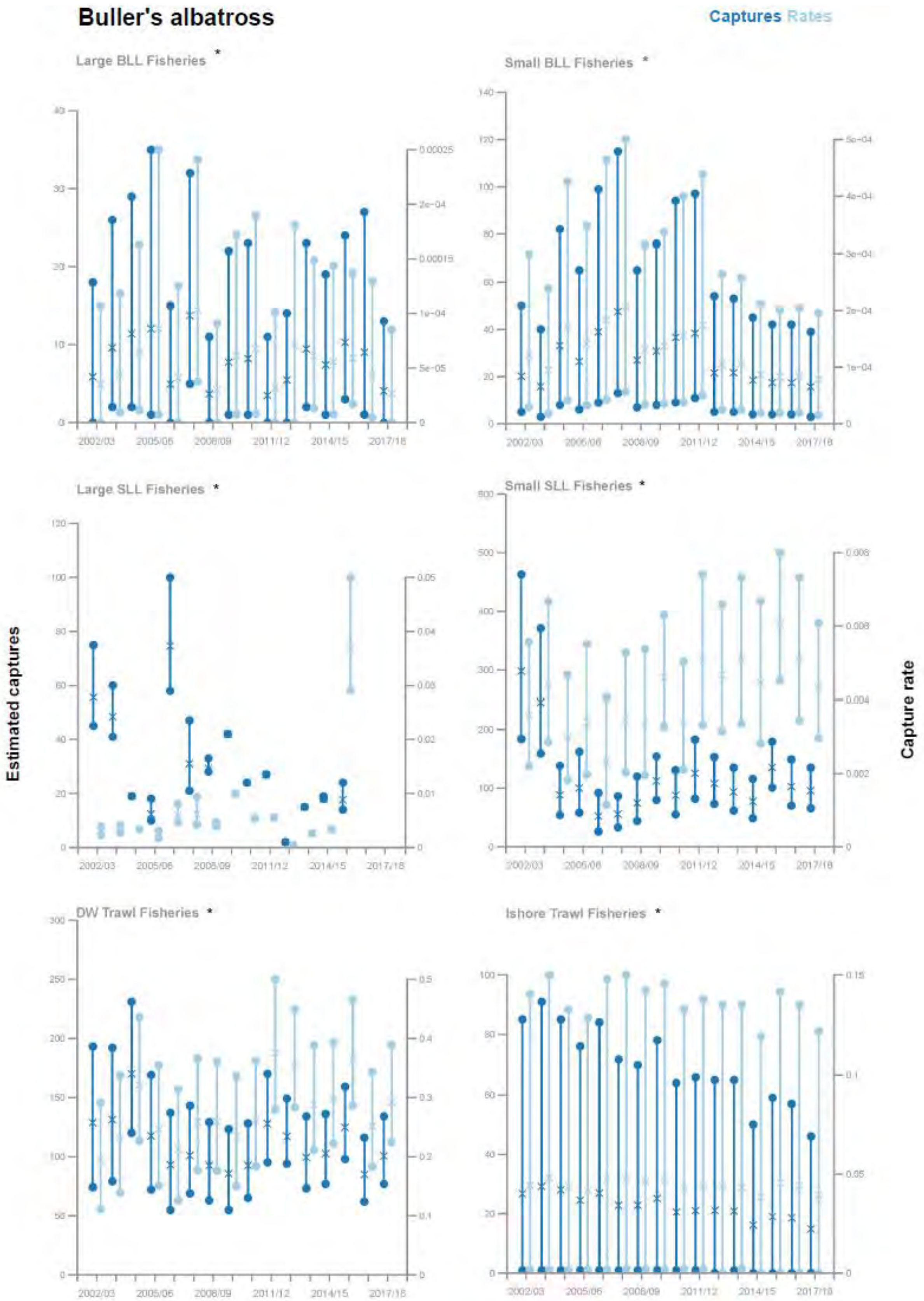


Figure 8.10: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of Buller's albatross in fisheries between 2002–03 and 2019–20. [Protected species bycatch (protectedspeciescaptures.nz)](https://protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

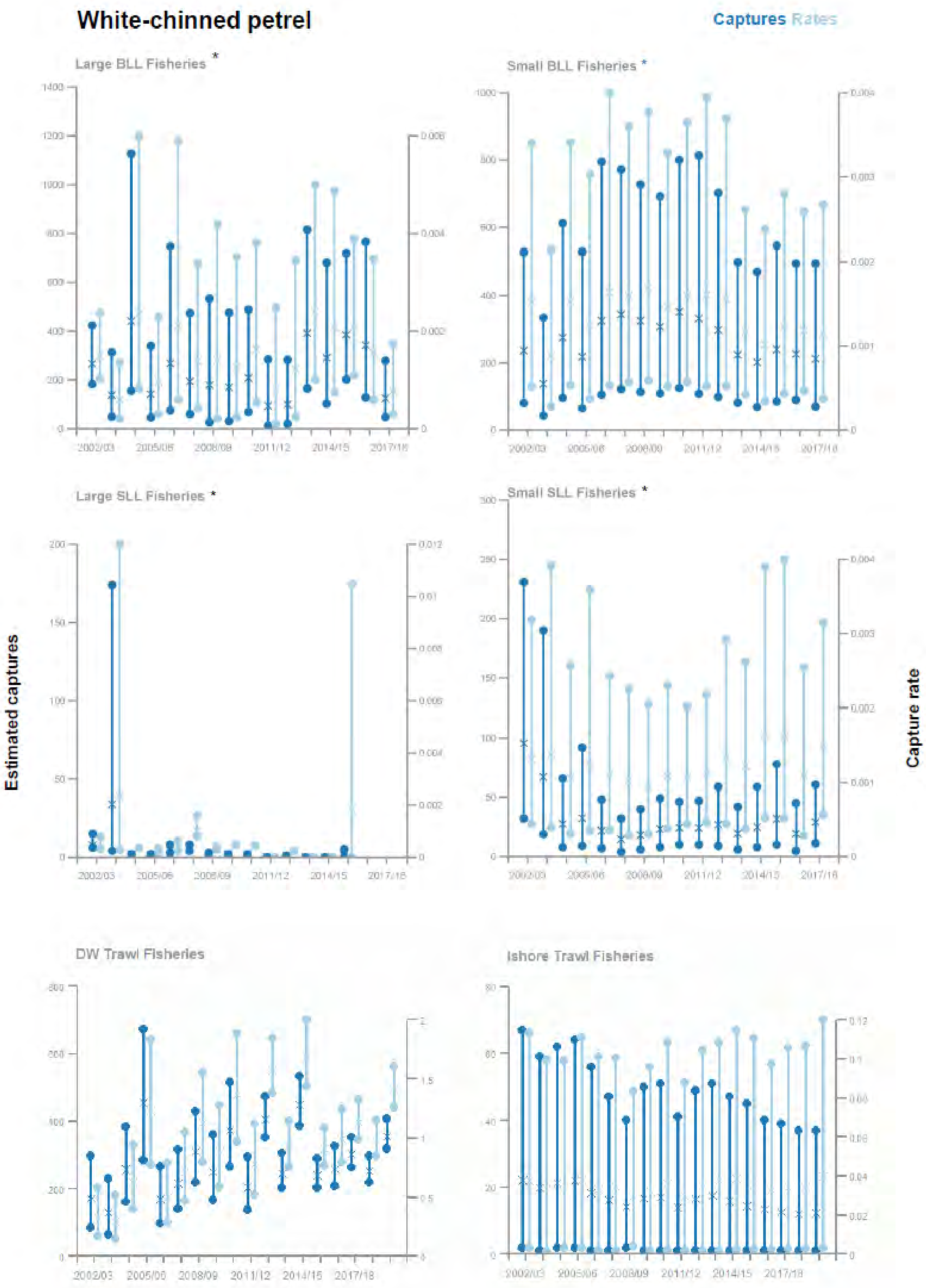


Figure 8.11: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of white-chinned petrels in fisheries between 2002–03 and 2019–20. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

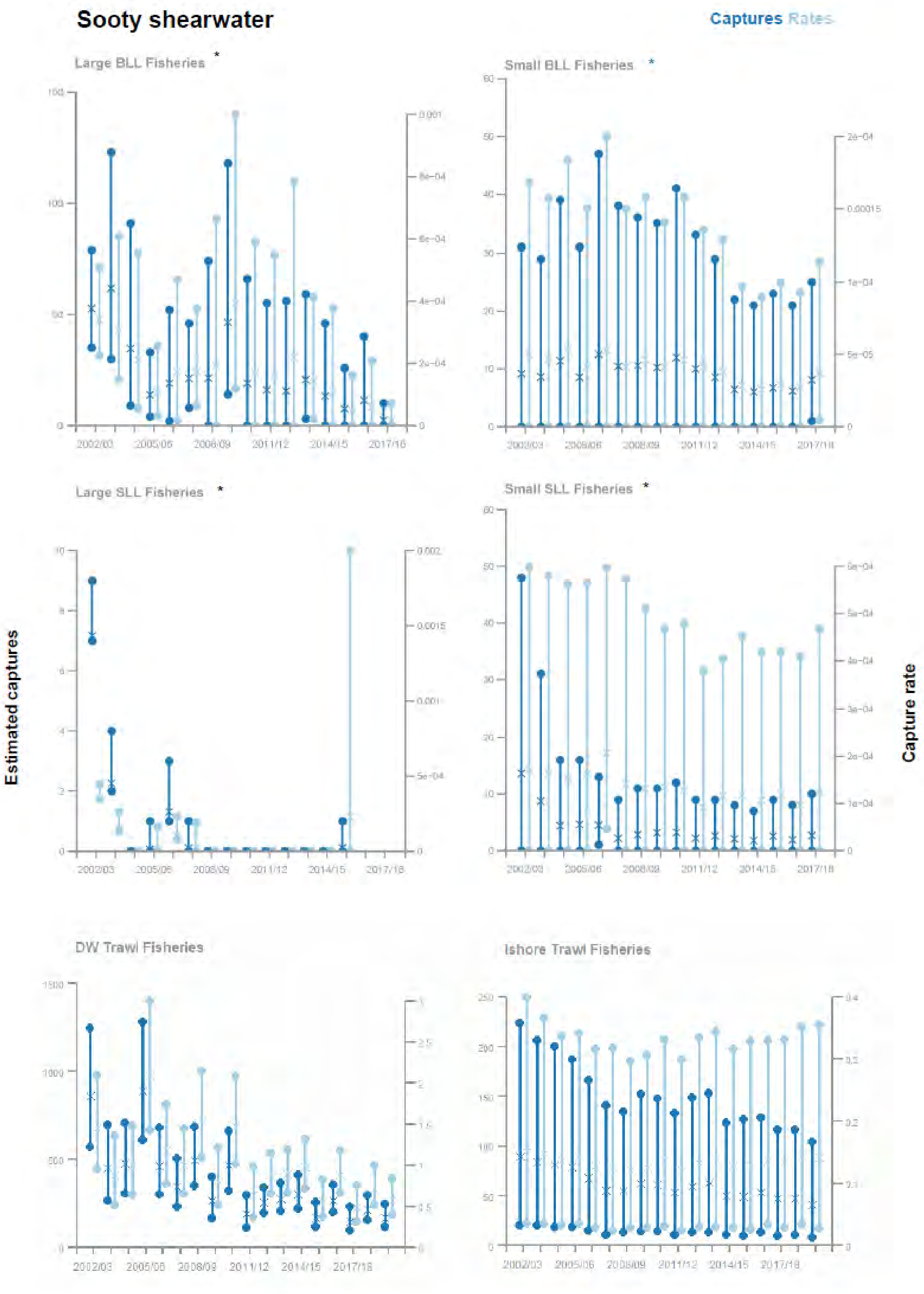


Figure 8.12: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of sooty shearwaters in fisheries between 2002–03 and 2019–20. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

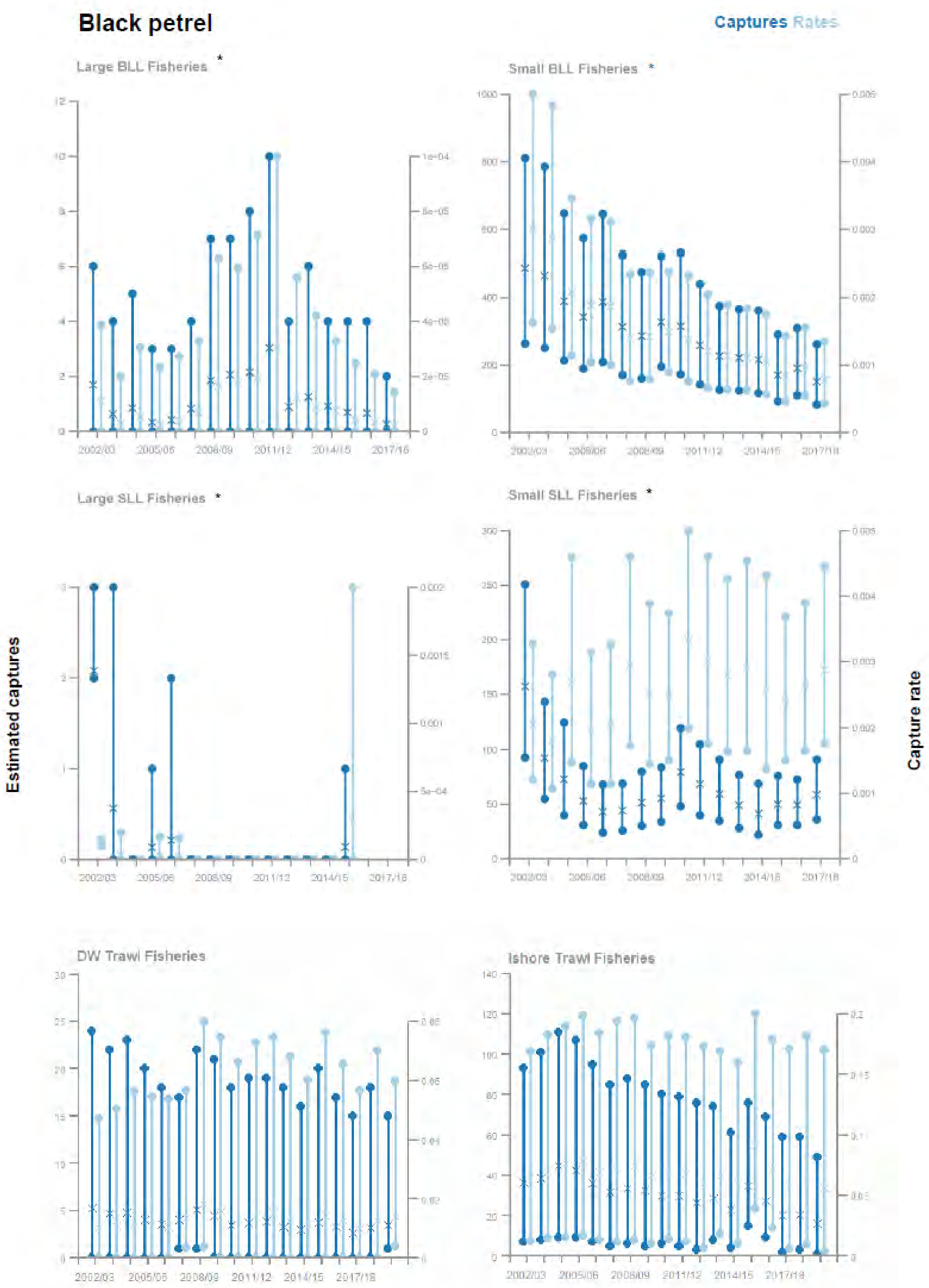


Figure 8.13: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of black petrels in fisheries between 2002–03 and 2019–20. [Protected species bycatch (protectedspeciescaptures.nz)](protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

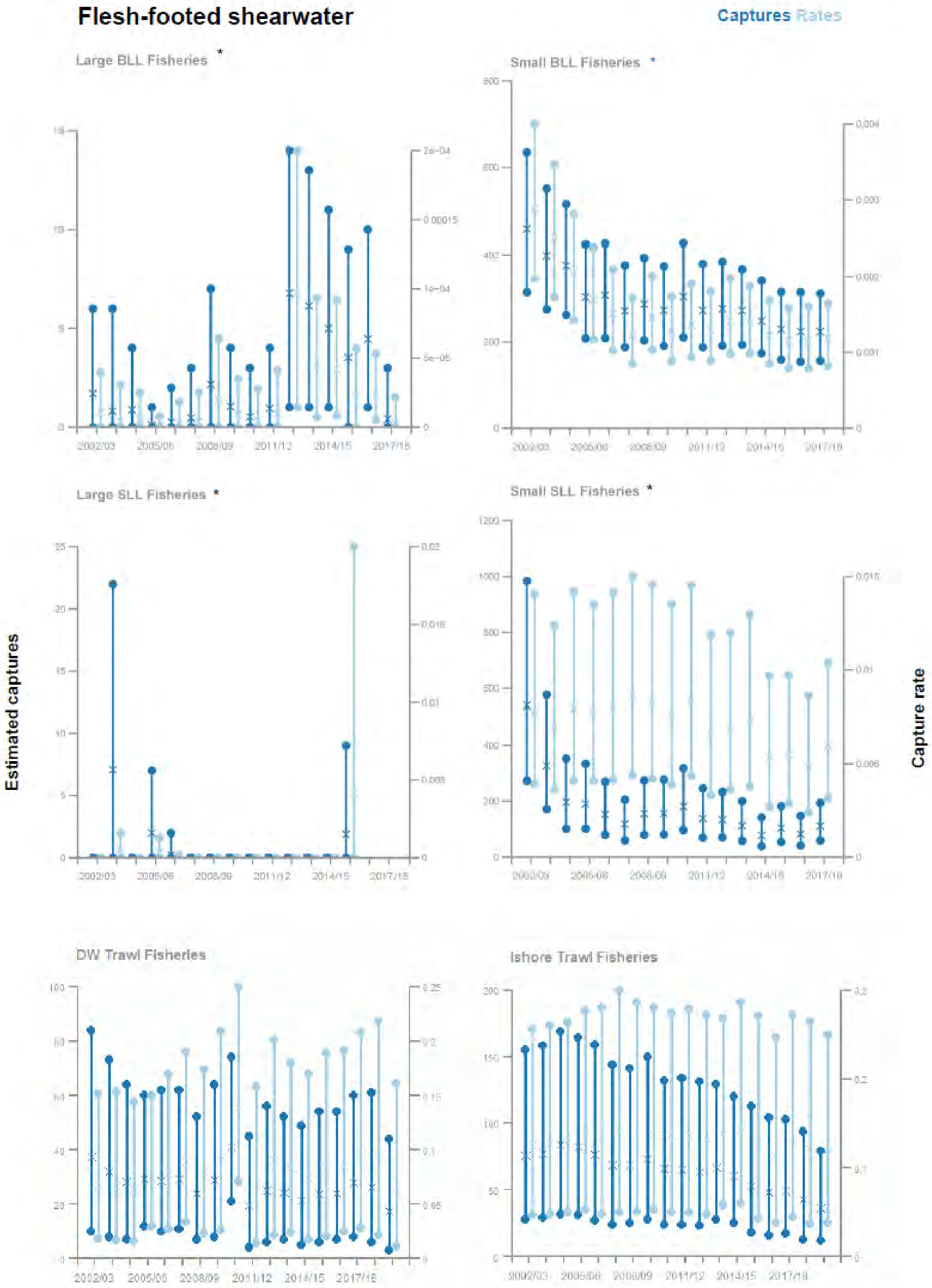


Figure 8.14: Model-based estimates of captures (dark blue) and capture rates (light blue, captures per 100 trawl tows or 1000 longline hooks) of flesh-footed shearwaters in fisheries between 2002–03 and 2019–20. Protected species bycatch (protectedspeciescaptures.nz). * partial update for the 2021–22 edition due to the unavailability of the data at time of publication.

8.4.2 MODELLING FISHERIES INTERACTIONS AND ESTIMATING RISK

8.4.2.1 HIERARCHICAL STRUCTURE OF RISK ASSESSMENTS

Hobday et al. (2007) described a hierarchical framework for ecological risk assessment in fisheries (see Figure 8.15). The hierarchy included three levels: Level 1 qualitative, expert-based assessments (often based on a Scale, Intensity, Consequence Analysis, SICA); Level 2 semi-quantitative analysis (often using some variant of Productivity Susceptibility Analysis, PSA); and Level 3 fully quantitative modelling including uncertainty analysis. The hierarchical structure is designed to 'screen out' potential effects that pose little or low risk for the least investment in data collection and analysis, escalating to risk treatment or higher levels in the hierarchy only for those potential effects that pose non-negligible risk. This structure relies for its effectiveness on a low potential for false negatives at each stage, thereby identifying and screening out activities that are 'low risk' with high certainty. This focuses effort on remaining higher-risk activities. In statistical terms, risk assessment tolerates Type I errors (false positives, i.e., not screening out activities that may actually present a low risk) to avoid Type II errors (false negatives, i.e., incorrectly screening out activities that actually constitute high risk), and it is important to distinguish this approach from normal estimation methods. Whereas normal estimation strives for a lack of bias and a balance of Type I and Type II errors, risk assessment is designed to answer the question 'how bad could it be?' The divergence between the risk assessment approach and normal, unbiased estimation approaches should diminish at higher levels in the risk assessment hierarchy, where the assessment process should be informed by good data that support robust estimation.

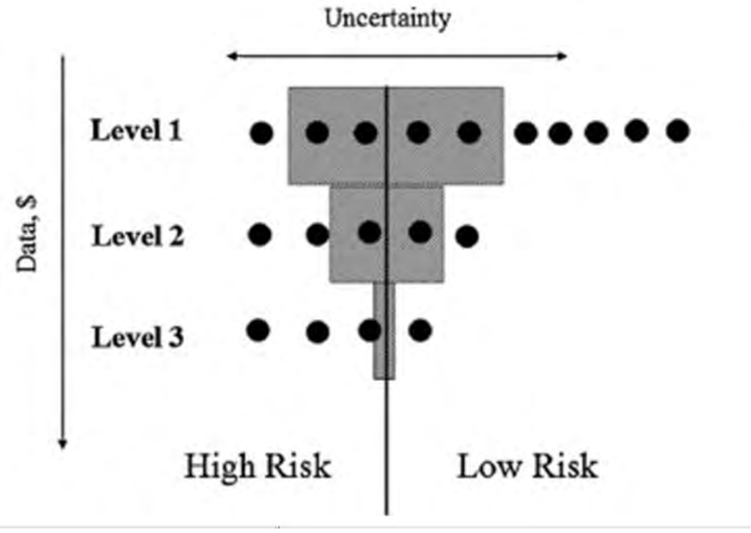


Figure 8.15: (from Hobday et al. 2007). Diagrammatic representation of the hierarchical risk assessment process where activities that present low risk are progressively screened out by assessments of increasingly high data content, sophistication, and cost.

8.4.2.2 QUALITATIVE (LEVEL 1) RISK ASSESSMENT

Rowe (2013) summarised an expert-based, qualitative (Level 1) risk assessment, commissioned by DOC, for the incidental mortality of seabirds caused by New Zealand fisheries. The main focus was on fisheries operating within the New Zealand EEZ and on all seabirds absolutely or partially protected under the Wildlife Act 1953. New Zealand flagged vessels fishing outside the EEZ were included, but risk from non-New Zealand fisheries and other human causes were not included.

The panel of experts who conducted the Level 1 risk assessment assessed the threat to each of 101 taxa posed by 26 fishery groups, scoring exposure and consequence independently (details are given by Rowe 2013). The risk for a given taxon posed by a given fishery was calculated as the product of exposure and consequence scores. Potential risk was estimated as the risk posed by a fishery assuming no mitigation was in place, and residual risk (called 'optimum risk' by Rowe 2013) was estimated assuming that mitigation was in place throughout a given fishery and deployed correctly. The panel also agreed a confidence score for each taxon-fishery interaction.

Total potential and residual risk for a seabird taxon was estimated by summing the scores across all fisheries (Table 8.16 shows taxa with an aggregate score of 30 or higher), and total potential and residual risk posed by a fishery group was estimated by summing the scores across all seabird taxa (Table 8.17 shows the results for all 26 fishery groups).

White-chinned petrel, sooty shearwater, black petrel, Salvin's albatross, white-capped albatross, and flesh-footed shearwater were all estimated by this procedure to have an aggregate risk score of 90 or higher (range 92 to 123) even if mitigation was in place and deployed properly across all fisheries. Of the 101 seabird taxa considered, the aggregate risk score was less than 30 for 70 taxa with respect to potential risk and for 72 taxa with respect to residual risk.

Table 8.16: Potential and residual risk scores for each seabird taxon with a potential risk score of 30 or more given by Rowe (2013). Residual risk ('optimal risk' of Rowe 2013, not tabulated therein for grey-faced petrel or light-mantled sooty albatross) is estimated assuming mitigation is deployed and correctly used throughout all interacting fisheries.

Taxon	Potential score	Residual	Percent reduction
White-chinned petrel	159	123	23
Sooty shearwater	126	108	14
Black petrel	139	106	24
Salvin's albatross	161	106	34
White-capped albatross	141	94	33
Flesh-footed shearwater	117	92	21
Southern Buller's albatross	123	85	31
Grey petrel	123	84	32
Black-browed albatross	114	80	30
Northern Buller's albatross	107	72	33
Chatham albatross	114	71	38
Campbell albatross	97	66	32
Westland petrel	89	59	34
Antipodean albatross	89	55	38
Gibson's albatross	89	55	38
Wandering albatross	89	55	38
Southern royal albatross	79	49	38
King shag	48	48	0
Pitt Island shag	46	46	0
Chatham Island shag	45	45	0
Hutton's shearwater	37	35	5
Northern giant petrel	62	35	44
Pied shag	35	35	0
Indian yellow-nosed albatross	58	34	41
Southern giant petrel	61	34	44
Fluttering shearwater	34	32	6
Spotted shag	31	31	0
Stewart Island shag	31	31	0
Yellow-eyed penguin	30	30	0
Grey-faced petrel	31	–	–
Light-mantled albatross	30	–	–

Set net and inshore trawl fisheries groups posed the greatest residual risk to seabirds (summed across all taxa); both had aggregate scores of over 200 and had no substantive mitigation. Surface and bottom longline fisheries and middle-depth trawl fisheries for finfish and squid also had aggregate risk scores of 100 or more. These risk scores were substantially reduced if mitigation was assumed to be deployed throughout these fisheries (reductions of 24 to 56%), but all remained above 100. Trawling for southern blue whiting and deepwater species, inshore drift net, various seine methods, ring net, diving, dredging, and hand gathering all had aggregate risk scores of 40 or less if mitigation was assumed to be deployed throughout these fisheries. Diving, dredging, and hand gathering were all judged by the panel to pose essentially no risk to seabirds.

Table 8.17: Cumulative potential risk and residual risk scores for all seabird taxa for each fishery from Rowe (2013). Residual risk ('optimal risk' of Rowe 2013) is estimated assuming mitigation is deployed and correctly used throughout a given fishery.

Fishery group	No. taxa	Potential risk	Residual risk	Percent reduction
Set net	42	374	374	0
Inshore trawl	44	225	225	0
Surface longline: charter	25	313	191	39
Surface longline: domestic	25	302	184	39
Bottom longline: small	33	354	154	56
Bottom longline: large	32	311	139	55
Mid-depth trawl: finfish	22	160	122	24
Mid-depth trawl: squid	21	156	118	24
Mid-depth trawl: scampi	23	94	94	0
Hand line	27	68	68	0
Squid jig	44	62	62	0
Dahn line	29	61	61	0
Pots, traps	17	61	61	0
Trot line	29	61	61	0
Pelagic trawl	27	63	51	19
Troll	23	50	50	0
Mid-depth trawl: southern blue whiting	21	53	40	25
Deepwater trawl	21	46	35	24
Inshore drift net	12	33	33	0
Danish seine	15	32	32	0
Beach seine	16	29	29	0
Purse seine	11	22	22	0
Ring net	12	13	13	0
Diving	0	0	0	–
Dredge	0	0	0	–
Hand gathering	0	0	0	–

8.4.2.3 SEABIRD SPATIALLY EXPLICIT FISHERIES RISK ASSESSMENT

The Spatially Explicit Fisheries Risk Assessment (SEFRA) approach used by Fisheries New Zealand was developed first for measuring the risk to multiple seabird species, starting in 2009. See Chapter 3 for more details.

The SEFRA method developed by the then Ministry for Primary Industries is a generalisation of the spatial overlap approach described by Kirby & Hobday (2007) and applies the 'exposure-effects' approach, where exposure refers to the number of fatalities arising from an activity, and effect refers to the consequence of that exposure for the population. The SEFRA approach arose initially from an expert workshop hosted by the then Ministry of Fisheries in 2008 and attended by experts with specialist knowledge of New Zealand fisheries, seabird-fishery interactions, seabird biology, population modelling, and ecological risk assessment. The overall framework is described by Sharp et al. (2011) (Figure 8.16) and has been variously applied and improved in multiple iterations for seabirds (Waugh et al. 2008a, 2008b, developed further by Sharp 2009, Waugh & Filippi 2009, Filippi et al. 2010, Richard et al. 2011, Richard & Abraham 2013b, Richard & Abraham 2015, Richard et al. 2017). The latest iteration of the risk assessment was reported by Richard et al. (2020) (Figure 8.17), how this update has affected the risk scores is shown in Figure 8.18.

Previous versions of this chapter, together with the references cited above, contain considerable detail about the evolution and refinement of the risk assessment approach and the reader is directed to those sources for a comprehensive record of the risk assessment framework development, and its outputs, over time. Here, the most recent outputs and summary details are provided, as reported by Richard et al. (2020).

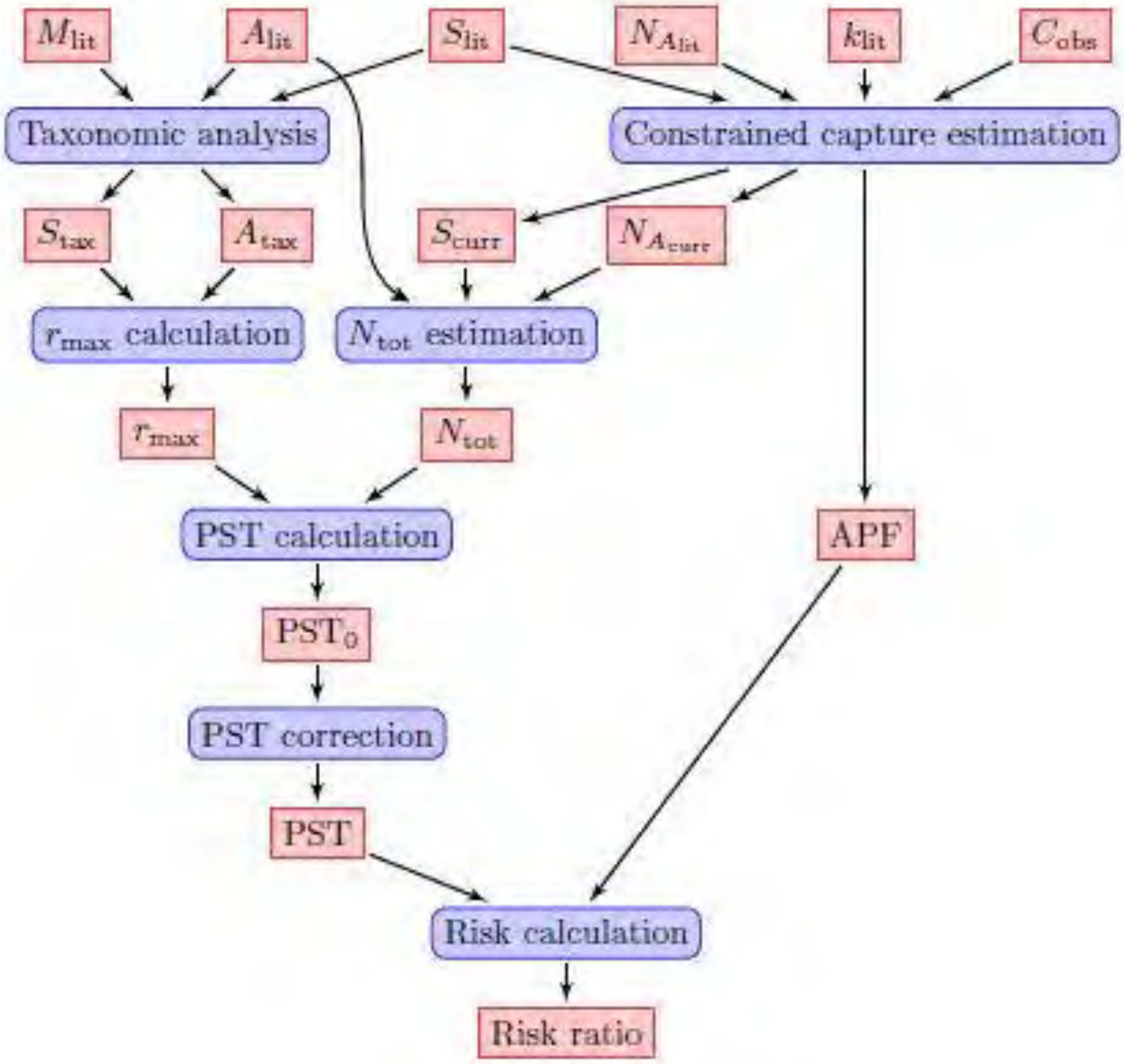


Figure 8.16: Schematic process of the estimation of risk in the current seabird risk assessment. M: body mass; A: age at first reproduction; S: adult survival rate; NA: adult population size; k: cryptic mortality multiplier; C: seabird captures; rmax: maximum net productivity rate; Ntot: total population size; PST: Population Sustainability Threshold; APF: annual potential fatalities. For the indices: lit: from the literature or expert-based; obs: recorded by observers; tax: from the taxonomic analysis; curr: representing current conditions, corrected by the model; tot: total; 0: prior to correction.

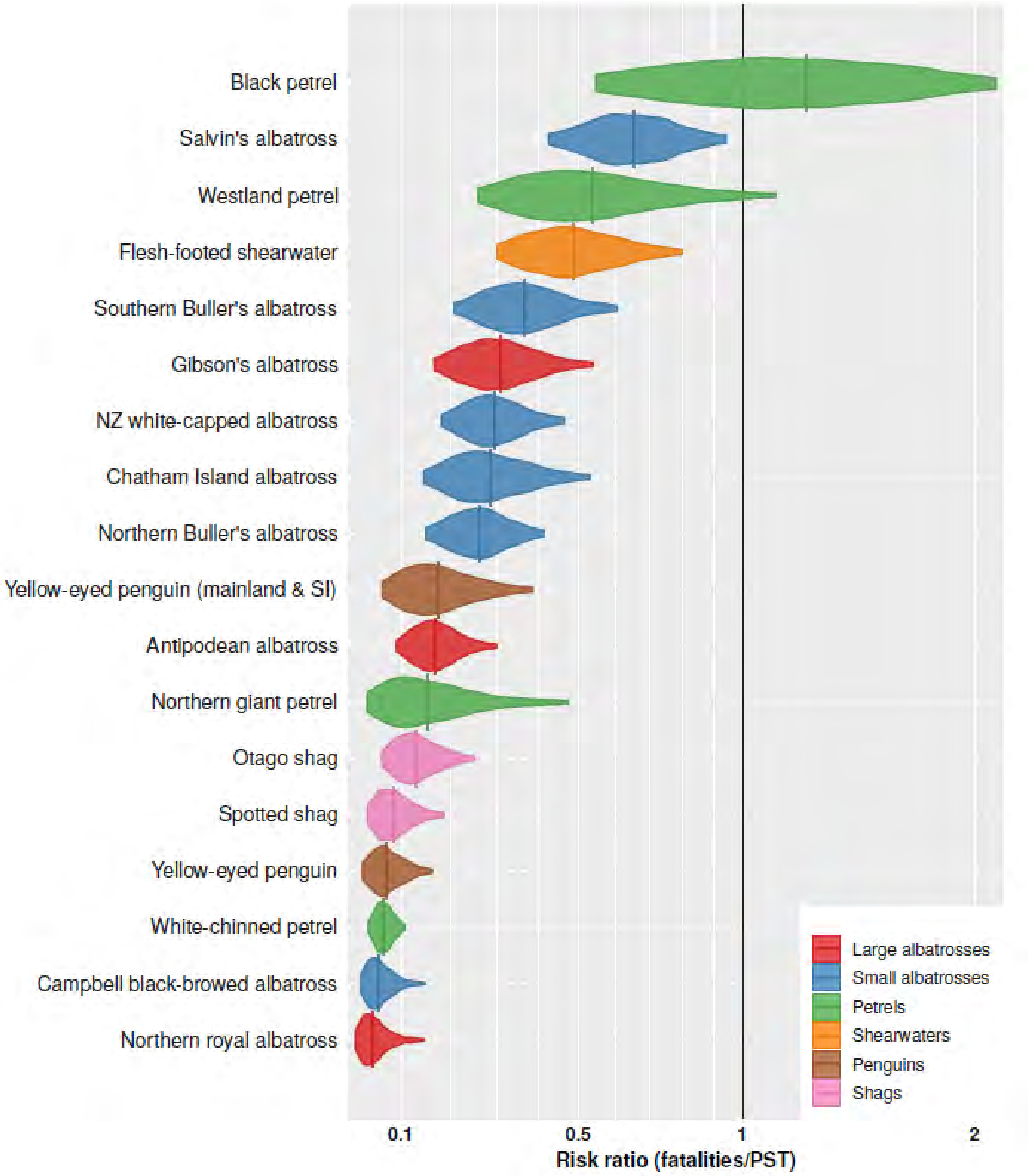


Figure 8.17: Risk ratio for different seabird taxa, based on data between 2006–07 and 2016–17. The risk ratio is displayed on a logarithmic scale, with the threshold of the number of potential bird fatalities equalling the Population Sustainability Threshold (PST) represented by the black vertical line, and the distribution of the risk ratios within their 95% credible interval, including the median risk ratio (vertical line). Seabird taxa are listed in decreasing order of the median risk ratio. Taxa with a risk ratio of almost zero were not included (95% upper limit less than 0.05). The risk ratio of yellow-eyed penguin refers to the mainland population only, based on the assumption that all estimated fatalities were of the mainland population, and the number of annual breeding pairs in 2016/17 was between 273 and 374 (YEP Trust pers. comm.).

Table 8.18: Population Sustainability Threshold (PST), total annual potential fatalities (D) in trawl, longline, and set net fisheries, risk ratio with *f* = 1 (RR = APF/PST), and the probability that APF > PST for seabird taxa in the current risk assessment. Taxa are ordered in decreasing order of the median risk ratio. The risk to yellow-eyed penguin was assessed for the entire New Zealand population, but also for the mainland population only, based on the assumption that all estimated fatalities were of the mainland population and the number of annual breeding pairs was between 600 and 800. Taxa names are coloured according to their risk category. Red: risk ratio with a median over 1 or upper 95% credible limit (u.c.l.) over 2; dark orange: median over 0.3 or u.c.l. over 1; light orange: median over 0.1 or u.c.l. over 0.3; yellow: u.c.l. over 0.1. PST and APF values were rounded to three significant digits.

	PST		D		Risk ratio		P(D > PST)
	Mean	95% c.i.	Mean	95% c.i.	Median	95% c.i.	
Black petrel	447	225–831	513	325–803	1.23	0.55–2.11	0.70
Salvin's albatross	3 460	2 630–4 730	2 250	1 640–3 060	0.65	0.42–0.94	0.01
Westland petrel	351	233–532	194	103–361	0.54	0.26–1.12	0.05
Flesh-footed shearwater	1 450	1 010–2 050	710	496–1 020	0.49	0.30–0.80	0.00
Southern Buller's albatross	1 360	896–2 160	486	358–664	0.37	0.21–0.60	0.00
Gibson's albatross	497	327–743	151	95–221	0.31	0.17–0.54	0.00
NZ white-capped albatross	10 800	7 680–15 700	3 160	2 290–4 330	0.29	0.18–0.46	0.00
Chatham Island albatross	428	292–632	123	69–196	0.28	0.14–0.53	0.00
Northern Buller's albatross	1 640	1 070–2 630	414	321–524	0.26	0.15–0.41	0.00
Yellow-eyed penguin (mainland)	120	79–180	21	8–41	0.17	0.06–0.38	0.00
Antipodean albatross	369	258–517	63	37–97	0.17	0.09–0.30	0.00
Northern giant petrel	337	159–792	51	16–113	0.15	0.04–0.47	0.00
Otago shag	283	184–418	37	20–58	0.13	0.06–0.25	0.00
Spotted shag	3 730	1 790–7 080	304	198–439	0.09	0.04–0.19	0.00
Yellow-eyed penguin	285	189–424	21	8–41	0.07	0.03–0.16	0.00
White-chinned petrel	25 800	16 100–41 300	1 680	1 390–2 010	0.07	0.04–0.11	0.00
Campbell black-browed albatross	2 000	993–3 570	117	65–223	0.06	0.03–0.15	0.00
Northern royal albatross	723	345–1 360	36	14–81	0.05	0.02–0.15	0.00
Foveaux shag	208	132–317	7	2–14	0.03	0.01–0.08	0.00
Grey petrel	5 460	3 190–9 130	139	86–217	0.03	0.01–0.05	0.00
Southern royal albatross	854	600–1 190	22	9–42	0.02	0.01–0.05	0.00
Snares Cape petrel	1 570	605–3 670	24	5–68	0.01	0.00–0.06	0.00
Fluttering shearwater	35 900	15 300–73 700	393	197–665	0.01	0.00–0.03	0.00
Northern little penguin	1 500	905–2 310	13	4–26	0.01	0.00–0.02	0.00
White-flippered little penguin	467	270–742	4	0–9	0.01	0.00–0.02	0.00
Little black shag	338	155–644	3	0–9	0.01	0.00–0.03	0.00
Pied shag	1 120	707–1 680	8	0–25	0.01	0.00–0.02	0.00
Grey-headed albatross	695	335–1 270	5	0–21	0.01	0.00–0.04	0.00
Fiordland crested penguin	626	283–1 180	4	0–15	0.00	0.00–0.03	0.00
Southern little penguin	1 500	910–2 380	7	1–15	0.00	0.00–0.01	0.00
Common diving petrel	137 000	46 900–309 000	383	63–1 430	0.00	0.00–0.01	0.00
Grey-faced petrel	30 000	19 200–50 200	62	27–117	0.00	0.00–0.00	0.00
Sooty shearwater	622 000	296 000–1 180 000	1 210	681–2 220	0.00	0.00–0.01	0.00
Light-mantled sooty albatross	873	668–1 140	2	0–13	0.00	0.00–0.01	0.00
Hutton's shearwater	14 900	9 160–23 300	17	3–68	0.00	0.00–0.00	0.00
Chatham Island little penguin	1 500	926–2 390	1	0–8	0.00	0.00–0.01	0.00
Buller's shearwater	56 200	34 300–102 000	17	6–35	0.00	0.00–0.00	0.00
Little shearwater	21 600	13 800–32 900	6	1–12	0.00	0.00–0.00	0.00
White-headed petrel	34 400	16 300–67 600	9	2–19	0.00	0.00–0.00	0.00
NZ white-faced storm petrel	331 000	139 000–683 000	85	17–239	0.00	0.00–0.00	0.00
Australasian gannet	9 400	4 120–18 500	3	0–12	0.00	0.00–0.00	0.00
Southern black-backed gull	333 000	138 000–689 000	54	18–117	0.00	0.00–0.00	0.00
Fairy prion	326 000	209 000–493 000	89	10–462	0.00	0.00–0.00	0.00
Snares crested penguin	6 860	4 800–9 660	1	0–5	0.00	0.00–0.00	0.00
Broad-billed prion	68 400	45 400–104 000	9	1–29	0.00	0.00–0.00	0.00
Black-bellied storm petrel	15 400	8 650–25 900	2	0–9	0.00	0.00–0.00	0.00
Cook's petrel	48 900	27 400–87 100	7	0–36	0.00	0.00–0.00	0.00
Antarctic prion	154 000	77 000–284 000	10	2–26	0.00	0.00–0.00	0.00
Mottled petrel	47 200	30 400–77 100	4	0–21	0.00	0.00–0.00	0.00
Auckland Island shag	485	198–988	0	0–1	0.00	0.00–0.00	0.00
Bounty Island shag	26	15–43	0	0–0	0.00	0.00–0.00	0.00
Subantarctic skua	67	44–103	0	0–0	0.00	0.00–0.00	0.00
Caspian tern	172	95–294	0	0–0	0.00	0.00–0.00	0.00
Chatham Island shag	76	47–116	0	0–3	0.00	0.00–0.05	0.00
Campbell Island shag	476	222–906	0	0–0	0.00	0.00–0.00	0.00
Eastern rockhopper penguin	11 100	6 800–17 300	1	0–3	0.00	0.00–0.00	0.00
Erect-crested penguin	17 800	12 600–24 700	1	0–4	0.00	0.00–0.00	0.00
White-bellied storm petrel	228	106–441	0	0–0	0.00	0.00–0.00	0.00
White tern	26	15–43	0	0–0	0.00	0.00–0.00	0.00
South Georgian diving petrel	10	5–18	0	0–1	0.00	0.00–0.07	0.00
NZ king shag	39	24–60	0	0–2	0.00	0.00–0.06	0.00
Kerm. storm petrel	12	4–26	0	0–0	0.00	0.00–0.00	0.00
Masked booby	53	28–94	0	0–0	0.00	0.00–0.00	0.00
NZ storm petrel	53	6–207	0	0–1	0.00	0.00–0.02	0.00
Pitt Island shag	103	63–161	0	0–2	0.00	0.00–0.02	0.00
Chatham petrel	42	23–76	0	0–0	0.00	0.00–0.00	0.00
Chatham Island taiko	2	1–4	0	0–0	0.00	0.00–0.00	0.00
Pycroft's petrel	412	247–718	0	0–1	0.00	0.00–0.00	0.00
Soft-plumaged petrel	497	136–1 290	0	0–0	0.00	0.00–0.00	0.00
Wedge-tailed shearwater	6 020	3 040–10 600	0	0–0	0.00	0.00–0.00	0.00
Kerm. petrel	779	500–1 300	0	0–1	0.00	0.00–0.00	0.00
White-naped petrel	7 080	3 340–14 200	0	0–0	0.00	0.00–0.00	0.00

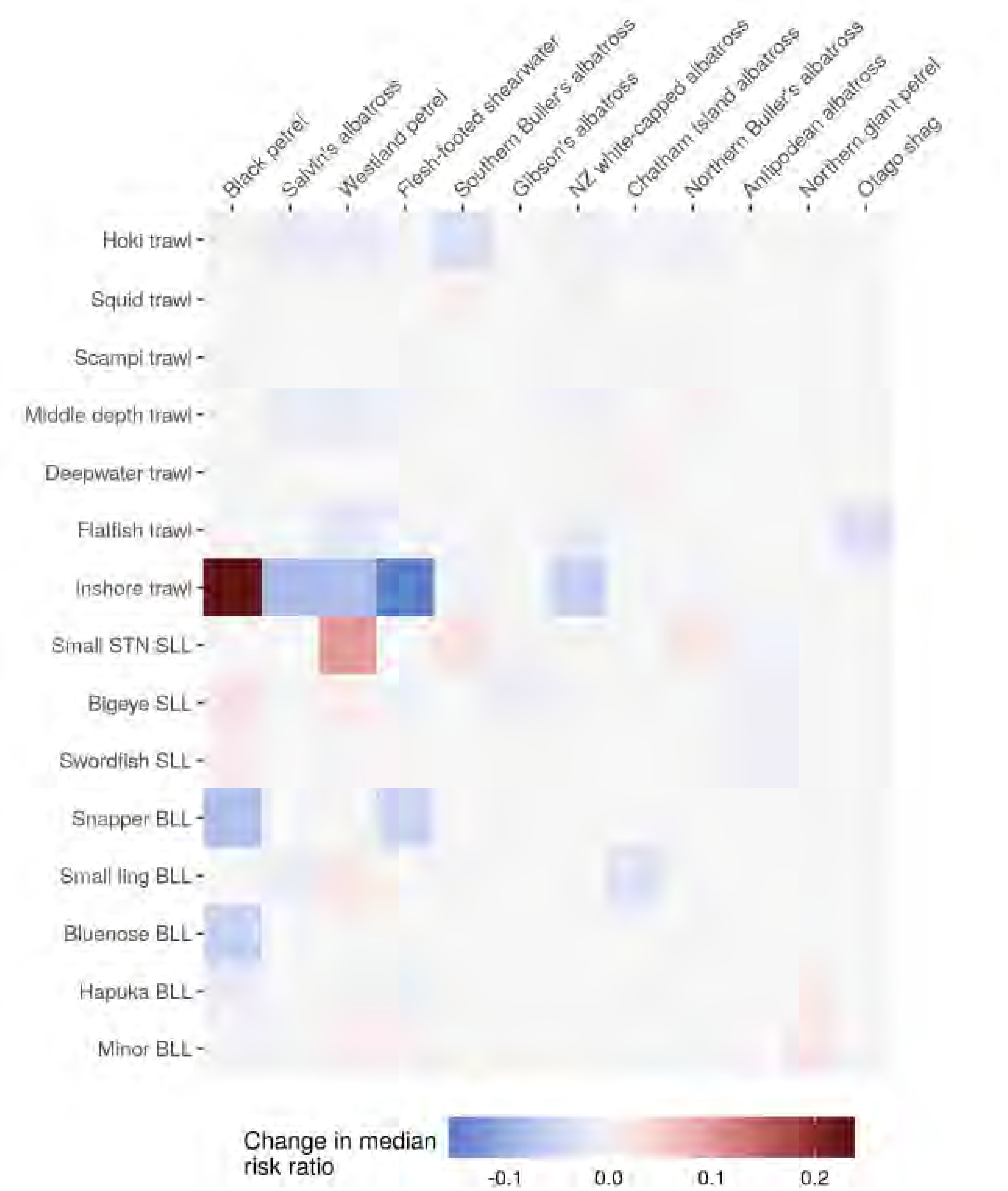


Figure 8.18: Change in median species-level risk estimates for seabird taxa, disaggregated by target fisheries. Highlighted cells (increasing red) identify fisheries that are responsible for an increasing proportion of species-level risk. Species are ordered by total risk ratio in descending order with only Very High, High, and Medium shown. Fisheries are grouped by type and ordered from highest effort to lowest effort. Target fisheries with zero risk to all species (rounded to two decimal places) are not shown; these include: albacore SLL, minor SLL, jack mackerel trawl, and grey mullet set net. STN is southern bluefin tuna.

8.4.2.4 SPECIES-SPECIFIC, FULLY QUANTITATIVE MODELLING

Fully quantitative population modelling has been conducted only for Chatham Island albatross, southern Buller’s albatross, black petrel, white-capped albatross, Gibson’s albatross, and Antipodean albatross. Data for other species or populations appear unlikely to be adequate for comprehensive population modelling. The poor estimates of observable and cryptic fishing-related mortality have restricted such work to comprehensive population modelling rather than formal assessment of risk. Summaries of the fully quantitative modelling outlined here are included in section 8.4.3 that focuses on the 13 most at-risk species (see Table 8.18: species with a risk ratio of 0.1 or greater from the most recent iteration of the seabird risk assessment, Richard & Abraham 2020).

8.4.2.5 OTHER QUANTITATIVE MODELS

This section is not intended to cover all quantitative modelling of seabird populations, rather to focus on recent studies that sought to assess the impact of fishing-related mortality.

Fletcher et al. (2008) sought to assess the potential impact of fisheries on Antipodean and Gibson’s wandering albatrosses, black petrel, and southern royal albatross *Diomedea epomophora*. Because of problems with the available fisheries and biological data, they were unable to use their models to predict the impact of a change in fishing effort on the population growth rate of a given species. Instead, they used the models to estimate the impact that changes in demographic parameters like annual survival are likely to have on population growth rate. They found that: reducing breeder survival rate by k percentage points will lead to a reduction in the population growth rate of about 0.3 k percentage points (0.4 for black petrel); and a reduction of k percentage points in the survival rate for each stage in the lifecycle (juvenile, pre-breeder, non-breeder, and breeder) will lead to a reduction in the population growth rate of approximately k percentage points. Fletcher et al. (2008) also made estimates of Potential Biological Removal for 23 New Zealand seabird taxa and summarised and tabulated non-fishing-related threats for 38 taxa.

Newman et al. (2009) combined survey data with demographic population models to estimate the total population of sooty shearwaters within New Zealand. They estimated the total New Zealand population between 1994 and 2005 to have been 21.3 (95% c.i.: 19.0–23.6) million birds. The harvest of ‘muttonbirds’ was estimated to be 360 000 (320 000–400 000) birds per year, equivalent to 18% of the chicks produced in the harvested areas and 13% of chicks in the New Zealand region. This directed harvest is much larger than estimates of captures in key fisheries or potential fatalities in the Level 2 risk assessment. Newman et al. (2009) did not assess the likely impact of fishing-related mortality and did not consider the different population-level impacts of adult mortality in fisheries and chick mortality in the directed harvest, but concluded that the much larger directed harvest was not an adequate explanation for the observed declines in the past three decades.

8.4.2.6 GENERAL CONCLUSIONS FROM QUANTITATIVE MODELLING

Fully quantitative modelling has now been conducted for five seabird populations for which apparently suitable data are available (see individual species sections below). This modelling suggests very strongly that one population had been increasing steadily (southern Buller’s albatross, but note that this trend may have since reversed), whereas a further population is declining quite rapidly (Antipodean albatross). White-capped albatross and black petrel were both assessed at the time of the modelling to be more likely to be declining than not but, even for these relatively data-rich populations, the conclusions were uncertain. Higher counts have been recorded for both species since the modelling was conducted. General conclusions from the modelling conducted to date, therefore, are summarised below.

- Very few seabird populations have sufficient data for fully quantitative modelling.
- Except for the most complete datasets (southern Buller’s albatross, Gibson’s albatross, and Antipodean albatross) it has been difficult to draw firm conclusions about trends in population size from model outputs.
- Information from surveys or census counts is much more powerful for detecting trends in

population size than data from the tagging programmes and plot monitoring implemented for New Zealand seabirds to date.

- The available information on incidental captures in fisheries have not allowed rigorous tests of the role of fishing-related mortality in driving population trends.
- Although comprehensive modelling provides additional information to allow interpretation, we will have to rely on Level 2 risk assessment approaches for much of our understanding of the relative risks faced by different seabird taxa and posed by different fisheries.

8.4.2.7 SOURCES OF UNCERTAINTY IN RISK ASSESSMENTS

There are several outstanding sources of uncertainty in modelling the effects of fisheries interactions on seabirds, especially for the complete assessment of risk to individual seabird populations.

8.4.2.7.1 SCARCITY OF INFORMATION ON CAPTURES AND BIOLOGICAL CHARACTERISTICS OF AFFECTED POPULATIONS

These sources of uncertainty can be explored within the analytical framework of the Level 2 risk assessment (Richard et al. 2011, Richard & Abraham 2013b, 2015), noting that the results of that exploration are constrained by the structure of that analysis. Richard et al. (2017) provided plots of such an exploration for nine taxa (Figure 8.19). It can be concluded from this analysis that better estimates of average adult survival would lead to substantially more precise estimates of risk for a wide variety of taxa, including most of the species estimated to be at most risk. More precise estimates of risk would be available for black petrel, Salvin's albatross, New Zealand white-capped albatross, Chatham Island albatross, and Antipodean albatross if better estimates of potential fatalities were available, and better estimates of survival would be useful for all nine taxa. This analysis was not applied at this iteration of the risk assessment to the spatial distribution of seabirds and fisheries, although it is acknowledged that this is extremely important for the proper implementation of any spatial overlap method. Noting this limitation, this type of sensitivity analysis is a powerful way of assessing the priorities for collection of new information, including research.

8.4.2.7.2 SCARCITY OF INFORMATION ON CRYPTIC MORTALITY

Cryptic mortality takes into account unobservable captures and is particularly poorly understood but has substantial influence on the results of the risk assessment. Richard et al. (2011) provided a description of the method used to incorporate cryptic mortality into their estimates of potential fatalities in the Level 2 risk assessment (their appendix B authored by B. Sharp, MPI). This method builds on the published information from Brothers et al. (2010) for longline fisheries and Watkins et al. (2008) and Abraham (2010a) for trawl fisheries. Brothers et al. (2010) observed almost 6000 seabirds attempting to take longline baits during line setting, of which 176 (3% of attempts) were seen to be caught. Of these, only 85 (48%) were retrieved during line hauling. They concluded that using only observed captures to estimate seabird fatalities grossly underestimates actual levels in pelagic longline fishing. Similarly, Watkins et al. (2008) observed 2454 interactions between seabirds and trawl warps in the South African hake fishery over 189.8 hours of observation. About 11% of those interactions (263) involved birds, mostly albatrosses, being dragged under the water by the warps, and 30 of those submersions were observed to be fatal. Of the 30 birds observed killed on the warps, only two (both albatrosses) were hauled aboard and would have been counted as captures by an observer in New Zealand. Aerial collisions with the warps were about eight times more common but appeared mostly to have little effect (although one white-chinned petrel suffered a broken wing, which would almost certainly have had fatal consequences). Parker (2013) presented some preliminary data on cryptic mortality associated with a fishing vessel operating around the Falkland Islands in the south Atlantic Ocean. Of a total of 2250 contacts between seabirds (almost all black-browed albatross *Thalassarche melanophris* and northern giant petrel *Macronectes halli*) and warp cables or bird scaring lines, 371 (17%) were considered heavy contacts, and of these 26 (7%) were recorded as being of unknown outcome. Overall, Parker (2013) estimated that at least 23% of total mortalities (including severe injuries that were deemed to ultimately be fatal) recorded were not observed from the fishing vessel.

Given the relatively small sample sizes in both these trials, there is substantial (estimable) uncertainty in the estimates from the trials themselves and additional (non-estimable) uncertainty related to the extent to which these trials are representative of all fishing of a given type, particularly as both trials were undertaken overseas. The binomial 95% confidence range (calculated using the Clopper-Pearson 'exact' method) for the ratio of total fatalities to observed captures in the Brothers et al. (2010) longline trial is 1.8–2.5 (mean 2.1), and that for the Watkins et al. (2008) trawl warp trial is 5–122 (mean 15.0 fatalities per observed capture). Abraham (2010a) estimated that there were 244 (95% c.i.: 190–330) warp strikes by large birds for each one observed captured, and 6440 (3400–20 000) warp strikes by small birds for each one observed captured (although small birds tend to be caught in the net rather than by warps). There is also uncertainty in the relative frequencies and consequences of different types of encounters with trawl warps in New Zealand fisheries (Abraham 2010a, Richard et al. 2011 appendix B). Some of this uncertainty is included and propagated in the most recent published risk assessment (Richard et al. 2020).

A review of available information on cryptic mortality was commissioned under CSP project INT2013-05 and supported by MPI project PRO2012-17 (Pierre et al. 2015). Pierre et al. (2015) recommended four 'nest steps' to progress the improvement of cryptic mortality scalars as applied to New Zealand fisheries.

- Amend the definition of cryptic mortality applied in New Zealand, such that the definition in use is better aligned with international approaches. Pierre et al. (2015) suggested the following definition: 'seabird mortalities that are unobserved or unobservable and directly or indirectly result from interactions with fishing gear or fishing operations';
- Examine existing datasets identified in this report, that are available internationally and in New Zealand, to improve estimates of cryptic mortality for New Zealand species, or species groups, caught in surface longline and trawl fisheries;
- Amend data collection protocols used by New Zealand fisheries observers such that potential cryptic mortalities will be documented routinely; and
- Develop a data collection programme to support the estimation of method-specific scalars for bottom longline fisheries, especially vessels less than 34 m in overall length.

In response, project PRO2019-10 will review and update the structural definitions used in current risk assessments, with a particular emphasis on cryptic mortalities and how they are disaggregated. The effects of these modifications to the seabird SEFRA model structure and model fits will then be evaluated.

8.4.2.7.3 MORTALITIES IN NON-COMMERCIAL FISHERIES

Little is known about the nature and extent of incidental captures of seabirds in non-commercial fisheries, either in New Zealand or globally (Abraham et al. 2010a). In New Zealand, participation in recreational fishing is high and 2.5% of the adult population are likely to be fishing in a given week (mostly using rod and line). Because of this high participation rate, even a low rate of interactions between individual fishers and seabirds could have population-level impacts. A boat ramp survey of 765 interviews at two locations during the summer of 2007–08 revealed that 47% of fishers recalled witnessing a bird being caught some time in the past. Twenty-one birds were reported caught on the day of the interview at a capture rate of 0.22 (95% c.i.: 0.13–0.34) birds per 100 hours of fishing. Observers on 57 charter boat trips recorded seabird captures at rate of 0.36 (0.09–0.66) birds per 100 fisher hours. The most frequently reported type of bird caught in rod and line fisheries were petrels and gulls. Captures of albatrosses, shags, gannets, penguins, and terns were also recalled.

The ramp surveys reported by Abraham et al. (2010a) were limited and covered only two widely separated parts of the New Zealand coastline. However, they also report two other pieces of information that suggest that non-commercial captures are likely to be very widespread. First, the Ornithological Society of New Zealand's beach patrol scheme records seabird hookings and entanglements as a common occurrence throughout New Zealand. Second, returns of banded birds caught in fisheries (separating commercial and non-commercial fisheries is very difficult) are very widely distributed around the coast (Figure 8.20).

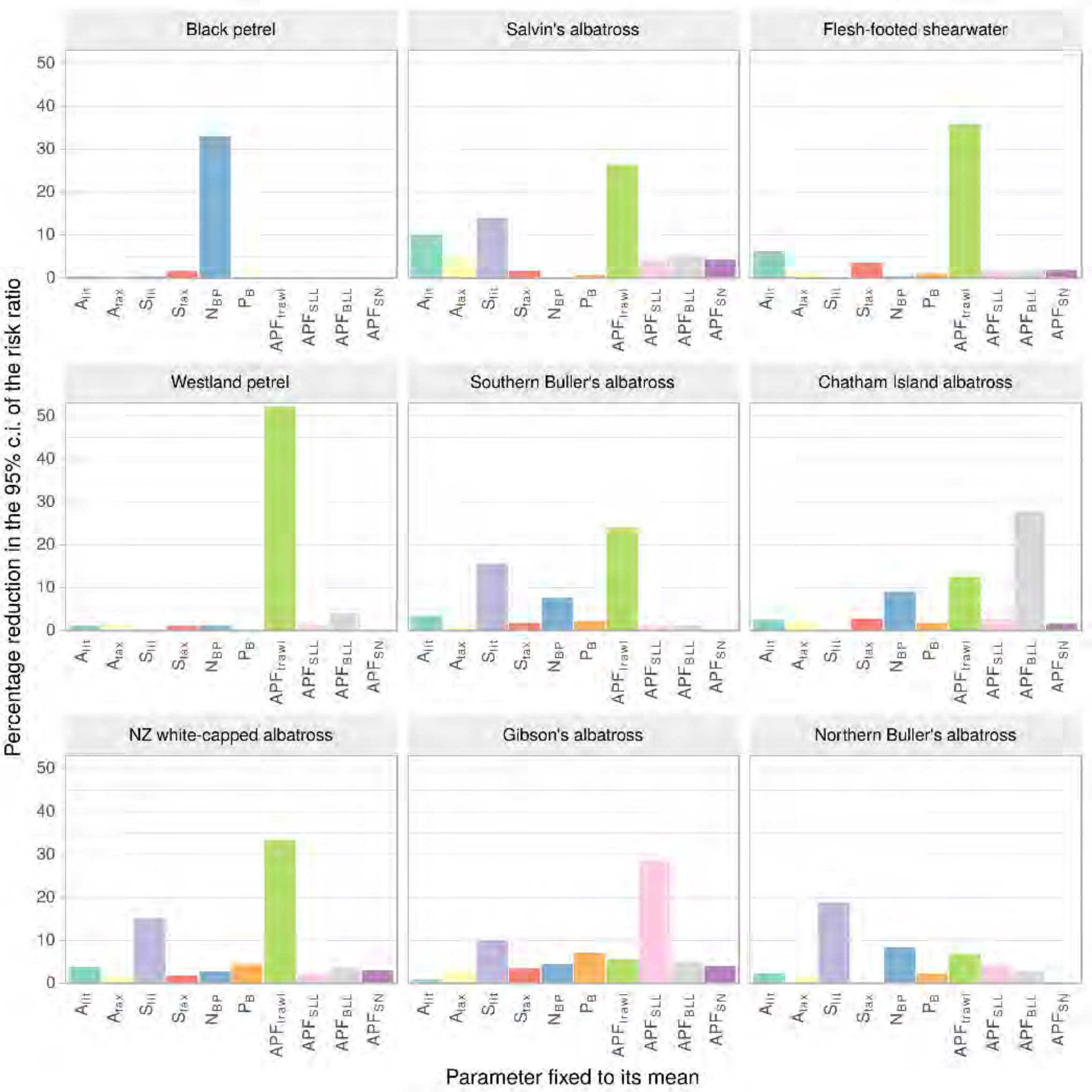


Figure 8.19: (reproduced from Richard et al. 2017). Sensitivity of the uncertainty in the risk ratio for the nine seabird species with the highest risk ratio. For each seabird type, the sensitivity to the uncertainty in the following parameters is considered: annual potential fatalities in trawl, bottom longline, surface longline and set net fisheries (TWL, BLL, SLL, SN, respectively); the cryptic multipliers (CM); age at first reproduction (A); adult survival (SA); the number of annual breeding pairs (N_{BP}); and the proportion of adults breeding (PB). The sensitivity is defined as the percentage of reduction in the 95% confidence interval of the risk ratio that occurs when the parameter is set to its arithmetic mean

Noting that our understanding of seabird capture rates in amateur fisheries is very sketchy, it is possible to make first-order estimates of total captures using information on fishing effort. For example, in the north-eastern region where most of the Abraham et al. (2010a) interviews were conducted, there were an estimated 4.8 (4.4–5.2) million fisher hours of rod and line fishing from trailer boats in 2004–05 (Hartill et al. 2007). Applying the Abraham et al. (2010a) capture rate leads to an estimate of 11 500 (6600–17 200) captures per year in this area. Based on estimates of nationwide recreational fishing effort, this could increase to as many as 40 000 bird captures annually. Most birds captured by amateur fishers were reported to have been released unharmed (77% of the incidents recalled) and only three people reported incidents where the bird died. Because of likely recall biases and the qualitative nature of the survey, the fate of birds that are captured by amateur fishers remains unclear.

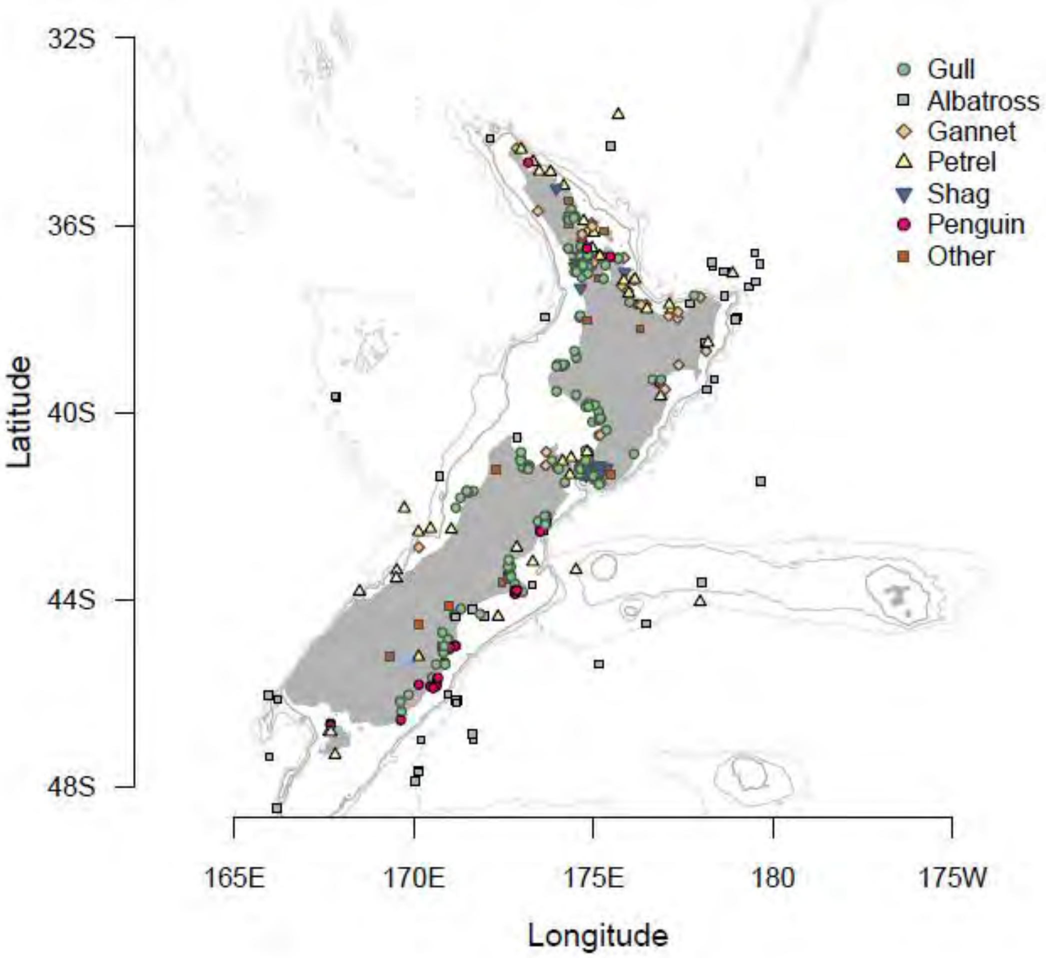


Figure 8.20: (from Abraham et al. 2010a). Distribution of the reported capture locations for banded seabirds reported as being captured in fishing gear, 1952–2007. Note, band recovery locations are reported with low spatial precision and some of the inland locations may be incorrect.

Non-commercial fishers are allowed to use set nets in New Zealand and two studies suggest that these have an appreciable bycatch of seabirds. A study of captures in non-commercial set nets in Portobello Bay, Otago Harbour, between 1977 and 1985 (Lalas 1991) suggested that spotted shags were the most frequently caught taxa (82 recorded, compared with 14 Stewart Island shags and 2 little shags). Lalas (1991) suggested that up to 800 spotted shags (20% of the local population) may have been caught in the summer of 1981–82. A broader-scale study of yellow-eyed penguin mortality in set nets in southern New Zealand (Darby & Dawson 2000) suggested non-negligible captures of this species by non-commercial fishers, also reporting other seabirds like spotted shags and little blue penguin.

8.4.2.7.4 OUT OF ZONE MORTALITY

Robertson et al. (2003) mapped the distribution of the 25 breeding (mainly endemic) New Zealand seabird taxa they considered most at risk outside New Zealand waters. These ranged widely: four used the South Atlantic; four the Indian Ocean; 22 Australian waters and the Tasman Sea; 15 used the South Pacific Ocean as far afield as Chile and Peru; and 6 used the North Pacific Ocean as far north as the Bering Sea. These taxa therefore use the national waters of at least 18 countries. For example, the Level 2 risk assessment described by Richard et al. (2011) includes only that part of the range of each taxon contained within New Zealand waters, but many, including commonly caught seabirds like white-capped albatross and white-chinned petrel, range much further and are vulnerable to fisheries in other parts of the world. For instance, fatalities of white-capped albatross outside the New Zealand EEZ greatly exceed fatalities within the zone (Baker et al. 2007a, Francis 2012, Table 8.19), and more than 10 000 white-chinned petrels are killed off South America each year (Phillips et al. 2006), noting that reliable records are not available for most of the fisheries involved. Note that white-chinned petrels also breed on Prince Edward Islands, Falkland Islands, South Georgia, Iles Crozet, and the Kerguelen group, so South American captures may be from populations other than New Zealand's. Based on similar analyses, Moore & Zydelis (2008) concluded that a population-based, multi-gear, and multi-national framework is required to identify the most significant threats to wide-ranging seabird populations and to prioritise mitigation efforts in the most problematic areas. To that end, the Agreement for the Conservation of Albatrosses and Petrels (ACAP) adopted a global prioritisation framework at the Fourth Session of the Meeting of the Parties (MoP4) in April 2012 (ACAP 2012).

On 2 July 2019 a female Antipodean albatross was captured while wearing a transmitting tag, which was subsequently recovered. The capture happened at 27° S by a longline fishing vessel operating in international waters (Elliott & Walker 2019). With seabird tracking and high seas effort availability increasing, a Southern Hemisphere Risk Assessment is planned to assess the total scale of impact these fisheries may be having.

Table 8.19: (from Francis 2012). Estimates of the number of white-capped albatrosses killed annually, by fishery. The first two columns are from Baker et al. (2007a) (mid-point where a range was presented), including their assessment of reliability (L = low, M-H = medium-high, H = high). Updated estimates are from Watkins et al. (2008, *) and Petersen et al. (2009, **). Estimates not already corrected for cryptic mortality are either doubled to allow for this (***) or replaced by estimates of potential fatalities from Richard et al. (2011, ****), noting that potential fatalities may considerably overestimate actual fatalities.

Fishery	From Baker et al. 2007a		Updated	Incl. cryptic mortality
South African demersal trawl	4 750	(L)	* 6 650	6 650
Asian distant-water longline	1 255	(L)	–	*** 2 510
Namibian demersal trawl	910	(L)	* 1 270	1 270
Namibian pelagic longline	180	(L)	** 195	*** 390
NZ hoki and squid trawl	513	(MH)	–	**** 4 920
NZ longline	60	(MH)	–	**** 199
Australian (line fisheries)	15	(MH)	–	*** 30
South African pelagic longline	570	(H)	** 570	*** 1 140
Total	8 210	–	–	17 110

8.4.2.7.5 OTHER SOURCES OF ANTHROPOGENIC MORTALITY

Taylor (2000) listed a wide range of threats to New Zealand seabirds including introduced mammals, avian predators (weka), disease, loss of nesting habitat, competition for nest sites, coastal development, human disturbance, commercial and cultural harvesting, volcanic eruptions, pollution, plastics and marine debris, oil spills and exploration, heavy metals or chemical contaminants, global sea temperature changes, marine biotoxins, and fisheries interactions. Relatively little is known about most of these factors, but the parties to ACAP have agreed a formal prioritisation process to address and prioritise major threats (ACAP 2012). Croxall et al. (2012) identified the main priorities as: protection of Important Bird Area (IBA) breeding, feeding, and aggregation sites; removal of invasive, especially predatory, alien species as part of habitat and species recovery initiatives. Lewison et al. (2012) identified similar research priorities (in addition to direct fishing-related mortality), including: understanding spatial ecology, tropho-dynamics, response to global change, and management of anthropogenic impacts such as invasive species, contaminants, and protected areas. Non-fishing-related threats to seabirds in New Zealand are largely the mandate of the Department of Conservation and a detailed description is beyond the scope of this document (although causes of mortality other than fishing are clearly relevant to the interpretation of risk assessment restricted to the direct effects of fishing). These threats are identified by DOC's Action Plan for Seabird Conservation in New Zealand (Taylor 2000) and various Threatened Species Recovery Plans.

8.4.2.8 DEVELOPMENT OF THE RISK ASSESSMENT FRAMEWORK

The following steps were identified in the NPOA-seabirds 2013 (MPI 2013) to improve the risk assessment framework that supported the implementation of the NPOA-seabirds 2013 and provides a framework for monitoring performance for the NPOA 2020 (Fisheries New Zealand 2020):

- implementation of a framework and process to consolidate different risk assessment and population monitoring results into an integrated assessment, including:
- checking the assessment results for the mulit-species risj assessment in particular high-risk species-fishery interactions, in light of other available data or identifiable structural biases on a case-by-case basis,
- a mechanism to incorporate issues associated with seabird mortalities outside the EEZ and recreational fisheries risk in future assessments, and
- the use of species population models or census data to constrain input parameters or interpret estimates of risk;
- routine update of the integrated fisheries risk assessment with relevant new information; and
- periodic review and update of risk management priorities in light of current risk estimates.

8.4.3 AT-RISK SEABIRDS: DEMOGRAPHIC, DISTRIBUTION, AND FULLY QUANTITATIVE MODELLING STUDIES

Previous versions of this chapter included species accounts for a smaller selection of species than that presented here. Furthermore, previous species accounts were included for those species that were included as study species in project MPI PRO200601. Here 13 species accounts are included for those seabirds that have a risk ration of 0.1 of greater in the latest iteration of the SEFRA (see Table 8.18, Abraham & Richard 2019), black petrel to Otago shag inclusive (those species coloured-coded red or shades of orange in Table 8.18)

8.4.3.1 BLACK PETREL

Black petrels are medium–sized endemic seabirds that only breed on Te Hauturu-o-Toi/Little Barrier Island and Great Barrier Island/Aotea in the Hauraki Gulf of New Zealand. Black petrels are known by the name of tākoketai by Ngāti Rehua Ngāti Wai ki Aotea, the tangata whenua and mana whenua of Great Barrier Island/Aotea.

Black petrels are ranked as Nationally Vulnerable under the New Zealand Threat Classification System and Vulnerable on the IUCN Red List of Threatened Species (Robertson et al. 2017, BirdLife International 2017). They are recognised as the seabird species that is at greatest risk of being adversely impacted by unsustainably high rates of bycatch in commercial fisheries within New Zealand's Exclusive Economic Zone (Richard et al. 2020). Of the 171 observed captures of black petrel recorded between 2002 and 2019, 55.6% of captures occurred in bottom longline fisheries, 26.3% in surface longline fisheries and 18.1% in trawl fisheries (https://protectedspeciescaptures.nz/PSCv6/; accessed 12/04/2021). Black petrels are also exposed to threats on land, principally depredation by cats (*Felis catus*), rats (*Rattus* spp.) and pigs (*Sus scrofa*) (Bell et al. 2013).

To monitor the ongoing population-level impacts of commercial fisheries on black petrels, it is necessary to quantify population parameters such as annual burrow occupancy rates, annual adult reproductive success as well as both adult and juvenile annual survival rates to create accurate assessments of population trends. To this end, a long-term research project aimed at quantifying these population parameters was initiated in 1995–1996 (Bell & Sim 1998). During this first season, three 40 m x 40 m study grids were set up within the largest known breeding colony on Mt Hobson/Hirakimata on Great Barrier Island/Aotea, and all burrows within the grids were marked and monitored. Additional burrows located within 10 m of the public walking tracks were also monitored. In 1998–1999, the number of study grids was increased to six, and then to nine in 1999–2000 (Bell & Sim 2000a, Bell & Sim 2000b). Over the years, additional burrows situated near the public walking tracks have continued to be added, so that by the 2020–2021 season a total of 476 study burrows were being monitored (Bell et al. 2022).

Burrow occupancy rate in the nine census grids provides the most consistent and representative measure of burrow occupancy across the study area. In the 2019–20 breeding season, in the 193 study burrows within the study grids, the mean percentage of study grid burrows occupied by breeding black petrels was 60.1%, 0.4% less than the 26-year average study grid burrow occupancy rate of 60.5% (Figure 8.21).

The breeding success rate observed during the 2020–21 season (76.8%) was 5% greater than the 26-year average of 71.8% (Figure 8.22).

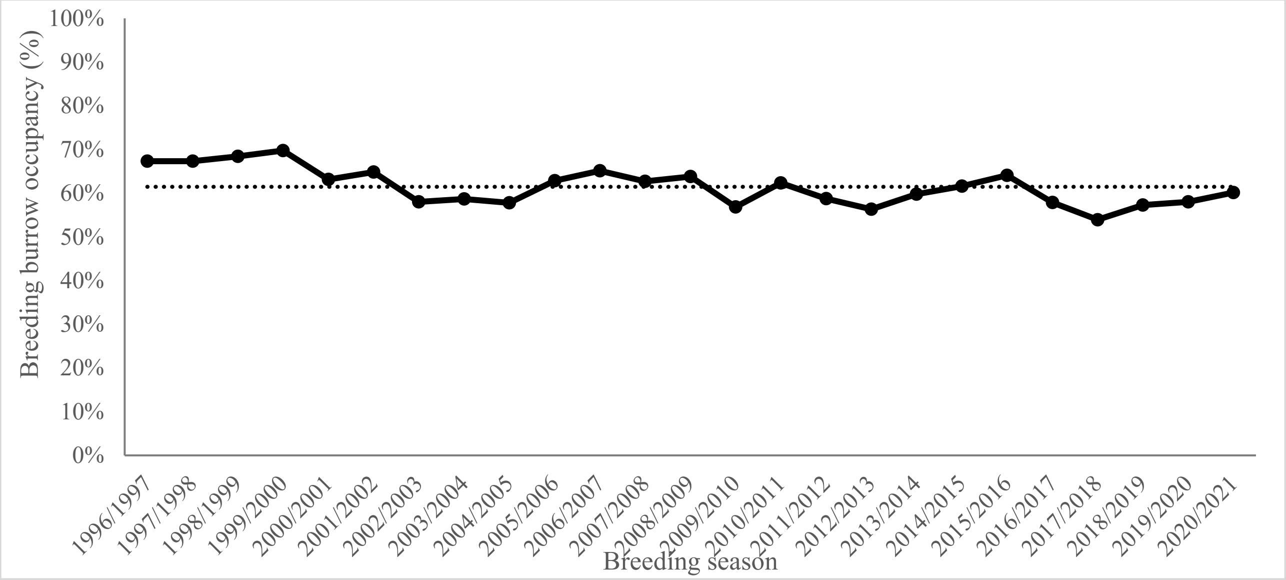


Figure 8.21: (from Bell et al. 2022). Percentage of census grid burrows occupied by breeding black petrels at Mt Hobson/Hirakimata on Great Barrier Island/Aotea between 1996 and 2021 (dotted line represents the mean occupation of census grid burrows over 26-years by breeding black petrels).

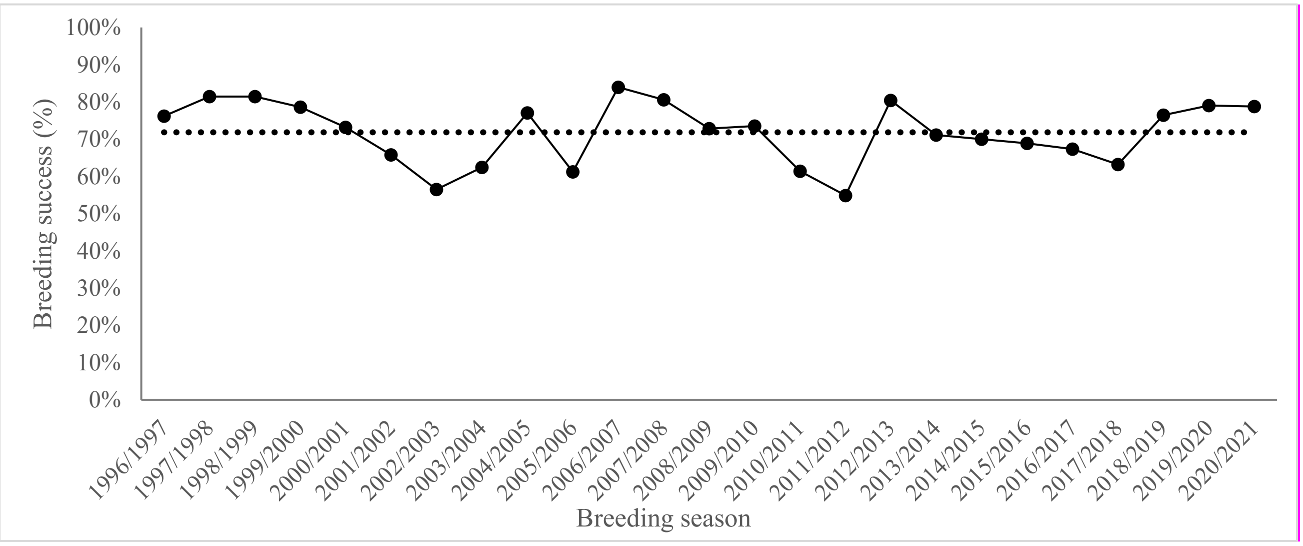


Figure 8.22: (from Bell et al. 2022). Breeding success (percentage of breeding burrows that fledge a chick) of all black petrel study burrows at Mt Hobson/Hirakimata on Great Barrier Island/Aotea between 1996 and 2021 (the dotted line represents the mean breeding success over 26-years by breeding black petrels).

Throughout the 2017–18 and 2018–19 breeding seasons, the at-sea distribution of black petrel was investigated using GPS and GLS loggers at various periods during the year. GPS tracking of birds during the incubation period showed that birds foraged to both the west and east of northern New Zealand (Figure 8.23). Average foraging trip duration was 10.7 days (± 5.7 SD, n = 29, range 2–26 days), and trip distance was 4383 km (± 2527 km SD, n = 26, range 522–10 275 km), with the mean maximum distance from Aotea/Great Barrier Island of 1029 km (± 718 km SD, n = 29, range 99–2872 km).

GPS tracking of birds during chick-rearing showed that trip durations varied greatly between individual birds, but the average was 8.6 days (± 5.7 SD, n = 37, range: 2–22 days). The average trip distance was 3633 km (± 2888 km SD, n = 32; range 825–9437 km) and the average maximum distance travelled from Aotea/Great Barrier Island was 846 km (± 679 km SD, n = 32; range 151–2902 km). As during incubation, birds foraged to both the west and east of northern New Zealand (Figure 8.24 and Figure 8.25).

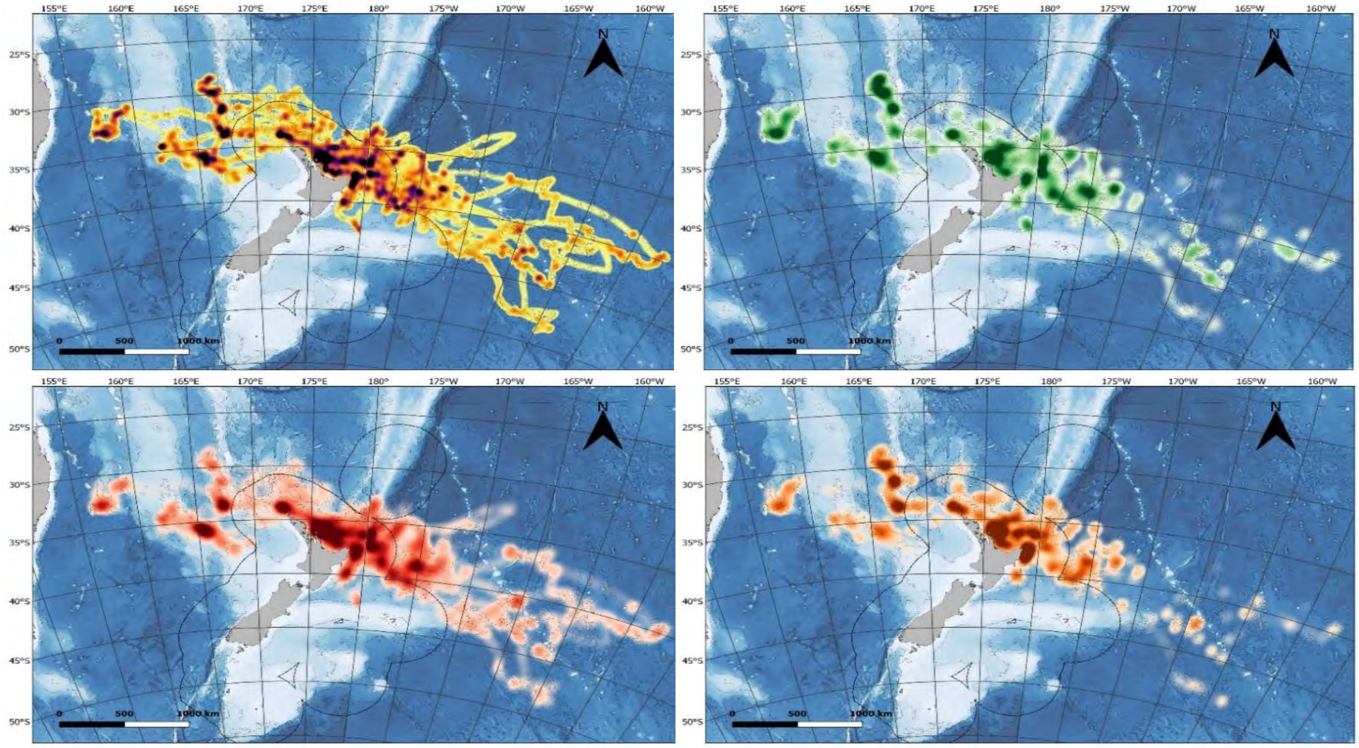


Figure 8.23: (from Bell et al. 2020). Kernel density maps of black petrel at-sea distribution and behaviour during incubation. Clockwise from top left: all behaviours combined (multi-coloured), foraging (green), rest/rafting (orange), and flight (red). Darker areas represent greater concentrations for each of the relative behaviour types. The black dashed line represents the boundary of the New Zealand EEZ.

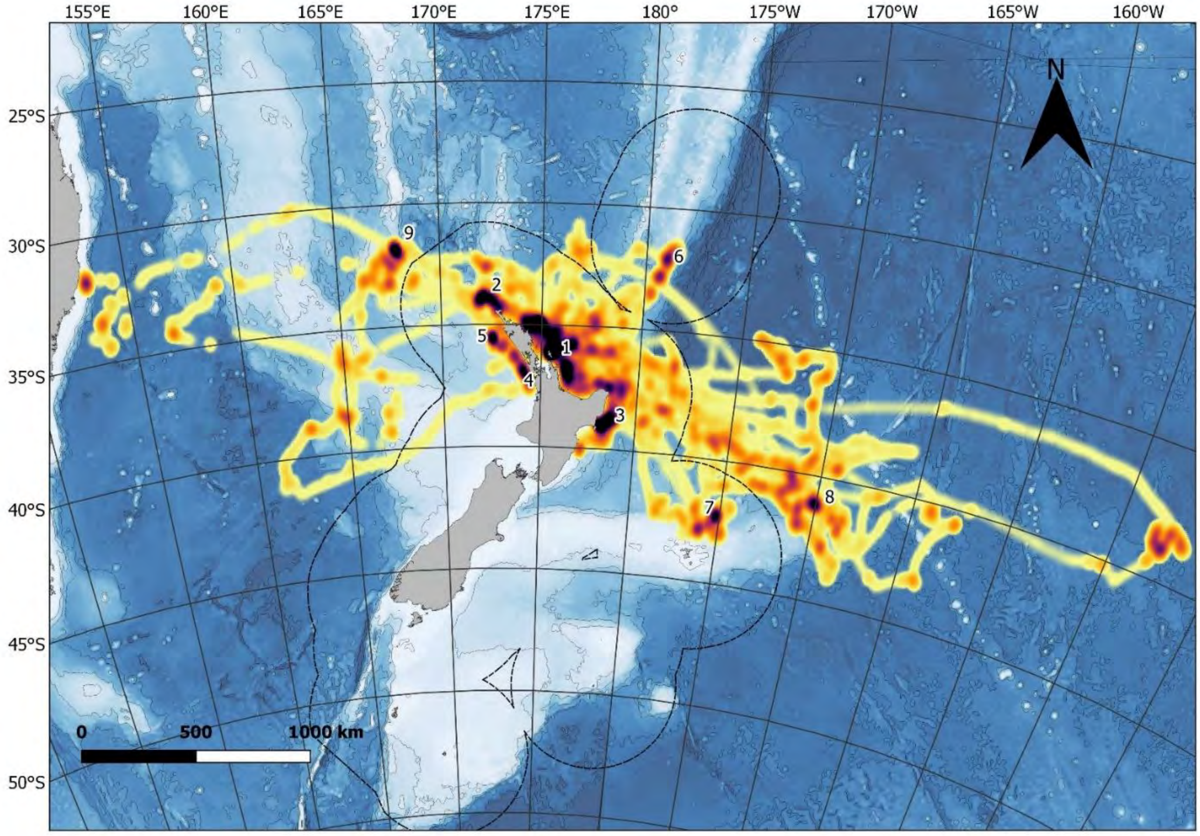


Figure 8.24: (from Bell et al. 2020). Kernel density map of all black petrel chick-provisioning trips from Aotea/Great Barrier Island recorded in March-April 2018. Darker areas represent greater concentrations of black petrel activity. The black dashed line represents the boundary of the New Zealand Exclusive Economic Zone.

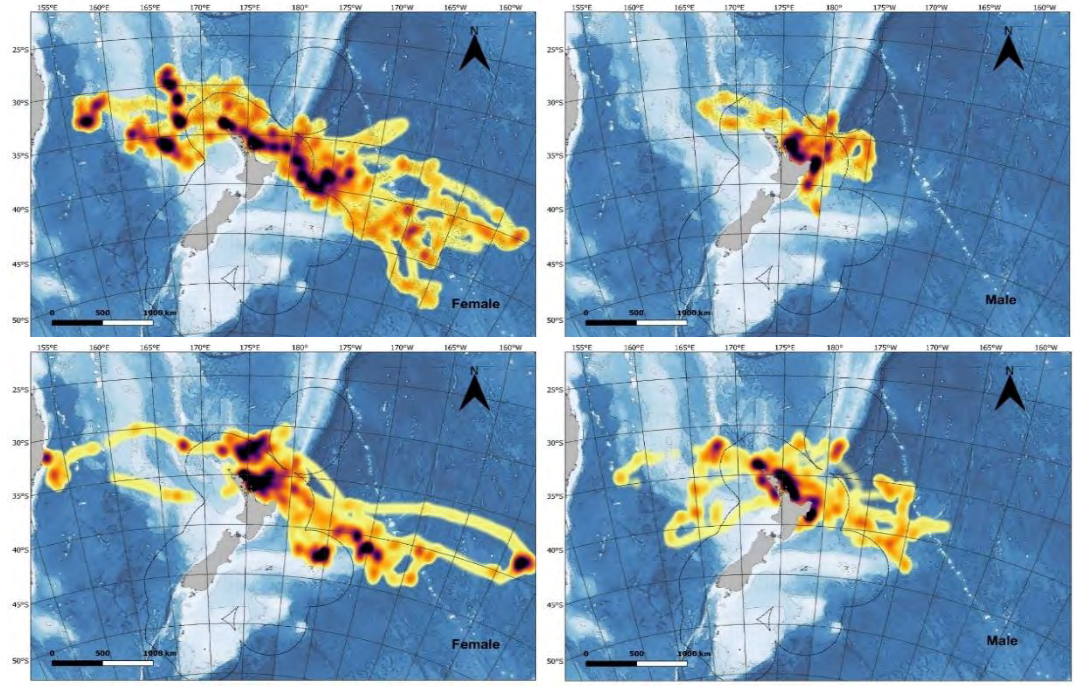


Figure 8.25: (from Bell et al. 2020) Kernel density maps comparing foraging trips for female (left-hand maps) and male (right-hand maps) black petrels breeding on Aotea/Great Barrier Island. The top two maps show foraging trips during incubation; the bottom two maps show chick-provisioning trips. Darker areas represent greater concentrations of black petrel activity. The black dashed line represents the boundary of the New Zealand EEZ.

The at-sea distribution of black petrel during the non-breeding season was investigated using GLS devices. Most black petrels tracked migrated in a band centred on the line between the points 170° W, 30° S and 110° W, 10° N. Generally, black petrels migrated east from New Zealand on a more southerly path and migrated back west toward New Zealand on a more northerly path. During the non-breeding season birds concentrated in two distinct areas. The first is centred off the coast of Ecuador and the second is due southwest of the Galapagos Islands stretching from the equator to approximately 1000 km south of the equator (Figure 8.26). They seem to be concentrated off the coast of Peru when they arrive between May and June and then move further north to sea areas off the coasts of Ecuador and Colombia and southwest of the Galapagos Islands as the season progresses (Figure 8.26). There was no apparent difference in the spatial distribution of male and female black petrels during the non-breeding season.

Breeding black petrels are unevenly distributed across Great Barrier Island/Aotea, with significantly higher densities of breeding birds found on high-altitude ridges under mature, unlogged, and unburnt native forest than at lower altitudes or in other vegetation types on the island (Marchant & Higgins 1990; WMIL unpublished data). Great Barrier Island/Aotea was stratified into high-, medium-, and low-grade black petrel habitat using all existing data on the presence and location of black petrel breeding burrows on the island using ArcMap (Table 8.20), with overlaid map layers describing altitude, vegetation type, and the presence and absence of feral pigs (a major threat to burrow-nesting shorebirds on land). Random start points of 100-m long line transects were generated within the high- and medium-grade habitat strata and surveyed during peak incubation using distance sampling.

Detection functions fitted to these line transect data provided a reasonable fit to the data, but the uniform key function with a cosine adjustment term was the model with the lowest AIC value, so those models were used to generate estimates of burrow detection probability and burrow density. Distance sampling proved to be a robust method to estimate black petrel breeding burrow density and breeding population size on Great Barrier Island/Aotea with all the key assumptions being met in the high-grade and core medium-grade habitat around Mt Hobson/Hirakimata. However, the very low numbers and scattered distribution of black petrel burrows in Glenfern and all other medium-grade habitat across Great Barrier Island/Aotea meant that the requirements for distance sampling could not be met and showed the limitations of the Distance modelling method in such habitat.

The total density of black petrel burrows in the high-grade habitat stratum is 28.649 burrows/ha and medium-grade habitat stratum is 2.8211 burrows/ha (Table 8.21. Multiplying these data by the proportion of these burrows used for breeding and by the total area of the habitat stratum yielded estimates of 1430 black petrel breeding pairs present in the high-grade habitat stratum and 2418 breeding pairs in the medium-grade habitat stratum, or 3064 breeding adults (2349–3995, 95% confidence limits) (Table 8.21).

Distance modelling produced valid models for the number of breeding black petrels within the 1019-ha core habitat around Mt Hobson/Hirakimata with a population estimate of 4336 breeding pairs (or 8672 birds) (Table 8.21). This suggests that the core black petrel habitat in the vicinity of Mt Hobson/Hirakimata supports most of the estimated global breeding population of black petrels, with the remainder breeding in isolated pockets elsewhere on Aotea/Great Barrier Island and on Te Hauturu-o-Toi/Little Barrier Island
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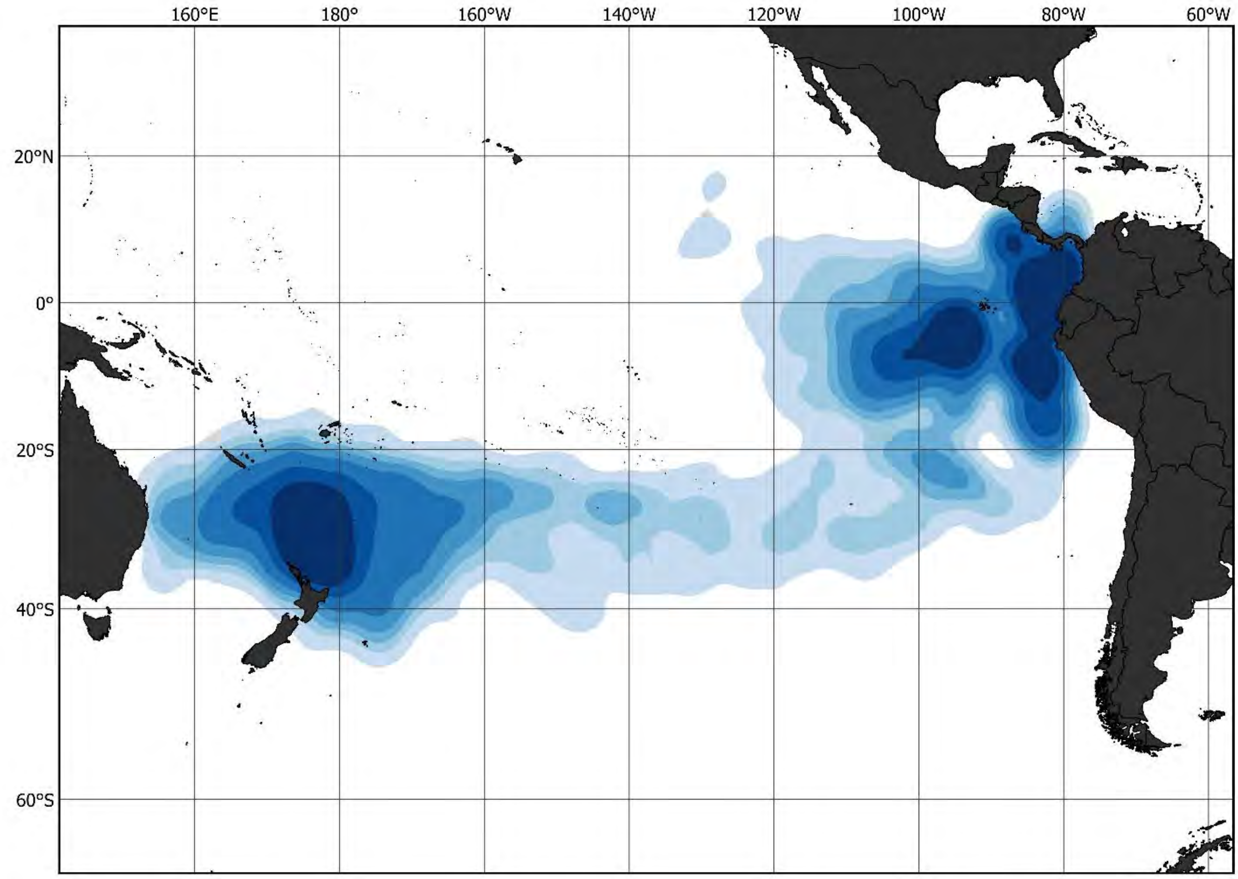


Figure 8.26: (from Bell et al. 2020). Kernel density map of GLS points for all black petrels between March 2018 and January 2019. Darker areas represent greater concentrations of black petrel activity.

Table 8.20: (from Bell et al. 2020). Definitions of high- medium- and low-grade habitat strata for black petrels on Aotea/Great Barrier Island.

Habitat stratum	Vegetation type	Altitude	Feral pigs	Example sites
High grade	Mature Forest	>400	Absent	Mt Hobson/Hirakimata and Mt Heale
Medium grade	Mature Forest	>400	Present	Te Paparahi Block
		250-400	Present	
		<250	Absent	Glenfern Sanctuary
	Mature Forest	>250	Present	
		<250	Absent	Glenfern Sanctuary
Low grade	Mature Forest	<250	Present	
	Shrubland	<250	Present	

Table 8.21: (from Bell et al. 2022). Model outputs for the population estimate for breeding black petrels on Great Barrier Island/Aotea using Conventional Distance Sampling Engine in Distance 7.2.

Where: ✓ = Model was valid with all assumptions met and × = model was not valid due to low level of burrow detections.

Habitat stratum	General area	Total 3D area (ha)	Burrow density/ha (truncated burrow density)	Breeding burrow occupancy rate	AIC	Population estimate (i.e., breeding pairs)	Truncated population estimate (truncated distance, m)	Validity
High grade	Mt Hobson/Hirakimata	108	28.649 (12.583)	0.4620	563.69952 (546.61798)	1430	626 (7 m)	✓
Medium grade	Core: Mt Hobson/Hirakimata	911	7.5138 (5.2093)	0.4524	642.48712 (525.93579)	3097	2147 (12 m)	✓
	Glenfern	246	0.01	0	N/a	N/a	–	×
	Te Paparahi (Northern) Block	590	0.002	0.6667	N/a	N/a	–	×
	The Needles	35	0.09	0.6667	N/a	N/a	–	×
	Ruahine	23	0	0.6667	N/a	N/a	–	×
	Te Ahumata/White Cliffs	87	0	0.6667	N/a	N/a	–	×
	Tramline	15	0	0.6667	N/a	N/a	–	×
	Windy Hill	10	0	0.6667	N/a	N/a	–	×
	Core & Glenfern combined	1157	5.6374 (4.2340)	0.4419	654.23914 (537.30853)	2882	2165 (12 m)	✓
	All medium grade combined	1917	2.8211 (2.8213)	0.4470	667.11908 (549.69745)	2418	2418 (12 m)	✓
High and medium grade combined	High and all medium grade combined	2025	7.0295 (6.8885)	0.4552	1268.8552 (1133.0770)	6480	6350 (12 m)	×
	High and core medium combined	1019	9.9240 (9.2967)	0.4577	1256.5188 (1121.0659)	4629	4336 (12 m)	✓

8.4.3.1.1 QUANTITATIVE MODELS FOR BLACK PETREL

Francis & Bell (2010) analysed data from the main population of black petrel at Great Barrier Island. Abundance data from transect surveys were used to infer that the population was probably increasing at a rate between 1.2% and 3.1% per year. Mark-recapture data were useful in estimating demographic parameters, like survival and breeding success, but contained little information on population growth rates. Fishery bycatch data from observers were too sparse and imprecise to be useful in assessing the contribution of fishing-related mortality. Francis & Bell (2010) suggested that, because the population was probably increasing, there was no evidence that fisheries posed a risk to the population at that time. They cautioned that this did not imply that there was clear evidence that fisheries do not pose a risk.

Subsequent analysis (Bell et al. 2012) included an additional line transect survey in 2009–10 in which the breeding population was estimated to be about 22% lower than in 2004–05 (the latest available to Francis & Bell 2010). Updating the model of Francis & Bell (2010) made little difference to estimates of demographic parameters such as adult survival, age at first breeding, and juvenile survival (which had 95% confidence limits of 0.67 and 0.91). The uncertainty in juvenile survival gave rise to uncertainty in the estimated population trend, with a mean rate of population growth over the modelling period ranging from -2.5% per year (if juvenile survival = 0.67) to +1.6% per year (if juvenile survival=0.91, close to the average annual survival rate for older birds) (Figure 8.27). Bell et al. (2012) concluded that the mean rate of change of the population over the study period had not exceeded 2% per year, though the direction of change was uncertain. The latest counts have increased, due mainly to increases in breeding rate (Bell et al. 2013), suggesting even more uncertainty about population trend than when the quantitative modelling was last updated.

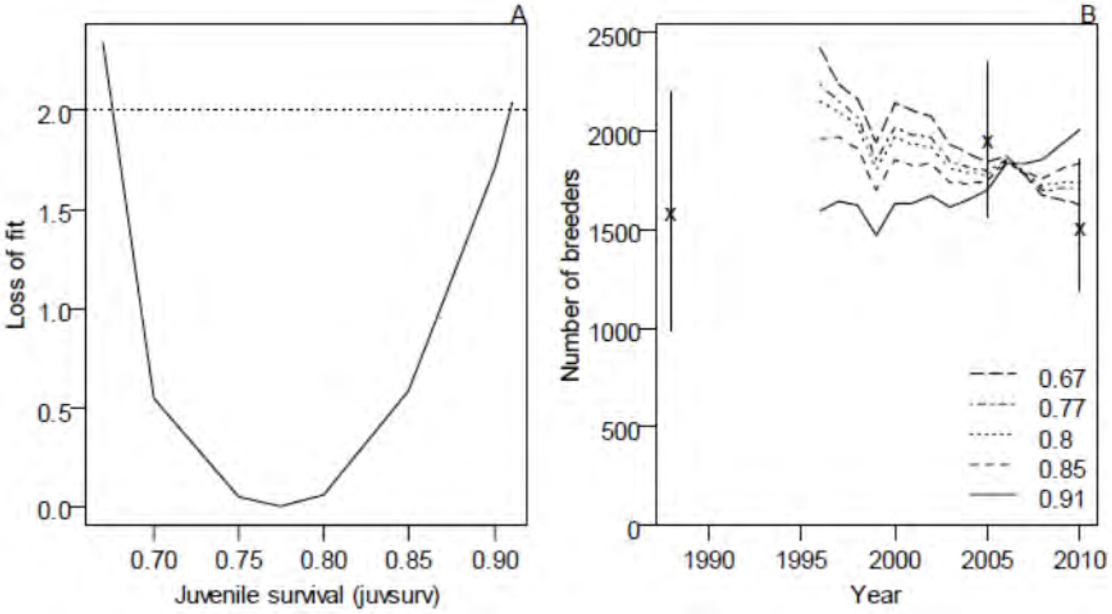


Figure 8.27: (from Bell et al. 2012). Likelihood profile for annual probability of juvenile survival of black petrel, showing: A, the loss of fit (the horizontal dotted line shows a 95% confidence interval for this parameter); and B, population trajectories corresponding to different values of juvenile survival, together with population estimates from transect counts (crosses with vertical lines indicate 95% confidence intervals). Note that the 1988 population estimate was not used in the model.

8.4.3.2 SALVIN'S ALBATROSS

8.4.3.2.1 BOUNTY ISLANDS

Salvin's albatross is endemic to New Zealand, breeding only on the Bounty Islands and the Western Chain of the Snares Islands. The Bounty Islands are a group of bare rocky islands/islets situated 659 km south-east of New Zealand's South Island. In October 2010, Baker et al. (2010a) completed an aerial survey of the Bounty Islands to photograph all albatross colonies. This was the first complete population survey of Salvin's albatross on the Bounty Islands. Photo montages were created from the aerial photography and the number of nesting birds was counted. From these data, Baker et al. (2010a) estimated the total count of nesting Salvin's albatrosses in the Bounty Islands in October 2010 to be 41 101 (95% c.i.: 40 696–41 506).

This estimate may be biased high by the presence of 'loafers' (non-breeding birds) because it was not possible to ground-truth the aerial photography or detect the proportion of loafers within the colony from close-up photography (because of the general lack of nest pedestals resulting from low availability of nesting material on the island). Conversely, the estimate may be biased low because aerial photography was not possible on some small areas of steep cliff where albatross nests may have been missed (Baker et al. 2012).

A review of existing ground counts was reported by Amey & Sagar (2013). To estimate population trends and examine the accuracy of ground counts, whole-island surveys of Salvin's albatross breeding at Proclamation Island, Bounty Islands, were undertaken during November in 1997, 2004, and 2011. These counts suggest that the numbers of Salvin's albatross nests on Proclamation Island declined by 14% between 1997 and 2004, by 13% between 2004 and 2011, and, overall, by 30% between 1997 and 2011. Counts of nests on Depot Island decreased by 10% between 2004 and 2011.

Baker et al. (2014a) conducted a repeat aerial survey of the Bounty Islands in October 2013. Using the same correction factor applied to the 2010 counts, they estimated the total annual breeding pairs at 39 995 (95% c.i.: 39 595–40 395) compared with the corrected estimate for 2010 of 31 786 (95% c.i: 31 430–32 143).

The DOC Conservation Services Programme has been reviewing the methodology for undertaking a survey of the Salvin's albatross on the Bounty Islands (Debski & Hjörvarsdóttir 2017). In October 2018, a ground count of breeding Salvin's albatross was completed at Proclamation Island, together with ground-truthing of an aerial survey and deployment of a suite of tracking devices. Further work was planned for October 2019.

8.4.3.2.2 WESTERN CHAIN, SNARES ISLANDS

In 2008, a three-year study of Salvin's albatrosses was initiated at the Western Chain of the Snares Islands. The three main objectives of the Salvin's albatross field work were:

- to estimate the breeding population size from counts of occupied nests;
- to determine foraging locations and activity by retrieving geolocator tracking devices deployed in 2008; and
- to estimate annual survival rates of banded adult birds from recapture analyses.

Totals of 1195 and 1116 breeding pairs were counted on Toru Islet and Rima Islet during October 2008 (Charteris et al. 2009) and September–October 2009, respectively (Carroll et al. 2010) (Table 8.22). Only Toru Islet was sampled in 2010.

To estimate the adult survival of Salvin's albatross, a total of 257 occupied nests were counted within a clearly defined study area established in October 2008 (Charteris et al. 2009). Within this area, 116 birds banded in previous years were recaptured, and a further 20 breeding birds were banded in the study area during October 2010. Among the recaptured birds were 13 that had been banded as chicks on Toru Islet during 1986, and 23 of the 123 birds banded as breeding adults in 1995. These recapture rates lead to an estimated adult survival probability of 0.967 for Salvin's albatross, one of the highest estimates for any species of annual-breeding albatross (Sagar et al. 2011), which is noteworthy given the location of the colony in an area of relatively high fishing activity.

Twenty-four of the 35 geolocation loggers deployed on breeding birds during October 2008 were retrieved. Data were processed by the British Antarctic Survey and a preliminary assessment of the distribution of Salvin's albatrosses during the entire year is shown in Figure 8.28. None of the 24 birds tracked was within the New Zealand EEZ during April; 23 were in South American waters between Tierra del Fuego and northern Peru and one was over eastern Bass Strait and along the eastern coast of Tasmania (Figure 8.28a). Birds began to return to New Zealand waters during May and this continued throughout June and July. The tracks of birds exiting South American waters originated from either the Peruvian or southern Chilean coasts. During this period, birds recently arrived in New Zealand waters were primarily east of the Chatham Islands, off Puysegur, and at the Stewart-Snares shelf (Figure 8.28b). Egg-laying starts in August and all the birds were within Australasian waters during August to October, primarily on the Challenger Plateau, off Puysegur, at the Stewart-Snares shelf, and at Campbell Plateau (Figure 8.28c). During this period these birds from the Snares Islands Western Chain occupy a relatively narrow longitudinal range between 160° E and 175° E and appear to avoid, or be excluded from, the area around the Bounty Islands, where there is another colony of Salvin's albatross. Beginning in mid-October, chicks hatch and, between November and March, presumed successful breeders foraged primarily on the Challenger Plateau, off Puysegur, at the Stewart-Snares shelf, and at Campbell Plateau (Figure 8.28d). There was some movement across the Pacific in each of the months between November and March with presumed failed breeders leaving the New Zealand EEZ during the earlier part of this period and presumed successful breeders migrating east during March (Sagar et al. 2011).

Further research has been recently conducted on the Salvin's albatross on the Snares Western Chain (Baker et al. 2015b). This research included a ground-based census, an aerial survey (including ground-truthing) and collection of information on tagged birds. The aerial survey estimated 1486 (95% c.i.: 1409–1563) annual breeding pairs in 2014–15, which was 32% higher than the ground counts undertaken on the same day of the aerial survey (Baker et al. 2015b).

Table 8.22 (from Sagar et al. 2011). Numbers of Salvin's albatross pairs breeding on Toru Islet and Rima Islet, Western Chain, Snares Islands, 2008–10. Failed nests are those assessed to contain fresh egg fragments. No count was made on Rima Islet in 2010.

Islet	Date	Adult + egg	Obvious failed nest	Total
Toru	6–7 October 2008	828	70	898
	2 October 2009	783	51	834
	28–29 September 2010	780	49	829
Rima	16 October 2008	279	18	297
	30 September 2009	265	17	282

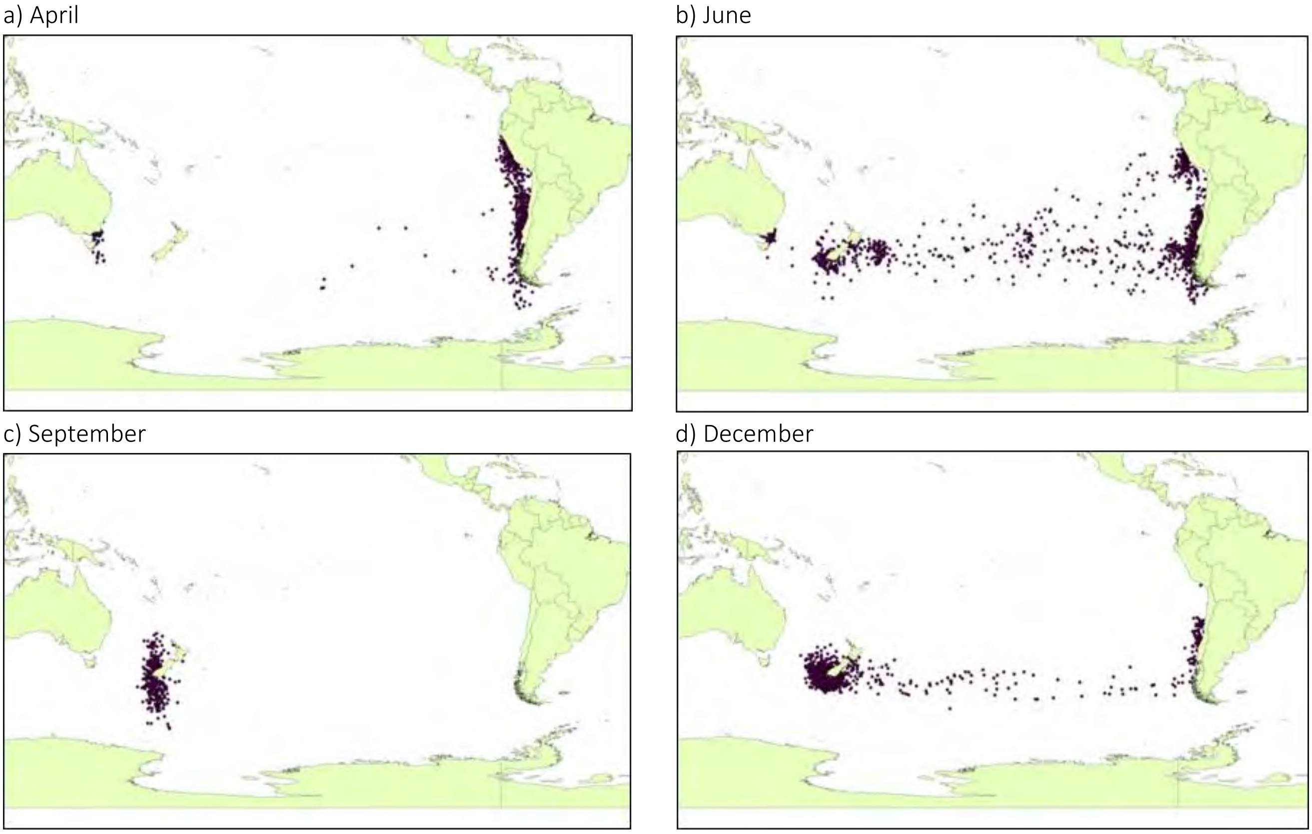


Figure 8.28: (from Sagar et al. 2011). Distribution of Salvin's albatrosses *Thalassarche salvini* from the Snares Islands Western Chain tagged with geolocators at four times of the year: a) April, after the completion of their breeding season, b) June, showing their return tracks from South American waters to New Zealand waters prior to egg laying, c) September, when their partners were incubating an egg, and d) December, the birds around New Zealand are presumed to be foraging for food for themselves and their chick, and the birds crossing the Pacific Ocean and in South American waters are presumed to be failed breeders.

8.4.3.3 WESTLAND PETREL

The Westland petrel is endemic to New Zealand and nests in burrows in dense rainforest near Punakaiki, Westland. This species is poorly studied, probably largely because the birds nest in burrows, inhabit dense forest, and attend their nests only at night. As for the flesh-footed shearwater, a survey methodology for estimating population size and assessing long-term trends for the Westland petrel was designed (Baker & Double 2007). Once a colony was located, Baker et al. (2007b, 2008a, 2011a) estimated population size through a three-stage process. First, burrow densities were determined in each colony by using 2-m wide strip 'colony transects', and burrows were mapped along each transect. These transects differed from search transects in that they were confined to identified colonies and were randomly placed within the colonies. Second, the proportion of active nests per burrow was estimated using burrow scopes and 'inspection by hand' (inserting an arm down burrows to determine occupancy and feel for eggs, chicks, adult birds, or nesting material). Finally, the area of each colony was measured by exploring the approximate boundaries on foot and mapping the densely inhabited area and this area multiplied was by the density to arrive at a population estimate for each colony.

Although Westland petrels breed throughout a 16-km^2 area near Punakaiki, which has been designated as a Special

Conservation Area, sampling effort was concentrated on estimating the population in high density areas, noting the challenges posed by the rugged terrain and often adverse weather conditions (Baker et al. 2007b, 2008a, 2011a). These authors estimated the number of potential burrows in all Westland petrel colonies to total 6846 (95% c.i.: 6389–7302) during 2007–11. Of these, an estimated 2827 (2143–3510) were occupied. The rugged terrain and inclement weather made it difficult to ensure that the permanent transects were replicated exactly each year and hence raises some doubts about the comparability of counts. Wood & Otley (2012) estimated colony and population sizes of Westland petrel between 2002 and 2005 and found between 2954 and 5137 breeding pairs, and an overall total of nearly 13 000 potential burrows.

Waugh et al. (2018) described the foraging ranges of Westland petrels during the pre-laying phase in 2011, incubation phase in 2011, 2012, 2015, and 2016, and during the chick-rearing phase in all years except 2011. Birds tended to use the same areas across years and breeding stages, with core areas off Hokitika, in Cook Strait, and to the south-west off Haast (Figure 8.29).

Landers et al. (2011) used light-based geolocation tags to describe the long-distance migration patterns of Westland petrel. Based on data from eight birds, Landers et al. (2011) found that post-breeding migration occurred in November and birds travelled east to waters off South America, covering approximately 7000 km in 6 days (range 4–7 days). The return trip to New Zealand ahead of the breeding season occurred in April, taking 10 days (range 8–13 days). Six of the eight birds remained off Chile during the non-breeding period, but the remaining two birds travelled to the east coast of South America to waters off southern Argentina (Landers et al. 2011).

8.4.3.3.1 QUANTITATIVE MODELS FOR WESTLAND PETREL

Waugh et al. (2015) modelled the population dynamics and demographic characteristics of Westland petrel using a data set spanning 1970 to 2014. They found that the population showed a slowly increasing trend from the early 1970s, underpinned by relatively high adult survival, high breeding success (60% of eggs laid produced chicks that fledged), a mean age of first return to the colony of 7.7 years, a recruitment of fledged chicks to first return of 36%, and negligible emigration. Annual adult survival for breeding birds was the same for both sexes (95.4%, 95% c.i. 91.8–97.5%) and was constant across years (Waugh et al. 2015).

8.4.3.4 FLESH-FOOTED SHEARWATER

Flesh-footed shearwaters breed around Australia and New Zealand and migrate to the northern hemisphere in the non-breeding season. In New Zealand, they nest in burrows on islands around the North Island and in Cook Strait. Of the breeding sites identified by DOC staff (G. Taylor unpublished) eight major breeding islands for the flesh-footed shearwater were chosen for re-survey: Lady Alice, West Chicken, Whatupuke, and Coppermine (Hen and Chickens Group); Green (Mercury Group), Ohinau (Ohena sub-group of Mercury Group), Karewa (Bay of Plenty), and Titi (Cook Strait). In addition, it is estimated that Middle Island (Mercury Group) held approximately 3000 pairs in 2003 (Waugh & Taylor 2012).

Baker & Double (2007) designed a survey methodology for estimating population size and assessing long-term trends for the flesh-footed shearwater. Surveys using this design were undertaken at the eight major breeding areas by Baker et al. (2008b, 2009a, 2010a, in press). Field work was focused on visiting all eight sites at least once during the five years of the study to estimate the number of pairs breeding at each site. A few sites were visited annually to estimate population trends. Baker et al. (2008b, 2009a, 2010a, 2011a) searched these sites by locating ridgelines and systematically searching from the ridgeline to the sea or, where unsuitable terrain such as a cliff was encountered, using a series of 2-m wide search transects. These transects were established by following a compass bearing downhill from the ridgeline. When potential burrows were located, the location of that burrow from the start point of the search transect was recorded, and the number of potential burrows subsequently found 1 m either side of the transect line counted. At some sites, colony transects were well marked to permit follow-up surveys in future years. The origin points for transects were randomly located along a central line or ‘backbone’, which was run through the colony. In practice, most colonies were centred on ridgelines or located on steep slopes, and the backbone was located along a ridgeline.

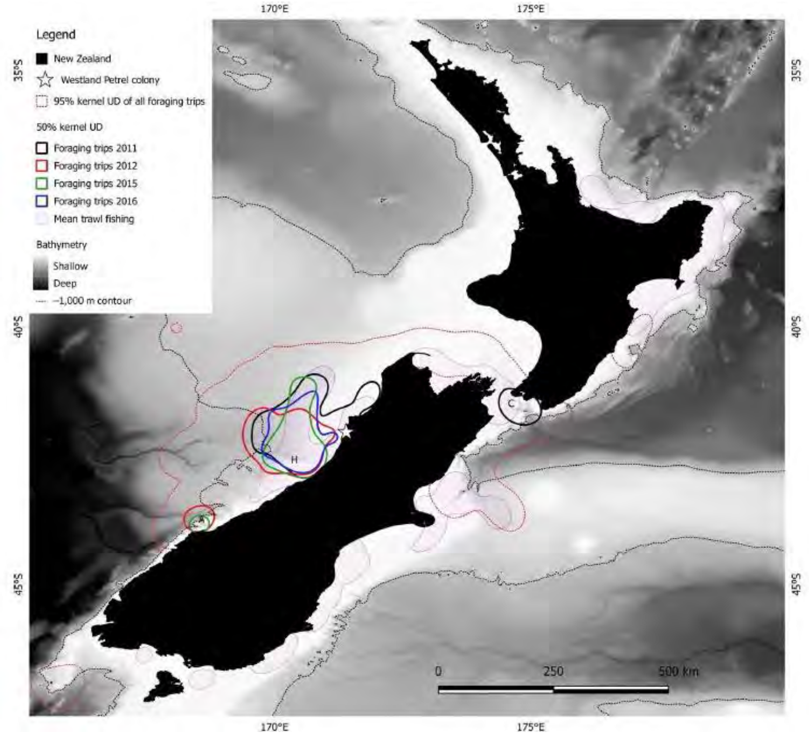


Figure 8.29: (from Waugh et al. 2018). Main foraging areas of Westland petrels, showing core foraging zones, bathymetry and the 1000 m depth contour. Areas of intensive usage are associated with steep bathymetric slopes south of the colony at Arawata and Haast canyons (A) (within the 2012 and 2015 50% kernel utilisation distributions [UD]; at Hokitika Canyon (H) in the central zone in all years; and in Cook Strait over Nicholson Canyon (C) between the North Island and South Island (2011 only).

All colony areas, with the exception of those on Karewa Island, were mapped by using transect data and a hand-held GPS. On Karewa Island, the sensitive nature of the substrate meant that sampling was curtailed to working from boards laid on the surface along a sandy track used by DOC for park management purposes. This access point was used as a long transect, with other shorter transects established either side as permitted by the terrain.

The density of potential burrows was scaled up to the estimated area of each colony to derive an estimate of the number of burrows for each colony (Table 8.23). Baker et al. (2011a) estimated the total count of burrows on the eight islands surveyed to be 20 945 (95% c.i.: 19 019–22 871), notably fewer than Taylor's (2000) estimate of 25 000–50 000 pairs. Baker et al. (2011a) stated that their estimates generally accord with the indicative population estimates developed by Graeme Taylor (cited by Baker et al. 2011) with the exception of that for Coppermine and Ohinau islands. The Baker et al. (2011a) estimate of 1425 occupied burrows (1059–1791) for Coppermine Island is much lower than Taylor's indicative estimate of 10 000 (presumably breeding pairs). In contrast, the Baker et al. (2011a) estimate of 2071 occupied burrows (943–3200) for Ohinau Island greatly exceeds Taylor's indicative estimate.

In January 2017, Bell & Boyle (2017) assessed the flesh-footed shearwater population at Middle Island in the Mercury Islands group and estimated 5822 (95% c.i. 2400–9244) breeding pairs (corrected for burrow occupancy), the largest breeding population of flesh-footed shearwaters in New Zealand. Bell & Boyle (2017) reappraised a 2003 population estimate for flesh-footed shearwater at Middle Island undertaken by Graeme Taylor. Bell & Boyle (2017) reanalysed Taylor's data and estimated 4441 breeding pairs in 2003.

Waugh et al. (2014) assessed the feasibility of gaining improved estimates of key flesh-footed shearwater population parameters and investigated the at-sea distribution of flesh-footed shearwaters. Study plots were established at Lady Alice Island/Mauimua, Titi Island, and Ohinau Island, with burrow mapping by GPS and hand-drawn maps. The occupancy of burrows and size of breeding population at each colony was assessed. Occupancy was assessed by burrow-scoping and through inspection of burrow contents through study hatches.

Table 8.23: (from Baker et al. 2011a). Estimated number of potential and occupied burrows for flesh-footed shearwater for eight New Zealand islands surveyed during 2007–08 to 2010–11. Note that some colonies on Lady Alice Island and Coppermine Island were visited in all years, and for these colonies the highest estimate was used to derive the island total. The number of occupied burrows can reasonably be considered an estimate of annual breeding pairs for each island.

Island	No. potential burrows	Lower 95% c.i.	Upper 95% c.i.	No. occupied burrows	Lower 95% c.i.	Upper 95% c.i.
West Chicken	193	-2	388	15	0	210
Lady Alice	2 763	2 079	3 447	921	237	1 605
Whatupuke	2 941	1 767	4 115	1 210	36	2 384
Coppermine	2 290	1 924	2 656	1 425	1 059	1 791
Titi	2 814	2 201	3 427	337	0	950
Green	132	82	182	74	24	124
Ohinau	3 883	2 755	5 011	2 071	943	3 200
Karewa	5 929	4 420	7 438	2 561	1 052	4 070
Total	20 945	19 019	22 871	8 614	6 689	10 540

Analysis of island-wide population survey information, collected from 2011–12 to 2013–14 compared with previous surveys conducted from 2007 to 2010 (Baker et al. 2008b, 2009a, 2010a, 2011a) indicated a probable decline for the population on Ohinau Island and stable populations on Lady Alice Island/Mauimua and Titi Island. Adult annual survival was within the range reported for other shearwaters, at 0.93 for Kauwahaia Island and 0.94 for burrow-caught birds at Lady Alice Island/Mauimua (Waugh et al. 2014, Jamieson & Waugh 2015). Crowe et al. (2017) determined an overall breeding success of 49.1% for flesh-footed shearwaters at Lady Alice Island/Mauimua and Ohinua Island for the 2016–17 breeding season. Competition for burrows from other seabirds may be an issues for flesh-footed shearwaters at some locations (G Taylor, DOC, pers. comm.).

Tracking of flesh-footed shearwaters from Ohinau and Titi islands using GPS loggers showed that birds were foraging several hundreds of kilometres from their breeding site over deep oceanic waters to the east of the New Zealand region during incubation. During the early chick-rearing period, the flesh-footed shearwaters contracted their range with a higher concentration of activity in waters near the breeding site and at zones of upwelling and relative high productivity within 400 km of the breeding site (Figure 8.30). The overlap of foraging activity with trawl, longline, and gillnet fisheries indicated highest intensity of overlap when the breeding birds were foraging close to the breeding site during early chick rearing (Waugh et al. 2014, 2016).

More recent GPS tracking work of breeding flesh-footed shearwaters at Lady Alice Island during January and February 2017, and January 2018, revealed a more northerly at-sea distribution, with the majority of foraging effort centred along the east and west coasts of the north of the North Island (Figure 8.31 and Figure 8.32, Kirk et al. 2017 and Crowe 2018, respectively).

The Department of Conservation has an ongoing project, POP2015-02, to update the population size, estimate demographic parameters, and gather at-sea distribution information for the flesh-footed shearwater (Mischler 2016).

8.4.3.5 SOUTHERN BULLER’S ALBATROSS

Southern Buller’s albatross is endemic to New Zealand and breeds at the Snares Islands (North East and Broughton islands) and the Solander Islands (main Solander and Little Solander) only. Most research effort has been focused at the Snares Islands, with study area population estimates conducted annually between 1992 and 2017 and whole-island counts conducted on five occasions (1969, 1992, 1997, 2002, and 2014). In contrast, whole-island population estimates for the Solander Islands have occurred on three occasions (1996, 2002, and 2016).

Sagar (2014) provided a summary of whole-island counts for the Snares Islands, which showed that the total breeding population, based on ground-based counts, increased from 4448 in 1969 to 7683 in 1992, 8242 in 1997, 8713 in 2002, and 8704 in 2014 (Table 8.24).

Thompson et al. (2017) provided a summary of counts of breeding southern Buller’s albatross at the Solander Islands

(Table 8.25). The total breeding population in February 2016 was estimated to be 5620 pairs, higher than the totals for 2002 (4912 pairs) and 1996 (4147 pairs). Combining the 2014 Snares population estimate of 8704 pairs (Sagar 2014) with the 2016 Solander population estimate of 5620 pairs (Thompson et al. 2017) gives an overall population estimate for southern Buller's albatross of approximately 14 300 annual breeding pairs.

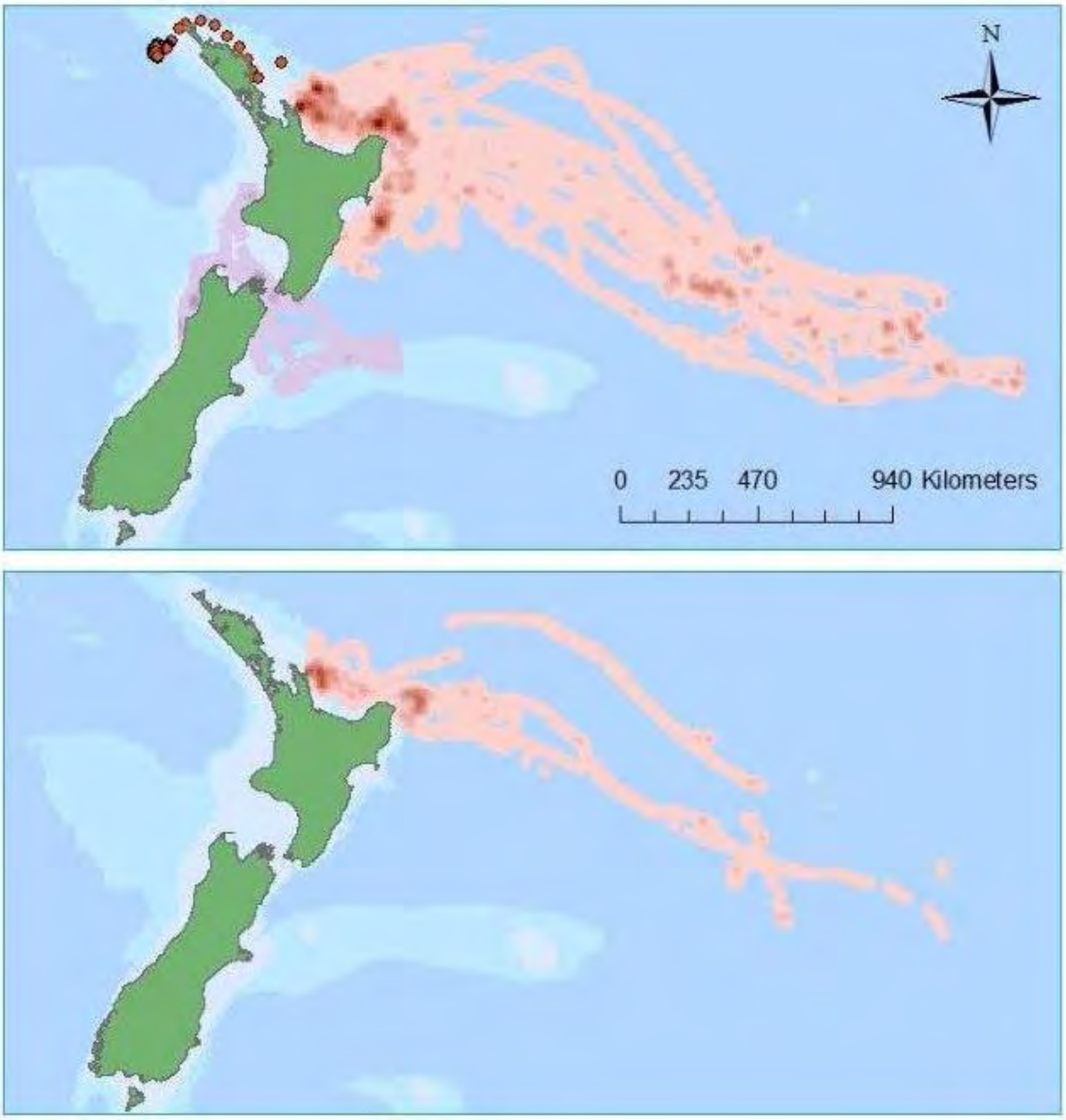


Figure 8.30: (from Waugh et al. 2014). Breeding season locations of foraging trips for flesh-footed shearwaters tracked with GPS loggers from Ohinau and Titi islands. The upper and lower figures represent kernelled density plots during the incubation period (January) and chick-rearing period (February), respectively. The dotted lines represent the 1000 m bathymetry contour.

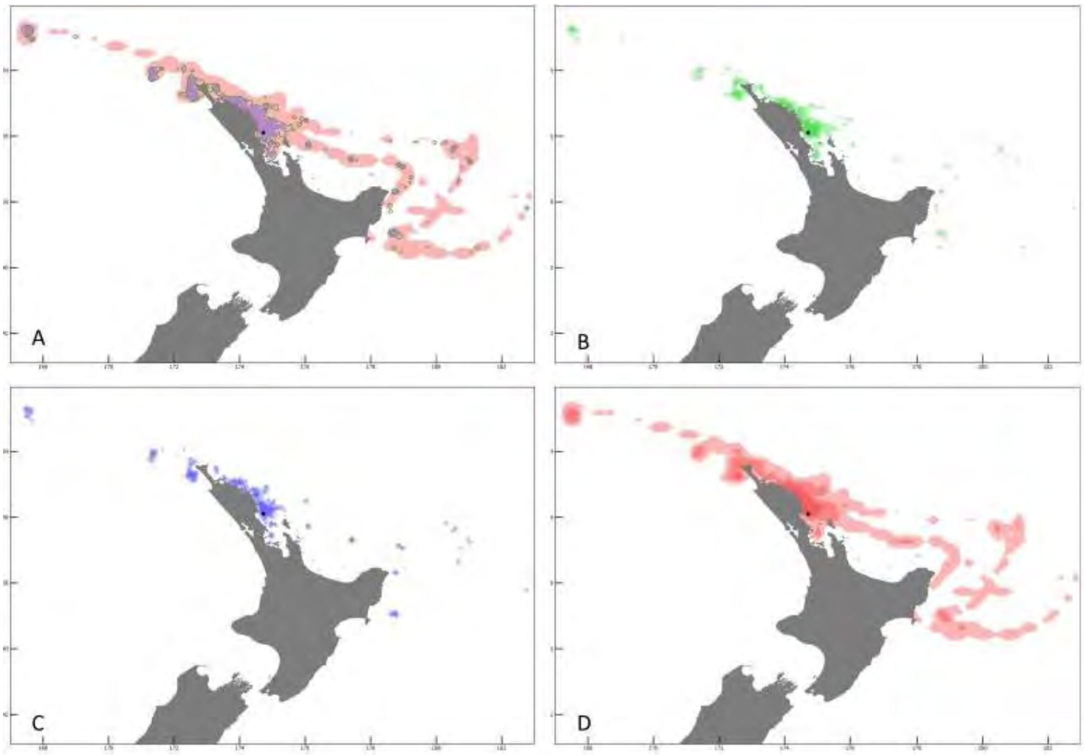


Figure 8.31: (from Kirk et al. 2017). Behavioural distributions of flesh-footed shearwaters tracked from Lady Alice Island in February 2017. A – 95% occupancy kernels for flight (red), rest (blue), and foraging (green line) behaviour. B, C, and D – Occupancy kernels for each behaviour at the 50%, 75%, and 95% levels (foraging behaviour is green, resting blue, and flight red). Location of Lady Alice Island is shown with a black dot.

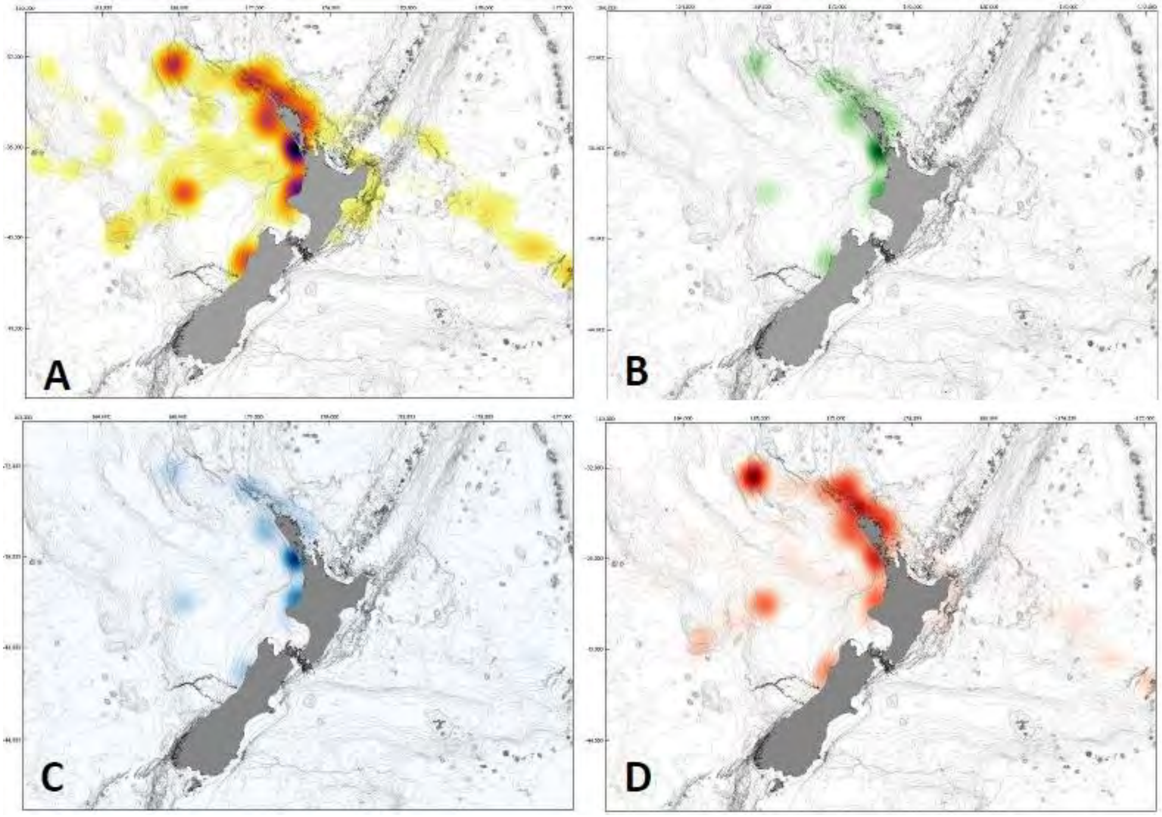


Figure 8.32: (from Crowe 2018). Heatmaps showing the behavioural distributions of flesh-footed shearwaters tracked from Lady Alice Island during the incubation period. Each behaviour type is represented by a different subset: All behaviour types combined (A), foraging (B), rest (C), and flight (D). Darker colours indicate higher occupancy.

Table 8.24: (from Sagar 2014). Numbers of occupied nests of southern Buller's albatross counted in different areas of the Snares Islands, 1969–2014. Values in parentheses assume rates of change on Broughton Island in 1969 and 2002, when no counts (NC) were made, are equal to those in the subtotal for North East Island in those years. Data for 1969–2002 are from Sagar & Stahl (2005).

Area/Year	1969	1992	1997	2002	2014
North Promontory	509	1 108	1 400	1 643	1 508
West Coast	121	262	317	205	146
North side, South-West Promontory	305	785	520	739	427
South side, South-West Promontory	763	1 236	1 410	1 025	1 201
Alert Stack	112	193	223	267	305
South Coast	1 425	2 095	2 161	2 554	2 425
East Coast	789	1 465	1 693	1 732	1 733
Total North East Island + Alert Stack	4 024	7 144	7 724	8 165	8 047
Broughton Island	NC	539	518	NC	657
Total	(4 448)	7 683	8 242	(8 713)	8 704

Table 8.25: (from Thompson et al. 2017). Numbers of breeding southern Buller's albatrosses at the Solander Islands in 2016, 2014, 2002, and 1996. Data for Little Solander Island in 2014 are from an aerial survey (Baker & Jensz 2014), those from 2002 and 1996 are from a combination of ground, vantage, and aerial counts (see Sagar & Stahl 2005).

Area/Year	2016	2014	2002	1996
Solander Island	5 280		4 579	3 885
East Bay	666		876	709
North East to North West headlands	778		1162	1 086
West Bay	819		489	387
West Bay to South West Bay	481		362	306
South West Bay to South East Peninsula	2536		1690	1 397
Little Solander Island	340	305	333	262
Totals	5 620		4 912	4 147

Stahl & Sagar (2000a), using satellite telemetry, described the movements and distributions of southern Buller's albatross from the Snares Islands in 1995–97. Birds made long trips to the Tasman Sea or the east coast of the South Island during incubation, but short trips east of the Snares Islands during the guard stage. Trip length increased again during chick-rearing. Torres et al. (2013) tracked southern Buller's albatross from the Snares Islands during the guard stage in 2008 to 2011 using GPS tags and found that generally birds travelled both to the east and west of the Snares Islands, but that in 2010 nearly all birds travelled to the east. Similarly, satellite telemetry of southern Buller's albatrosses at the Solander Islands revealed long trips to the Tasman Sea and both coasts of the South Island during incubation, with shorter trips during the guard stage (Stahl & Sagar 2000b). Males tended to travel east from the Solander Islands, whereas females tended to travel west from the Solander Islands (Stahl & Sagar 2000b). Waugh et al. (2017) used GPS tags to track breeding southern Buller's albatrosses from the Solander Islands in May 2016 and found that some birds travelled north to Cook Strait and the southern half of the North Island.

8.4.3.5.1 QUANTITATIVE MODELS FOR SOUTHERN BULLER'S ALBATROSS

Francis et al. (2008, see also Francis & Sagar 2012) assessed the status of the Snares Islands population of southern Buller's albatross. They estimated (see also Sagar & Stahl 2005) that the adult population had increased about five-fold since about 1950 (Figure 8.33) at a rate of about 2% per year and concluded from this that the risk to the viability of this population posed by fisheries had been small. This conclusion depends critically on the reliability of the first census of nesting birds conducted in 1969, but Francis et al. (2012) gave compelling reasons to trust that information. In summary, the later censuses did not find any concentrations of nests that were not present on the maps prepared during the 1969 census and the increase in counts after 1969 occurred in all census subareas and in five colonies where counts were made in many non-census years.

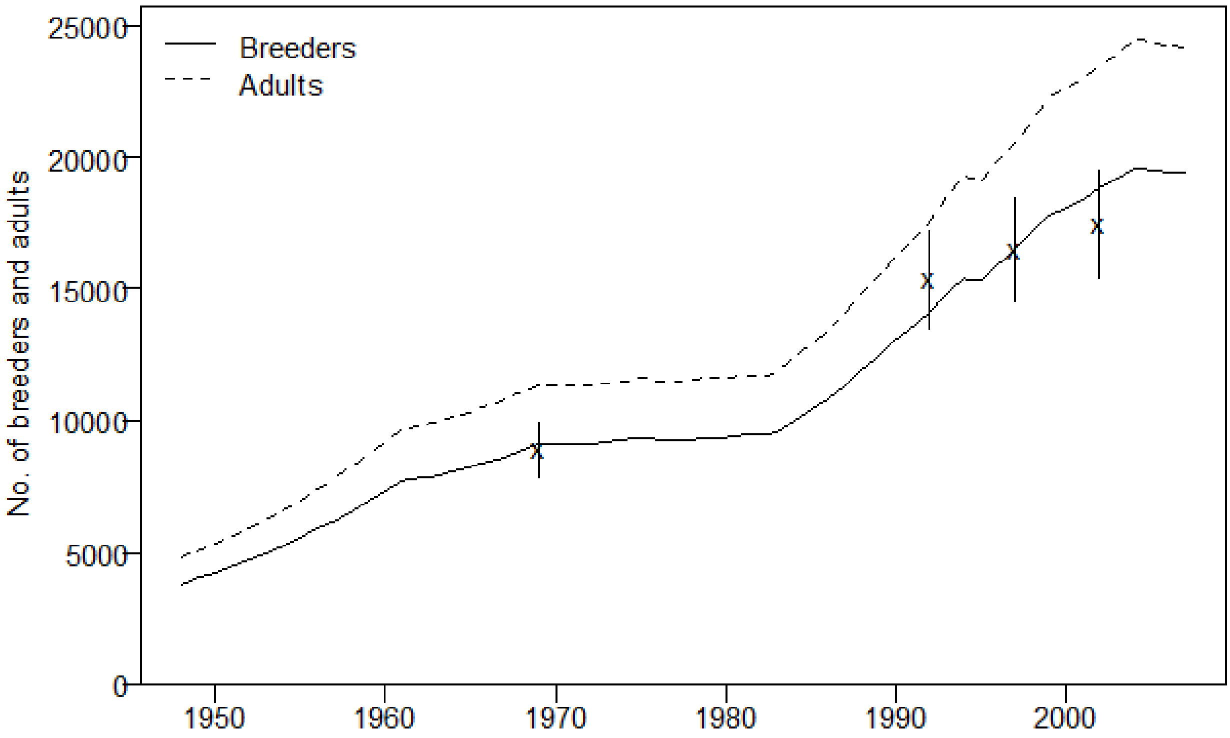


Figure 8.33: (from Francis et al. 2008). Estimates from model SBA21 of numbers of breeders (solid line) and adults (broken line) of southern Buller's albatross in each year. Also shown are the census observations (after Sagar & Stahl 2005) of numbers of breeders (crosses), with assumed 95% confidence intervals (vertical lines).

Model SBA3 estimated no increase in the size of the breeding population between 2002 and 2014, and therefore concurred with Francis et al. (2008) that population growth may have stopped. The adult survival rate is likely to have declined since 1990, but for the most recent years (2008 onwards) has increased slightly. Since adult survival was the only year-varying demographic parameter, a decline in adult survival was the only possible demographic explanation for the changing population trajectory inferred by the latest census. The estimates of breeding rate (0.83 and 0.57 for breeders and non-breeders, respectively) and probability of breeding success (0.73) are similar to those of Francis et al. (2008). Forward projection assuming current demographic rates suggested that the population is likely to increase by 5.7% in the next 10 years.

There was considerable variability in some of the key demographic rates (e.g., breeding rate), and it is not known if this was due to noise in the data or natural variation in demographic processes. The ability to quantify these variabilities was hindered by a large reduction in resighting effort since 2006. Changes in resighting methods after 2006 precluded meaningful year-varying estimates of breeding parameters in recent years and so their potential effect on changing population was not properly investigated. Also, since 2008, there have been changes in the monitoring of breeding status and the cessation of monitoring of breeding success, precluding an assessment of temporal variation in reproductive rates. Also, the numbers of non-breeders may have been overstated from the mark-recapture field study during this period. These are likely to have biased the estimates of some of the demographic rates.

Fishery discards are an important component of the diet of chicks (James & Stahl 2000), but Francis et al. (2008) were not able to assess whether the associated positive effect on population growth (e.g., from increased breeding success) is greater or less than the negative effect of fishing-related mortality.

8.4.3.6 GIBSON'S ALBATROSS

Gibson's albatross is confined as a breeding taxon to the Auckland Islands: approximately 95% of the population breeds on Adams Island, with the remainder on Disappointment Island and main Auckland Island. The population on Adams Island has been the subject of research since 1991 (Walker & Elliott 2002), and the most recent report (Walker et al. 2017) provided summary data spanning most of the period 1991 to 2017.

Gibson's albatross nest numbers on Adams Island (Table 8.26) reveal a marked reduction in 2005 with only a slow increase in nest numbers subsequently.

Nesting success also declined sharply from approximately 50% to 75% from 1991 to 2004 to approximately 25% in 2006 and increased to 68% in 2016, the highest level recorded for 20 years (Figure 8.34, Walker et al. 2017). Estimates of sex-specific adult survival also showed marked reductions, especially so for females, over the period 2006–2008 (Figure 8.35, Walker et al. 2017).

Walker et al. (2017) also provided a summary of distribution data for Gibson's albatross, covering the periods 1996 to 2004 (satellite telemetry data acquired from 57 birds, also summarised by Walker & Elliott 2006) and from 2009 (light-based geolocation data from 77 birds). Walker et al. (2017) concluded that the range of breeding birds had changed relatively little between the two periods but noted that distributions have extended further to the north post 2004 (Figure 8.36).

For non-breeding birds, Walker et al. (2017) reported that the ranges of both sexes had increased between the two periods, with tracked birds from 2009 now utilising the South Australian Bight, a pattern that was particularly evident for females (Figure 8.37).

8.4.3.6.1 QUANTITATIVE MODELS FOR GIBSON'S ALBATROSS

Francis et al. (2015) concluded that there is cause for concern about the status of the population of Gibson's albatross on the Auckland Islands. Since 2005, the adult population has been declining at 5.7% per year (95% c.i.: 4.5–6.9%) because of sudden and substantial reductions in adult survival, the proportion of adults breeding, and the proportion of breeding attempts that are successful (Figure 8.38). Forward projections showed that the most important of these to the future status of this population is adult survival (Figure 8.39).

The population in 2011 was 64% (58–73%) of its estimated size in 1991. The breeding population dropped sharply in 2005, to 59% of its 1991 level, but has been increasing since 2005 at 4.2% per year (2.3–6.1%). The 2011 breeding population is estimated to be only 54% of the average of 5831 pairs estimated by Walker & Elliott (1999) for 1991–97.

Francis et al. (2015) found it difficult to assess the effect of fisheries mortality on the viability of this population because, although some information exists about captures in New Zealand and Australian waters, the effect of fisheries in international waters is unknown. Three conclusions are possible from the available data: most fisheries mortality of Gibson's is caused by surface longlines; mortality from fishing within the New Zealand EEZ is now probably lower than it was; and there is no indication that the sudden and substantial drops in adult survival, the proportion breeding, and breeding success were caused primarily by fishing.

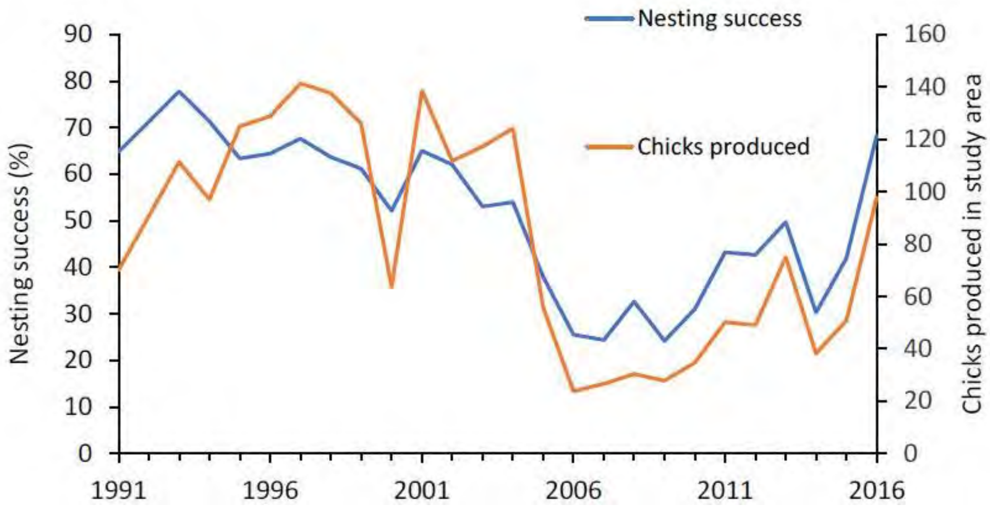


Figure 8.34: (from Walker et al. 2017). Nesting success and the number of chicks fledged from the study area on Adams Island.

Table 8.26: (from Walker et al. 2017). The number of Gibson's wandering albatross nests in late January in three census blocks on Adams Island during 1998–2017. Corrected total is the estimated number of nests in the three blocks taking account of the number of failed and un-laid nests at the time of counting. Estimated total population is the estimated number of nests on the island, based on the number of nests in the three counted blocks in 1997 when the last whole island count was undertaken.

Year	Rhys' Ridge (low density)	Amherst-Astrolabe (medium density)	Fly Square (high density)	Total no. of nests	Corrected total	Estimated total population
1998	60	483	248	781	798	7 875
1999	60	446	237	743	746	7 367
2000	45	284	159	488	497	4 904
2001	64	410	201	675	706	6 969
2002	60	408	246	675	740	7 303
2003	71	496	217	784	791	7 809
2004	77	501	284	862	884	8 728
2005	34	323	72	412	452	4 467
2006	15	185	79	279	341	3 371
2007	38	230	132	400	430	4 245
2008	26	201	91	318	341	3 371
2009	28	238	120	386	426	4 211
2010	32	237	114	383	392	3 872
2011	33	255	137	425	438	4 323
2012	35	224	120	379	418	4 131
2013	39	315	138	492	519	5 120
2014	29	267	134	430	473	4 669
2015	39	237	105	381	406	4 010
2016	34	332	153	519	545	5 385
2017	32	252	140	424	448	4 423

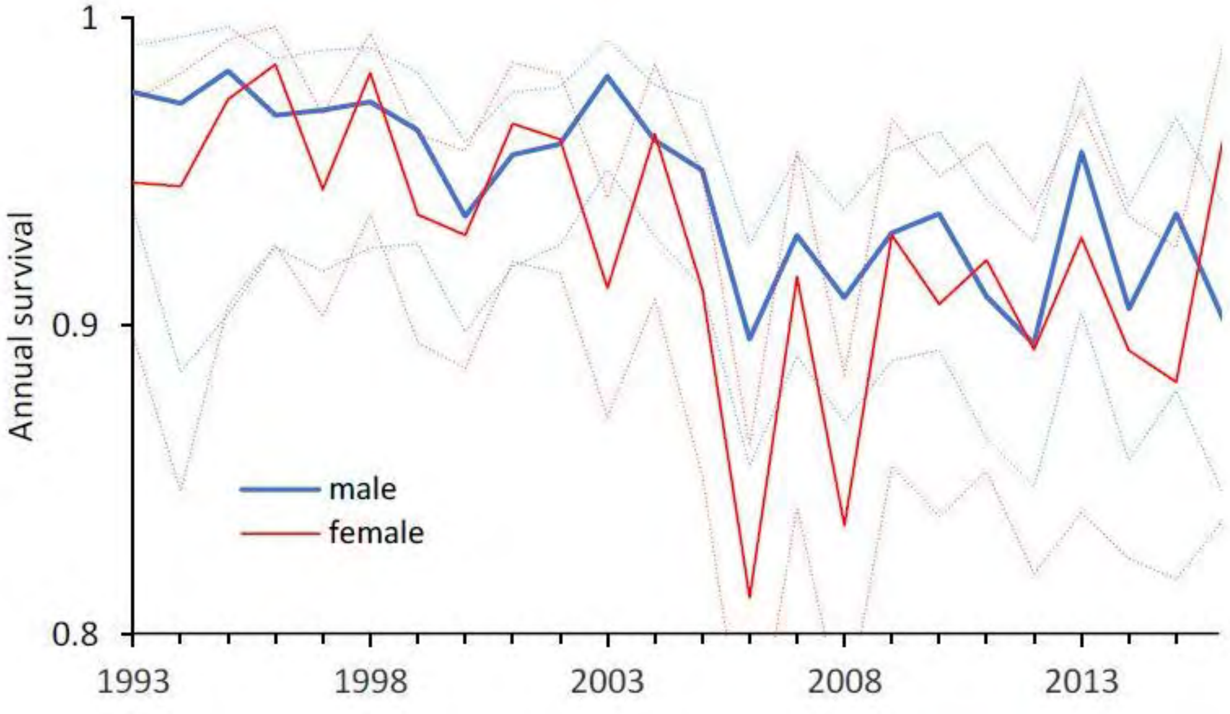


Figure 8.35: (from Walker et al. 2017). Annual survival of birds in the study area on Adams Island estimated by mark-recapture. The dotted lines are 95% confidence intervals.

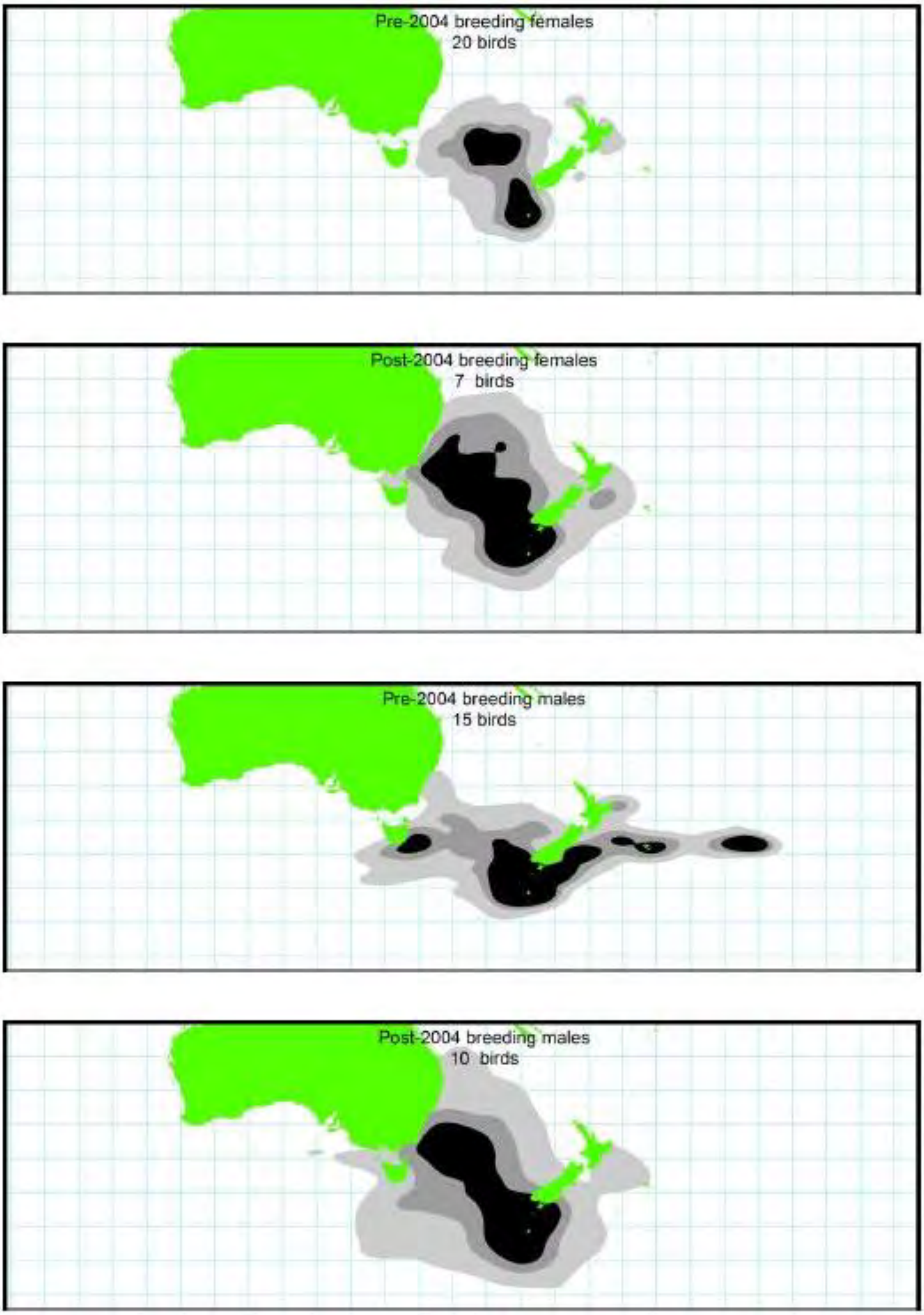


Figure 8.36: (taken from Walker et al. 2017). Kernel density plots of breeding Gibson's wandering albatrosses tracked in 1996–2004 and in 2009–17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

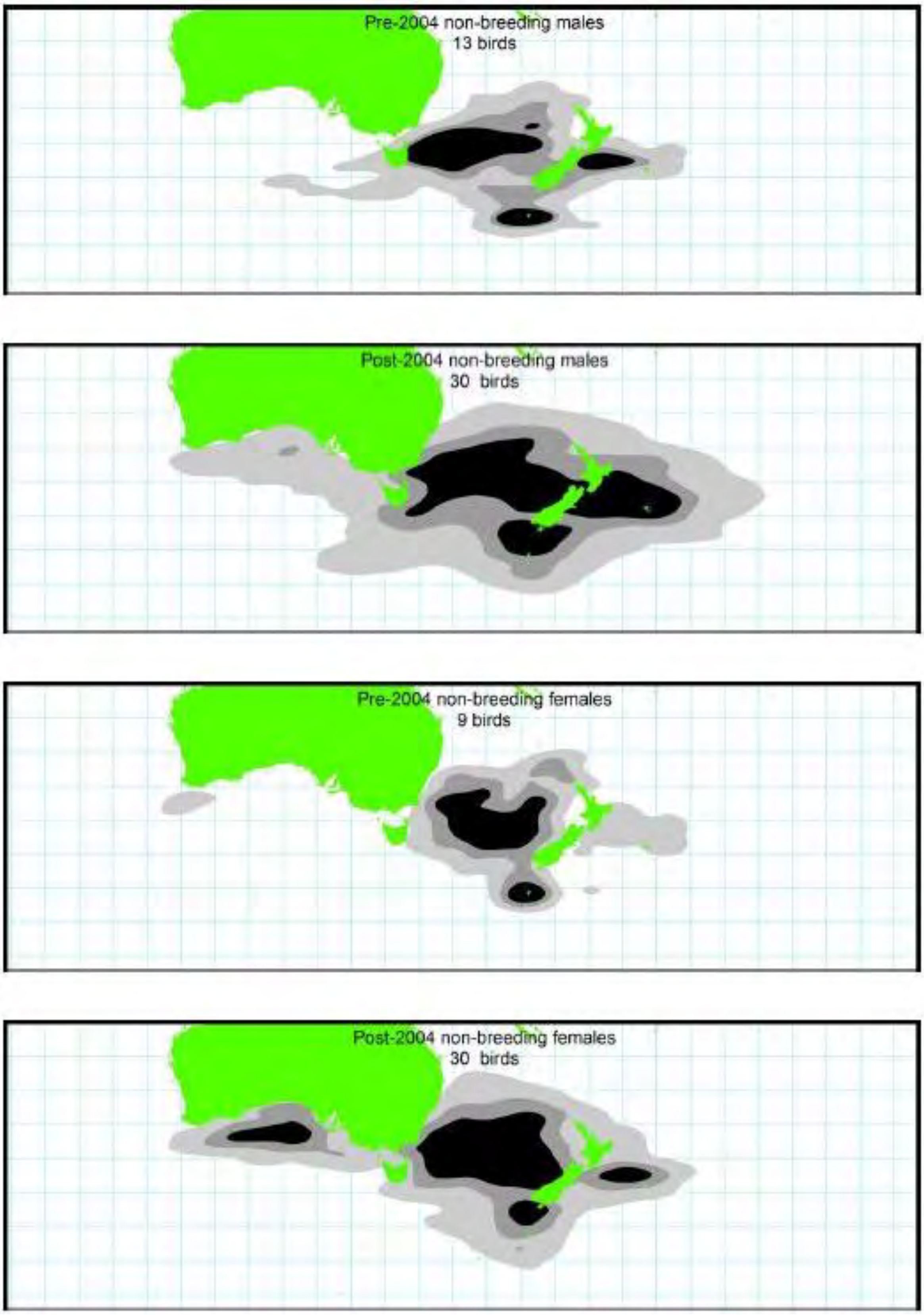


Figure 8.37: (taken from Walker et al. 2017). Kernel density plots of non-breeding Gibson's wandering albatrosses tracked in 1996–2004 and in 2009–17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

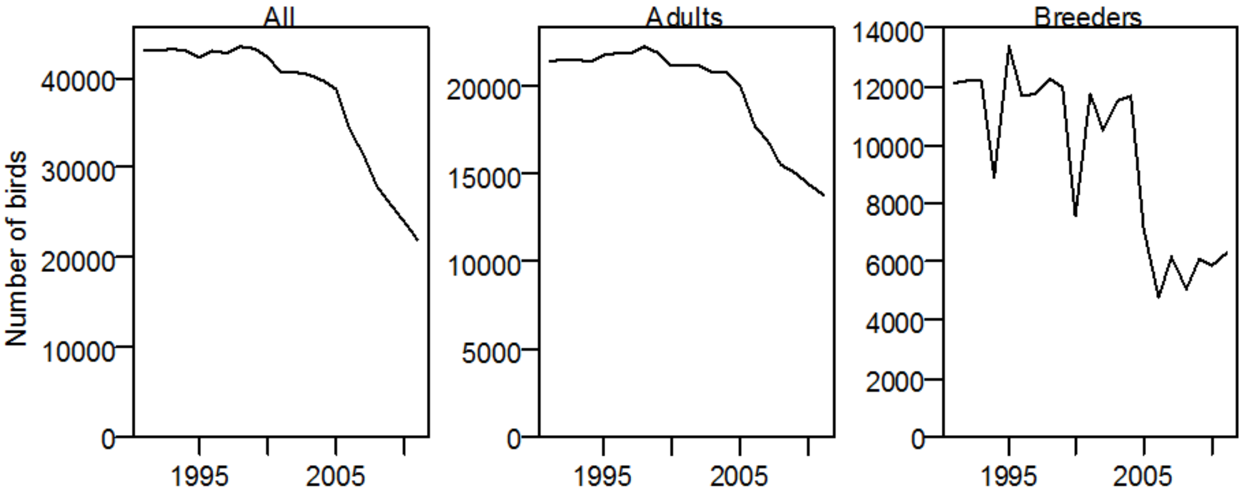


Figure 8.38: Estimated population trajectories for the whole Auckland Islands population of Gibson's wandering albatross. These were calculated by scaling up the Francis et al. (2015) GIB5 trajectories to match the Walker & Elliott (1999) estimate for the whole population.

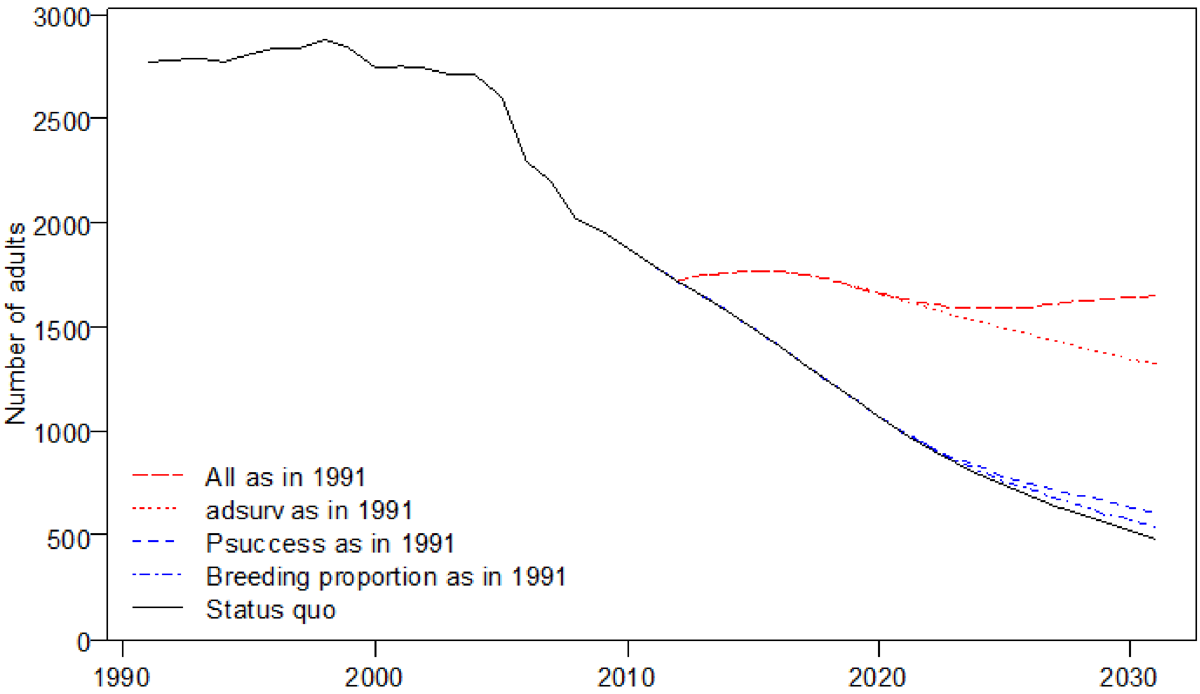


Figure 8.39: Estimated population trajectory for Gibson's albatross adults from the Francis et al. (2015) model GIB5 with 20-year projections under five alternative scenarios about three demographic parameters: adult survival (adsurv), breeding success (Psuccess), and proportion of adults breeding. These scenarios differ according to whether each parameter remains at its status quo (i.e., 2011) level or recovers immediately to its 1991 level.

8.4.3.7 CHATHAM ISLAND ALBATROSS

The Chatham Island albatross breeds only at The Pyramid, a small southern islet in the Chatham Island group (note that a translocation project began in early 2014 transferring chicks to the main Chatham Island in an attempt to establish a second breeding site). To index the population size of the Chatham Islands albatross, nest counts are conducted on The Pyramid. The islet is divided into 19 areas and, within each, every accessible nest site is counted and its status recorded (Scofield et al. 2008a, Fraser et al. 2009b, 2010b).

Nest counts have been conducted when the birds are in the early stages of chick rearing. The total number of Chatham Island albatross nest sites counted in the most recent trip was 5296 (Bell et al. 2017). This result compared closely with previous counts (which have ranged from 5194 to 5407 in late November and early December (Fraser et al. 2011, Table 8.27) indicating a relatively stable number of occupied nests on The Pyramid. The average number of nest sites from 1999–2016 was 5294 (range 5194–5407) (Bell et al. 2017). Chatham Island albatrosses have been banded on The Pyramid since 1974 and, at each visit, the recaptures have added to the growing number of known-aged birds. This banding record enables an assessment of annual adult mortality. A total of 304 banded Chatham Island albatross were recaptured between 19 November and 2 December 2010 on The Pyramid and a further 50 new Chatham Island albatross were banded during the 2010 trip (Fraser et al. 2011).

To determine foraging movements and behaviour of Chatham Island albatross during the incubation and early chick rearing stages of the breeding season, GPS loggers were applied to breeding birds for the duration of one foraging trip. Where possible, birds were also tagged with a geolocator logger to record activity (i.e., salt-water immersion) during foraging trips. The resulting distributional range of Chatham Islands albatross during incubation and early chick rearing from these tracking studies from November to December 2007–09 is shown in Figure 8.40 (Fraser et al. 2010b). Deppe et al. (2014) provided a year-by-year analysis of these tracking data.

To track the birds on a longer timescale during the non-breeding season, geolocation loggers (GLS) were used. These devices have a life span of up to about six years and are intended to remain on the birds for at least one year. They were applied to each banded bird's leg using a plastic band to which the loggers were attached with glue and a cable tie.

MPI has a current project to undertake population specific modelling of adult survival of the Chatham Island albatross (PRO2017-05B).

Table 8.27: (from Fraser et al. 2011 and Bell et al. 2017). Counts of Chatham Island albatross nest sites for the years: 2007 (19–29 November); 2008 (22 November–7 December); 2009 (9–12 December); 2010 (24–30 December); and 2016 (9-14 November).

	2007	2008	2009	2010	2016
Total nests counted	5 247	5 407	5 194	5 245	5 296

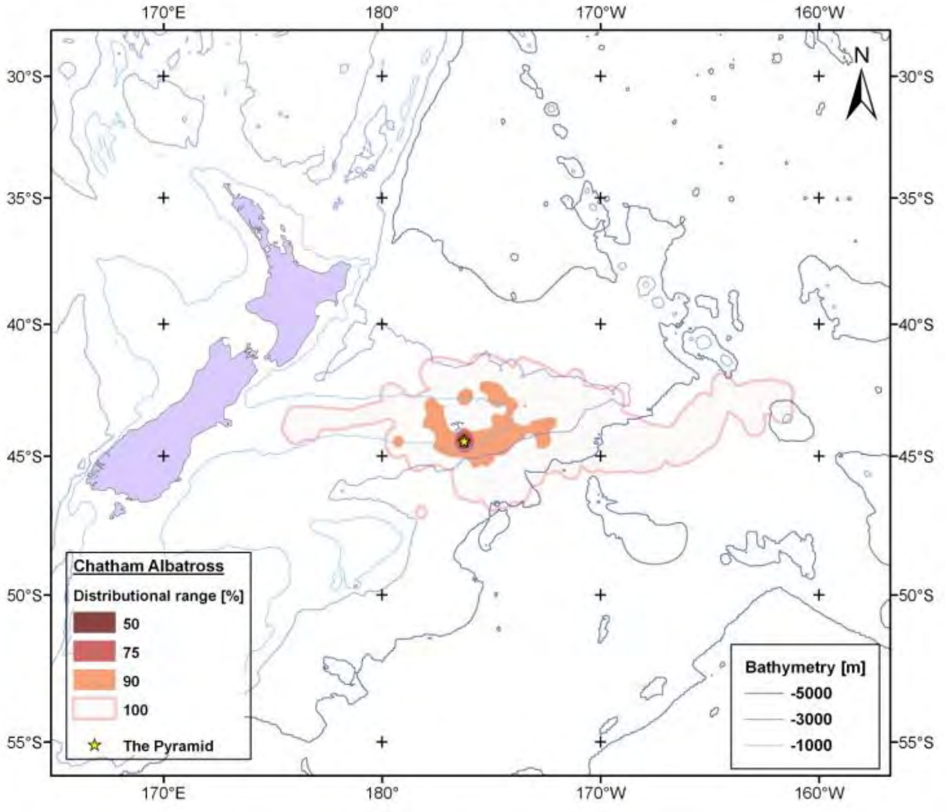


Figure 8.40: (from Fraser et al. 2010b). Distributional range of Chatham Island albatross during incubation and early chick rearing as derived from tracking studies in November–December 2007–09 (n=51 tracks).

8.4.3.8 WHITE-CAPPED ALBATROSS

Repeated population censuses of the white-capped albatrosses breeding in the Auckland Islands were conducted in the month of December between 2006 and 2010, and the month of January in 2012 and 2013, using aerial photography (Baker et al. 2007b, 2008a, 2009b, 2011b, 2013). These population censuses were carried out to estimate population size and track population trends. Photo montages were created from the aerial photography and counted by an observer. Counts of photo montages in all years except 2006 were undertaken by one observer only. Multiple counts of photo montages from the December 2006 census were undertaken to estimate counter variability associated with miscounting and misidentifying white spots on the ground as birds. Ground-truthing was conducted to determine the number of birds sitting or standing on nests, the number of pairs (partners accompanying an incubating bird), and the number of loafers present in the colony.

2006–10 census. In 2010, the total count of nesting white-capped albatrosses was estimated to be 72 635 (95% c.i.: 72 096–73 174), 4370 (4238–4502) and 117 (95–139) annual breeding pairs at Disappointment Island, South West Cape, and Adams Island, respectively, giving a total for these sites of 77 122 (76 567–77 677) breeding pairs (Table 8.28). The counts of nesting white-capped albatross over the previous four years were significantly lower than the counts taken in 2006, when a total of 117 197 breeding pairs were present at the Auckland Islands. These differences in counts may represent normal inter-annual variation in breeding rather than indicating a decline in numbers due to fisheries mortalities (Baker et al. 2011b).

2011–16 census. Surveys suggested 99 776 breeding pairs in 2011, 118 098 in 2012, 95 278 in 2013, 101 798 in 2014, 96 623 in 2015, and 91 117 in 2016. However, evidence from a series of close-up photographs taken each year over the entire series indicates that the number of non-breeding birds present in the colonies differed somewhat between December and January. The proportion was very low in December counts (1–2% of birds present) compared with 7% and 15% for the January counts taken in 2012 and 2013, respectively. Estimated annual counts for all three breeding sites in the Auckland Islands were adjusted to account for the presence of non-breeding birds (Table 8.28). These adjusted figures were used as inputs into models used for assessment of population trend. The population size estimates computed from a TRIM model indicate an average growth rate of -1.73% per year (λ = 0.9827 ± 0.001); assessed by TRIM as moderate decline. However, a simple linear trend analysis, as performed by TRIM is not well suited to a dataset with high inter-annual variability. Trend analysis using regression splines is more appropriate to such datasets and showed no evidence for systematic monotonic decline over the nine years of the study, therefore providing support to the null hypotheses of no trend (stability) in the total population. Full details are provided by Baker et al. (2013, 2014b, 2015a, in press).

Further aerial surveys were conducted in 2016 and 2017 (Baker & Jensz 2017). The resulting aerial photographs are being analysed under MPI project SEA2016-29.

A marked population of breeding adult white-capped albatross was established at Disappointment Island during 2015–18 to estimate their demographic parameters in the long term, including adult survival (Parker et al. 2016, Rexer-Huber et al. 2018). To date, a total of 521 breeding white-capped albatrosses have been banded in the four annual visits to Disappointment Island.

Torres et al. (2011) tracked breeding white-capped albatrosses at South West Cape (main Auckland Island) during the guard stage using GPS tags and found that foraging effort was concentrated at an area to the east of the Auckland Islands over the edge of the shelf (approximately 250 m water depth).

Table 8.28: (after Baker et al. 2013, 2014b, 2015a, in press). Aerial-photographic counts of breeding pairs of white-capped albatrosses on three islands in the Auckland Islands group in December 2006–14. (Continued on next page)

Year	Adams	Disappointment	SW Cape	Total	95% limits	Adjusted for loafers
2006	–	110 649	6 548	117 197	116 570–117 823	116 025
2007	79	86 080	4 786	90 945	90 342–91 548	90 036
2008	131	91 694	5 264	97 089	96 466–97 712	96 118
2009	132	70 569	4 161	74 862	74 315–75 409	73 838

Year	Adams	Disappointment	SW Cape	Total	95% limits	Adjusted for loafers
2010	117	72 635	4 370	77 122	76 567–77 677	76 119
2011	178	93 752	5 846	99 776	99 144–100 408	92 692
2012	215	111 312	6 571	118 098	117 411–118 785	102 273
2013	184	89 552	5 542	95 278	94 661–95 895	74 031
2014	193	96 864	4 741	101 798	101 160–102 436	95 894
2015	176	91 554	4 893	96 623	90 949-92 159	87 057
2016	171	85 510	5 436	91 117	84 925-86 095	82 005

8.4.3.8.1 QUANTITATIVE MODELS FOR WHITE-CAPPED ALBATROSS

Francis (2012) described quantitative models for white-capped albatross, New Zealand’s most numerous breeding albatross, and the most frequently captured, focusing on the population breeding at the Auckland Islands. After a correction for a probable bias introduced by sampling at different times of day in one of the surveys, aerial photographic counts by Baker et al. (2007b, 2008b, 2009b, and 2010b) suggested that the adult population declined at about 9.8% per year between 2006 and 2009. However, this estimate is imprecise and is not easily reconciled with the high adult survival rate (0.96) estimated from mark-recapture data. Francis (2012) also compared the trend with his estimate of the global fishing-related fatalities of white-capped albatross (slightly over 17 000 birds per year, about 30% of which is taken in New Zealand fisheries) and found that fishing-related fatalities were insufficient to account for the number of deaths implied by a decline of 9.8% per year (roughly 22 000 birds per year over the study period). The scarcity of information on cryptic mortality makes these estimates and conclusions uncertain, however. Since this modelling was conducted, further counts of white-capped albatross have been conducted (Figure 8.41 compiled from data presented by Baker et al. in press), which showed considerable annual variation. These authors consider that the substantial year to year variation in counts is real, that trend analyses appropriate in this situation support the null hypothesis of no trend in the population, and that the trend should be considered uncertain.

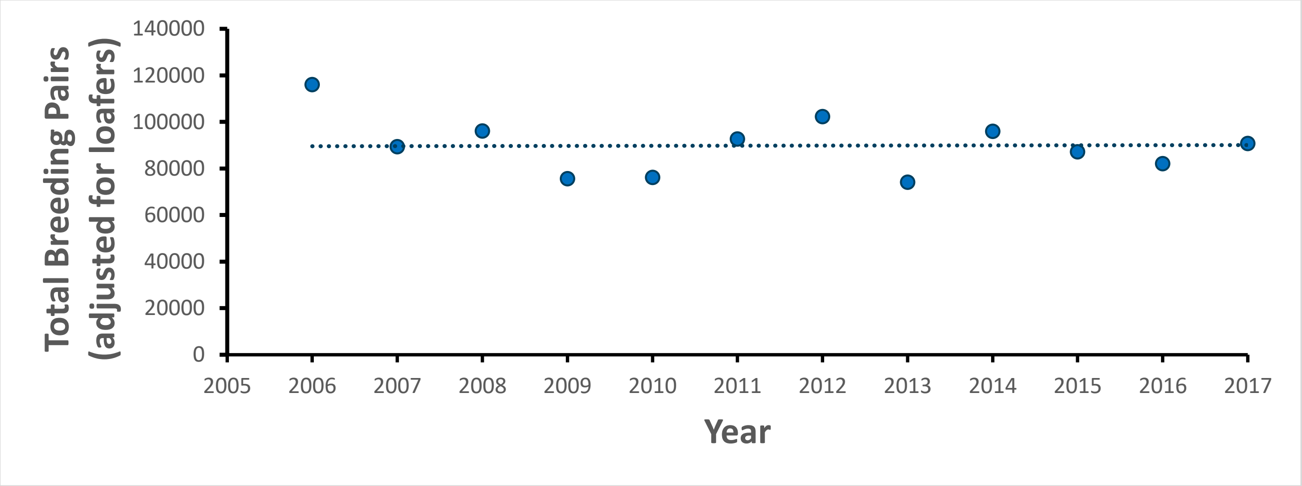


Figure 8.41: (Data from Baker et al. unpublished progress report). Total counts of white-capped albatross at the Auckland Islands (as adjusted for the presence of non-breeding birds). Line represents the long term average.

8.4.3.9 NORTHERN BULLER’S ALBATROSS

The Forty-Fours, a small group of islands located about 35 km east of Chatham Island, are home to the main breeding population of northern Buller’s albatross *Thalassarche bulleri platei*. The northern Buller’s albatross nest estimate on the Forty-Fours for 2007 was 15 238 (Scofield et al. 2008b), for 2008 was 14 674 (Fraser et al. 2009a), and for 2009 was 14 185 (Fraser et al. 2010a). Fixed grids sampled each year also confirmed the consistent population count (Fraser et al. 2010a).

In 2016, aerial photography was compared with ground counts to determine the population size of northern Buller's albatross on the Forty-Fours and The Sisters. The aerial survey estimated a total of 17 969 breeding pairs (2646 pairs at The Sisters and 15 322 pairs at the Forty-Fours) after correction with aerial close-ups, and 16 138 breeding pairs (2366 pairs at the Sisters and 13 771 pairs at the Forty-Fours) after correction using ground counts (Baker et al. 2017). A ground survey of the Forty-Fours in December 2016 updated the population size for northern Buller's albatross with an estimate of 17 682 nest sites, although this total included empty nests and nests for which contents were not determined. The total for nests with either an egg or with signs of breeding was 16 492 (Bell et al. 2017).

In November–December 2017 a full census of northern Buller's albatross at The Sisters revealed a total of 3158 nests with evidence of breeding (Bell et al. 2018). The overall Chatham Islands population of northern Buller's albatross is close to 20 000 pairs (Bell et al. 2018).

Additionally, a small population of northern Buller's albatross breeds at Rosemary Rock within the Three Kings Islands group. Frost et al. (2018) provided a summary of all counts for this population and estimated 34 active nests in November 2017 using aerial photography.

8.4.3.10 ANTIPODEAN ALBATROSS

Antipodean albatross is an endemic taxon breeding primarily at Antipodes Island, with a very small number of pairs breeding at Campbell Island and the Chatham Islands. Monitoring of the Antipodes Island population began in 1994 and has been continued on a near-annual basis since inception. The most recent report of this work (Elliott & Walker 2019) provided summary information on the Antipodes Island population spanning 1994 to 2019 for most statistics.

The Antipodean albatross population at Antipodes Island mirrored the trajectory of Gibson's albatross at Adams Island in that following a period of population growth up to 2004 the population declined markedly from 2005, and, although the rate of decline has slowed, the population in 2017 was at the lowest level since the study began, after there was a slight increase to 2019 (Table 8.29, Figure 8.42).

Adult survival was generally high (approximately 0.96) and consistent between sexes up to 2004, from which point survival becomes highly variable and has declined to 0.90-0.95 for males and to approximately 0.81-0.92 for females (Figure 8.43, Elliott & Walker 2019). Figure 8.44 shows the decrease in the density of nests in the study site between 2004 and 2017 (Elliott & Walker 2019).

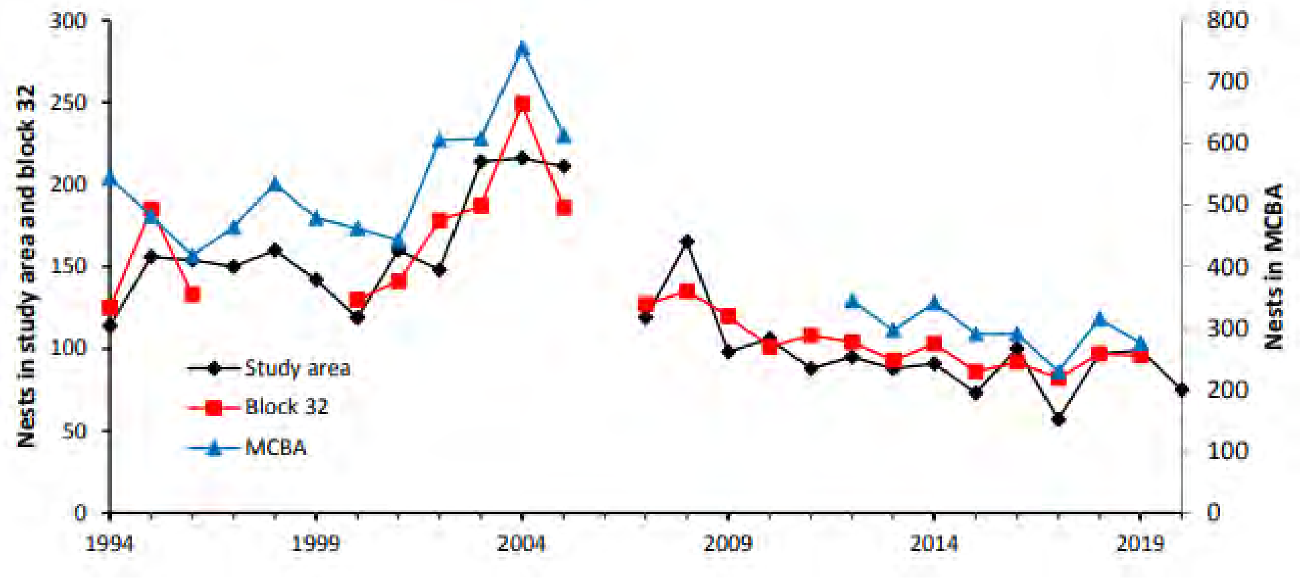


Figure 8.42: (taken from Elliott & Walker 2020). The number of Antipodean wandering albatross nests in three blocks on Antipodes Island since 1994. MCBA = Marked Census Block A.

Table 8.29: (from Elliott & Walker 2017). Antipodean wandering albatross nests with eggs in February in three areas on Antipodes Island between 1994 and 2017. MCBA = Marked Census Block A.

Year	Study area	Block 32	Subtotal	MCBA	Total	Estimated nests on island
1994	114	125	239	544*	783	5233
1995	156	185	341	482*	823	5500
1996	154	133	287	418*	705	4712
1997	150	–	–	464*	–	5463
1998	160	–	–	534	–	5827
1999	142	–	–	479	–	5172
2000	119	130	249	462	711	4752
2001	160	141	301	443	744	4972
2002	148	178	326	605	931	6222
2003	214	187	401	608	1009	6743
2004	216	249	465	755	1220	8153
2005	211	186	397	613	1010	6750
2006	–	–	–	–	–	–
2007	119	127	246	–	–	4368
2008	165	135	300	–	–	5327
2009	98	120	218	–	–	3871
2010	106	101	207	–	–	3676
2011	88	108	196	–	–	3480
2012	95	104	199	345	543	3629
2013	88	93	181	297	478	3195
2014	91	103	194	341	535	3576
2015	73	86	159	291	450	3007
2016	100	92	192	291	483	3228
2017	57	82	139	230	369	2466
2018	97	97	194	315	509	3402
2019	99	96	195	276	471	3148

* estimated (see Walker & Elliott 1999).

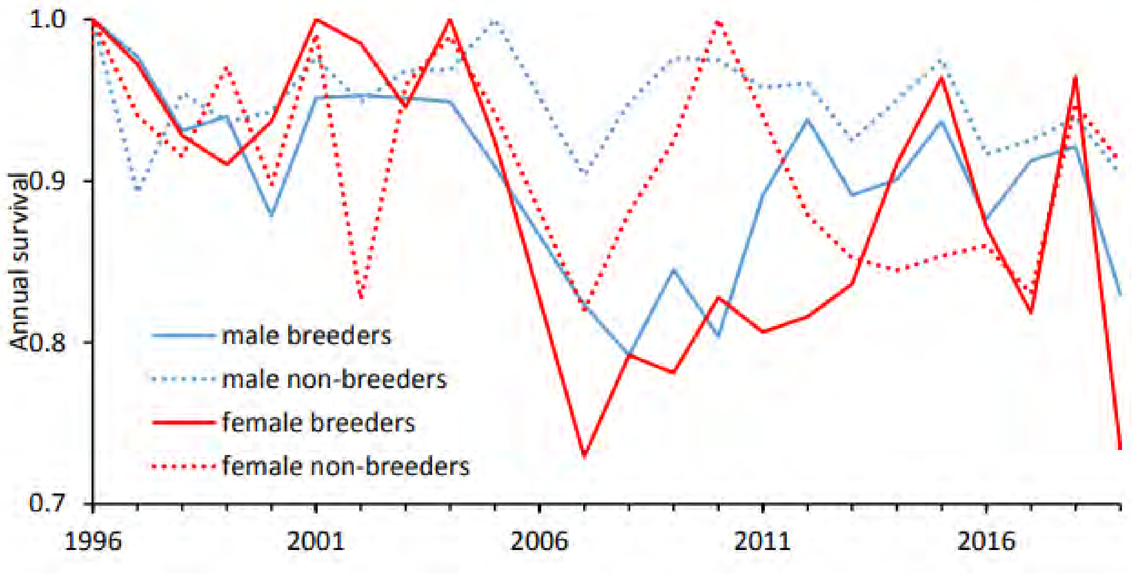


Figure 8.43: (from Elliott & Walker 2019). Estimated annual survival of Antipodean wandering albatross on Antipodes Island since 1994. Note that because the island wasn't visited in 2006, survival estimates for 2006 and 2007 were estimated from the survival over a 2-year period and then equally apportioned amongst the two years.

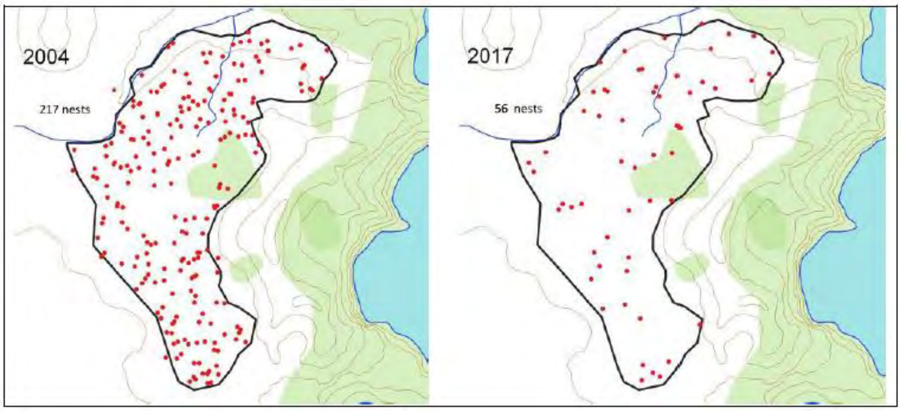


Figure 8.44: (from Elliott & Walker 2019). Location of wandering albatross nests (red dots) in the study area on Antipodes Island in February 2004, just before the population crashed, and in February 2017 when the lowest number of nests so far was recorded.

The at-sea distribution of Antipodean albatross has also shifted over the course of the study. A total of 65 birds were tracked using satellite telemetry between 1996 and 2004 (Walker & Elliott 2006) and the data from this earlier period have been compared with light-based geolocation data from 50 birds tracked from 2011 to 2017. For breeding birds, females have expanded their range to the north post-2004, whereas males appear to range in all directions post-2004 (Figure 8.45). For non-breeding Antipodean albatrosses, the differences in distribution prior to 2004 and post-2004 were more marked. Post-2004 females travelled to waters off South America, a strategy that was not apparent before 2004, and males post-2004 utilised an area in the mid-Pacific Ocean and a much larger area off South America compared with birds before 2004 (Figure 8.46, Elliott & Walker 2017).

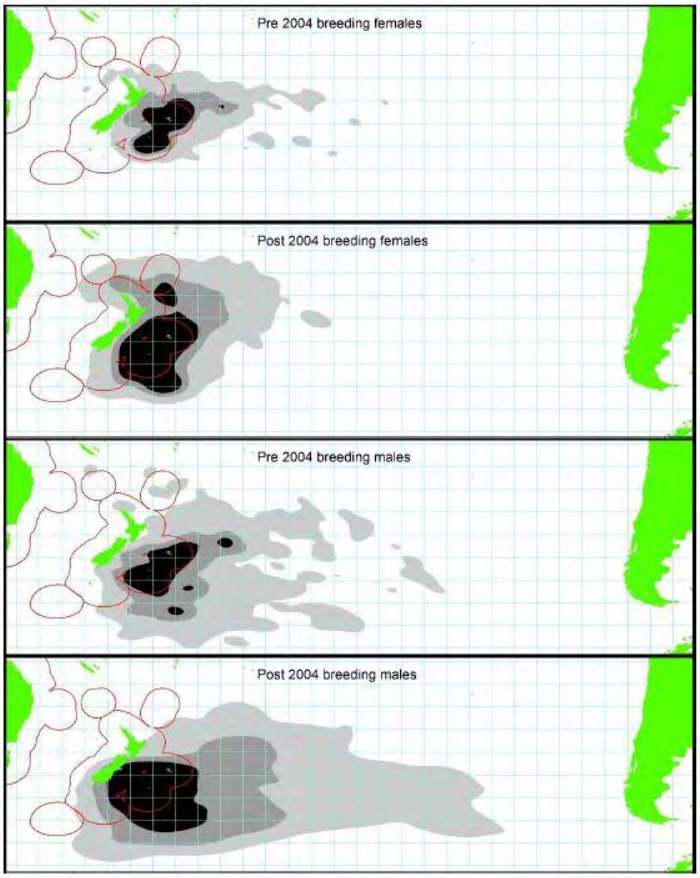


Figure 8.45: (from Elliott & Walker 2017). Kernel density plots of breeding Antipodean wandering albatrosses tracked in 1996–2004 and in 2011–17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

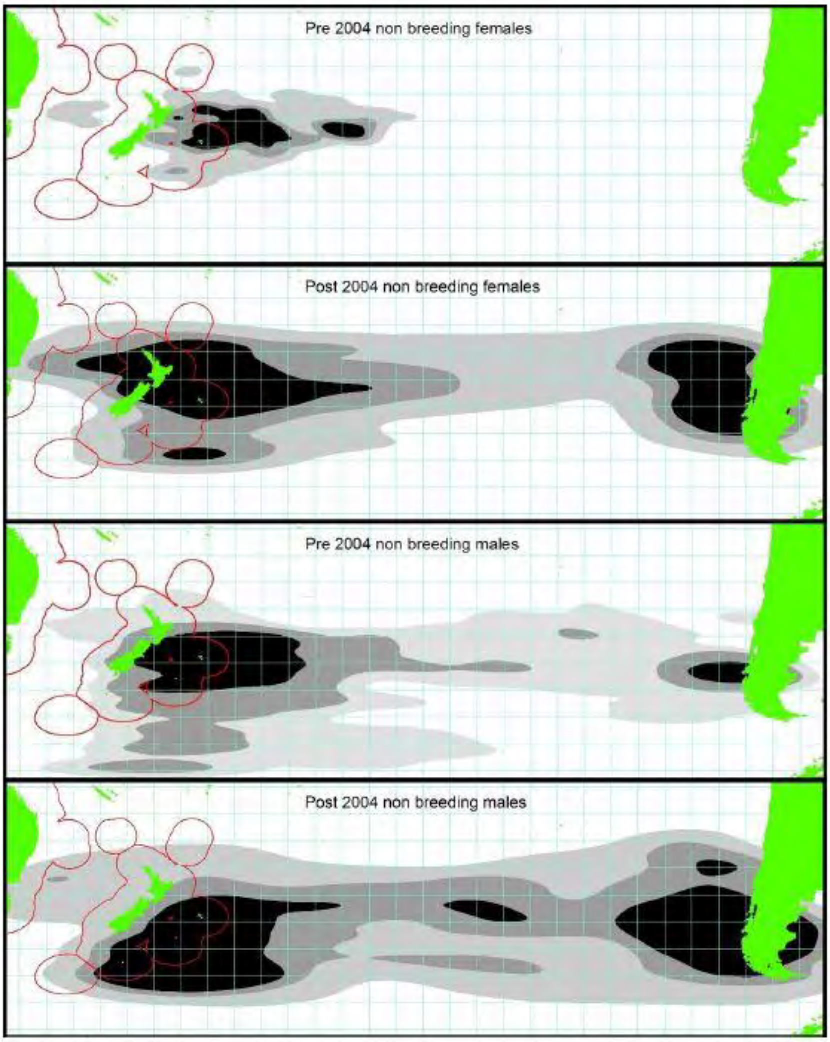


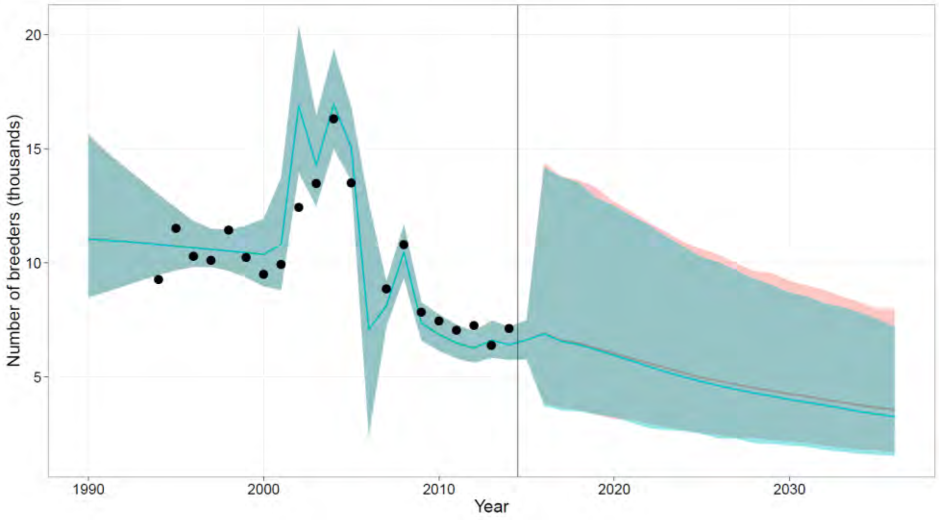
Figure 8.46: (from Elliott & Walker 2017). Kernel density plots of non-breeding Antipodean wandering albatrosses tracked in 1996–2004 and in 2011–17. Black indicates the 50% contour, dark grey the 75% contour, and light grey the 95% contour.

8.4.3.10.1 QUANTITATIVE MODELS FOR ANTIPODEAN ALBATROSS

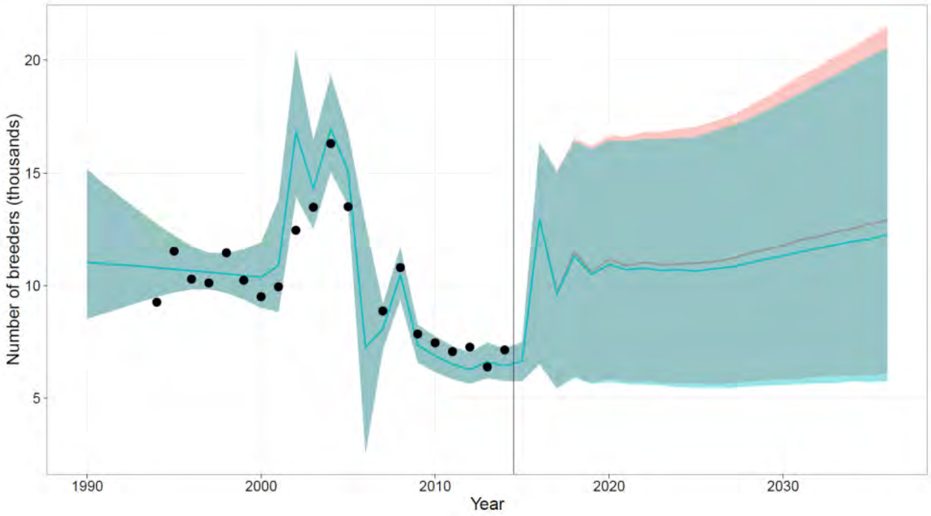
Edwards et al. (2017) developed a quantitative demographic model for the Antipodean albatross to estimate vital rates and predict population changes into the future given the observed declines in the population since 2005 (Elliott & Walker 2014). The model was parameterised using extensive mark-recapture and census data, which allowed the estimation of time-variant survivorship and breeding parameters.

Edwards et al. (2017) found that although the survivorship has changed over time, it was apparent from both the modelling and an empirical review of the data that changes in the breeding rate, breeding success, and age of first breeding are primarily responsible for temporal changes in the population abundance. The model predicted that the population has declined in numbers since 2007 and will continue to do so unless these demographic vital rates recover (Figure 8.47). Furthermore, it predicted that reduced adult survivorship as a result of fishing-induced mortalities within New Zealand waters is likely to be having a negligible impact, although the impact of unquantified mortalities arising from potential species misidentification or captures outside New Zealand waters could not be evaluated due to a lack of data (Edwards et al. 2017). The model demonstrates that domestic captures of Antipodean albatrosses are insufficient to be the cause of the population decline. Capture of Antipodean albatrosses outside New Zealand is a key factor that needs to be addressed to reverse the current population trajectory.

(a) Base case model with vital rates sampled from 2004–13



(b) Base case model with vital rates from 1995–2004



(c) Female-only model with vital rates sampled from 2004–13

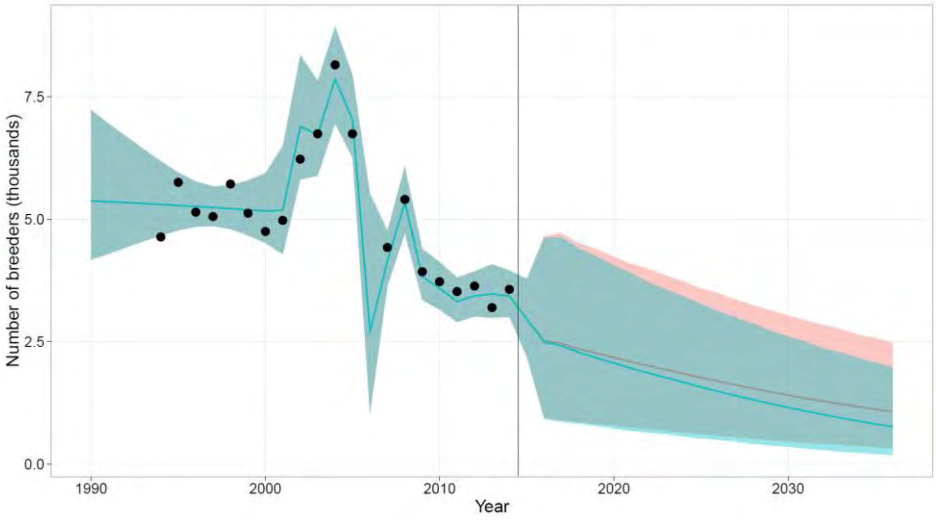


Figure 8.47: Predicted population dynamics for the whole island, showing the number of Antipodean albatross breeders from 2015–34. Predicted dynamics with and without fishing pressure are superimposed. The median and 95% confidence intervals are shown for each scenario. The vertical line indicates the limit of the empirical data, with subsequent dynamics representing an extrapolation. Empirical census data are also shown.

8.4.3.11 NORTHERN GIANT PETREL

The Forty-Fours in the Chatham Islands support the largest breeding colony of northern giant petrel in New Zealand. Northern giant petrels nest mainly in the north-eastern part of the island along the cliff tops. Estimates of nests with live chicks in two areas assessed over several years were: 431 in November 2007 (Scofield et al. 2008b), 335 in November 2008 (Fraser et al. 2009a), 270 in December 2009 (Fraser et al. 2010a), and 398 in December 2016 (Bell et al. 2017). It should be noted that because these counts were carried out during the chick-rearing phase of the breeding season they represent minimum estimates of breeding totals for the areas assessed, because some breeding attempts will have failed by the time the counts were made. Bell et al. (2017) conducted a complete count of nests with chicks at the Forty-Fours in December 2016 (1235 nests) and corrected this total using a breeding success rate of 60.1% to calculate a total breeding population of 1977 pairs of northern giant petrel.

Elsewhere in New Zealand, C.J.R. Robertson (cited by Imber 1994) estimated 336 nests at the Sisters (Chatham Islands) in 1976, and C.J.R. Robertson (cited by Hunter 1984) estimated a total of 2150 breeding pairs for the Chatham Islands, no date specified. Wiltshire & Scofield (2000) and Wiltshire & Hamilton (2003), based on complete nest counts, estimated totals of 234 breeding pairs at Campbell Island in 1996–97 and 233 breeding pairs at Antipodes Island in 2000–01, respectively. Taylor (2000) concluded that at least 50 pairs breed at the Auckland Islands. However, this estimate did not include Adams Island where there were approximately 35 breeding pairs in 2003–04, centred on Fairchild’s Garden towards the western end of the island (K. Walker pers. comm.). More recently, Parker et al. (2016) recorded 216 northern giant petrel chicks across eight of 15 islands in the Auckland Islands group in December 2015 and January 2016, with Enderby Island supporting the largest total of 96 chicks. Using a correction factor to account for breeding success, Parker et al. (2016) estimated an Auckland Islands breeding population of approximately 340 pairs.

8.4.3.12 YELLOW-EYED PENGUIN

The yellow-eyed penguin is an endemic species with two distinct populations. The northern population breed on mainland New Zealand/Aotearoa, Stewart Island/Rakiura and outliers, and the southern population breed on the sub-Antarctic Auckland Islands/Motu Maha and Campbell Island/Motu Ihupuku. There is very little movement between the northern and southern populations and they should be managed as separate units (Boessenkool et al. 2009). It is also worth noting that more is known about the northern population of hoiho due to regular monitoring, whereas the status of the southern population remains largely unknown. Estimates for the northern population of yellow-eyed penguins have been collated by the Department of Conservation up until 2019–20, from monitoring carried out by a number of different organisations (DOC unpublished data) for the four main breeding areas: South Island, Stewart Island and outliers, the Auckland Islands, and Campbell Island.

Breeding on the South Island has been further partitioned into four distinct regions. In the north, six nests were found on Banks Peninsula in 2011–12 (Ellenberg & Mattern. 2012), but in 2019–20 only one nest was found (DOC unpublished data). However, there have only ever been small numbers of hoiho at Banks Peninsula with low breeding success (Ellenberg & Mattern 2012). In North Otago, nest numbers peaked in 2008–09 at 77 and by 2019–20 had declined to 41 nests (DOC unpublished data). The decline here has however been slower than in other regions in part due to intensive management over a longer period of time. In contrast and overall, nest numbers on the Otago Peninsula have declined significantly since 1992, with the strongest decline, of about -60%, observed at Sandfly Bay. Nest numbers peaked at approximately 385 pairs in 1996–97 (Ellenberg & Mattern 2012) and by 2019–20 were only 74 (DOC unpublished data). Censuses were undertaken in the Catlins approximately every 5 years (1997–98, 2001–02, 2007–08, 2011–12, 2016–17); however in the last few years monitoring has occurred annually at the majority of hoiho colonies. These data suggested a stable population with between 190 and 240 breeding pairs until 2013–14. Since then the population declined steadily until 2018–19. In 2019–20 there was a 50% decline from the previous season down to only 51 nests (DOC unpublished data).

Darby (2003) reported estimates of 170–230 pairs of yellow-eyed penguin on Stewart Island and its outliers and of 50–80 pairs on Codfish Island during the period 1984–1994. Massaro & Blair (2003) reported the numbers of yellow-eyed penguin pairs for the period 1999–2001, with totals of 79 pairs for Stewart Island and 99 pairs on outlying islands, including 61 pairs on Codfish Island. In 2008–09 King et al. (2012) reported 77 breeding pairs throughout Stewart

Island, but noted that along the northern coast of Stewart Island nest numbers declined by 27% between 1999–2001 and 2008–09, and on Whenua Hou/Codfish Island yellow-eyed penguin numbers declined from 61 pairs in 2001–02 to only 12 pairs in 2019–20 (King 2020).

At the Auckland Islands, Moore (1992) counted yellow-eyed penguins at landing sites at islands in the north of the archipelago and at Adams Island in the south in 1989 and estimated a population of 420–470 pairs. Moore (1992) further suggested that inclusion of breeding pairs along the eastern coast of main Auckland Island could take the overall population to 520–570 breeding pairs. Muller et al. (2020) estimated an average of 577 breeding pairs at the Auckland Islands using nest search data from part of Enderby Island from 2015–16 to 2017–18 and corrected beach count data from main Auckland Island and outliers between 2012–13 and 2017–18. There is no evidence of an increase or decline in hoiho numbers since the 1990s.

Moore et al. (2001) reported numbers of yellow-eyed penguins at Campbell Island as 2277 individuals in 1988 and 1347 birds in 1992, a decrease of 41%. Counts of penguins at 11 landing sites indicated that the Campbell Island population remained relatively low until 1994 and then began to increase through to 1998 (Moore et al. 2001).

Based on the best available data the total number of breeding pairs are estimated to be 168 for mainland New Zealand and 103 for Stewart Island and its outliers (DOC unpublished data), about 577 for the Auckland Islands (Muller et al. 2020), and 350–540 for Campbell Island (Moore et al. 2001). On this basis, it is likely that there are only an estimated 1300 breeding pairs remaining and that approximately 80% of hoiho are found in the subantarctic.

Mattern et al. (2007) described foraging routes and benthic foraging behaviour in yellow-eyed penguins tracked from Bushy Beach near Oamaru in 2003–05 and found that birds performed daily trips ranging in distance from 12 to 20 km from the coast, with shorter (less than 7 km) trips in the evening. Of all dives, 87% were benthic, and birds exhibited consistent foraging routes and destinations across consecutive foraging trips. Mattern & Ellenberg (2018) further explored foraging in yellow-eyed penguins, tracking birds from the Otago Peninsula, and found that some birds foraged benthically (> 87% of dives) along straight trajectories (Figure 8.48).

Disease, sometimes aggravated by starvation or relatively poor dietary provisioning, has been demonstrated in mortality events and population declines in yellow-eyed penguins (Houston 2005, King et al. 2012, Argilla et al. 2013, Alley et al. 2016. Mattern et al. (2017) modelled climate effects on the fluctuations of a yellow-eyed penguin population at the Otago Peninsula and found that sea surface temperature accounted for 33% of the variation in penguin numbers, affecting both adult and fledgling survival; increasing sea surface temperatures since the mid-1990s was accompanied by a reduction in survival rates and a population decline.

8.4.3.12.1 QUANTITATIVE MODELS FOR YELLOW-EYED PENGUIN

Maunder et al. (2007) sought to assess the impact of commercial fisheries on the Otago Peninsula yellow-eyed penguins using mark-recapture data within a population dynamics model. They found the data available at that time inadequate to assess fisheries impacts but evaluated the likely utility of additional information on annual survival or an estimate of bycatch for a single year. Including auxiliary information on average survival in the absence of fishing allowed estimation of the fishery impact, but with poor precision. Including an estimate of fishery-related mortality for a single year improved the precision in the estimated fishery impact. The authors concluded that there was insufficient information to determine the impact of fisheries on yellow-eyed penguins and that quantifying fishing-related mortality over several years was required to undertake such an assessment using a population modelling approach.

Mattern et al. (2017) used a Bayesian modelling approach to assess the influence of climate change on population trends of yellow-eyed penguin over the last 30 years at the Otago Peninsula. They found that sea surface temperature (SST) was the key factor influencing survival of both adult birds and fledglings. Increasing SST since the mid-1990s was accompanied by a reduction in survival rates and a population decline. The population model showed that 33% of the variation in population numbers could be explained by SST alone. Mattern et al. (2017) suggested that the effect of SST significantly increased pressure on the penguin population such that the population becomes less resilient to non-climate related impacts, such as fisheries interactions, habitat degradation, and human disturbance.

Mattern et al. (2017) noted, however, that due to the absence of quantifiable data on fisheries impacts, habitat changes, and human disturbance, it was difficult to delineate the extent of the contribution of these factors to a declining population.

8.4.3.13 OTAGO SHAG

Based on analyses of genetic and morphological evidence, Rawlence et al. (2016) argued that Stewart Island shag *Leucocarbo chalconotus* be split into two species, Foveaux shag *Leucocarbo stewartia* and Otago shag *Leucocarbo chalconotus*. Otago shag is distributed along the south-eastern coast of South Island, from The Sisters rocks just offshore on the north side of Teahimate Bay in the south to Oamaru in the north (Lalas & Perriman 2009), but with occasional sightings extending further north as far as Lake Ellesmere (Crossland 2012). Lalas & Perriman (2009) identified six breeding sites for Otago shag (north to south: Maukiekie Island, Taiaroa Head, Wharekakahu, Gull Rocks, Green Island, and Kinakina Island), with a maximal overall population of approximately 1900 nests in 1987, falling to approximately 1150-1300 nests in 2005–07, although these totals did not include the KinaKina Island breeding site, which Lalas & Perriman (2009) estimated could support 200–300 nests.

8.4.4 MANAGING FISHERIES INTERACTIONS

New Zealand had taken steps to reduce incidental captures of seabirds before the advent of the IPOA in 1999, NPOA 2004, NPOA 2013 and NPOA 2020. For example, regulations were put in place under the Fisheries Act to prohibit drift net fishing in 1991 and prohibit the use of netsonde monitoring cables (‘third wires’) in trawl fisheries in 1992. The use of tori lines (streamer lines designed to scare seabirds away from baited hooks) was made mandatory in all tuna longline fisheries in 1992.

The fishing industry also undertook several initiatives to reduce captures, including funding research into new or improved mitigation measures and adopting voluntary codes of practice and best practice fishing methods. Codes of practice have been in place in the joint-venture tuna longline fishery since 1997–98, requiring, among other things, longlines to be set at night and a voluntary upper limit on the incidental catch of seabirds. That limit was steadily reduced from 160 ‘at risk’ seabirds in 1997–98, to 75 in 2003–04. Most vessels in the domestic longline tuna fishery had also voluntarily adopted night setting by 2004. A code of practice was in place for the ling auto-line fishery by 2002–03. Other early initiatives included reduced deck lighting, the use of thawed rather than frozen baits, sound deterrents, discharging of offal away from setting and hauling, weighted branch lines, different gear hauling techniques, and line shooters. Current regulated and voluntary initiatives are summarised by fishery in Table 8.30.

In 2002, MFish, DOC, and stakeholders began working with other countries to reduce the incidental catch of seabirds. As a result, a group called Southern Seabird Solutions was formed and formally established as a Trust in 2003 (https://www.catchfishnotbirds.nz) and received royal patronage in 2012. Southern Seabird Solutions exists to promote responsible fishing practices that avoid the incidental capture of seabirds in New Zealand and the Southern Ocean. Membership includes representatives from the commercial fishing industry, environmental and conservation groups, and government departments. The Trust’s vision is that: *All fishers in the Southern Hemisphere avoid the capture of seabirds*, and this is underpinned by the strategic goals on: Culture Change, Supporting Collaboration, Mitigation Development and Knowledge Transfer, Recognising Success, and Strengthening the Trust.

Building on these initiatives, New Zealand’s NPOAs established a more comprehensive framework for reducing incidental captures across all fisheries (because focusing on longline fisheries, like the IPOA does, was considered neither equitable nor sufficient).

The 2020 NPOA included four objectives that set the overall direction:

1. Effective bycatch mitigation practices are implemented in New Zealand Fisheries;
2. Direct effects of New Zealand fishing do not threaten seabird populations or their recovery;
3. Information to effectively manage direct fisheries effects on seabirds is continuously improved; and
4. New Zealand actively engages internationally to promote measures and practices that reduce impacts on New Zealand seabirds.
5. .

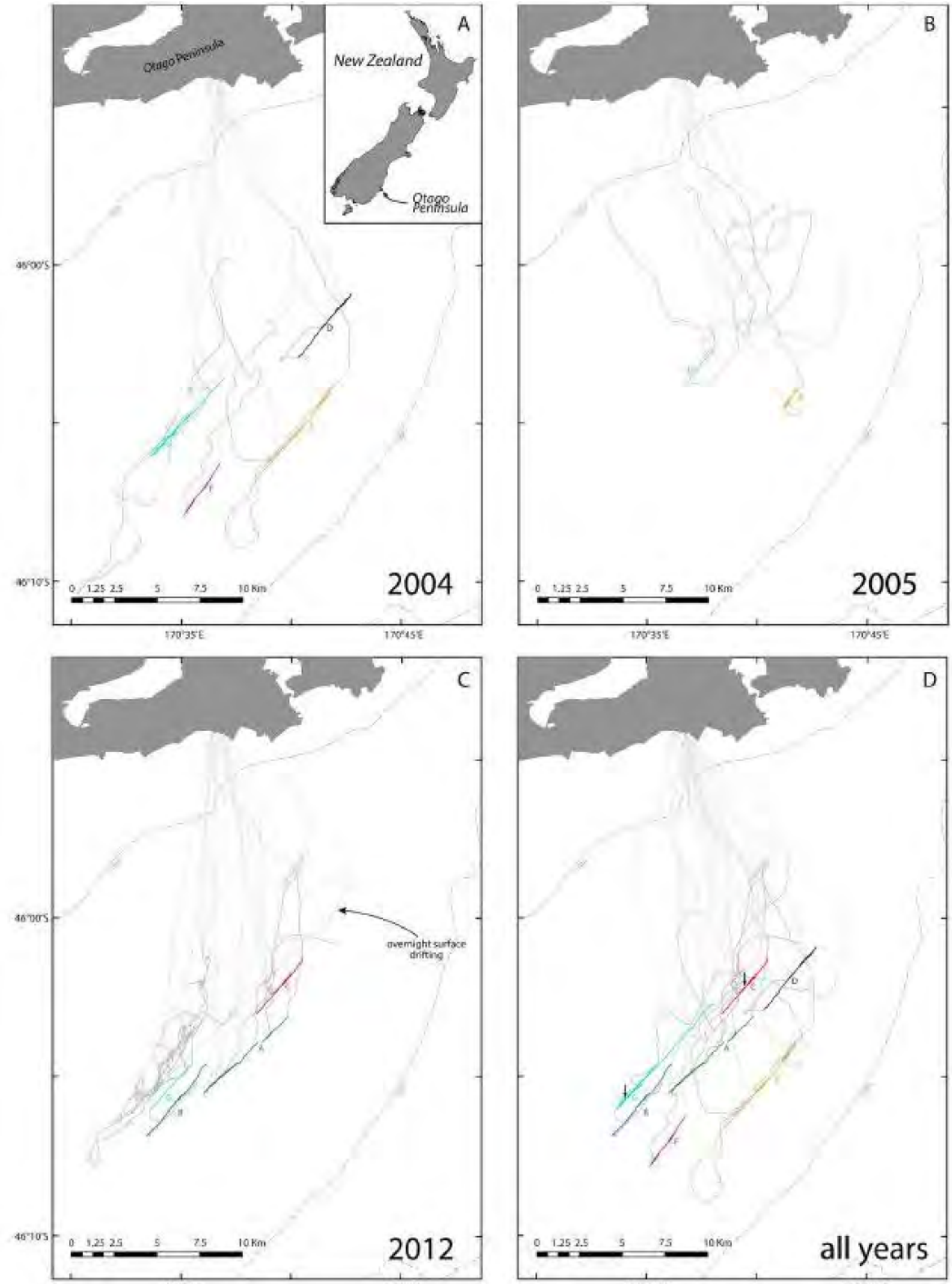


Figure 8.48: (from Mattern et al. 2013). Mid-shelf foraging tracks of yellow-eyed penguins recorded in 2004 (A), 2005 (B), and 2012 (C) that feature straight-line patterns. Foraging track segments in light grey represent outgoing and incoming stages of foraging trips; dark grey segments highlight the foraging stage. Dashed line segments indicate where linearity of the track is a result of interpolation. Track portions that met line criteria are highlighted in different colours; line identifiers shown in capital letters of the same colour. Small arrows in (D) indicate sites of ROV deployments in February 2013. Trips with lines from all three seasons are combined in (D).

Table 8.30: (from MPI 2013, NPOA-seabirds). Summary of current mitigation measures applied to New Zealand vessels fishing in New Zealand waters to avoid incidental seabird captures. R, regulated; SM, required via a self-managed regime (non-regulatory, but required by industry organisation and audited independently by government); V, voluntary with at least some use known; N/A, measure not relevant to the fishery; years in parentheses indicate year of implementation; *, part of a vessel management plan (VMP). Note, this table may not capture all voluntary measures adopted by fishers.

Mitigation measure	Surface longline	Bottom longline	Trawl ≥ 28 m	Trawl < 28 m	Set net	Notes
Netsonde cable prohibition	N/A	N/A	R (1992)	R (1992)	N/A	Netsonde cables also called third wires
Streamer (tori) lines	R	R	N/A	N/A	N/A	
Additional streamer line	–	–	N/A	N/A	N/A	
Night setting	R (or line weighting)	R (or line weighting)	–	–	–	Longlines must use night setting if not line weighting, or *vice-versa*
Line weighting	R (or night setting)	R (or night setting)	N/A	N/A	N/A	
Seabird scaring device	N/A	N/A	R (2006)	R?	N/A	To prevent warp captures and collisions
Additional bird scaring device	N/A	N/A	SM (2008)*	–	N/A	
Dyed bait	V	–	N/A	N/A	N/A	
Offal management	V	R	SM (2008)*	–	–	
VMPs			SM (2008)	V	–	Some VMPs developed for vessels < 28m
Code of Practice	V	–	VMP	–	–	–

Note: A vessel management plan (VMP) is a vessel-specific seabird risk management plan that specifies seabird mitigation devices to be used, operational management requirements to minimise the attraction of seabirds to vessels, and incident response requirements and other techniques or processes in place to minimise risk to seabirds from fishing operations.

Together the two goals established the NPOA as a long-term strategy. The second goal was designed to build on the first goal by promoting and encouraging the reduction of incidental catch beyond the level that is necessary to ensure long-term viability. The goals recognised that, although seabird deaths may be accidentally caused by fishing, most seabirds are absolutely protected under the Wildlife Act. The second goal balances the need to continue reducing incidental catch against the factors that influence how this can be achieved in practice (e.g., advances in technology and the costs of mitigation). The scope of the 2004 NPOA included:

- all seabird species absolutely or partially protected under the Wildlife Act;
- commercial and non-commercial fisheries;
- all New Zealand fisheries waters; and
- high seas fisheries in which New Zealand flagged vessels participate, or where foreign flagged vessels catch protected seabird species.

Specific objectives were established in the 2004 NPOA as follows:

- Implement efficient and effective management measures to achieve the goals of the NPOA, using best practice measures where possible.;
- Ensure that appropriate incentives and penalties are in place so that fishers comply with management measures;
- Establish mandatory bycatch limits for seabird species where they are assessed to be an efficient and effective management measure and there is sufficient information to enable an appropriate limit to be set;
- Ensure that there is sufficient, reliable information available for the effective implementation and monitoring of management measures;
- Establish a transparent process for monitoring progress against management measures;
- Ensure that management measures are regularly reviewed and updated to reflect new information and developments, and to ensure the achievement of the goals of the NPOA;
- Encourage and facilitate research into affected seabird species and their interactions with fisheries;

- Encourage and facilitate research into new and innovative ways to reduce incidental catch;
- Provide mechanisms to enable all interested parties to be involved in the reduction of incidental catch; and
- Promote education and awareness programmes to ensure that all fishers are aware of the need to reduce incidental catch and the measures available to achieve a reduction.

The 2004 NPOA-seabirds set out the mix of voluntary and mandatory measures that would be used to help reduce incidental captures of seabirds, noted research into the extent of the problem and the techniques for mitigating it, and outlined mechanisms to oversee, monitor, and review the effectiveness of these measures. It was not within the scope of the NPOA to address threats to seabirds other than fishing. Such threats are identified in DOC's Action Plan for Seabird Conservation in New Zealand (Taylor 2000) and their management is undertaken by DOC.

Since publication of the NPOA in 2004, more progress has been made in the commercial fishing sector, including:

- in the deepwater fishing sector:
  - industry has implemented vessel specific risk management plans (VMPs) comprising non-mandatory seabird scaring devices, offal management, and other measures to reduce risks to seabirds,
  - the government has implemented mandatory measures to reduce risk to seabirds (e.g., use and deployment of seabird scaring devices), and
  - industry has taken a proactive stance in resourcing a 24/7 liaison officer to undertake incident response actions, mentoring, VMP and regime development and reviewing, and fleet-wide training;
- in the bottom and surface longline sectors:
  - the government has implemented mandatory measures including tori lines, night setting, line weighting, and offal management,
  - a number of research projects have been or are currently being undertaken by government and industry into offal discharge, efficacy of seabird scaring devices, line weighting, and longline setting devices, and
  - workshops organised by both industry bodies and Southern Seabird Solutions are being held for the inshore trawl and longline sectors.

Mitigation has developed substantially since FAO's IPOA was published and a number of recent reviews consider the effectiveness of different methods (Bull 2007, 2009) and summarise currently accepted best practice (ACAP 2011). In December 2010, FAO held a Technical Consultation where International Guidelines on bycatch management and reduction of discards were adopted (FAO 2010). The text included an agreement that the guidelines should complement appropriate bycatch measures addressed in the IPOA-seabirds and its Best Practice Technical Guidelines (FAO 2009). The Guidelines were subsequently adopted by FAO in January 2011.

In 2013 the Ministry for Primary Industries released a revised and updated version of the NPOA-seabirds. This revision sought to address recommendations from the IPOA/NPOA Seabirds Best Practice Technical Guidelines (FAO 2009). The scope of the revised New Zealand NPOA-seabirds 2013 is as follows:

- all seabird species absolutely or partially protected under the New Zealand Wildlife Act 1953;
- commercial, recreational and customary non-commercial fisheries in waters under New Zealand fisheries jurisdiction;
- all fishing methods that capture seabirds, including longlining, trawling, set netting, hand lining, trolling, purse seining, and potting;
- all waters under New Zealand fisheries jurisdiction;
- high seas fisheries in which New Zealand flagged vessels participate, and, as appropriate and relevant, where foreign flagged vessels catch New Zealand seabirds; and

- other areas in which New Zealand seabirds are caught.

The long-term objective of the 2013 NPOA-seabirds is: *'New Zealand seabirds thrive without pressure from fishing related mortalities, New Zealand fishers avoid or mitigate against seabird captures and New Zealand fisheries are globally recognised as seabird friendly.'*

The high-level subsidiary objectives of the NPOA-seabirds 2013 are:

i. Practical objective: All New Zealand fishers implement current best practice mitigation measures relevant to their fishery and aim through continuous improvement to reduce and where practicable eliminate the incidental mortality of seabirds.
ii. Biological risk objective: Incidental mortality of seabirds in New Zealand fisheries is at or below a level that allows for the maintenance at a favourable conservation status or recovery to a more favourable conservation status for all New Zealand seabird populations.
iii. Research and development objectives:
    a. the testing and refinement of existing mitigation measures and the development of new mitigation measures results in more practical and effective mitigation options that fishers readily employ;
    b. research and development of new observation and monitoring methods results in improved cost effective assurance that mitigation methods are being deployed effectively; and
    c. research outputs relating to seabird biology, demography, and ecology provide a robust basis for understanding and mitigating seabird incidental mortality.
iv. International objective: In areas beyond the waters under New Zealand jurisdiction, fishing fleets that overlap with New Zealand breeding seabirds use internationally accepted current best practice mitigation measures relevant to their fishery.

Areas identified in the NPOA-seabirds 2013 that clearly require additional progress include:

i. mitigation measures for, and education, training, and outreach in, commercial set net fisheries and inshore trawl fisheries;
ii. implementation of spatially and temporally representative at-sea data collection in inshore and some Highly Migratory Species (HMS) fisheries;
iii. mitigation measures for net captures for deepwater trawl fisheries;
iv. the extent of any cryptic mortality (seabird interactions that result in mortality but are unobserved or unobservable); and
v. mitigation measures for, education, training, and outreach in, and risk assessment of, non-commercial fisheries (in particular the set net and hook and line fisheries).

The most important factor influencing contacts between seabirds and trawl warp cables is the discharge of offal (Wienecke & Robertson 2002, Sullivan et al. 2006b, ACAP 2011). Offal management methods used to reduce the attraction of seabirds to vessels include mealing, mincing, and batching. ACAP recommends (ACAP 2011) full retention of all waste material where practicable because this significantly reduces the number of seabirds feeding behind vessels compared with the discharge of unprocessed fish waste (Wienecke & Robertson 2002, Abraham 2009, Favero et al. 2010) or minced waste (Melvin et al. 2010). Offal management has been found to be a key driver of seabird bycatch in New Zealand trawl fisheries (Abraham 2007, 2010b, Abraham & Thompson 2009b, Abraham et al. 2009, Pierre et al. 2010, 2012a, 2012b). Other best practice recommendations (ACAP 2011) are the use of bird scaring lines to deter birds from foraging near the trawl warps, use of snatch blocks to reduce the aerial extent of trawl warps, cleaning fish and benthic material from nets before shooting, minimising the time the trawl net is on the surface during hauling, and binding of large meshes in pelagic trawl before shooting.

In New Zealand, the three legally permitted devices used for mitigation by trawlers are tori lines (e.g., Sullivan et al. 2006a), bird bafflers (Crysel 2002), and warp scarers (Carey 2005). Middleton & Abraham (2007) reported experimental trials of mitigation devices designed to reduce the frequency of collisions between seabirds and trawl warps on 18 observed vessels in the squid trawl fishery in 2006. The frequencies of birds striking either warps or one of three mitigation devices (tori lines, 4-boom bird bafflers, and warp scarers) were assessed using standardised protocols during commercial fishing. Different warp strike mitigation treatments were used on different tows according to a randomised experimental design. Middleton & Abraham (2007) confirmed that the discharge of offal was the main factor influencing seabird strikes; almost no strikes were recorded when there was no discharge, and strike rates were low when only sump water was discharged (see also Abraham et al. 2009). In addition to this effect, tori lines were shown to be most effective mitigation approach and reduced warp strikes by 80–95% of their frequency without mitigation. Other mitigation approaches were only 10–65% effective. Seabirds struck tori lines about as frequently as they did the trawl warps in the absence of mitigation, but the consequences are unknown.

Recommended best practice for surface (pelagic) longline fisheries and bottom (demersal) longlines (ACAP 2011) includes weighting of lines to ensure rapid sinking of baits (including integrated weighted line for bottom longlines), setting lines at night when most vulnerable birds are less active, the proper deployment of bird scaring lines (tori lines) over baits being set, and offal management (especially for bottom longlines). A range of other measures are offered for consideration.

In 2016, ACAP revised its best practice recommendations for surface longline fishing, to modify the line weighting configurations and add approved hook shielding devices as stand-alone measures (ACAP 2016).

A review of the implementation of the 2013 NPOA-seabirds was scheduled to occur after four years. MPI commenced this review in April 2017 with significant input from a multi-stakeholder Seabird Advisory Group (SAG) established under the NPOA-seabirds and administered by MPI. The SAG is reflective of the multi-sector interests in seabirds and those that were involved in the development of the NPOA-seabirds, including the Crown’s Treaty partner in fisheries matters, commercial industry, recreational sector, NGOs, and the relevant government departments (DOC, MPI, and MFAT).

The review examined the extent to which the five-year objectives of the NPOA-seabirds have been achieved and identifies key actions as priority for the next NPOA-seabirds which was released in 2020.

A new NPOA-seabirds was released in 2020 (Fisheries New Zealand & Department of Conservation 2020). The 2020 NPOA included a vision “New Zealanders work towards zero fishing-related seabird mortalities”, the vision of the NPOA Seabirds 2020 sets out the desired future state for the management of the impacts of fishing on seabirds.

Underlying the vision, four objectives have been developed for a range of key focus areas. The objectives are intended to be achieved within the lifespan of this plan, but it is acknowledged that some may flow through to subsequent versions.

These objectives are:

1. Effective bycatch mitigation practices are implemented in New Zealand Fisheries;
2. Direct effects of New Zealand fishing do not threaten seabird populations or their recovery;
3. Information to effectively manage direct fisheries effects on seabirds is continuously improved; and
4. New Zealand actively engages internationally to promote measures and practices that reduce impacts on New Zealand seabirds.

Fisheries New Zealand and DOC are committed to achieving the NPOA Seabirds 2020’s objectives. Their specific priorities for the next five years will be guided by the multi-year Seabird Implementation Plan. This will be regularly updated through an annual planning and review process.

Each year, Fisheries New Zealand and DOC will review what actions have been taken, and what progress has been made against the NPOA Seabirds 2020’s objectives. The results of this review will be published in a Seabird Annual Report and inform actions that need to be taken the following year. These actions will be transferred to the multi-year Seabird Implementation Plan, and will guide the work for the following year.

8.5 INDICATORS AND TRENDS

<table>
<tr><td>Population size</td><td>Multiple species and populations: see Taylor (2000)</td></tr>
<tr><td>Population trend</td><td>Multiple species and populations: see Taylor (2000)</td></tr>
<tr><td>Threat status</td><td>Multiple species and populations: see Robertson et al. (2017)</td></tr>
<tr><td>Number of interactions[1]</td><td>In the 2017–18 October fishing year, there were an estimated 3329 seabird captures (excluding cryptic mortalities) across all trawl and longline fisheries (http://data.dragonfly.co.nz/psc, data version 2019v1). About 46% of the estimated captures from these fisheries (other fisheries such as set net are excluded) were in trawl fisheries, 19% in surface longline fisheries (there are no large SLL vessels operating in New Zealand’s EEZ), and 36% in bottom longline fisheries:
<table>
<tr><th>Bird group</th><th>DW Trawl</th><th>Inshore Trawl</th><th>Large BLL</th><th>Small BLL</th><th>Large SLL</th><th>Small SLL</th><th>All these methods</th></tr>
<tr><td>White-capped albatross</td><td>184</td><td>127</td><td>1</td><td>18</td><td>0</td><td>132</td><td>462</td></tr>
<tr><td>Salvin’s albatross</td><td>181</td><td>107</td><td>10</td><td>45</td><td>0</td><td>9</td><td>352</td></tr>
<tr><td>Buller’s albatross</td><td>101</td><td>15</td><td>4</td><td>16</td><td>0</td><td>96</td><td>232</td></tr>
<tr><td>Other albatrosses</td><td>31</td><td>13</td><td>12</td><td>61</td><td>0</td><td>132</td><td>249</td></tr>
<tr><td>Sooty shearwater</td><td>153</td><td>47</td><td>2</td><td>8</td><td>0</td><td>3</td><td>213</td></tr>
<tr><td>White-chinned petrel</td><td>302</td><td>13</td><td>125</td><td>212</td><td>0</td><td>29</td><td>681</td></tr>
<tr><td>Black petrel</td><td>4</td><td>28</td><td>0</td><td>151</td><td>0</td><td>59</td><td>242</td></tr>
<tr><td>Grey petrel</td><td>9</td><td>4</td><td>12</td><td>95</td><td>0</td><td>19</td><td>139</td></tr>
<tr><td>Flesh footed shearwater</td><td>36</td><td>55</td><td>0</td><td>223</td><td>0</td><td>111</td><td>425</td></tr>
<tr><td>Other birds</td><td>49</td><td>46</td><td>17</td><td>173</td><td>0</td><td>46</td><td>331</td></tr>
<tr><td>All birds combined</td><td>1 051</td><td>456</td><td>184</td><td>1 002</td><td>0</td><td>636</td><td>3 329</td></tr>
</table>
</td></tr>
<tr><td>Trends in interactions</td><td>Captures of all birds combined show a decreasing trend between 2002–03 and 2016–17, except white-chinned petrel (http://data.dragonfly.co.nz/psc, data version 2019v1). Captures of several species have decreased following high total estimates in the early portion of the period, including white-capped albatross, Buller’s albatross, sooty shearwater, black petrel, and flesh-footed shearwater:

Capture rate trends (excluding cryptic mortalities) are described for deepwater trawl, inshore trawl, large bottom longline, small bottom longline, large surface longline, and small surface longline. For white-capped albatross, captures rates in the trawl fisheries for deepwater and midwater species declined after 2004–05. For Salvin’s albatross, captures rates have fluctuated without trend in all fisheries except small surface longline vessels where they showed a decrease between 2002–03 and 2016–17, and deepwater trawl where there has been a slight increase. Capture rates were unusually high in the deepwater trawl fisheries in 2004–05, and capture rates have increased from 2008–09. Together, trawl fisheries account for 81% of all estimated captures of Salvin’s albatross in the 2017–18 fishing year. For Buller’s albatross captures and capture rates fluctuated with no trend in bottom longline fisheries and deepwater trawl fisheries. For white-chinned petrel, captures rates increased between 2002–03 and 2016–17 for deepwater trawlers and exhibited large inter-annual variations in both capture rates and captures. For black petrels, captures rates decreased proportionally with captures in the small bottom longline fisheries between 2002–03 and 2016–17.

The most recent SEFRA by Richard et al. (2020) ranks black petrel as the most at risk seabird, with a median risk ratio of 1.23. This is followed by the Salvin’s albatross, Westland petrel, flesh-footed shearwater, southern Buller’s albatross, and Gibson’s albatross. These species make up the Very High and High risk species.</td></tr>
</table>

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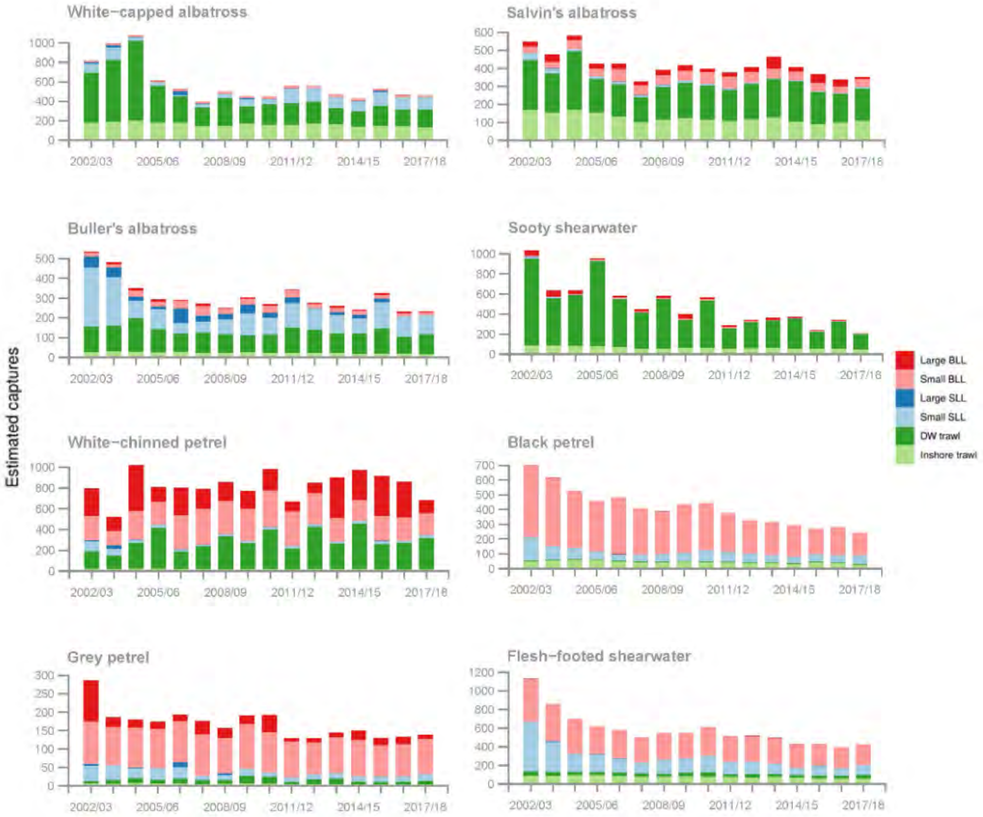
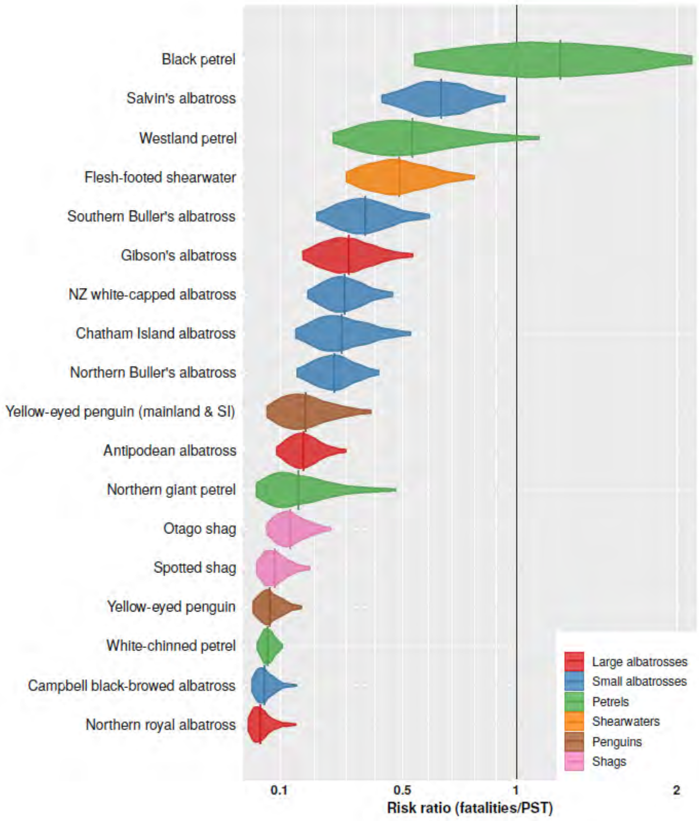
[1] For more information, see: http://data.dragonfly.co.nz/psc.

Trends in interactions





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THEME 2: NON-TARGET FISH AND INVERTEBRATE CATCH

Chapter 9: Non-target fish and invertebrate catch - Technical Summary

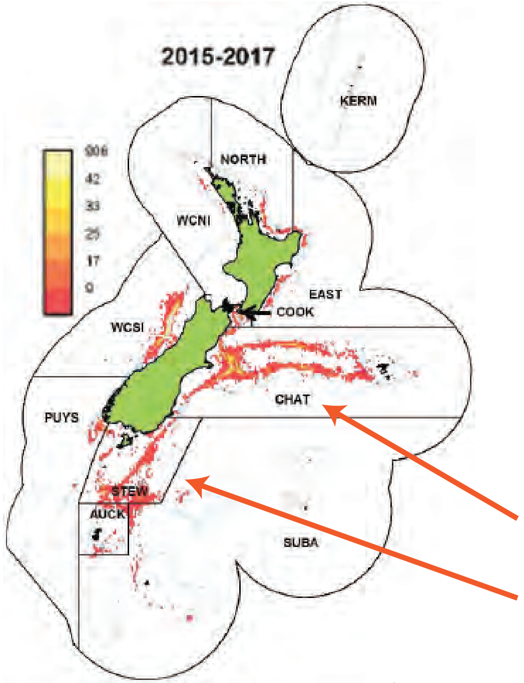
Capture of non-target fish and invertebrates

Estimates of non-target catch and discards at sea

1. THE ISSUE IN BRIEF

- Non-target catch (also known as bycatch) refers to all fish and invertebrates caught while fishing, that were not the intended target species for a given fishing event
- In New Zealand over 800 non-target species were identified by observers in the hoki/hake/ling trawl fishery
- Non-target catch is not always unwanted, and so is often landed. Only the unwanted portion is legally discarded
- Non-target catch is extremely variable and fluctuates periodically in different areas and with different gears
- Despite their small share of the non-target catch, some sharks (due to their low productivity), and some benthic invertebrates (for their habitat-forming characteristic), could be of particular concern (see box 3 below)

2. WHERE AND HOW MUCH?



- Models based on observer data are used to estimate non-target catch and discards in offshore fisheries (in areas >12 nautical miles from the coast). Records of non-target catch in inshore fisheries (in territorial waters, <12 nm from the coast) are being improved to allow the application of similar models.

- The latest (2017) overall estimate of non-target catch in offshore fisheries stands at nearly 65,000 tonnes per year. About 24% (16,000 tonnes) of this non-target catch was legally discarded

Offshore areas with highest effort, target and non-target catch:

- Chatham rise (CHAT)
  hoki/hake/ling, silver and white warehou trawl fisheries
- Stewart-Snares shelf (STEW)
  arrow squid and hoki/hake/ling trawl fisheries

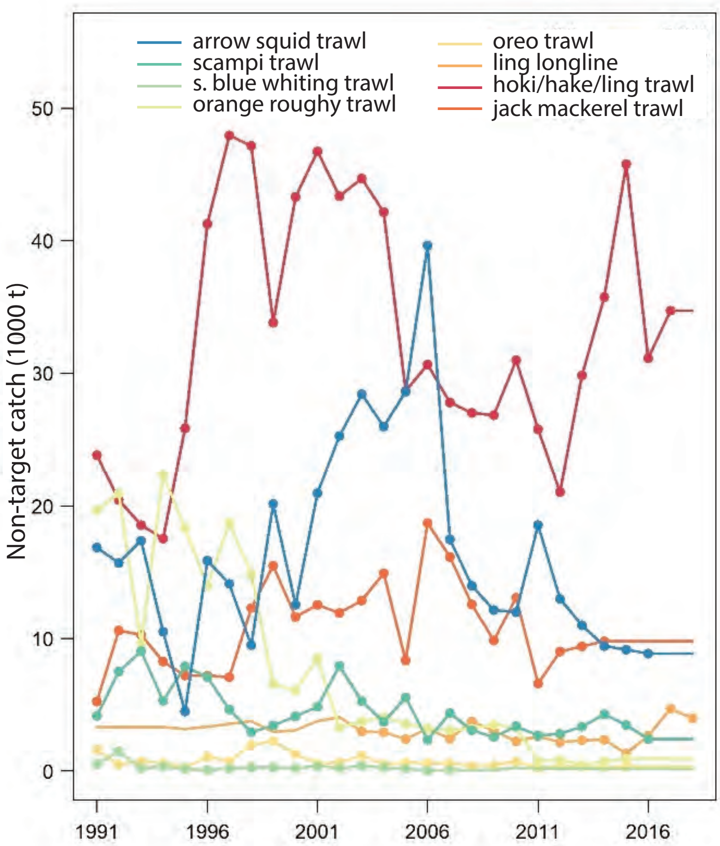
The map represents, in shades of red and yellow, the fishing effort (tows per year) of the hoki, hake, ling, silver warehou, and white warehou fisheries

3. SHARKS AND INVERTEBRATES

- Survival of sharks caught and returned to the sea alive is poorly known, but thought to be low
- The risk posed by fishing to shark populations in NZ is being investigated
- Read more in Chapter 10

- With the exception of squid, non-target catch of invertebrate species is less than 0.05% of the hoki/hake/ling trawl fishery catch
- The impact of fishing on habitat-forming invertebrates, e.g. corals, cannot be estimated from non-target catch data alone; further studies are needed to evaluate the effect of fisheries
- Read more in Chapter 11

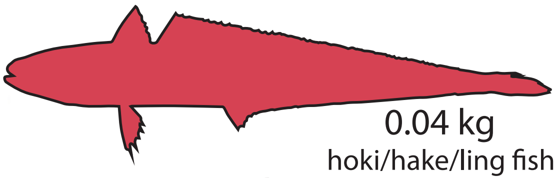
4. NON-TARGET CATCH AND DISCARDS



Time trends for non-target catch in offshore fisheries
Note that only a small portion of non-target catch is discarded

- Estimated tonnage of non-target catch in different fisheries is informative, but it needs to be scaled to the size of the fishery (see box 5 below)
- Furthermore, because a large portion of the non-target catch is retained, it is useful to consider how much is also discarded
- Discards are estimated as a fraction of the target catch harvested for the latest four years of fishing, where data are available

Mean utilisation rate (kilograms of non-target catch discarded per kg of target catch harvested) in some selected fisheries

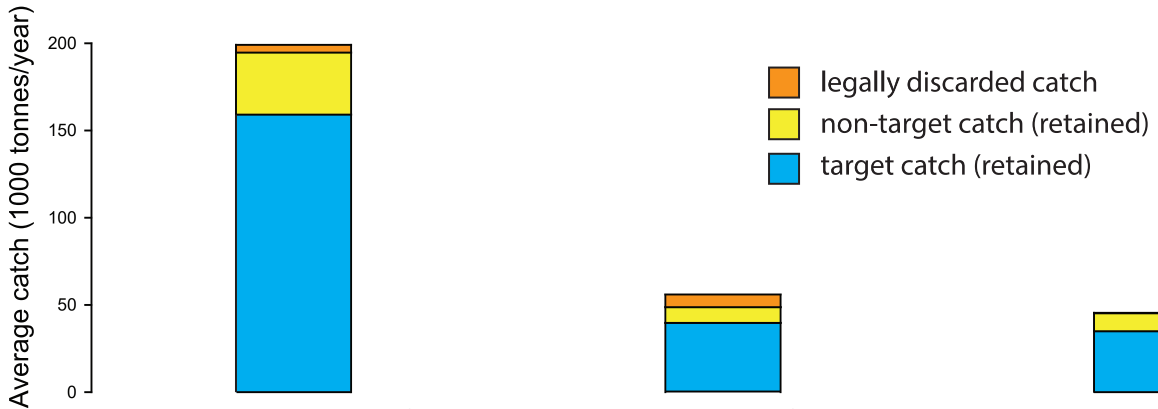


0.04 kg
hoki/hake/ling fishery

0.14 kg
arrow squid fishery

3.83 kg
scampi fishery

5. EXAMPLES FROM OFFSHORE TRAWL FISHERIES



hoki/hake/ling trawl fishery
22% non-target catch and
6% discards (2016-2017)

arrow squid trawl fishery
23% non-target catch and
11% discards (2015-2016)

jack mackerel trawl fishery
29.6% non-target catch and
0.6% discards (2013-2014)

6. ONGOING RESEARCH

- Exploration of ways to estimate non-target catch in inshore fisheries more accurately
- Research on non-target catch spatial and temporal patterns
- Future quantitative risk assessment for sharks might use non-target catch as input (see Chapter 10)

9 NON-TARGET FISH AND INVERTEBRATE CATCH

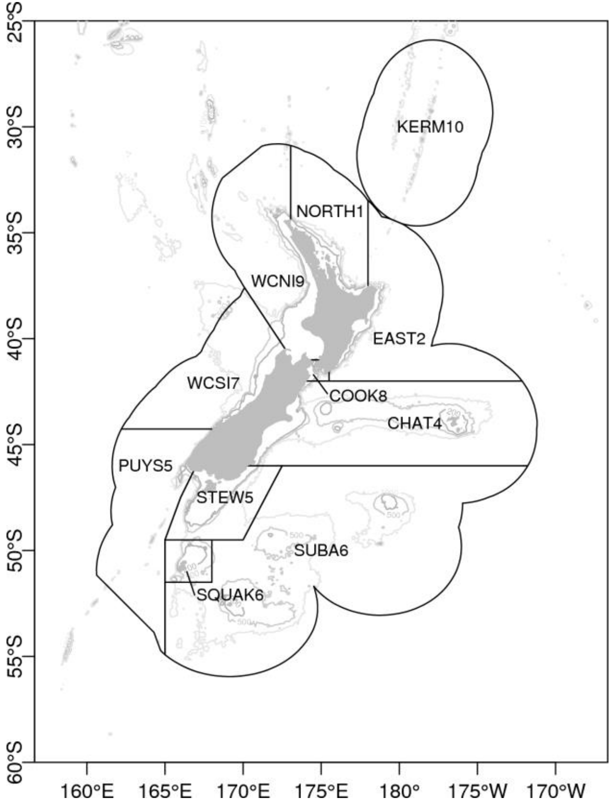
<table>
<tr><td>Status of chapter</td><td>This chapter has had minor updates for AEBAR 2021.</td></tr>
<tr><td>Scope of chapter</td><td>This chapter outlines the main fish and invertebrate species caught as non-target species catch in New Zealand’s major offshore fisheries (>12 nautical miles from the coast), with summaries of the amounts caught and discarded.
This chapter incorporates a re-assessment of historical analyses for offshore fisheries, with stratification aligned to standardised areas, thus providing estimates of non-target species catch and discarding across all offshore fisheries within separate regions of New Zealand fisheries.
Since 2013, continued research analysed non-target species catch over time for each of the Tier 1 offshore fisheries at a frequent but irregular rate. This chapter presents the latest available information; however, the last date of detailed analysis differs between fisheries, e.g., the southern blue whiting fishery was updated to 2006–07, and the jack mackerel fishery has been updated to 2018–19 (see Table 9.1 for more details).
Much of the research in this field has been conducted fishery by fishery with no spatial breakdown of annual catch totals, but information by fishery is available for:
<table>
<tr><th>Trawl fisheries</th><th>Longline fisheries</th><th>Other fisheries</th></tr>
<tr><td>Arrow squid</td><td>Ling (bottom)</td><td>Albacore tuna troll</td></tr>
<tr><td>Hoki/hake/ling</td><td>Tuna (surface)</td><td>Skipjack tuna purse seine</td></tr>
<tr><td>Jack mackerel</td><td></td><td></td></tr>
<tr><td>Southern blue whiting</td><td></td><td></td></tr>
<tr><td>Orange roughy</td><td></td><td></td></tr>
<tr><td>Oreo</td><td></td><td></td></tr>
<tr><td>Scampi</td><td></td><td></td></tr>
</table>
</td></tr>
<tr><td>Areas</td><td>Total annual non-target catch and discards are summarised for 10 of the 11 fishery areas shown in Figure 9.1. Non-target catch and discards in the Kermadec area are not addressed (due to the negligible fishing effort in this area).


Figure 9.1: Standardised assessment areas for estimation of total non-target fish and invertebrate catch in offshore fisheries. Note that inshore areas (within 12 nautical miles from the coastline) are excluded from this assessment.</td></tr>
</table>

Focal localities	**Trawl fisheries** *Arrow squid*: Auckland Islands and Stewart-Snares shelf (80–300 m). *Hoki/hake/ling*: Chatham Rise, West Coast of the South Island, Campbell Plateau, Puysegur Bank, and Cook Strait (200–800 m). *Jack mackerel*: West coast of the North Island and South Island, Chatham Rise, and Stewart-Snares shelf (0–300 m). *Southern blue whiting*: Campbell Plateau and Bounty Plateau (250–600 m). *Orange roughy*: The entire New Zealand region (700–1200 m). *Oreos*: South Chatham Rise, Pukaki Rise, Bounty Plateau, and Southland (700–1200 m). *Scampi*: East coasts of the North Island and South Island, Chatham Rise (300–500m), and Auckland Islands (350–550 m). **Longline fisheries** *Ling (bottom)*: Chatham Rise, Bounty Plateau, and Campbell Plateau (150–600 m). *Tuna (surface)*: East coast of the North Island and both east and west coasts of the South Island. **Other fisheries** *Albacore tuna troll*: West coasts of the North Island and South Island. *Skipjack tuna purse seine*: Northern North Island
Key issues	• Lack of data on non-target catch and discards for most inshore (0–200 m) fisheries because of low observer coverage, and simpler reporting requirements prior to 1 October 2007, which saw most catch and effort data aggregated per day and by statistical area (Catch Effort and Landing Return). • Collection of more detailed fishing event catch and effort data for smaller trawl (6–28 m), longline, and setnet vessels began on 1 October 2007. • Rollout of electronic monitoring tools on inshore vessels is completed for data collection, which should help address this issue.
Emerging issues	• Trends showing increased rates and levels of catch and discarding of several non-target species or species categories, especially some non-QMS fish species and invertebrates.
Fisheries New Zealand research (current)	BYC2021-03 *Bycatch monitoring and quantification in deepwater fisheries*
New Zealand government research (current)	DOC19301-POP2018-01 *Improved habitat suitability modelling for protected corals in New Zealand waters*
Related chapters/issues	Chondrichthyans (sharks, rays, and chimaeras), Benthic (Seabed) Impacts

1.1 CONTEXT

Bycatch is usually defined as composed of unwanted zero- or low-value species but, in some context like the New Zealand one, this definition is extended to include all species that were not the stated target for that fishing event. Some non-target species may have commercial value and are thus retained and landed. Discards are usually defined as the portion of the catch that is not landed (discarded at sea, legally or illegally) and may include both target (e.g., damaged or too small to process individuals) and non-target species (of zero- or low-value). Both bycatch and discards are significant issues in many fisheries worldwide.

For this chapter *non-target species catch* is equivalent to *bycatch* and refers to all fish and invertebrates caught that were not the intended or stated target species for that fishing event, whether or not they were discarded (McCaughran 1992). *Discarded catch* (or *discards*) is defined as “all the fish, both target and non-target species, which are returned to the sea whole as a result of economic, legal, or personal considerations” (McCaughran 1992). Discards can be legal (e.g., under Schedule 6 of the Fisheries Act 1996), are sometimes counted against the vessel quota, returned to sea under observer approval, and also include fish returned alive (but for which survival is largely unknown). *Discarded catch* in this report also includes discards of invertebrate species and estimates of any fish lost from the net at the surface.

OFFSHORE TRAWL AND BOTTOM-LONGLINE FISHERIES

The management of non-target fish and invertebrate catch in the offshore (deepwater and middle-depth) fisheries is described in the National Fisheries Plan for Deepwater and Middle-depth Fisheries (Fisheries New Zealand 2019). Under the National Deepwater Plan 2019, the objective most relevant for management of non-target fish and invertebrate catch is Management Objective 6: *Manage deepwater and middle-depth fisheries to avoid, remedy or mitigate adverse effects on associated or dependent or incidentally caught fish species*. Specific objectives for the management of non-target species catch are outlined in the fishery-specific chapters of the National Deepwater Plan. Estimation of non-target species catch is carried out for each of the Tier 1 Deepwater fisheries on a rotational basis, with each of the following fisheries updated about every 4–5 years:

- arrow squid trawl
- ling bottom longline
- hoki/hake/ling trawl
- jack mackerel trawl
- southern blue whiting trawl
- orange roughy/oreo trawl
- scampi trawl.

SURFACE LONGLINE, TROLL, AND PURSE-SEINE FISHERIES

Non-target fish species catch in the fisheries for Highly Migratory Species (HMS) is addressed in the HMS fish plan. Tuna fisheries incidental bycatch is examined, with updates every 1–3 years as required by the relevant international commissions for the management of tuna stocks. Some data on bycatch in the albacore tuna troll fishery and the skipjack tuna purse seine fishery are also available.

INSHORE FISHERIES

The three draft National Fisheries Plans for Inshore species (finfish, shellfish, and freshwater fisheries) also include objectives that address non-target species catch, but analyses to inform these objectives have yet to be conducted. However, summaries of the main bycatch species have occasionally been included in reports from fisheries characterisation projects, for example school shark, red gurnard, and elephant fish (Starr et al. 2010a, 2010b, 2010c, Starr & Kendrick 2012, Starr & Kendrick 2013).

1.2 GLOBAL UNDERSTANDING

Few fisheries are completely without bycatch or discards and this issue has been the subject of many studies and international meetings. Saila (1983) made the first comprehensive global assessment and estimated, albeit with very poor information, that at least 6.7 million tonnes were discarded each year. Alverson et al. (1994) extended that work and estimated the global bycatch at 27.0 (range 17.9–39.5) million tonnes each year. An update by Kelleher (2005) suggested global bycatch of about 8% of the global catch, or 7.3 million tonnes, in 1999–2001. Perceptions of target and non-target catch vary widely and pose a real challenge for the estimation of global bycatch. Defining bycatch as “catch that is either unused or unmanaged”, Davies et al. (2009) estimated that bycatch represents approximately 40.4% of global marine trawl catches.

Tropical shrimp trawl fisheries typically have very fine net mesh size and the highest levels of unwanted bycatch, with an average discard rate of 62% (Kelleher 2005), accounting for about one-quarter to one-third of global bycatch. Tuna longline fisheries have the next largest contribution and tend to have greater unwanted bycatch than other line fisheries (Kelleher 2005). Discard rates in demersal trawl fisheries targeting finfish are much lower but, because they are so widespread, comprise a considerable proportion of total global discards (Kelleher 2005).

Since the Alverson et al. (1994) estimate, the global level of discards has declined, but differences in the methodology and definition of bycatch used (see Kelleher 2005, Davies et al. 2009) make it difficult to appropriately quantify this decline. The main reasons for the estimated decline in discards may be due to a combination of higher retention rates, better fisheries management, more selective/targeted fishing methods, and an overall reduction in catches (despite an increasing effort, see e.g., Anticamara et al. 2011, Zeller et al. 2005).

Bycatch and discard estimation is frequently very coarse, and estimates of rates based on occasional surveys are often scaled up to represent entire fisheries and applied across years, or even to other fisheries (e.g., Bellido et al. 2011). Data from dedicated fisheries observers are also frequently used for individual fisheries, and these are considered to provide the most accurate results, providing that discarding is not illegal (leading to bias due to ‘observer effects’; Fernandes et al. 2011). Ratio estimators similar to those historically applied in some New Zealand fisheries were frequently used to scale observed bycatch and discard rates to the wider fishery, and the methods historically used in New Zealand fisheries were broadly similar to those used elsewhere (e.g., Fernandes et al. 2011, Borges et al. 2005). A new methodology has recently been developed for New Zealand fisheries, which is now replacing the ratio method. This method uses multiple predictor variables in a model-based estimation process fitted using Bayesian methods and has shown in simulation studies to provide estimates with less bias and improved precision (Edwards et al. 2015). This modeling approach has been used alongside the ratio method in two assessments (Anderson et al. 2017a, 2017b), and as the sole method in the most recent assessments (Anderson & Edwards 2018, Anderson et al. 2019, Finucci et al. 2020, Finucci et al. in press).

Discard data are increasingly incorporated into fisheries stock assessments and management decision-making, especially with the move towards an Ecosystem Approach to Fisheries (EAF) (Bellido et al. 2011), and as third-party fishery certification schemes more closely examine the effects of fishing on the ecosystem. These data have also been used to assess impacts on non-target species overseas (e.g., Pope et al. 2000, Casini et al. 2003).

1.3 NEW ZEALAND OVERVIEW

Estimation of annual catch and discard levels of non-target species in selected New Zealand fisheries have been undertaken frequently, but at irregular intervals since 1998 (Table 9.1).

Table 9.1: Summary of research into bycatch and discards in New Zealand fisheries.

Trawl fisheries	Report	Trawl fisheries	Report
Arrow squid trawl (SQU)	Anderson et al. (2000) Anderson (2004b) Ballara & Anderson (2009) Anderson (2013a) Anderson (2013b) Anderson (2014b) Ballara (2015) Edwards et al. (2015) Anderson (2017) Anderson & Edwards (2018) Finucci et al. (2019)	Oreo trawl (OEO)	Clark et al. (2000) Anderson (2004a) Anderson (2011) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Anderson et al. (2017a) Finucci et al. (2019) Anderson & Finucci (in press)
Hoki trawl (HOK)	Clark et al. (2000) Anderson et al. (2001) Anderson & Smith (2005) Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (2019) Finucci et al. (2019)	Scampi trawl (SCI)	Clark et al. (2000) Anderson (2004a) Ballara & Anderson (2009) Anderson (2012) Anderson (2013b) Anderson (2014b) Edwards et al. (2015) Anderson (2017) Anderson & Edwards (2018) Finucci et al. (2019)
Hake trawl (HAK)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (2019) Finucci et al. (2019)	Southern blue whiting trawl (SBW)	Clark et al. (2000) Anderson (2004a) Anderson (2009b) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017) Finucci et al. (2019)
Ling trawl (LIN)	Ballara et al. (2010) Anderson (2013b) Anderson (2014b) Ballara (2015) Ballara & O'Driscoll (2015) Anderson (2017) Anderson et al. (2019) Finucci et al. (2019)	Orange roughy trawl (ORH)	Clark et al. (2000) Anderson et al. (2001) Anderson & Clark (2003) Anderson (2009a) Anderson (2011) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson (2017)

			Anderson et al. (2017a) Finucci et al. (2019) Anderson & Finucci (in press)
Jack mackerel trawl (JMA)	Anderson et al. (2000) Anderson (2004b) Anderson (2007) Anderson (2013b) Anderson (2014b) Ballara (2015) Anderson et al. (2017b) Anderson (2017) Finucci et al. (2019) Finucci et al. (2022)		
Other fisheries	**Report**	**Other fisheries**	**Report**
Albacore tuna troll	Griggs et al. (2014)	Ling longline (LLL)	Anderson et al. (2000) Anderson (2008) Anderson (2013a) Anderson (2013b) Anderson (2014a) Anderson (2014b) Ballara (2015) Edwards et al. (2015) Anderson (2017) Finucci et al. (2019) Finucci et al. (2020)
Skipjack tuna purse seine	Anon (2013) Anon (2017)	Tuna longline	Griggs et al. (2013) Griggs et al. (2018) Griggs et al (2021)

TRAWL AND BOTTOM-LONGLINE FISHERIES

The estimation process for the trawl and bottom longline fisheries used rates of bycatch and discards in various categories, i.e., in recent analyses ‘all QMS species combined (QMS)’, ‘all non-QMS fish species combined (non-QMS)’, and ‘all non-QMS invertebrate species combined (INV)’. It also used information from fishery strata in the observed fraction of the fishery, together with effort statistics from the wider fishery, to calculate annual bycatch and discard levels. The ratio-based approach estimates precision by incorporating a multi-step bootstrap algorithm, which considers the effect of correlation between trawls in the same observed trip and stratum, whereas the statistical model method estimates uncertainty from the 95% credibility interval of the posterior distribution of model estimates. For this report, additional estimates of annual bycatch and discards within standardised areas (Figure 9.1) were re-calculated from archived data where possible (and where necessary), but without estimates of precision. The original analyses were based on a stratification using different sets of areas and, in some, additional strata such as depth or gear type. For this re-calculation, the estimated values for each area were scaled so as to have the same annual total as the published values. To enable totals to be calculated across all fisheries within each area, bycatch and discard estimates for years/fisheries where data have yet to become available were assumed to be equal to that of the last year for which an estimate has been published.

Estimates of the annual bycatch (and discards) of a wide range of individual species were also made in the most recent analysis of the jack mackerel fishery (Finucci et al. in prep), ling longline fishery (Finucci et al. 2020), hoki/hake/ling fishery (Anderson et al. 2019), orange roughy and oreo fisheries (Anderson et al. 2017a, Anderson & Finucci in press), and the scampi and arrow squid fisheries (Anderson & Edwards 2018) as well as in a more simplified manner for the remaining Deepwater Tier 1 fisheries (Finucci et al. 2019).

In some analyses, the apparent increase or decrease in bycatch of a species is likely to be the result of external factors including the introduction of new species to the QMS, new species-specific 3-letter codes to replace generic codes, and improvements in species identification over time; e.g., the increase in recorded bycatch of floppy tubular sponge in the hoki/hake/ling trawl fishery reflects the improved identification of sponges in more recent years, and use of the species-specific code for giant spider crab (GSC) instead of unspecified crabs (CRB) in the hoki/hake/ling trawl fishery. Some codes may also have been misused, e.g., among paddle crab species in the arrow squid fishery where the increase in recorded bycatch of the smooth red swimming crab (*Nectocarcinus bennetti*, NCB) appears to be at the expense of bycatch of the similar-looking *Ovalipes catharus*, PAD), which has a code which may be mistaken for a generic species code for paddle crabs.

The approach used in these analyses has relied heavily on an appropriate level and spread of observer effort being achieved, and this was examined in detail in each published report. Although details of bycatch and discards were recorded directly by vessel skippers for all fishing events through catch-effort forms, these data were generally inadequate for precise measurement of annual totals because the forms only require the top five or eight catch species to be reported, discard information is often not required, and they generally lacked the accuracy of identification and precision of observer data. Despite these inadequacies, annual bycatch totals were usually derived from catch-effort data but presented only as secondary estimates.

SURFACE LONGLINE FISHERIES

The estimation process used for surface longline fisheries up until the 2014–15 year was similar to that used for trawl and bottom longline fisheries, with each species assessed separately. Thus, CPUE was calculated as the number of fish observed caught per 1000 hooks set, stratified by fishing year, fleet (Foreign-licensed, Foreign-chartered, and Domestic), and area. CPUE was expressed using a ratio of means estimator (see Bradford 2002, Ayers et al. 2004). The total number of each species caught in each stratum was estimated by scaling up the CPUE to the total number of hooks set. These numbers were then summed across strata to give total annual catch estimates. An analytical estimator was used to calculate variance, using an adjustment to account for correlation between variance and the mean of the effort variable (after Thompson 1992). Additional estimates of annual bycatch within the standardised areas used for the offshore trawl and bottom longline fisheries are currently not available.

TROLL AND PURSE SEINE FISHERIES

Fish bycatch analyses in these fisheries are limited to annual summaries of observer recorded species catches, without any attempts to scale/apply observed catch rates to the total commercial fishery.

INSHORE FISHERIES

Some bycatch information is available from fishery characterisation studies (see Section 9.1) but there were no detailed analyses of bycatch and discards from inshore fishing principally because of the lack of observer data. Most of the analyses of bycatch and discards for offshore fisheries were reliant on observer data, e.g., Anderson 2012, 2013a, and similar analyses for inshore fisheries are not currently possible. Past observer coverage of inshore fisheries has been low (e.g., fewer than 2% of trawl tows were observed in 2009–10; Ramm 2012) and coverage has often been issue-focused – e.g., around 6% of set nets were observed in 2018–19 (Fisheries New Zealand data) to monitor Hector’s and Māui dolphin interactions – rather than to be representative of a fishery. There are also practical and logistical problems with placing observers on smaller inshore vessels, and other options are being explored to improve the monitoring of these fisheries.

Detailed fishing event data for inshore fishing, e.g., tow-by-tow catch and effort, were not collected by all vessels before 1 October 2007 using the statutory reporting system. Before 1 October 2007, smaller trawl (6–28 m), longline, and set net vessels used the Catch Effort and Landing Return (CELR) to collect daily summary catch-effort and landings data by General Statistical Area. From 1 October 2007 onwards, detailed data for each fishing event were collected using the new Trawl Catch and Effort Return (TCER), and this will be used to support analyses of bycatch in inshore trawl fisheries.

Electronic reporting is now implemented across all New Zealand fisheries (including for the inshore) and electronic monitoring is being implemented in a phased manner. Some progress has been made with estimating the bycatch of undersized fish, such as sub-legal-sized snapper (SNA), but some issues need to be resolved before electronic monitoring can provide all the information required to estimate fish and invertebrate bycatch.

1.4 INFORMATION BY FISHERY AREA

1.4.1 CHATHAM RISE (CHAT4)

The Chatham Rise is an important region for all the major offshore fisheries, except for southern blue whiting. Total non-target catch from offshore fisheries in this area has ranged from about 11 000 t to about 46 000 t, with generally decreasing amounts after 2000–01, then rising again after 2009–10 (Figure 9.2). In each year since 1995–96 the combined trawl fishery targeting hoki, hake, ling, silver warehou, and white warehou has been the main contributor to total non-target catch in this area. Prior to that most of the non-target catch from offshore fisheries was attributed to the orange roughy fishery. The arrow squid and scampi fisheries also contributed substantially in some years.

Total discards on the Chatham Rise from offshore fisheries has ranged from about 4000 t to about 25 000 t, with generally decreasing amounts after 2002–03 (Figure 9.3). In most years the largest contributor to discards by volume in this area was the hoki/hake/ling trawl fishery, although recent low discards in this fishery have increased the relative contribution of the arrow squid trawl, scampi trawl, and ling longline fisheries. In the orange roughy fishery, bycatch/discard rates were relatively low and progressively decreased in recent years; a large part of the bycatch in that fishery is composed by commercially valuable species – oreos in particular.

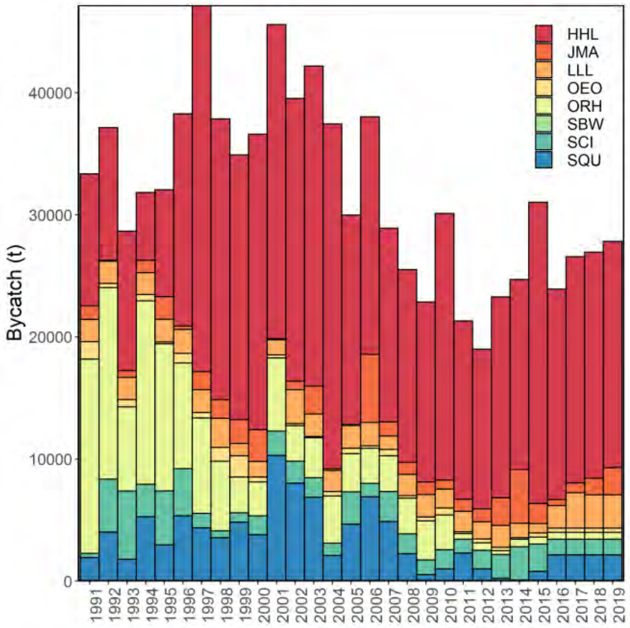


Figure 9.2: Estimated total annual non-target catch (source fisheries shown by bar colouration) in Chatham Rise offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

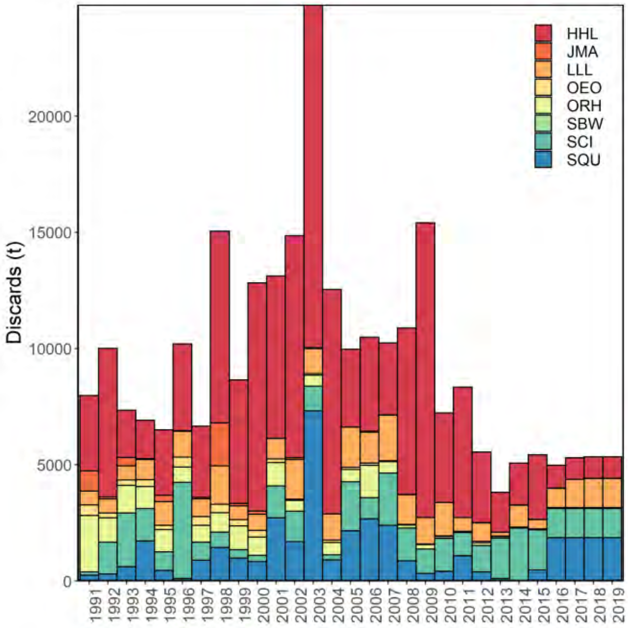


Figure 9.3: Estimated total annual discards (source fisheries shown by bar colouration) in Chatham Rise offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.2 SUB-ANTARCTIC (SUBA6)

The Sub-Antarctic is an important region for all the major offshore fisheries except for jack mackerel. Total non-target catch from offshore fisheries in the area has ranged from about 300 t to about 4500 t, with variable levels but generally lower since 2004–05 (Figure 9.4). In the past, major contributors have been the ling longline, and southern blue whiting, orange roughy, and hoki/hake/ling trawl fisheries. Most recently the hoki/hake/ling (including longline) fisheries have been the greatest contributors.

Total discards in the Sub-Antarctic from offshore fisheries have ranged from about 200 t to about 2200 t, with generally decreasing levels over time, especially after 2002–03 (Figure 9.5). Discards in the southern blue whiting fishery are high relative to bycatch due to the discarding of the target species (discarding of smaller fish is legal but is counted against the quota). Currently total discards are mostly split between the southern blue whiting and hoki/hake/ling trawl fisheries, and the ling longline fishery.

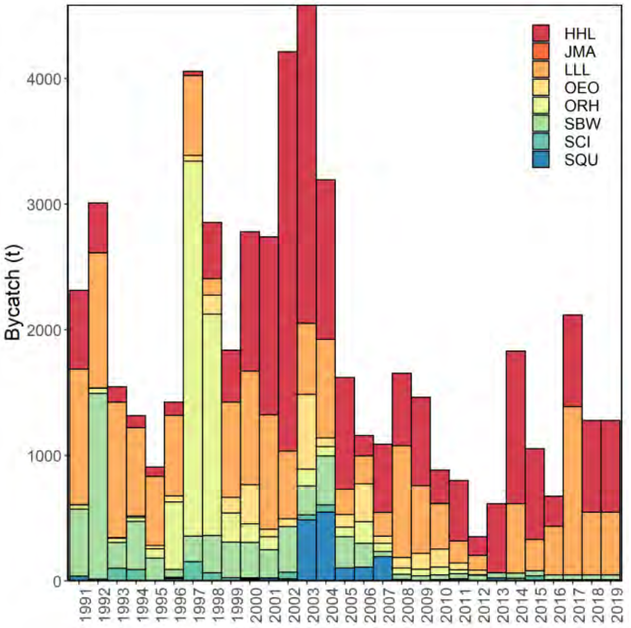


Figure 9.4: Estimated total annual non-target catch (source fisheries shown by bar colouration) in Sub-Antarctic offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.3 STEWART-SNARES SHELF (STEW5)

The Stewart-Snares shelf is an important region for the hoki/hake/ling and arrow squid trawl fisheries, as well as, to a minor extent, jack mackerel trawl fisheries, with smaller trawl fisheries also operating for oreo and orange roughy and longlines for ling. Total non-target catch in the Stewart-Snares shelf area from deepwater and middle-depth fisheries has ranged from about 3000 t to about 32 000 t per year, with the lowest values in the mid-1990s, but lower levels also after 2005–06 (Figure 9.6). The majority of this non-target catch, in all years except for 1994–95, has been from the arrow squid fishery, with most of the remainder coming from the hoki/hake/ling fishery and notable, but intermittent, contributions from the jack mackerel fishery.

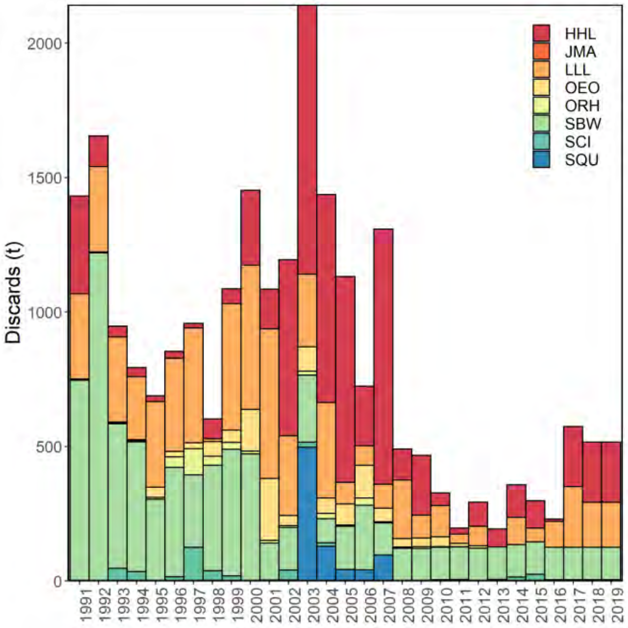


Figure 9.5: Estimated total annual discards (source fisheries shown by bar colouration) in Sub-Antarctic offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

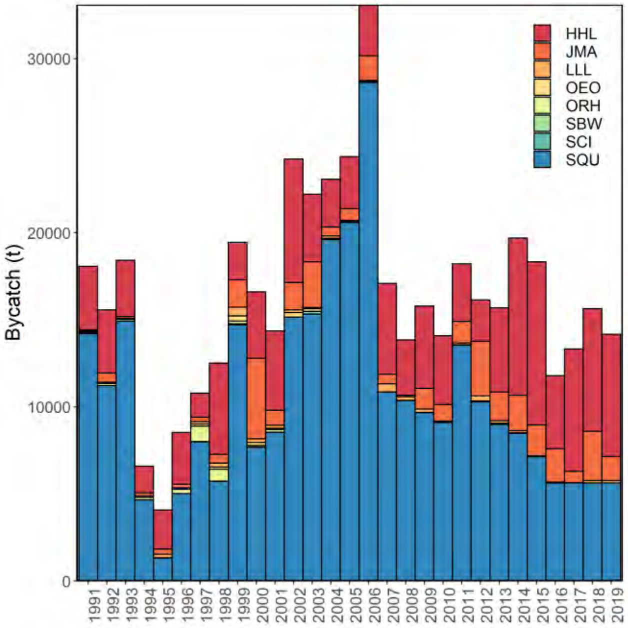


Figure 9.6: Estimated total annual non-target catch (source fisheries shown by bar colouration) in Stewart-Snares shelf offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

Total discards in the Stewart-Snares shelf area from offshore fisheries has ranged from about 500 t to about 7000 t, with lower values in the mid-1990s (Figure 9.7). Currently discarding in this area is mostly attributed to the arrow squid and hoki/hake/ling fisheries.

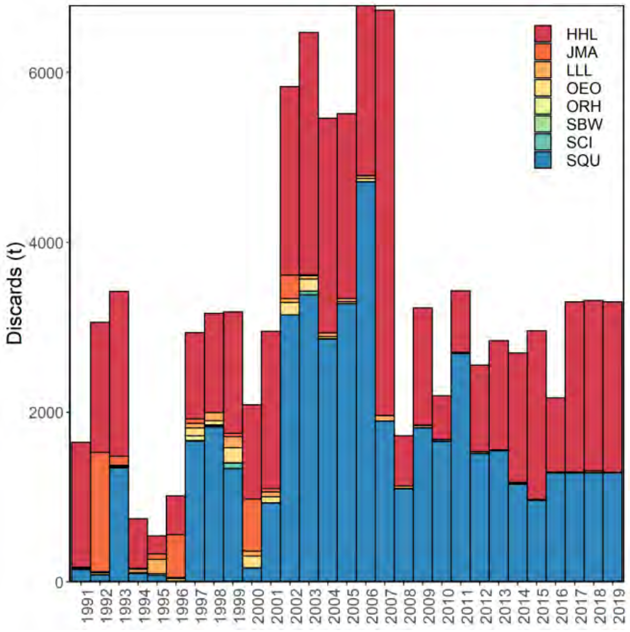


Figure 9.7: Estimated total annual discards (source fisheries shown by bar colouration) in Stewart-Snares shelf offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.4 AUCKLAND ISLANDS (SQUAK6)

The main fisheries currently operating in the Auckland Islands region are the scampi and arrow squid trawl fisheries, with smaller fisheries for hoki/hake/ling also present. An orange roughy fishery operated in the region from the mid-1990s to the early 2000s but has been very minor in recent years. Total non-target catch in the Auckland Islands area from offshore fisheries has ranged from about 750 t to about 7500 t per year but has fluctuated between about 1800 t and 4200 t since 1997–98 (Figure 9.8). The main contributors to non-target catch in this area have been the scampi and arrow squid fisheries, as well as the orange roughy fishery during the 1990s. Currently the main contributing fishery is the arrow squid fishery.

Total discards in the Auckland Islands area from offshore fisheries have ranged widely, from about 100 t to about 3000 t per year, resulting mostly from the scampi and arrow squid fisheries (Figure 9.9). Current levels are typical of the last 15 years, at about 1600 t per year.

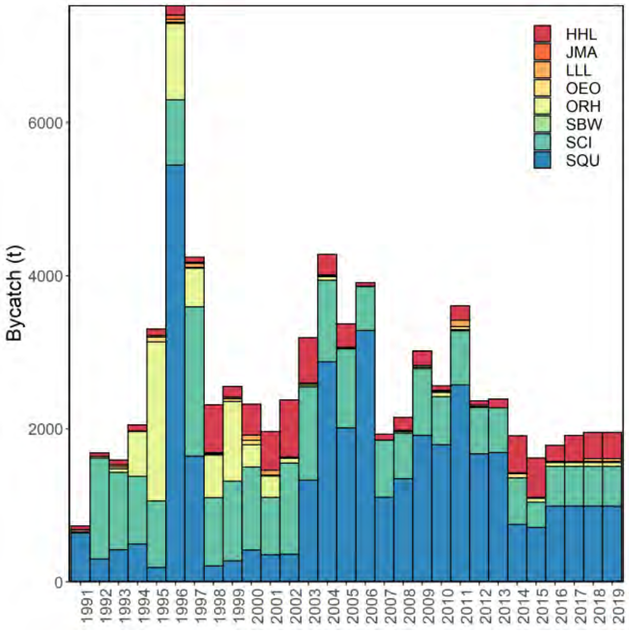


Figure 9.8: Estimated total annual non-target catch (source fisheries shown by bar colouration) in the Auckland Islands offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

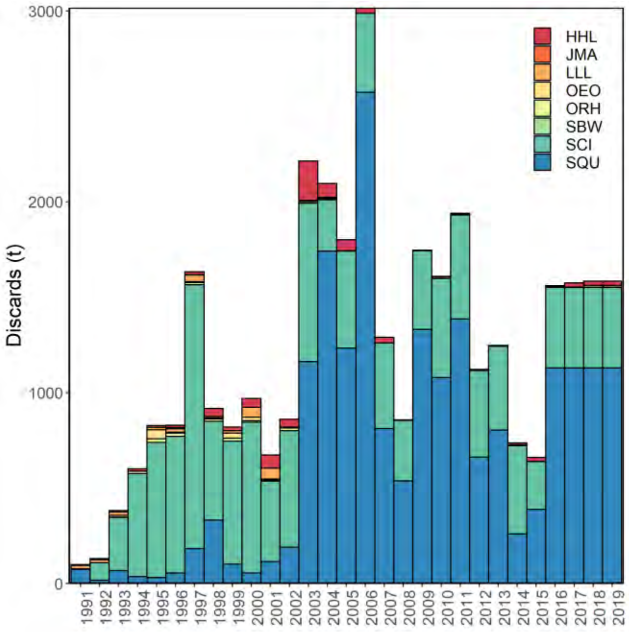


Figure 9.9: Estimated total annual discards (source fisheries shown by bar colouration) in the Auckland Islands offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.5 PUYSEGUR (PUYS5)

Most offshore fisheries have operated at some time in the Puysegur area, with non-target catch mainly attributed to the orange roughy fishery in the early 1990s, to the arrow squid fishery in the early 2000s, and to the hoki/hake/ling fishery since the mid-2000s. Total non-target catch in the area from offshore fisheries has ranged from about 600 t to about 4600 t per year, with generally decreasing amounts between about 2000–01 and 2007–08 followed by a consistent level of about 1500–2000 t per year since (Figure 9.10). Annual non-target catch is currently attributed mostly to the hoki/hake/ling trawl fishery, with smaller contributions from the arrow squid and orange roughy trawl fisheries and ling longline fisheries.

Total discards in the Puysegur area from offshore fisheries have ranged from about 100 t to about 4200 t per year with mostly lower amounts after 2002–03, a year in which increased effort in the arrow squid fishery coupled with some large discards resulted in a high estimate for that fishery (Figure 9.11). Discards were otherwise mostly attributable to the hoki/hake/ling fishery except for a few years in the early 2000s when the arrow squid fishery was operating more in this area.

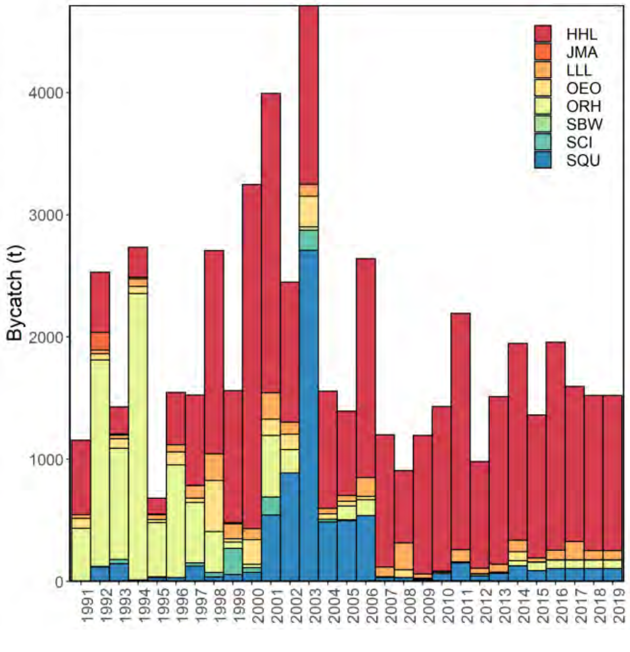


Figure 9.10: Estimated total annual bycatch (source fisheries shown by bar colouration) in Puysegur offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

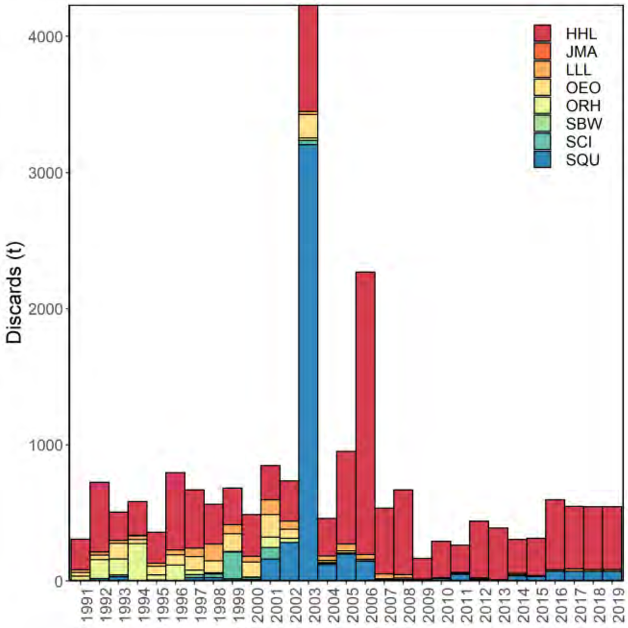


Figure 9.11: Estimated total annual discards (source fisheries shown by bar colouration) in Puysegur offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.6 WEST COAST SOUTH ISLAND (WCSI7)

The main fisheries in this area are the trawl fisheries for hoki/hake/ling and jack mackerel, as well as a small ling longline fishery and, occasionally, orange roughy and arrow squid trawl fisheries. Currently most non-target catch can be attributed to the hoki/hake/ling fishery. Total non-target catch in the west coast South Island area from offshore fisheries has ranged from about 3000 t to about 20 000 t per year, with generally decreasing amounts between about 2000–01 and 2009–10, then increasing levels over the last few years (Figure 9.12).

Total discards in the west coast South Island area from offshore fisheries has ranged from about 700 t to about 9000 t per year, with generally decreasing amounts since about 1994–95 and a relatively low contribution from the jack mackerel fishery compared with bycatch (Figure 9.13). Total discards have been below 4500 t per year since 1997–98, attributed mostly to the hoki/hake/ling fishery, with small and variable contributions from the jack mackerel and orange roughy trawl fisheries and ling longline fisheries.

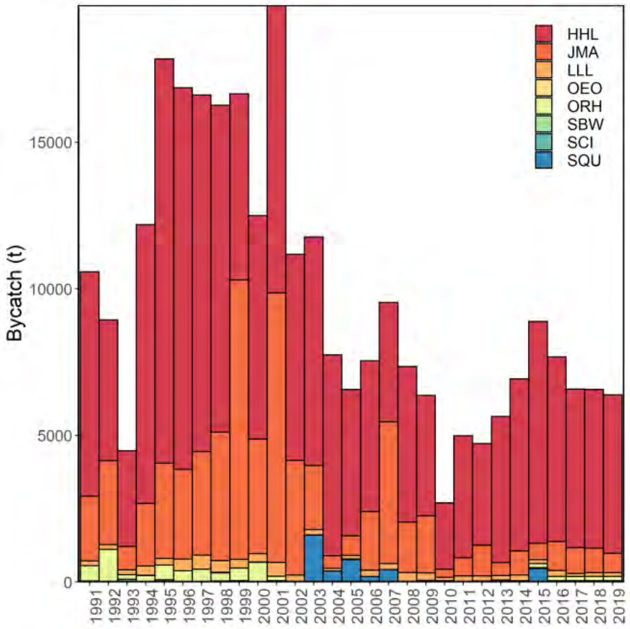


Figure 9.12: Estimated total annual non-target catch (source fisheries shown by bar colouration) in west coast South Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

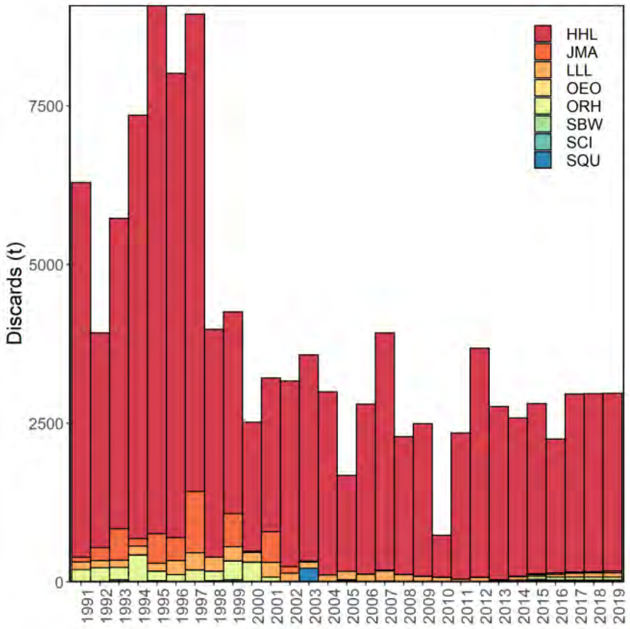


Figure 9.13: Estimated total annual discards (source fisheries shown by bar colouration) in west coast South Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.7 WEST COAST NORTH ISLAND (WCNI9)

The dominant offshore fishery off the west coast North Island region is currently the jack mackerel trawl fishery, with fisheries for orange roughy and arrow squid operating mostly before 2003–04. Total non-target catch in the west coast North Island area from offshore fisheries has ranged from about 1100 t to about 13 000 t per year, with decreasing amounts from 2003–04 to 2009–10 and then stable afterwards (Figure 9.14). In most years almost all of the non-target catch can be attributed to the jack mackerel fishery, but with moderate contributions from the orange roughy and arrow squid fisheries in some earlier (pre 2004) years.

Total discards in the west coast North Island area from offshore fisheries has ranged from about 20 t to about 1400 t per year, with generally stable levels of 100–300 t per year since 2003–04 (Figure 9.15). The jack mackerel fishery contributes relatively less to total discards than it does to bycatch, because of a high proportion of QMS species in the catch, but still dominates in most years.

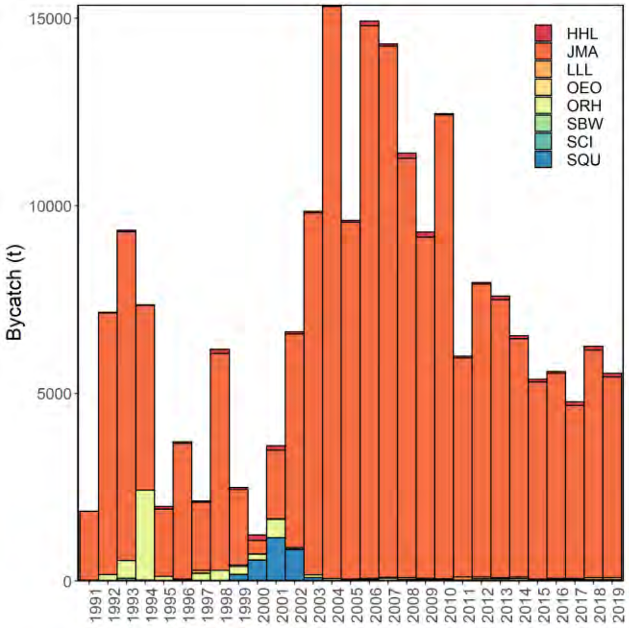


Figure 9.14: Estimated total annual bycatch (source fisheries shown by bar colouration) in west coast North Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

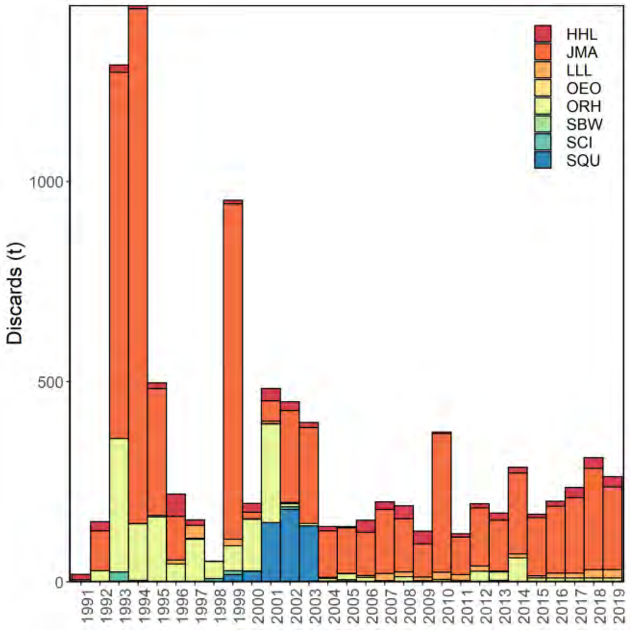


Figure 9.15: Estimated total annual discards (source fisheries shown by bar colouration) in west coast North Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.8 NORTHLAND (NORTH1)

Offshore fisheries in the Northland region are mainly limited to a trawl fishery for scampi, and smaller fisheries for orange roughy and hoki/hake/ling. Total non-target catch in the area from offshore fisheries has ranged from about 500 t to about 4000 t per year, but with generally stable levels of less than 1400 t per year since about 1997–98 (Figure 9.16). In most years non-target catch was mainly associated with the scampi fishery, with smaller amounts from the hoki/hake/ling fishery and a large contribution from the orange roughy fishery in 1996–97. Other offshore fisheries are minor in this area and currently total annual non-target catch is less than 1000 t, split mostly between the scampi and hoki/hake/ling fisheries.

Total discards in the Northland area from offshore fisheries have ranged from about 300 t to about 1000 t per year, with levels of about 300–600 t per year over the last several years (Figure 9.17). Discards in this area are dominated by the scampi fishery in all years except 1996–97, when the hoki/hake/ling fishery contributed more.

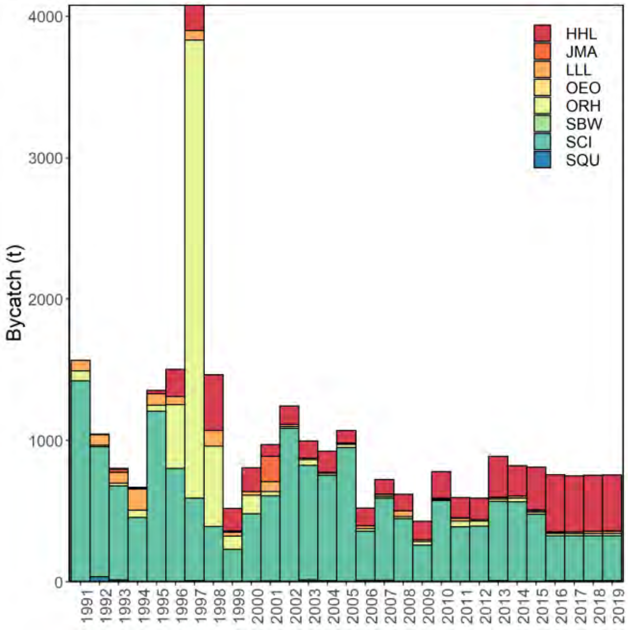


Figure 9.16: Estimated total annual non-target catch (source fisheries shown by bar colouration) in Northland offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

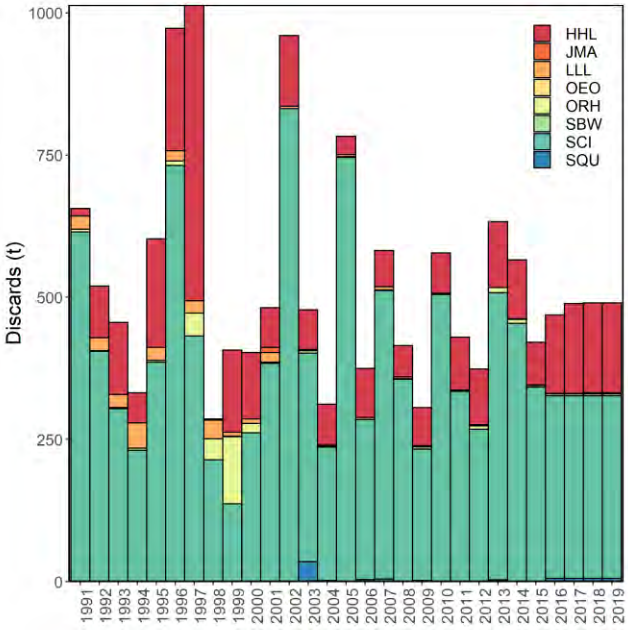


Figure 9.17: Estimated total annual discards (source fisheries shown by bar colouration) in Northland offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.9 EAST COAST NORTH ISLAND (EAST2)

The main offshore fisheries operating in the East Coast North Island area have been the scampi, hoki/hake/ling, and orange roughy trawl fisheries and ling longline fisheries. Total non-target catch in the area from offshore fisheries has ranged from about 1000 t to about 7500 t per year, with generally decreasing levels after 1997–98 (Figure 9.18). Most of the non-target catch comes from the hoki/hake/ling and scampi fisheries, with larger contributions from the orange roughy fishery before 2001–02, and lower contributions from the ling longline fishery.

Total discards in the East Coast North Island area from offshore fisheries has ranged from about 300 t to about 2700 t per year, with generally lower levels after 2001–02 (Figure 9.19). The scampi and orange roughy fisheries contributed more to discards than to non-target catch in this area, and in most years only a small proportion of total discards was attributable to the hoki/hake/ling fishery. Current annual discards are about 400–500 t, mostly associated with the scampi fishery.

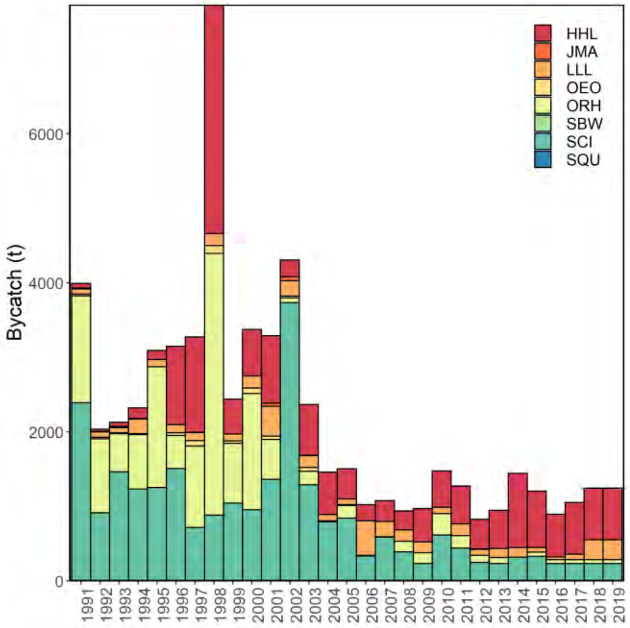


Figure 9.18: Estimated total annual non-target catch (source fisheries shown by bar colouration) in East Coast North Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

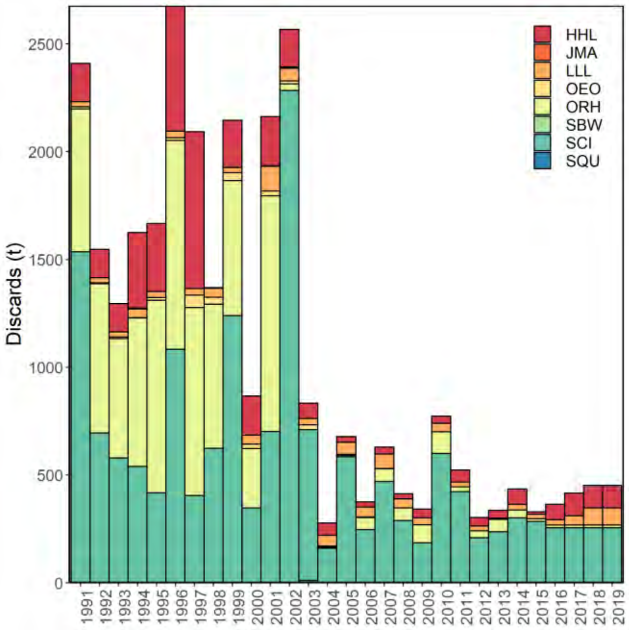


Figure 9.19: Estimated total annual discards (source fisheries shown by bar colouration) in East Coast North Island offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.4.10 COOK STRAIT (COOK8)

The main fishery in the Cook Strait area has been the hoki/hake/ling trawl fishery, with this fishery contributing the great majority of non-target catch in most years. Total non-target catch in the Cook Strait area from offshore fisheries has ranged from about 200 t to about 6200 t per year, with generally decreasing levels after 1995–96 (Figure 9.20). The orange roughy fishery operating on the fringes of this area also contributed substantially to total annual non-target catch during the early 1990s. Currently total annual bycatch is less than 500 t, almost all from the hoki/hake/ling fishery.

Total discards in the Cook Strait area from offshore fisheries have ranged from about 200 t to about 4000 t per year, with generally decreasing levels since about 1995–96 (Figure 9.21). Discards in this area have virtually all been associated with the hoki/hake/ling fishery. Current discard levels are about 200–300 t per year.

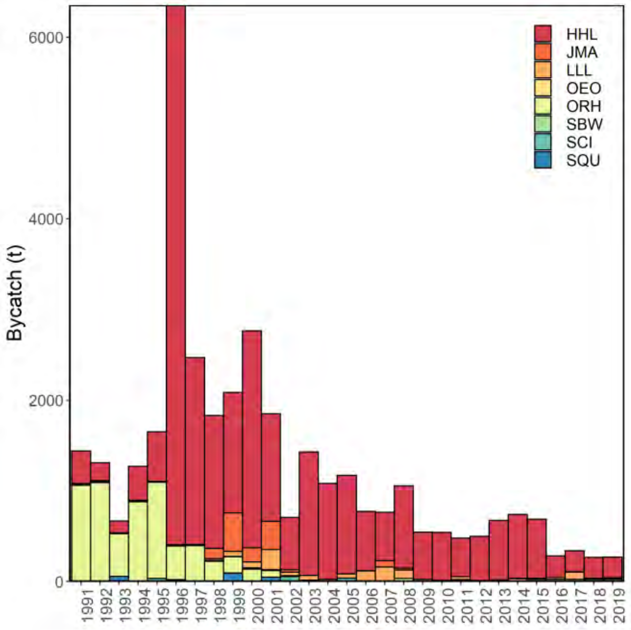


Figure 9.20: Estimated total annual non-target catch (source fisheries shown by bar colouration) in Cook Strait offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

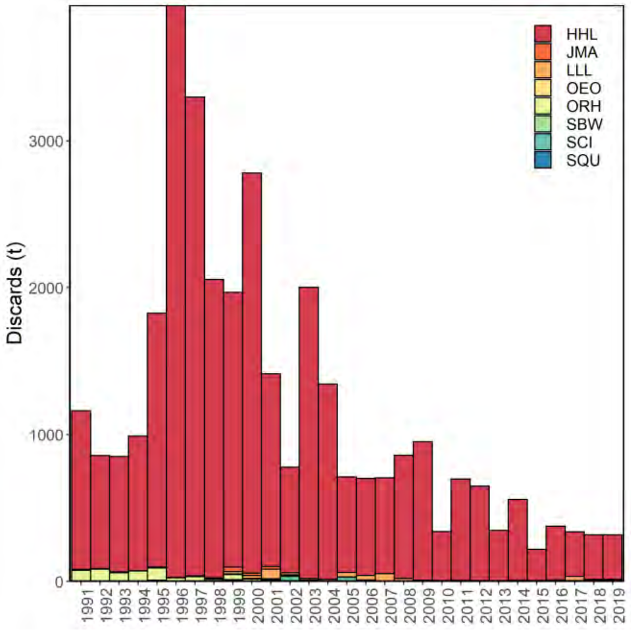


Figure 9.21: Estimated total annual discards (source fisheries shown by bar colouration) in Cook Strait offshore fisheries. For fisheries abbreviations see Table 9.1. HHL is hoki/hake/ling mixed trawl fishery.

1.5 INFORMATION BY FISHERY

1.5.1 ARROW SQUID TRAWL FISHERY

Since 1990–91 the level of observer coverage in this fishery was 6–97% of the total annual catch and was relatively high, 28–40%, from 2006–07 to 2010–11 due to the management measures imposed for the protection of New Zealand sea lions (*Phocarctos hookeri*) (Ministry for Primary Industries 2012). After 2011–12 and up until 2018–19, the observer coverage further increased to 90–97%, due to 100% coverage requirements for Foreign Charter Vessels (FCVs). After 2016, FCVs could only operate in this area under a New Zealand flag, but 100% observer coverage is still applied to Foreign Owned Vessels (FOV) flagged to New Zealand. This coverage was well spread across the fleet and annually 10–71% of all vessels targeting arrow squid were observed, with this fraction increasing over time. Although observers covered the full size range of vessels operating in the fishery, the smallest vessels were slightly undersampled and the largest oversampled.

The observer effort was mostly focused on the main arrow squid fisheries around the Auckland Islands Shelf and Stewart-Snares shelf, but the smaller fisheries on the Puysegur Bank and off Banks Peninsula were also covered, although less consistently. Observer coverage was more focused on the central period of the arrow squid season, February to April, than the fleet was in general – with fishing in January and May slightly undersampled. The most recent assessment of non-target catch and discards in this fishery (Anderson & Edwards 2018) was based on a statistical model approach using a combination of standard areas, fishing years, net type, and meal plant usage as model covariates, and covered the period from 2002–03 to 2015–16. The key categories of catch/discards examined were: all QMS species combined, all non-QMS species combined, and all invertebrate species combined, with membership of these categories adjusted from year to year as species were added to the QMS. Since 1990–91, nearly 600 bycatch species or species groups were identified by observers in this fishery, most being non-commercial species (including invertebrate species) caught in low numbers. Arrow squid accounted for about 79% of the total estimated catch recorded by observers. The main non-target catch species or species groups were the QMS species barracouta (9.1%), silver warehou (3.3%), spiny dogfish (1.7%), and red cod (1.2%); and of these only spiny dogfish were generally discarded (Figure 9.22), which is legally allowed under Schedule 6 of the Fisheries Act. Total annual non-target catch in the arrow squid fishery for 2002–03 to 2015–16 was about 9000–40 000 t, with a significant downward trend (Figure 9.23). The majority of the non-target catch comprised QMS species, with less than 1000 t of non-QMS species and invertebrate species non-target catch in most years. Of the other (non-squid) invertebrate groups, crustaceans (1.2%), in particular smooth red swimming crab (*Nectocarcinus bennetti*) (0.8%), were caught in the greatest amounts and were mostly discarded. Smaller amounts of octopus and squid, sponges, cnidarians, and echinoderms were also often caught and discarded.

When combined into broader taxonomic groups, bony fish (excluding rattails, tuna, flatfish, and eels) contributed the most non-target catch (15.9% of the total catch), followed by sharks and dogfishes (1.9%), morid cods (1.2%), crustaceans (1.2%), and rattails (0.3%).

More than 75% of the sharks, dogfishes, and rattails were discarded, whereas most of the catch of the other groups was retained. The fish species discarded in the greatest amounts were spiny dogfish, rattails, and silver dory. Of the invertebrates, most were discarded, but crustaceans, octopuses, and other molluscs were sometimes retained.

Estimated total annual discards ranged from about 1300 t in 2013–14 to about 16 000 t in 2002–03 and, like bycatch, showed a significant decline over time (Anderson & Edwards 2018). Discards were an even mix of QMS species (about 44% for all years) and non-QMS species (41%), with lesser amounts of invertebrate species (15%) and arrow squid (8%) (Figure 9.24)

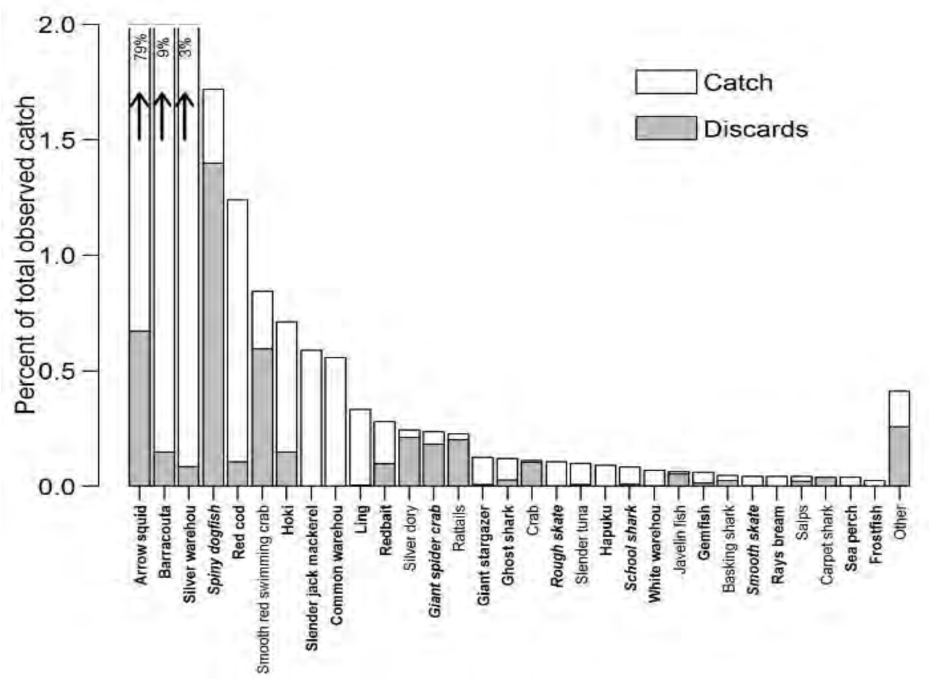


Figure 9.22: Percentage of the total catch contributed by the main non-target catch species (those representing 0.02% or more of the total catch; white) in the observed portion of the arrow squid fishery, and the percentage discarded (grey), 1 October 2001 to 30 September 2016 (Anderson & Edwards 2018). The 'Other' category is the sum of all non-target catch species representing less than 0.02% of the total catch. QMS species are shown in bold, Schedule 6 species are in italics.

TRENDS IN ESTIMATED BYCATCH BY SPECIES FROM THE ARROW SQUID TRAWL FISHERY

Finucci et al. (2019) estimated the level of the main individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were barracouta (*Thyrsites atun*, BAR), silver warehou (*Seriolella punctata*, SWA), and spiny dogfish (*Squalus acanthias*, SPD).
- Of the 347 bycatch species examined, 15 showed a significant decrease in catch over time and 29 had a significant increase in catch.
- The species showing the greatest declines were paddle crab (*Ovalipes catharus*, PAD), jack mackerels (*Trachurus* spp., JMA), and thresher shark (*Alopias vulpinus*, THR) (Figure 9.25).
- The species showing the greatest increases were giant spider crab (*Jacquinotia edwardsii*, GSC), smooth red swimming crab (*Nectocarcinus*

bennetti, NCB), and GON (*Gonorynchus forsteri* & *G. greyi*) (Figure 9.25).

- The decrease in PAD and the corresponding increase in NCB is a reporting artefact caused by the use of different codes following the introduction of PAD to the QMS.

1.5.2 HOKI/HAKE/LING TRAWL FISHERY

Earlier analyses were limited to the hoki target fishery but were subsequently expanded to cover non-target catch and discards from hoki, hake, and ling target fisheries combined, and most recently included silver warehou and white warehou in the definition of the target fishery; hoki nevertheless dominates this fishery, accounting for over 90% of the catch (Anderson et al. 2019). The most recent non-target catch and discard report for this fishery included a more rigorous analysis of data, with the use of additional covariates (gear type and vessel class) (Anderson et al. 2019). Between 2002–03 and 2016–17, observer sampling levels have been highest off the west coast South Island, in the Sub-Antarctic, and at Puysegur, with lower levels in the Chatham Rise area. These areas comprise the majority of the fishery in any year; little sampling has occurred outside these main fishery areas. Observer coverage was spread throughout the year, with modest increases in effort during the hoki spawning season (June to early September), meaning that the spawning period was relatively undersampled and the off-season was oversampled in many years. Observer effort was found to be more closely matched to overall effort in some years between 2010–11 and 2015–16, which is likely to be due to monitoring requirements for foreign charter vessels. Hoki, hake, and ling accounted for 85% (73%, 6.7%, and 5.2%, respectively) of the total observed catch from trawls targeting hoki, hake, and ling between 2002–03 and 2016–17. The remaining 15% comprised a large range of species, in particular silver warehou (3.9%), javelinfish (1.9%), rattails (1.6%), spiny dogfish (1.4%), and white warehou (1.3%) (Figure 9.26). In total, over 800 species or species groups were identified by observers, the majority of these species were non-QMS species caught in low numbers. Chondrichthyans in general, often unspecified but including shovelnose dogfish and Baxter's dogfish, accounted for much of the non-commercial catch. Echinoderms, squids, crustaceans, and other unidentified invertebrates were also well represented in the non-target catch of this fishery.

Total non-target catch in the hoki, hake, and ling fishery between 1990–91 and 2016–17 was 18 000–50 000 t per year (compared with the combined total landed catch of hoki, hake, and ling of about 100 000–300 000 t) (Anderson et al. 2019). Overall, total non-target catch increased during the 1990s to a peak in the early 2000s, then declined slowly. Annual non-target catch for the 1990–91 to 2016–17 period was also estimated for QMS species, non-QMS species, and invertebrates. Roughly similar amounts of QMS species and non-QMS species were caught overall and each showed a similar pattern over time to total non-target catch; invertebrate catch was less than 1000 t in most years, but peaked at about 1800 t in 2001–02 (Figure 9.27).

Total annual discard estimates for 1990–91 to 2016–17 were 5000–25 000 t per year, and the main species observed discarded included spiny dogfish, rattails, javelinfish, and hoki (Anderson et al. 2019). Estimated annual discards of the target species combined ranged from 76–2340 t per year, with increasing levels since 2007–08. Estimates of total annual discards were variable but have generally declined over time, from 25 000 t in 2002–03 to 5000–8000 t in the most recent five years of the time series (Figure 9.28). Discard rates have been shown to be strongly influenced by the use of fishmeal plants on fishing vessels; with discards of non-commercial species on factory vessels without meal plants up to twice the level of discards for vessels with meal plants (Ballara et al. 2010). Vessel class is also very influential, with greater rates of discarding attributed to FOVs (former FCVs), compared with other vessel types (probably through the lack of fishmeal plants on board of vessels owned/chartered by some nations).

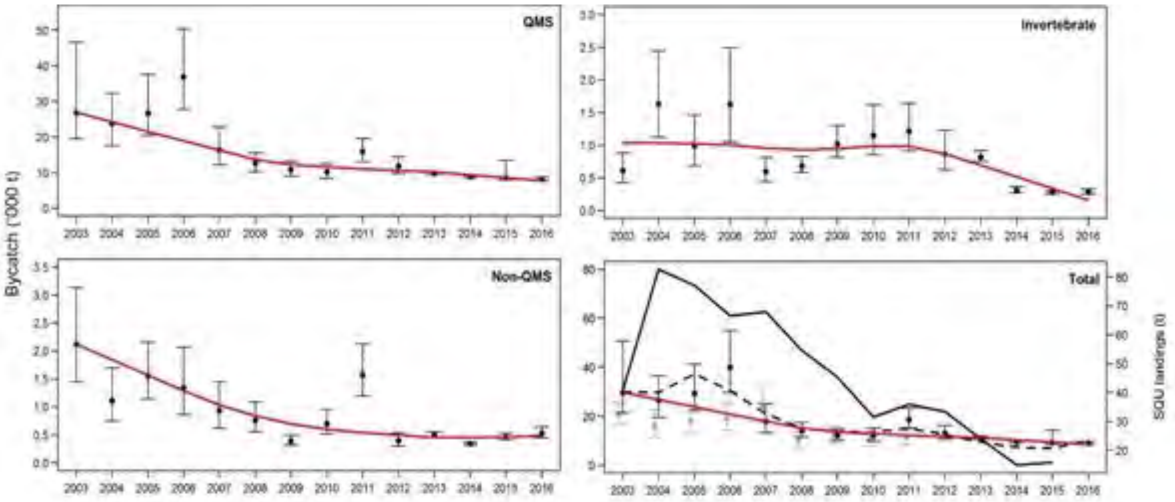


Figure 9.23: Annual estimates of non-target catch in the arrow squid trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards 2018). Also shown (in grey) are estimates of total non-target catch calculated for 2002–03 to 2010–11 (Anderson 2013a). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported trawl-caught landings of arrow squid, and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

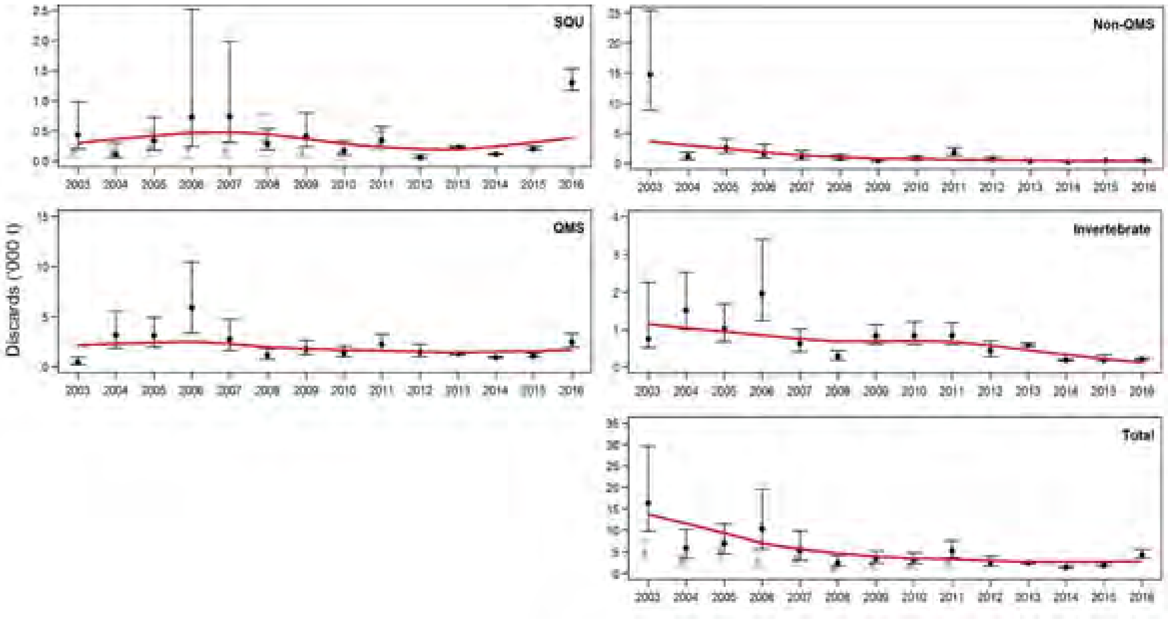


Figure 9.24: Annual estimates of discards in the arrow squid trawl fishery, for arrow squid (SQU), QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards 2018). Also shown (in grey) are estimates of arrow squid and total discards calculated for 2002–03 to 2010–11 (Anderson 2013a). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

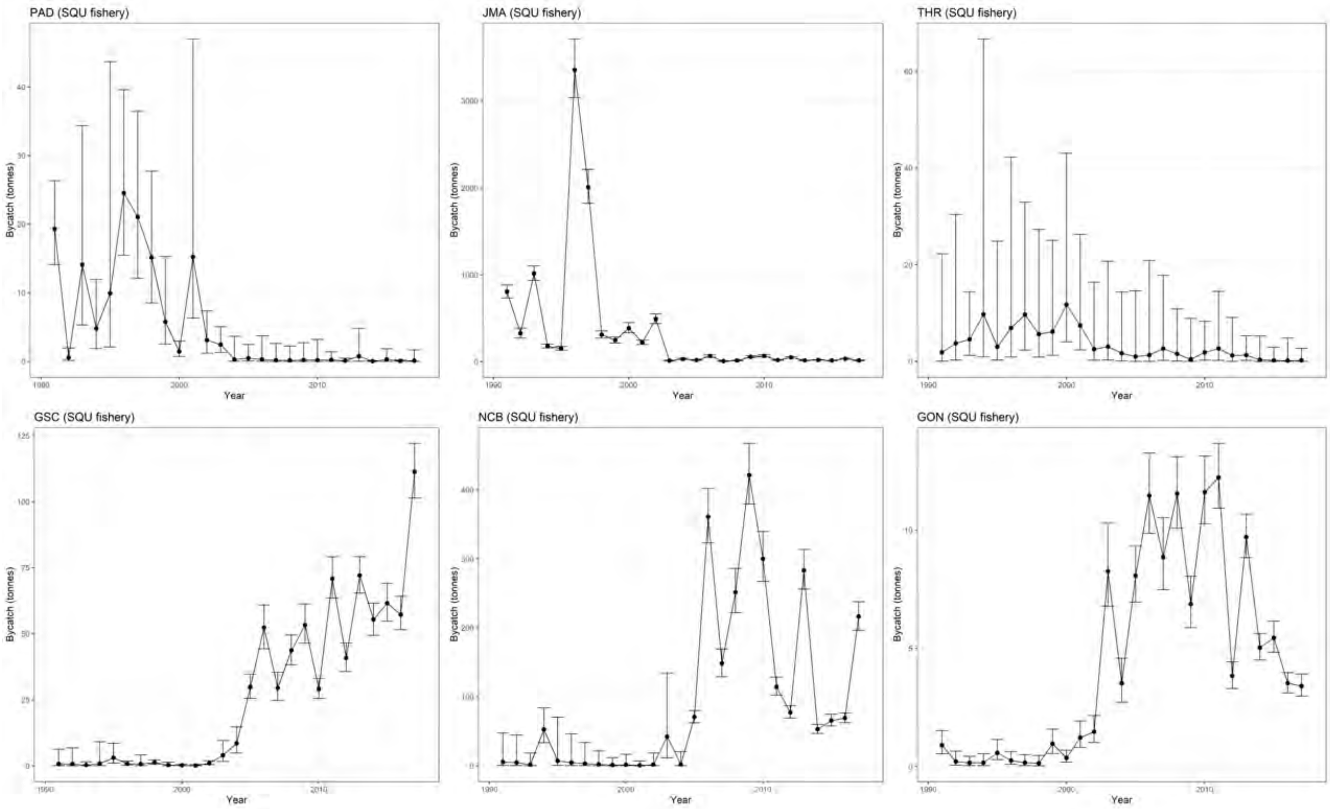


Figure 9.25: Annual non-target catch estimates in the target arrow squid trawl fishery for the species that have the most non-target catch between 1990–91 and 2016–17, with 95% CIs, in descending order of total catch (Finucci et al. 2019). See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots.

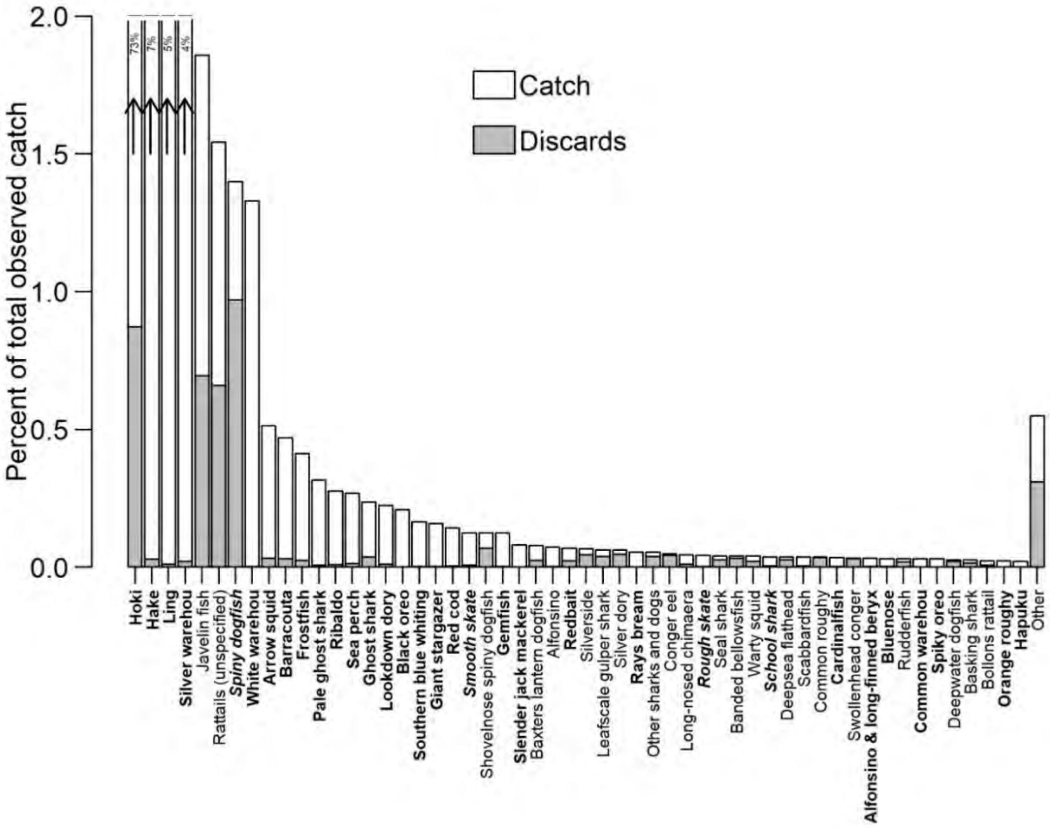


Figure 9.26: Percentage of the total catch contributed by the main non-target catch species (those representing 0.05% or more of the total catch; white) in the observed portion of the hoki/hake/ling fishery (1990–91 to 2012–13) and the percentage discarded (grey; Anderson et al. 2019). QMS species are shown in bold.

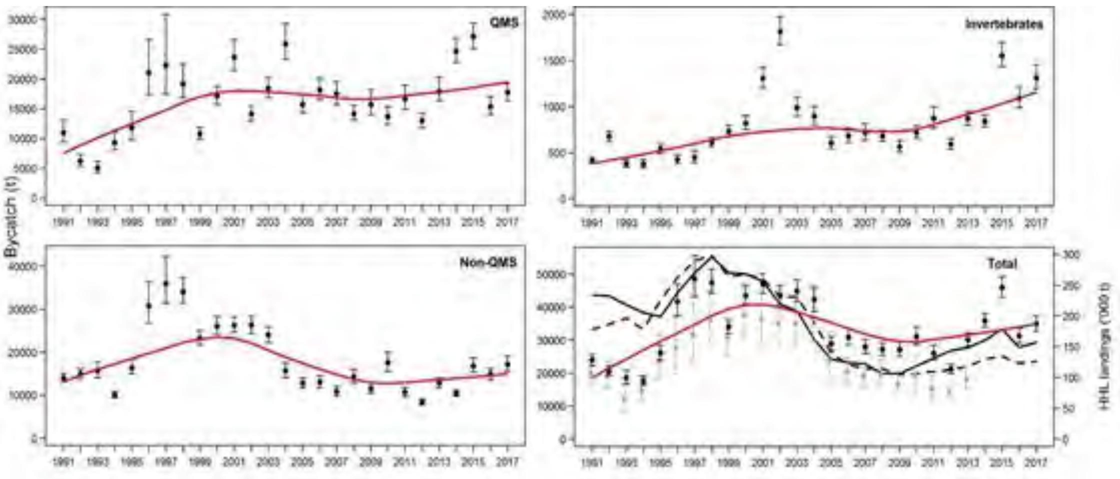


Figure 9.27: Annual estimates of non-target catch in the target hoki, hake, and ling trawl fishery, calculated for QMS species, non-QMS species, invertebrates, and overall for 1990–91 to 2016–17 (Anderson et al. 2019). Also shown (in grey) are earlier estimates of non-target catch (Ballara & O'Driscoll 2015). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom right panel the solid black line shows the total annual reported trawl-caught landings of hoki, hake, or ling and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

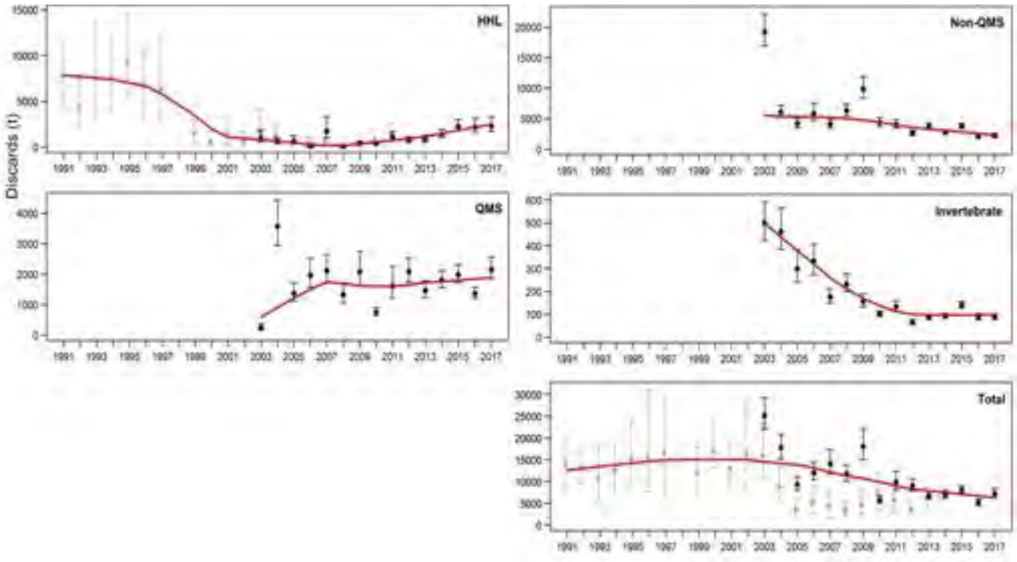


Figure 9.28: Annual estimates of discards in the hoki, hake, ling, silver warehou, or white warehou mixed target trawl fishery, by species category, for 2002–03 to 2016–17 (black dots) (Anderson et al. 2019). Also shown (in grey) are earlier estimates of the mixed target fishery and total discards calculated for 1991–92 to 2012–13 (Ballara & O'Driscoll 2015). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards

TRENDS IN BYCATCH BY SPECIES FROM THE HOKI, HAKE, AND LING TRAWL FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species non-target catch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were javelinfish (*Lepidorhynchus denticulatus*, JAV), unspecified rattails (Macrouridae, RAT), and silver warehou (*Seriolella punctata*, SWA).
- Of the 493 non-target catch species examined, 35 had a significant decrease in catch over time and 83 a significant increase in catch.
- The species showing the greatest decline were unspecified skates (SKA), lanternshark (*Etmopterus* spp., ETM), and moonfish (*Lampris guttatus*, MOO) (Figure 9.29). Notably SKA and ETM are generic

codes that have been replaced by more specific codes, which probably explains these declines.

- The species showing the greatest increase were umbrella octopus (*Opisthoteuthis* spp., OPI) Tam O'Shanter sea urchins (Echinothuriidae & Phormosomatidae, TAM), and floppy tubular sponge (*Hyalascus* sp., HYA) (Figure 9.29).

1.5.3 JACK MACKEREL TRAWL FISHERY

Estimates of annual non-target catch in this fishery are available for fishing years up to 2018–19, with the most recent analysis focusing on the 2002–03 to 2018–19 period using the statistical model methods of estimation (Finucci et al. (2022). The annual level of observer coverage in this fishery was 8–39% of the target fishery catch before 2007–08 but rapidly increased to be 70–95% after 2010–11. This elevated level of coverage was due to a commitment by the Ministry to full observer coverage on FCVs (and subsequently FOVs), which have historically taken a large part of the catch in this fishery (Ministry for Primary Industries 2013b), and for the purpose of better monitoring the incidental captures of common dolphin (*Delphinus delphis*). Observer effort in each year has generally been focused on the main fishery, off the west coasts of the North Island and the northern part of the South Island, with some additional coverage on the Stewart-Snares shelf and Chatham Rise fisheries. In the past this was variable, however, and in 2003–04 and 2004–05 there were only 12 trawls observed outside the western fishery (notably since 2002–03 over 90% of the effort in this fishery has been in the west coast fisheries). The fishery occurs mostly in October–February and April–August, and observer coverage has been well matched to the main fishing periods in recent years.

Jack mackerel species comprised 78% of the total observed catch from all trawls targeting jack mackerel from 2002–03 to 2018–19. The remaining 22% mostly comprised other QMS species: especially barracouta (11%), blue mackerel (3.1%), frostfish (3.0%), and redbait (2.4%) (Figure 9.30). Overall, about 370 species or species groups were identified by observers during this period, many of which were non-QMS species caught in small numbers. The species most discarded was the spiny dogfish[1] (which entered the QMS and Schedule 6 in October 2004), comprising about 0.2% of the total catch. There has been an increasing proportion of Schedule 6 species discarding attributed to kingfish since 2015–16. Of the invertebrates, only molluscs (mostly arrow squid) were observed caught in substantial amounts (about 1151 t) and these were mostly retained. Lesser amounts of cnidarians, sponges, and echinoderms were observed caught (about 16 t in total), and almost all were discarded.

Total non-target catch in the jack mackerel trawl fishery from 2002–03 to 2018–19 was 7105–24 269 t annually. Non-target catch has mainly comprised QMS species, which shows a declining trend over time (Figure 9.31).

Total annual discards annually varied between 93 t and 359 t between 2002–03 and 2018–19, with some increasing trend since 2012–13 (Finucci et al. 2022). Estimates were generally well matched with the previous analysis (Anderson et al. 2017b). Discards comprised mainly QMS species, which have also shown an increasing trend in recent years (Figure 9.32).

TRENDS IN BYCATCH BY SPECIES FROM THE JACK MACKEREL TRAWL FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were barracouta (BAR), frostfish (*Lepidopus caudatus*, FRO), and blue mackerel (*Scomber australasicus*, EMA).
- Of the 163 bycatch species examined, 30 showed a significant decrease in catch over time and 7 showed an increase.
- Species with significant declines included dark ghost shark (*Hydrolagus novaezealandiae*, GSH), red cod (*Pseudophycis bachus,* RCO), and sea perch (*Helicolenus* spp., SPE) (Figure 9.33).

Species showing significant increases included albacore tuna (*Thunnus alalunga*, ALB), pilchard (*Sardinops sagax*, PIL), and kingfish (*Seriola lalandi*, KIN) (Figure 9.33)

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[1] Notably it is legal to discard spiny dogfish under Schedule 6 of the Fisheries Act. Dogfish (and other sharks) can be only partly used in fishmeal operations, because their ammonia content tends to spike the meal if added in large quantities.

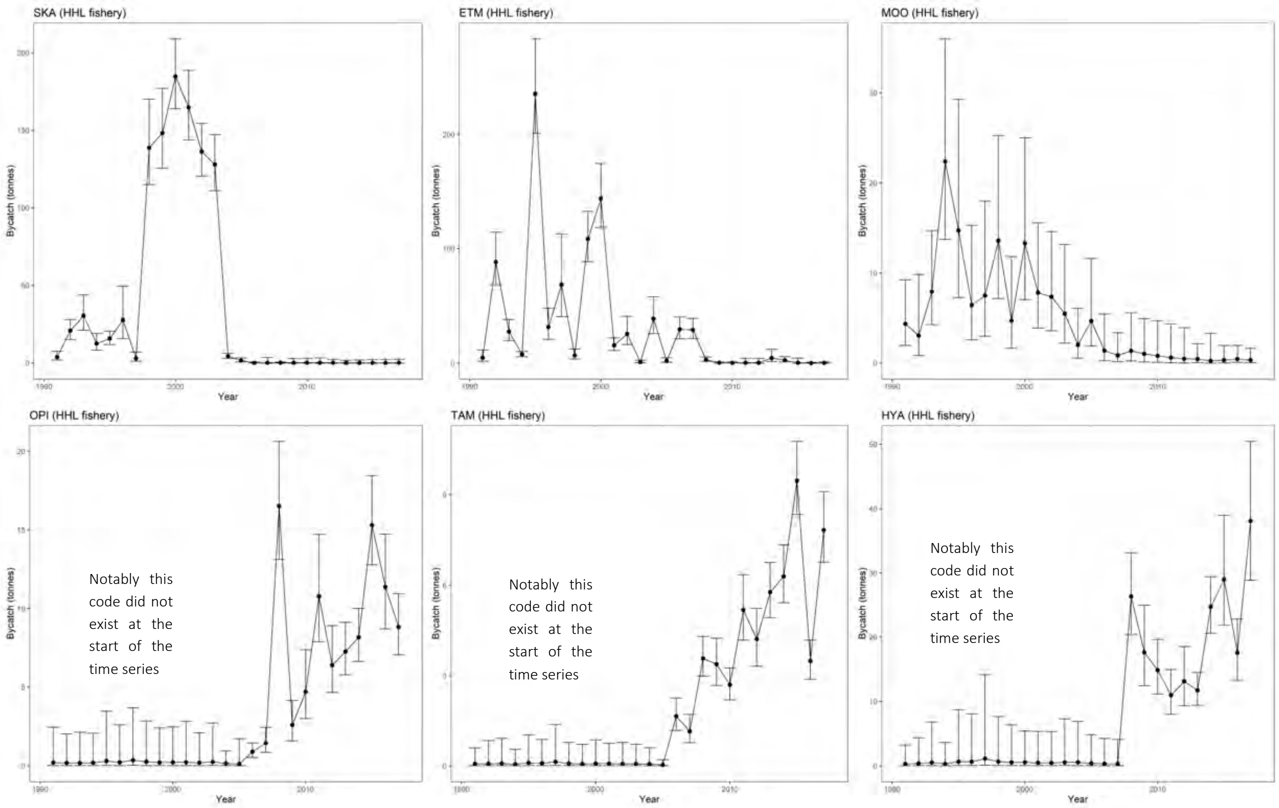

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Figure 9.29: Annual bycatch estimates in the hoki, hake, and ling trawl fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. 2019). Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3) and may be area-specific (see text above). See text above for species codes.

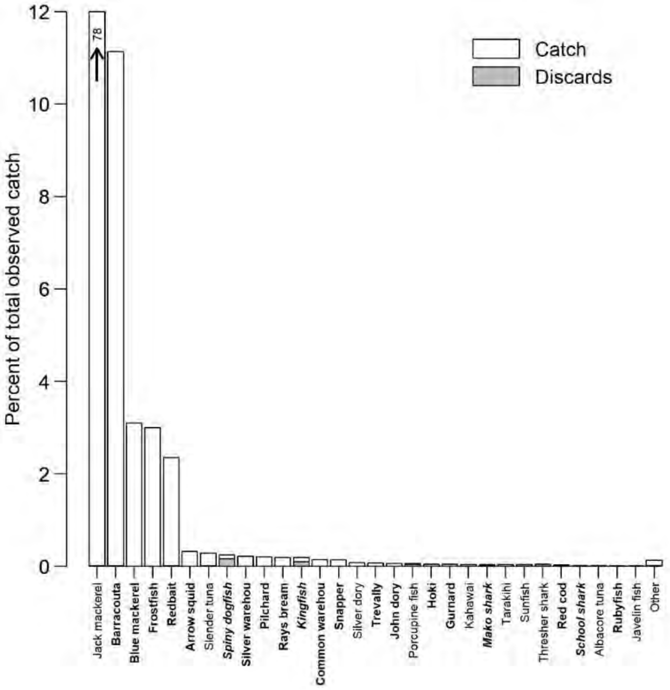

Figure 9.30: Percentage of the total catch contributed by the main non-target species (those representing 0.01% or more of the total catch; white) in the observed portion of the jack mackerel trawl fishery between 2002–03 and 2018–19, and the percentage discarded (grey; Finucci et al. 2022). The ‘Other’ category is the sum of all non-target species representing less than 0.01% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996)

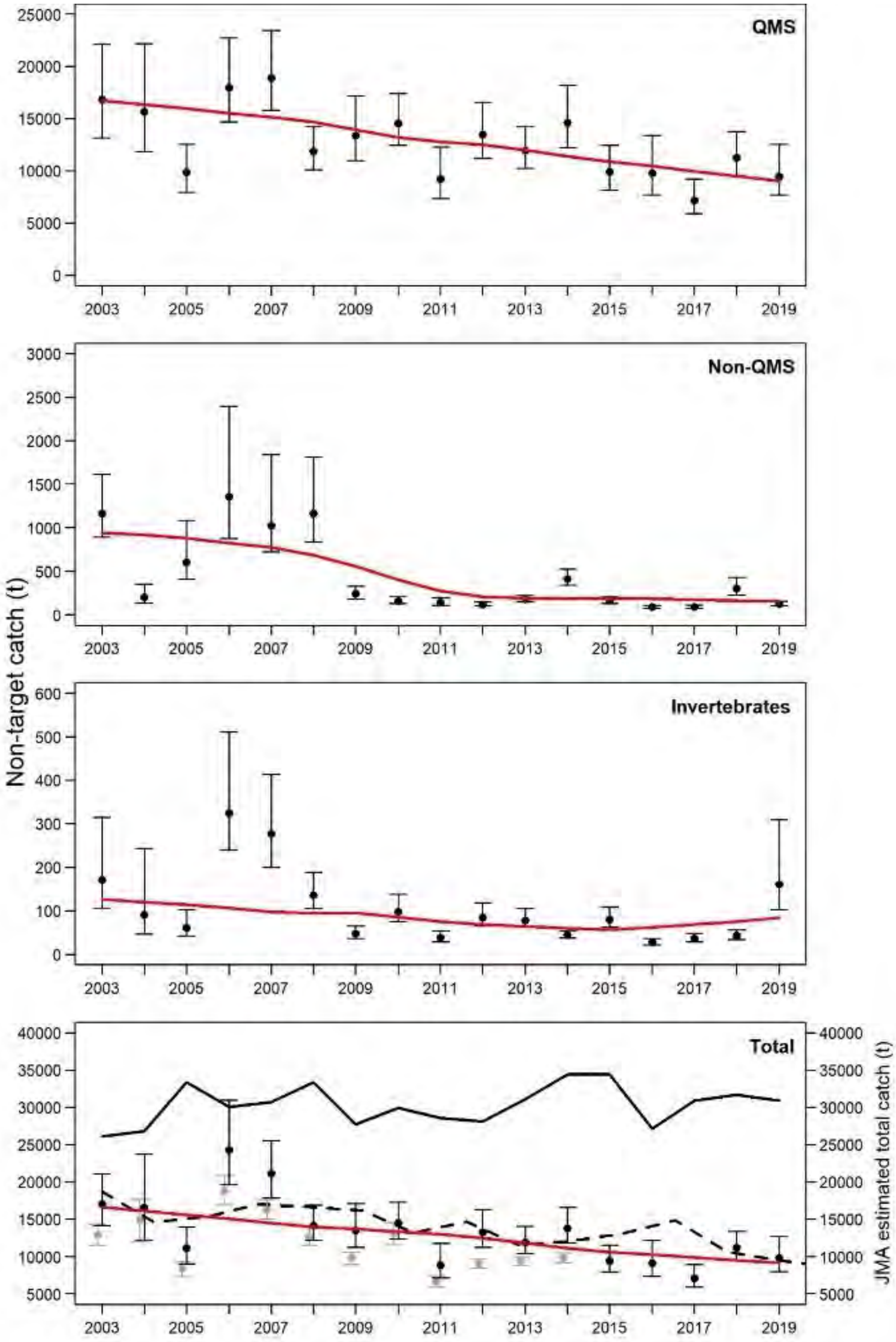

Figure 9.31: Annual estimates of non-target catch in the jack mackerel trawl fishery, for QMS species, non-QMS species, invertebrates, and overall for 2002–03 to 2018–19 (Finucci et al. in prep). Also shown (in grey) are earlier estimates of total bycatch calculated for 2002–03 to 2013–14 (from Anderson et al. 2017b). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual non-target catch. In the bottom panel the solid black line shows the total annual reported landings of jack mackerels, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total non-target catch.

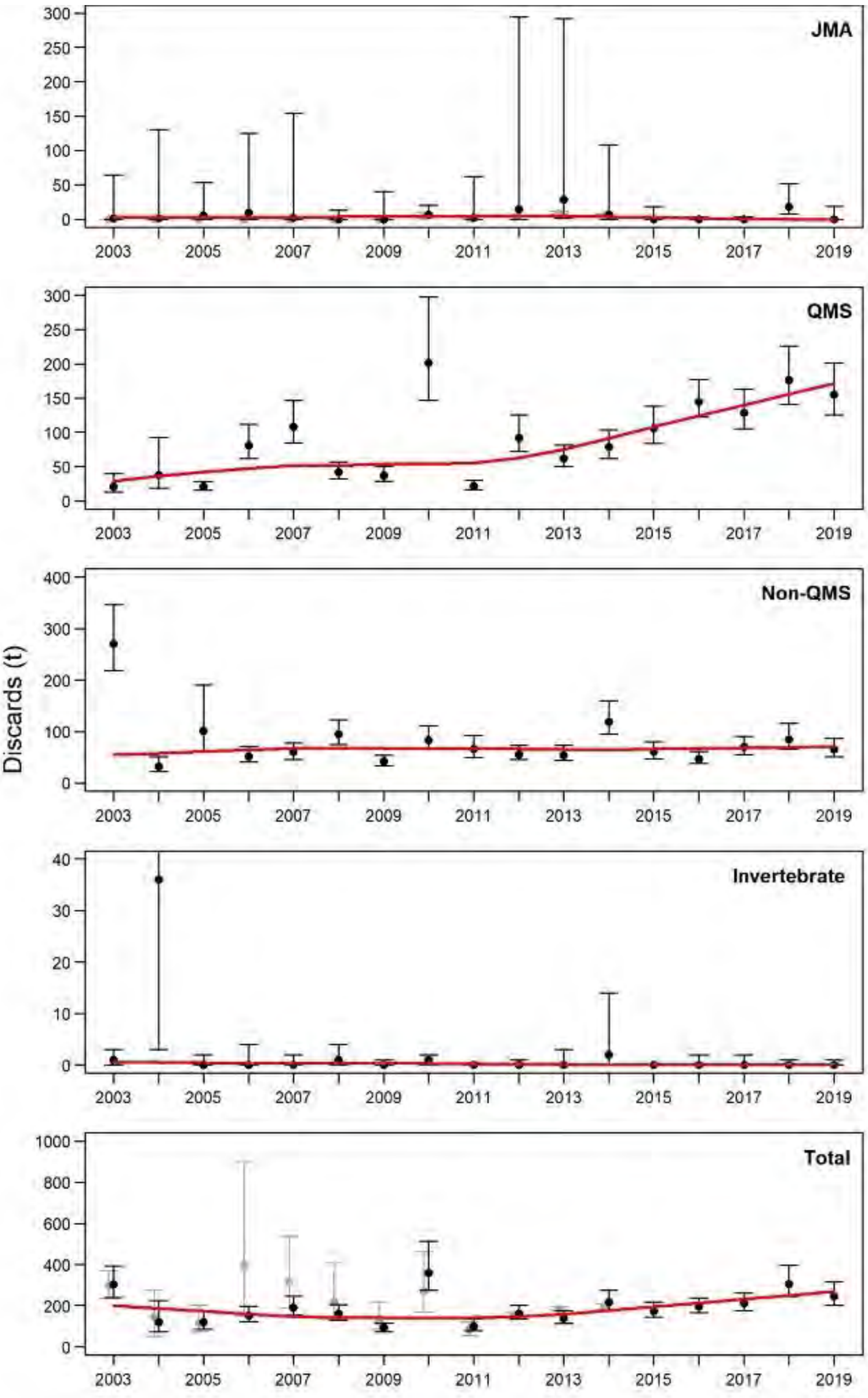

Figure 9.32: Annual estimates of discards in the jack mackerel trawl fishery, for jack mackerels (JMA), QMS species, non-QMS species, invertebrates, and overall (Total) for 2002–03 to 2018–19 (Finucci et al. in prep). Also shown (in grey) are earlier estimates of total discards calculated for 2002–03 to 2013–14 (from Anderson et al. 2017b). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards

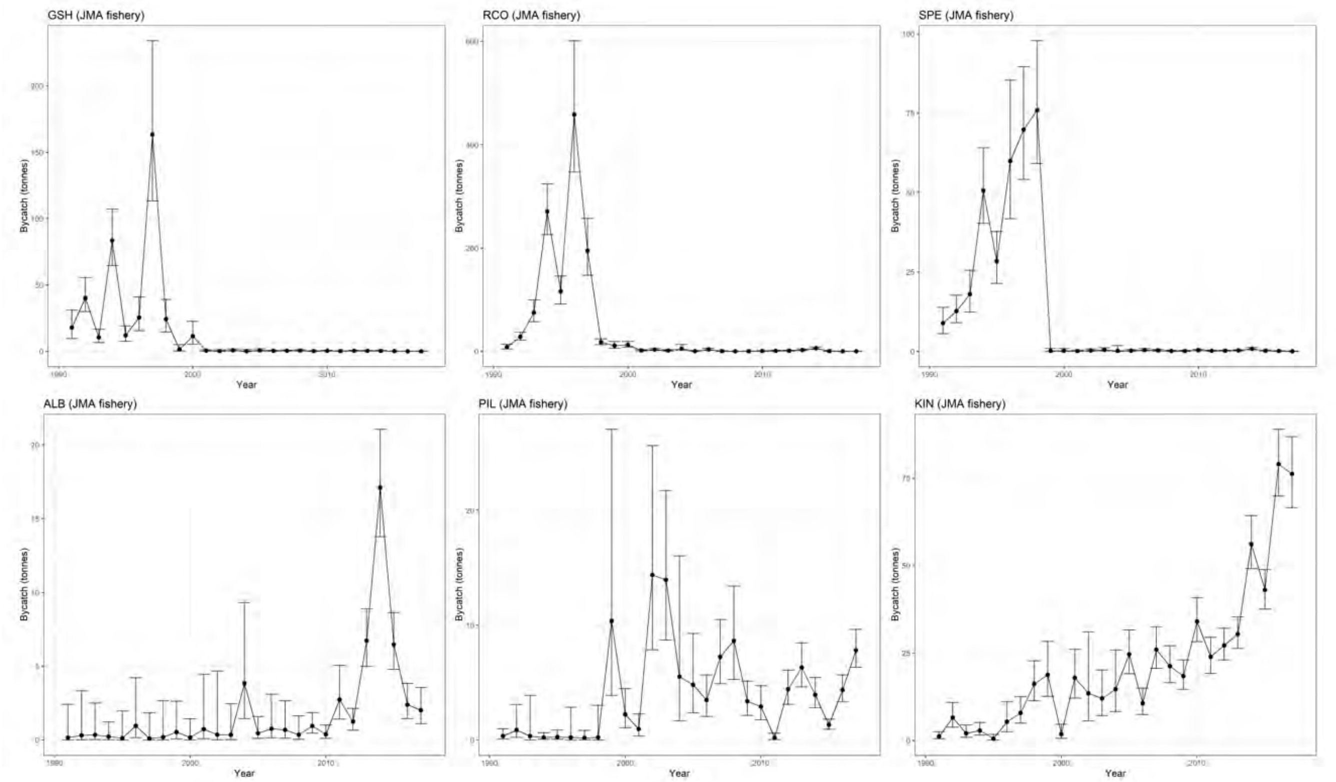

Figure 9.33: Annual non-target estimates in the jack mackerel trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. 2019). See text above for explanation of the species codes. Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

1.5.4 SOUTHERN BLUE WHITING TRAWL FISHERY

In a study that covered data from 2002–03 to 2006–07, the ratio estimator used to calculate bycatch and discard rates in this fishery was based on trawl duration (Anderson 2009b). Linear mixed-effect models (LMEs) identified fishing depth as the key variable influencing bycatch rates and discard rates in this fishery, and regression tree methods were used to optimise the number of levels of this variable in order to stratify the calculation of annual bycatch and discard totals in each catch category.

The key categories of catch/discards examined were: southern blue whiting, other QMS species combined, commercial species combined (as defined above for hoki/hake/ling), non-commercial species combined, and three commonly caught individual species, hake, hoki and ling.

Fisheries observers covered 22–53% of the target fishery catch from 2002–03 to 2006–07, and similar coverage levels were reported from 1990–91 to 2001–02. The spread of observer data, across a range of variables, had no obvious shortcomings, due to a combination of the highly restricted distribution of the southern blue whiting fishery over space and time of year, a stable and uniform fleet composition, and a high level of observer effort.

Southern blue whiting comprised more than 99% of the total estimated catch from all observed trawls targeting southern blue whiting from 2002–03 to 2006–07. About half the remaining total catch was made up of ling (0.2%), hake (0.1%), and hoki (0.1%) (Figure 9.34). These three species, along with other QMS species, comprised over 80% of the total bycatch. In all, over 120 species or species groups were identified by observers, most were non-commercial species caught in low numbers. Porbeagle sharks (introduced into the QMS in 2004), javelinfish and other rattails, and silverside accounted for much of the remaining bycatch. Invertebrate species (mainly sponges, crabs, and echinoderms) were also recorded by observers, but no taxon accounted for more than 0.01% of the total observed catch.

Estimated total annual bycatch from 2002–03 to 2006–07 was 40–390 t, compared with approximate target species catches in the same period of about 22 000 to 42 000 t. This bycatch was split between commercial species (55%) and non-commercial species (45%), although QMS species accounted for about 80% of the total bycatch during this period. Total annual bycatch decreased during the period, to an all-time low of 40 t in 2006–07. Total annual bycatch estimates for 1990–91 to 2001–02, from earlier reports, were mostly 60–500 t but reached nearly 1500 t in 1991–92 (Figure 9.35). This year immediately preceded the introduction of southern blue whiting into the QMS, and the effort and catch were exceptionally high.

TRENDS IN BYCATCH BY SPECIES FROM THE SOUTHERN BLUE WHITING TRAWL FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were ling (*Genypterus blacodes*, LIN), hake (*Merluccius australis*, HAK), and hoki (*Macruronus novaezelandiae*, HOK).
- Of the 109 bycatch species examined, six had a significant decrease in catch over time and one had a significant increase in catch.
- The species showing the greatest decline were moonfish (*Lampris guttatus*, MOO), unspecificed rattails (RAT), and dark ghost shark (*Hydrolagus novaezealandiae*, GSH) (Figure 9.37).
- The species showing the greatest increase were opah (*Lampris immaculatus,* PAH), Ray's bream (*Brama brama*, RBM), and pale ghost shark (*Hydrolagus bemisi*, GSP) (Figure 9.37).

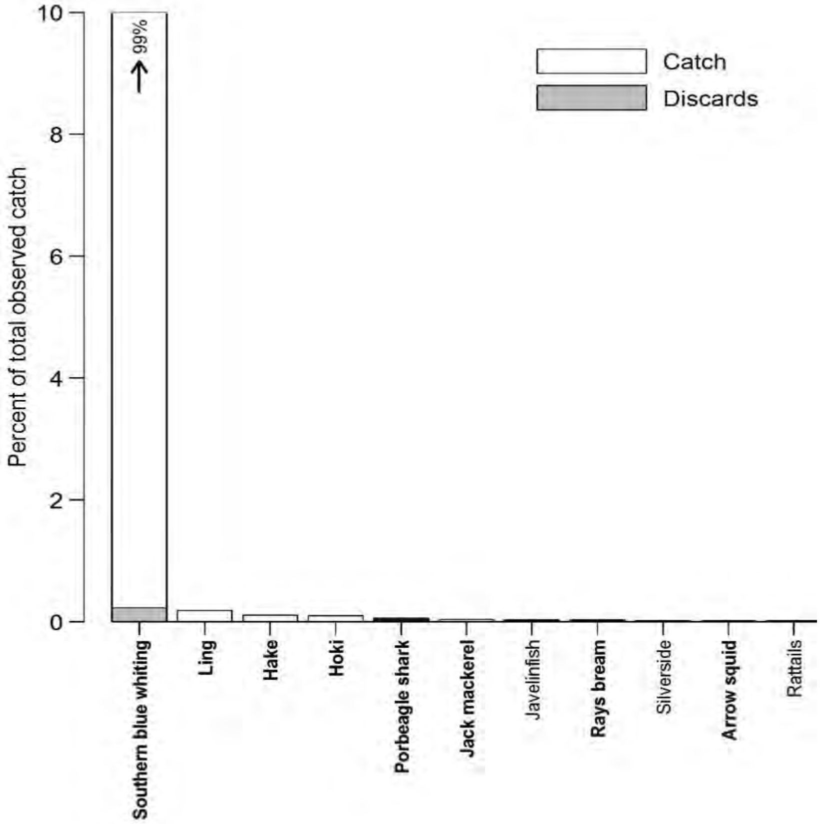

Figure 9.34: Percentage of the total catch contributed by the main bycatch species (those representing 0.05% or more of the total catch; white) in the observed portion of the southern blue whiting fishery, 2002–03 to 2006–07, and the percentage discarded (grey; Anderson 2009b). QMS species are shown in bold.Total annual discard estimates from 2002–03 to 2006–07 were 90–250 t per year (Anderson 2009b). Discard amounts sometimes exceeded bycatch due to the large contribution of the target species (50–230 t per year) to total discards — the result usually of fish losses during recovery of the trawl. Discarding of commercial species was virtually non-existent in most years and discards of non-commercial species amounted to only 10–50 t per year. The main species discarded were southern blue whiting, rattails, and porbeagle sharks. Total annual discard estimates for 1990–91 to 2001–02, from earlier reports, were mostly 140–750 t but were about 1200 t in 1991–92 (Figure 9.36). Discards of southern blue whiting (and therefore total discards) decreased substantially at the end of the 1990s and remained at low levels, below 250 t per year, up to 2006–07

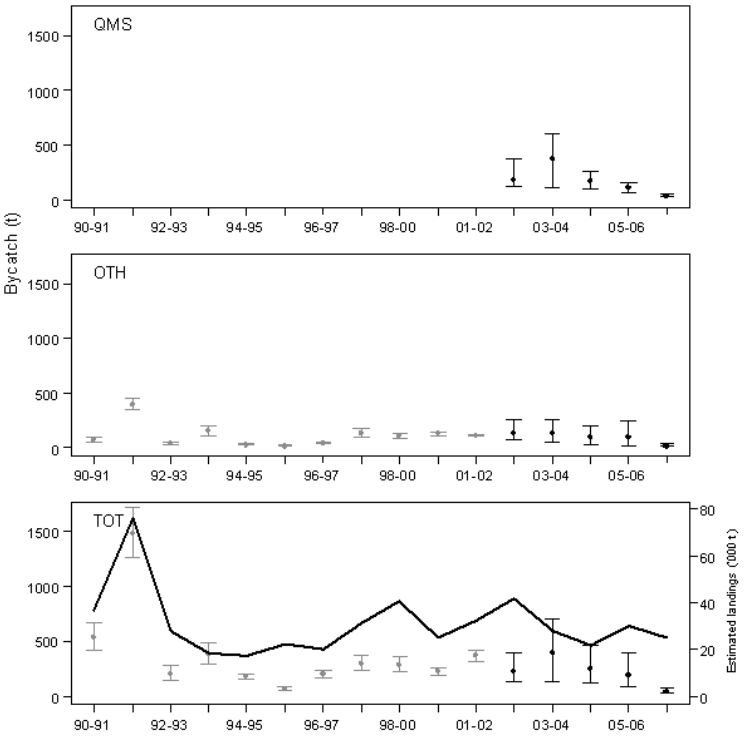

Figure 9.35: Annual estimates of fish bycatch in the southern blue whiting trawl fishery, calculated for QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black) (Anderson 2009b). Also shown (in grey) are estimates of bycatch in each category (excluding QMS) for 1990–91 to 2001–02 (Anderson 2004a). Error bars show the 95% confidence intervals. Note: the 98–00 fishing year encompasses the 18 months between September 1998 and March 2000, the transitional period during a change from an Oct–Sep to Apr–Mar fishing year. The dark line in the bottom panel shows the total annual estimated landings of SBW (Ministry for Primary Industries 2013a).

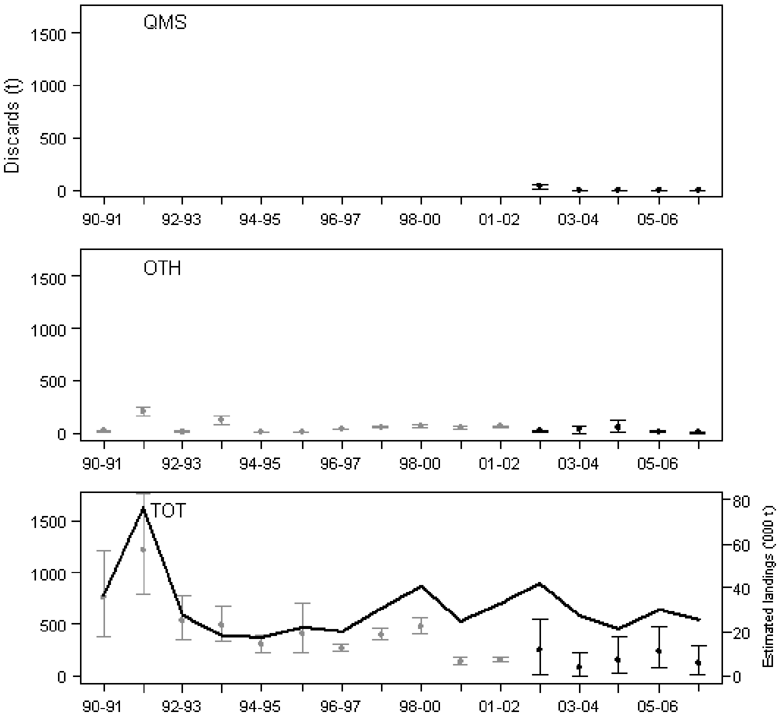

Figure 9.36: Annual estimates of fish discards in the southern blue whiting trawl fishery, calculated for the target species (SBW), QMS species, non-commercial species (OTH), and overall (TOT) for 2002–03 to 2006–07 (in black) (Anderson 2009b). Also shown (in grey) are estimates of discards in each category (excluding QMS) calculated for 1990–91 to 2001–02 by Anderson (2004a). Error bars show the 95% confidence intervals. The dark line shows the total annual estimated landings of SBW (Ministry for Primary Industries 2013a).

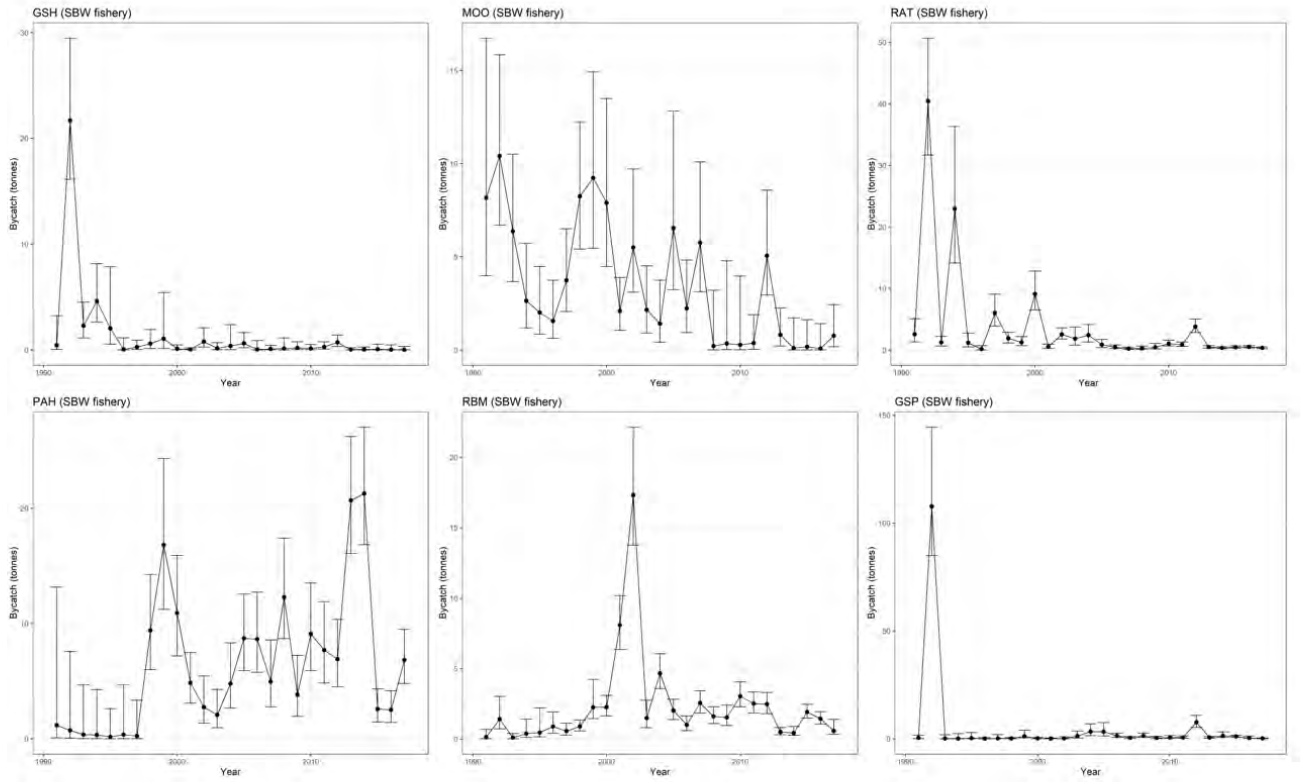

Figure 9.37: Annual bycatch estimates in the southern blue whiting trawl fishery for the species that had the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. 2019). Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3) and may be area-specific (see text above). See text above for species codes.

1.5.5 ORANGE ROUGHY TRAWL FISHERY

The most recent published analysis of this fishery covered the period 2001–02 to 2014–15 and used both the ratio estimator and the statistical model method (Anderson et al. 2017a), although a new analysis is immenent (Anderson & Finucci *in press*).

The key categories of catch/discards examined were: orange roughy, other QMS species combined, non-QMS species combined, and invertebrate species combined.

The level of observer coverage in this fishery since 1990–91 has been over 10% of the total fishery catch in all but one year, and over 50% in some years; between 2001–02 and 2014–15 coverage averaged 37% and was over 50% in five years. This coverage was relatively well spread across the orange roughy fishery, but some undersampling occurred of smaller vessels, off the east coast fisheries in QMAs ORH 2A, ORH 2B, and ORH 3A (where mainly small vessels operated), and oversampling occurred of fisheries outside the EEZ (where vessels are normally required to carry an observer).

Since 2001–02, orange roughy has comprised about 85% of the total observed catch. Much of the remainder of the total catch (about 9%) comprised oreo species: mainly smooth oreo (7%) and black oreo (1.6%). Rattails (various species, 0.7%) and shovelnose spiny dogfish (*Deania calcea*, 0.6%) were the species most caught by this fishery, with over 50% discarded (Figure 9.38). Other fish species frequently caught, and usually discarded, included offshore dogfishes (family Squalidae), especially *Etmopterus* species, the most common was probably Baxter's dogfish (*Etmoptertus baxteri* – which is the most common species in the *Etmopterus* genus), slickheads, and morid cods, especially Johnson's cod (*Halargyreus johnsonii*) and ribaldo (*Mora moro*). In total, over 700 bycatch species or species groups were observed, most were non-commercial species, including invertebrate species, caught in low numbers. Squid (mostly warty squid, *Onykia* spp.) were the largest component of invertebrate catch, followed by various groups of protected corals, echinoderms (mainly starfish), and crustaceans (mainly king crabs, family Lithodidae).

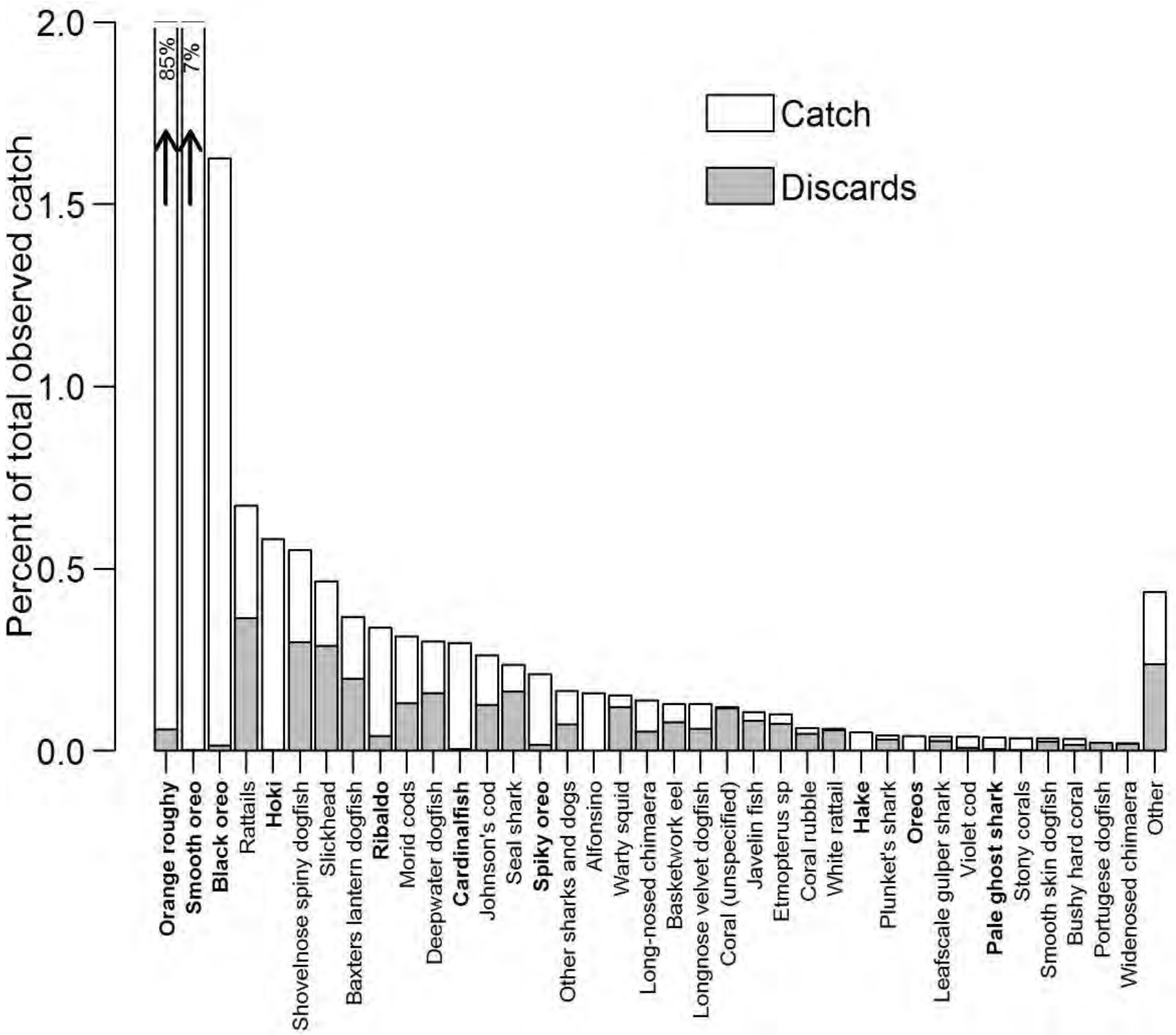

Figure 9.38: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch; white) in the observed portion of the target orange roughy trawl fishery for fishing years from 2001–02 to 2014–15, and the percentage discarded (grey; Anderson et al. 2017a). The 'Other' category is the sum of all bycatch species representing less than 0.02% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996).

Total annual bycatch in the orange roughy fishery since 2001–02 was highly variable, with greater levels (3093–6075 t per year) before 2009–10 and decreasing levels thereafter (706–1080 t per year), in line with decreasing orange roughy landings (Figure 9.39). Bycatch comprised similar amounts of QMS and non-QMS species, with invertebrate species bycatch below 200 t in most years and below 50 t since 2010–11.

Estimated total annual discards also decreased over time, from about 3400 t in 1990–91 (Anderson 2011) to less than 500 t since 2007–08 (Figure 9.40). Since about 2000, discards have comprised mostly non-QMS species. Large discards of orange roughy and other QMS species, more prevalent early in the fishery, were often due to fish lost from torn nets during hauling (and are accounted for in stock assessments). In more recent times, improved fishing gear and techniques have substantially lowered the level of discards/losses in these categories.

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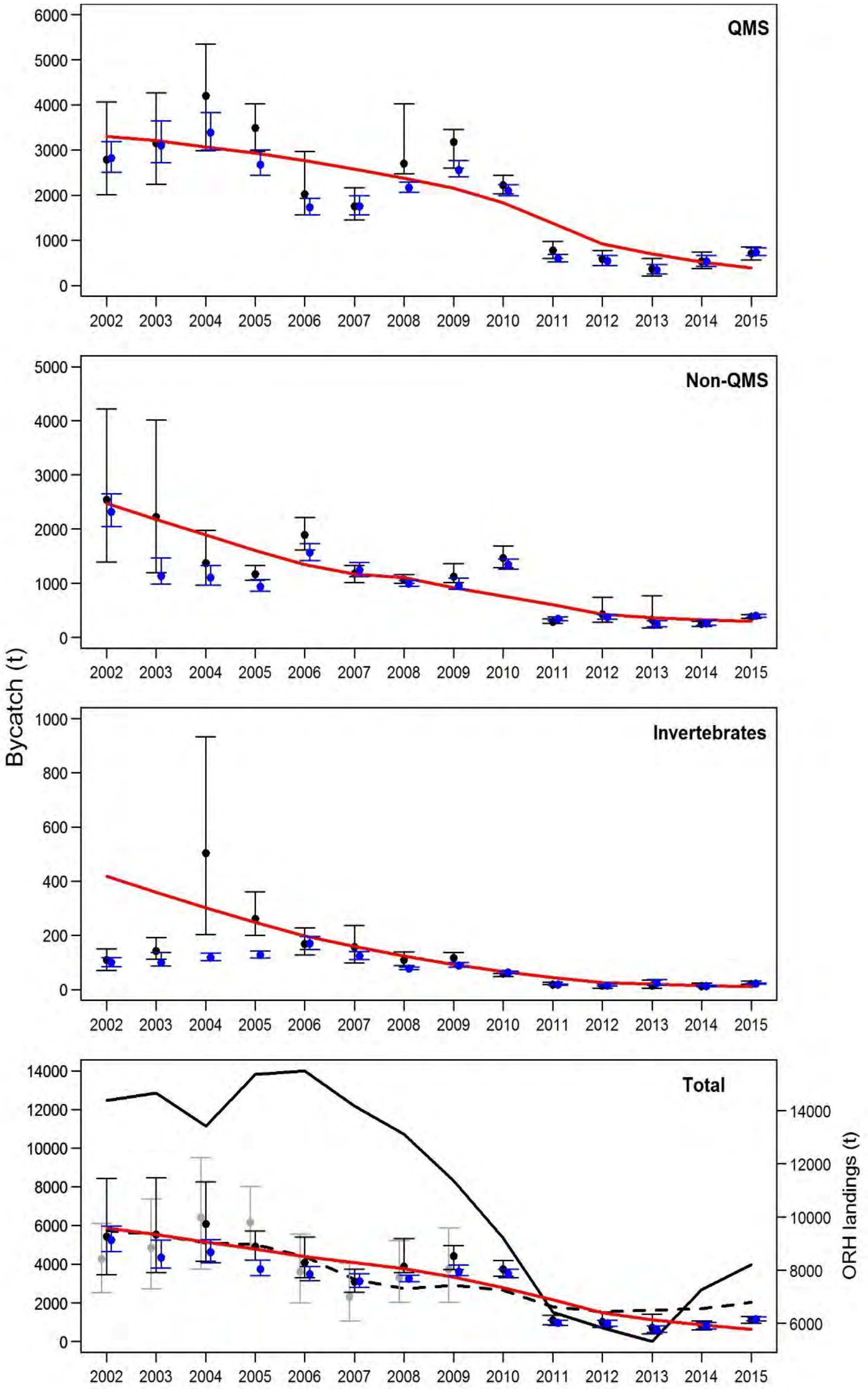

Figure 9.39: Annual estimates of bycatch (t) in the target orange roughy trawl fishery, species categories for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of orange roughy, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

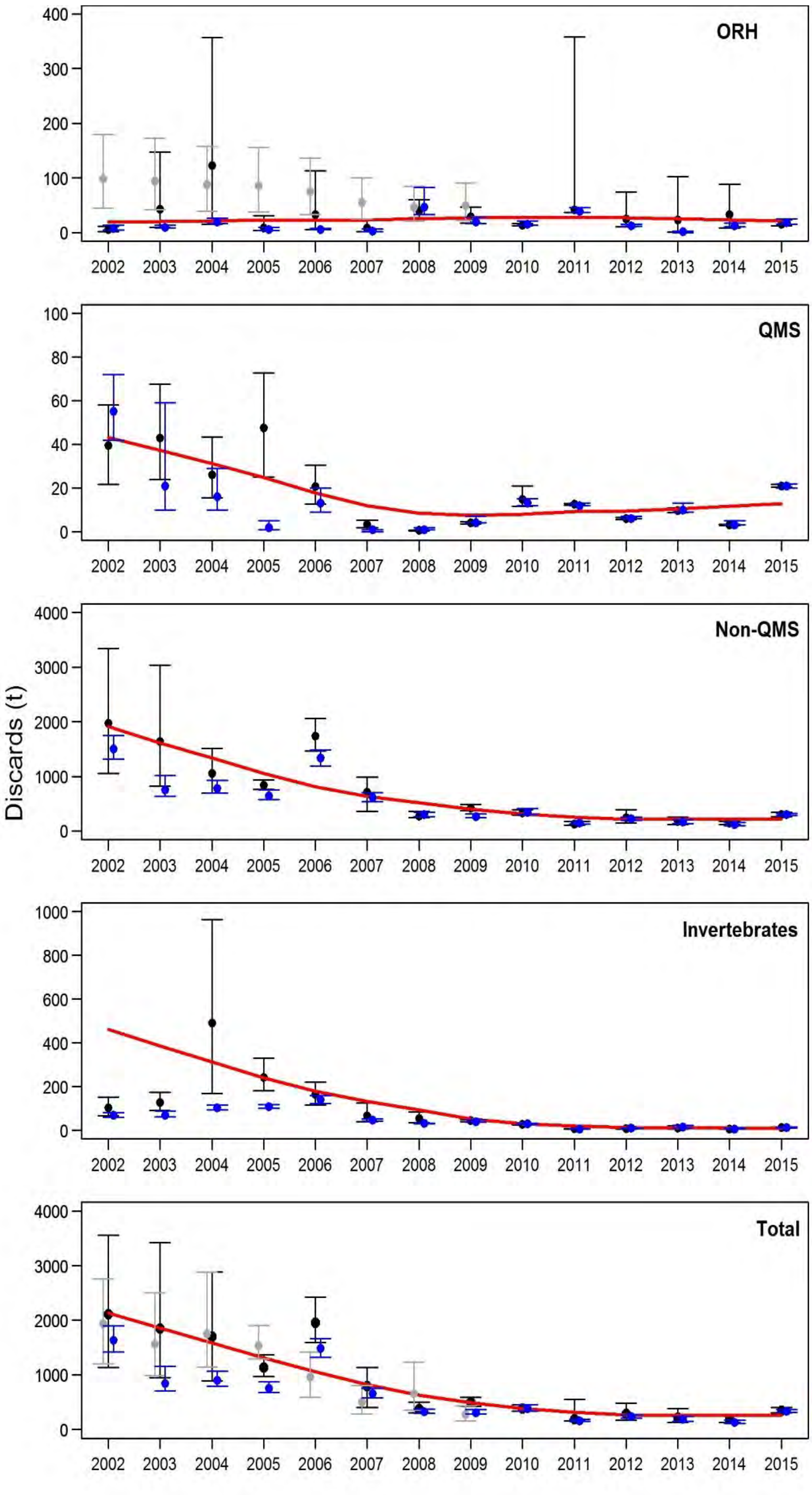

Figure 9.40: Annual estimates of discards (t) in the target orange roughy trawl fishery, for species categories for 2001–02 to 2014–15 (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

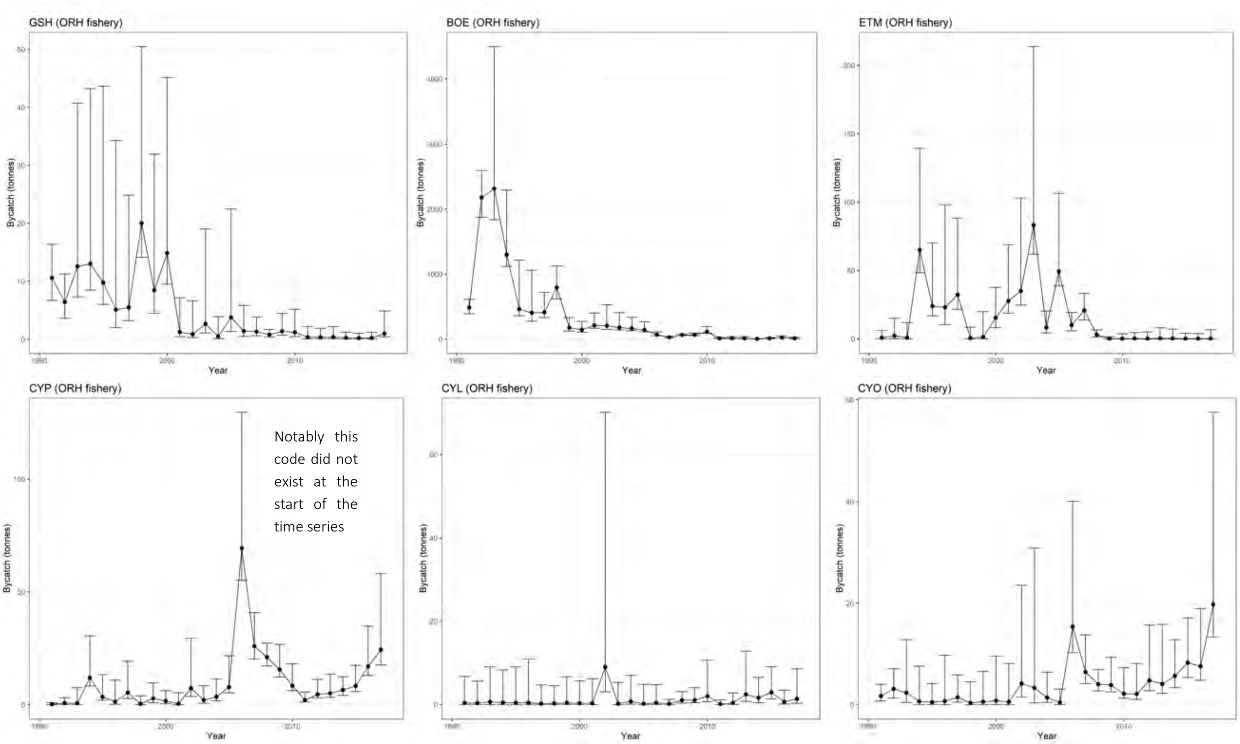

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Figure 9.41: Annual bycatch estimates in the orange roughy trawl fishery for the species that have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. 2019). See text above for explanation of the species codes. Some apparent changes in bycatch may be due to improvements in observer identifications (see Section 9.3). Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years.

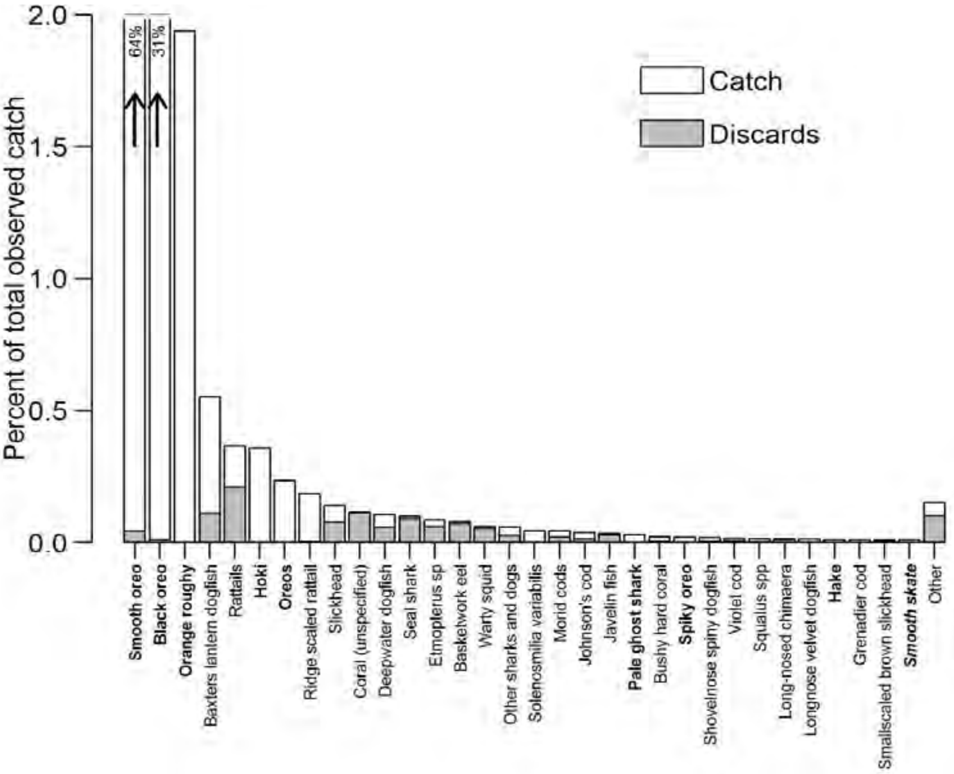

Figure 9.42: Percentage of the total catch contributed by the main bycatch species (those representing 0.01% or more of the total catch; white) in the observed portion of the oreo trawl fishery between 2001–02 and 2014–15, and the percentage discarded (grey; Anderson et al. 2017a). The 'Other' category is the sum of all bycatch species representing less than 0.01% of the total catch. Names in bold are QMS species, names in italics are QMS species that can be legally discarded under Schedule 6 of the Fisheries Act (1996)

1.5.6 OREO TRAWL FISHERY

The most recent published analysis of this fishery covered the period 2001–02 to 2014–15 and used both the ratio estimator and the statistical model method (Anderson et al. 2017a), although a new one is imminent (Anderson & Finucci *in press*).

The key categories of catch/discards examined were: Orange roughy, other QMS species combined, non-QMS species combined, and invertebrate species combined.

The oreo fishery is strongly linked to the historically larger and more widespread orange roughy fishery, with an earlier study showing that about a third of observed trawls targeting oreos were from trips that predominantly targeted orange roughy (Anderson 2011). The observer coverage of the oreo fishery is therefore partly, if not largely, determined by the operations of the orange roughy fishery.

The annual number of observed trawls in the oreo fishery ranged from 30 in 1991–92 to 1011 in 2006–07 and the number of vessels observed ranged from 2 to 12. The level of coverage remained at a relatively consistent level after the mid-1990s, despite a decrease in the total catch and effort, but declined after 2009–10 to a level of about 140–210 tows per year between 2012–13 and 2014–15. As a fraction of the total catch, observer coverage has been over 12% since 1999–2000 and approached 50% in a few years in the mid-2000s. Observer coverage has been mostly restricted to the main fisheries on the south Chatham Rise and further south. Within this region, few locations were not covered by observers during the period most recently examined, although the south Chatham Rise was undersampled and some southern fisheries oversampled in a few years. The full range of vessel sizes (mainly between 300 t and 3000 t, median length per area ranged from 26 to 66 m) was covered by observers, although small vessels were under-represented and large vessels over-represented.

Oreo species accounted for about 95% of the total estimated catch from all observed trawls targeting oreos after 1 October 2001. Orange roughy (1.9%) was the main bycatch species, with no other species or group of species accounting for more than 0.6% of the total catch. Baxter's dogfish was the next most common bycatch species, followed by rattails (which were mainly discarded) and hoki (Figure 9.42). In total, over 500 species or species groups were identified by observers in the target fishery, including numerous invertebrates. Corals (accounting for about 0.1% of the total catch), squids, and echinoderms were the main invertebrate groups caught. Total bycatch in the oreo fishery has fluctuated in recent years with higher levels from 2001–02 to 2009–10 (range 579–1575 t per year), followed by lower levels from 2010–11 (352–535 t per year) (Figure 9.43). Bycatch was split almost evenly between commercial and non-commercial species overall, the ratio fluctuating without any trend over time.

Discards in the oreo fishery have slowly decreased over time, with the 14 t estimated for 2014–15 the lowest recorded for any year since 1990–91 (Figure 9.44). Discards mainly comprised non-QMS species but included a varying amount of the target species in most years.

TRENDS IN BYCATCH BY SPECIES FROM THE OREO TRAWL FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were orange roughy (*Hoplostethus atlanticus*, ORH), unspecified sharks (SHA), and Baxter's, dogfish (*Etmopterus granulosus*, ETB).
- Of the 228 bycatch species examined, 40 showed a significant decrease in catch over time and 9 showed a significant increase in catch.
- The species showing the greatest decline were dark ghost shark (*Hydrolagus novaezealandiae*, GSH), unspecified sharks (SHA)[2] and lanternshark (*Etmopterus* spp., ETM) (Figure 9.45).
- The species showing the greatest increase were longnose velvet dogfish (*Centroselachus crepidater*, CYP), ridge scaled rattail (*Macrourus carinatus*, MCA), and Baxter's dogfish (*Etmopterus granulosus*, ETB) (Figure 9.45).

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[2] Notably SHA is a generic code and its decline is probably due to better species level identification of sharks over time.

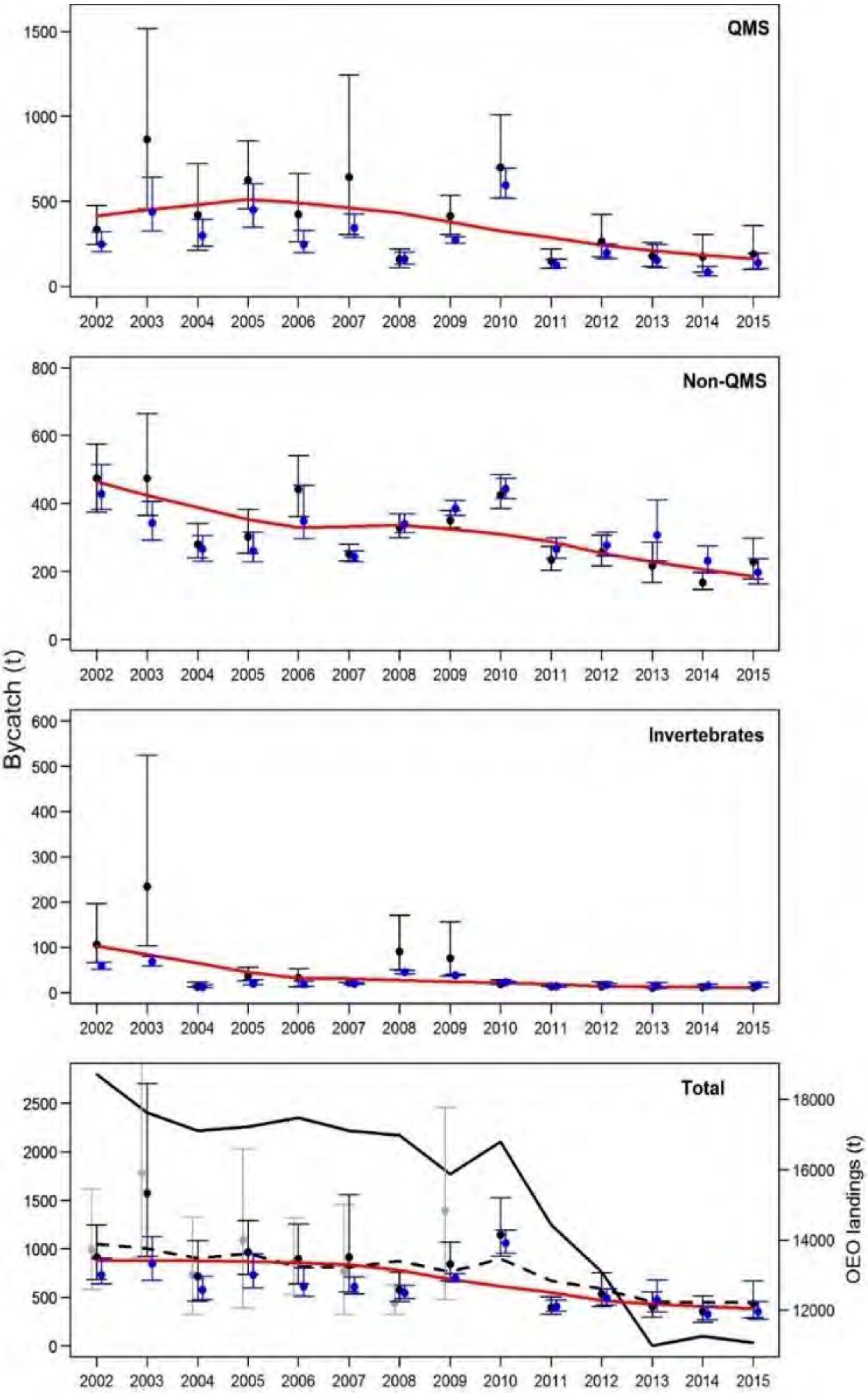

Figure 9.43: Annual estimates of bycatch in the oreo target trawl fishery for 2001–02 to 2014–15: black dots, ratio method; blue dots, statistical model method (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total bycatch calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of oreos, and the dashed line shows annual effort (number of tows), scaled to have mean equal to that of total bycatch.

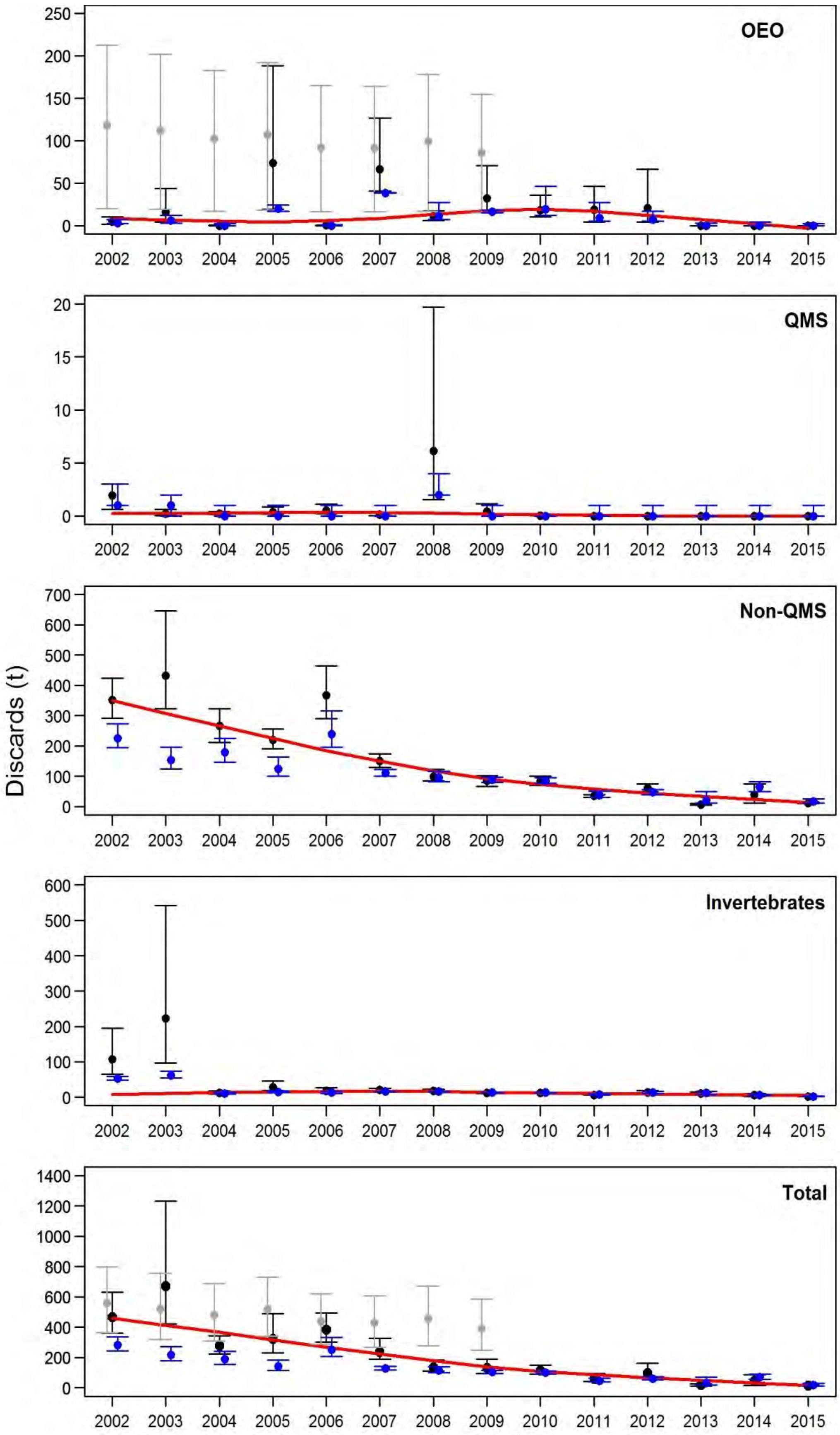

Figure 9.44: Annual estimates of discards in the oreo fishery, for species categories for 2001–02 to 2014–15 (Anderson et al. 2017a). Also shown (in grey) are earlier estimates of total discards calculated for 2001–02 to 2008–09 (Anderson 2011). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards.

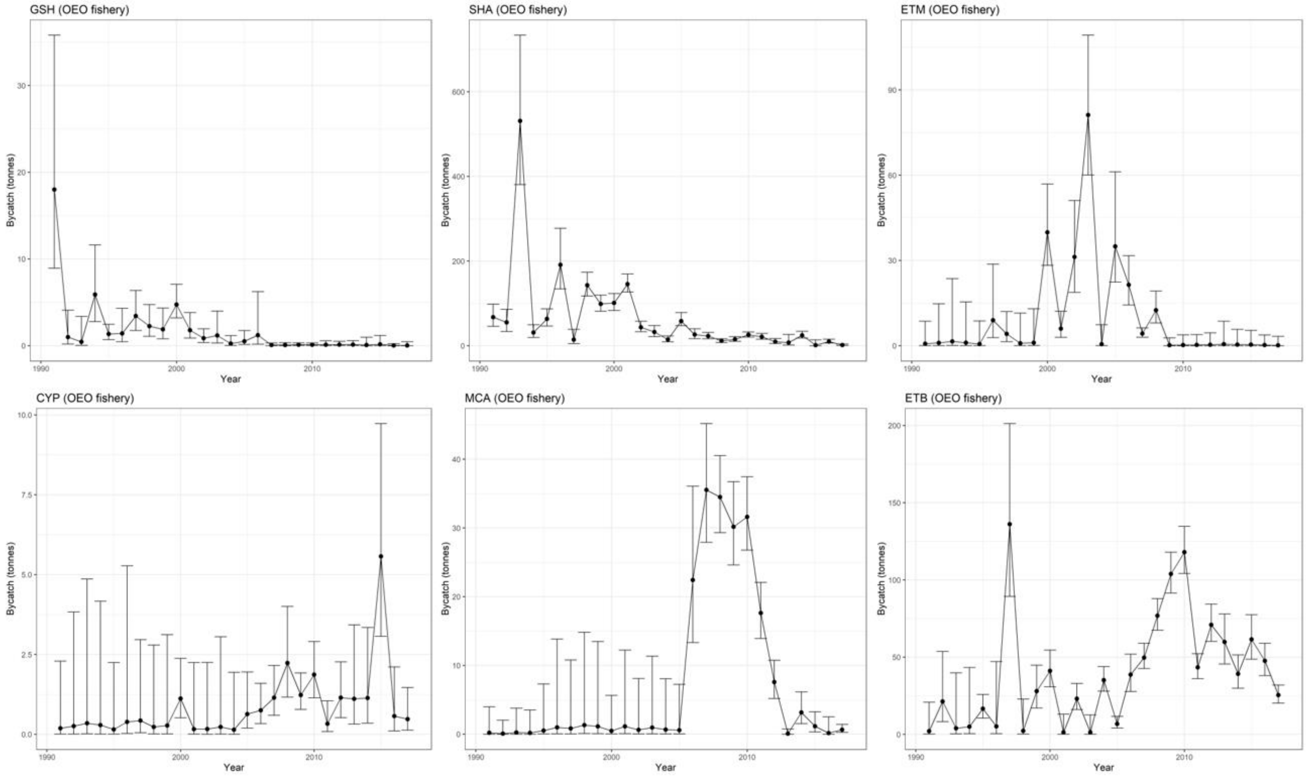

Figure 9.45: Annual bycatch estimates in the oreo trawl fishery for the species which have shown the greatest decrease (top) and greatest increase (bottom) between 1990–91 and 2016–17 (Finucci et al. 2019). See text above for explanation of the species codes. Sharks (SHA) may have been identified to an increasingly higher taxonomic level over time; rattails such as MCA may not have been well identified in earlier years. Note: the scale changes on the y-axis between plots; lines are joined only where there are data points for consecutive years

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1.5.7 SCAMPI TRAWL FISHERY

A detailed analysis of this fishery from 2002–03 to 2015–16 used the statistical model approach to calculate bycatch and discard levels in the scampi fishery, with effort based on the number of trawls (Anderson & Edwards 2018).

The key categories of catch/discards examined were: all QMS species combined, all non-QMS species combined, and all invertebrate species combined, with membership of these categories adjusted from year to year as species were added to the QMS.

Observer coverage in the scampi fishery has been relatively low compared with most of the other assessed offshore fisheries. The long-term level of observer coverage in the orange roughy, oreo, arrow squid, and southern blue whiting trawl fisheries and ling longline fisheries has covered more than 18% of the targeted catch (and over 40% for southern blue whiting) whereas in the scampi fishery (and also in the jack mackerel fishery) long-term coverage has been less than 12% of the targeted catch. For the 2002–03 to 2015–16 period most recently assessed, annual coverage was below 10% in 8 of the 14 years, reaching a maximum of 17% in 2002–03.

The annual number of observed tows in the fishery ranged from 142 to 535 but was over 300 tows in most years (2.7 to 15.5% of the total number of scampi tows). The number of vessels observed in each year ranged from 3 to 8 (equivalent to 33–75% of the fleet, composed of approximately 10 vessels) and was relatively constant, at 5 or 6 vessels in most years. Analysis of the spread of observer effort compared with that of the scampi fishery as a whole, across a range of variables, indicated that this coverage was reasonably well spread. Although some less important regions of the fishery received relatively low coverage (e.g., the eastern Chatham Rise, Puysegur, and west coast South Island), the main scampi fisheries were consistently sampled throughout the period examined. Vessels were mostly of a similar size, and the small amount of effort by larger vessels was adequately covered, as was the full depth range of the fishery and (despite highly intermittent sampling in several years) all periods of this year-round fishery.

Nearly 500 bycatch species or species groups were observed in the scampi target fishery catch, most being non-commercial species, including invertebrate species, caught in low numbers. Scampi accounted for about 19% of the total estimated catch from all observed trawls targeting scampi since 1 October 2002. The main bycatch species or species groups were javelinfish (18%), other (unidentified) rattails (12%), sea perch (*Helicolenus* spp., 10%), hoki (5%), and ling (4%). The first three of these bycatch groups were mostly discarded (Figure 9.46). Of the other invertebrate groups, unidentified crabs (0.9%) and unidentified starfish (0.8%) were caught in the greatest amounts. When combined into broader taxonomic groups, bony fish (excluding rattails and morid cods) contributed the most to total bycatch (33%), followed by rattails (30%), rays and skates (3.5%), sharks and dogfish (3.2%), chimaeras (3.1%), crustaceans (2.9%), echinoderms (1.6%), and morid cods (1.8%). A large percentage of the bycatch in these groups was discarded — over 80% for rattails, sharks, eels, crustaceans, echinoderms, octopuses, and other invertebrates.

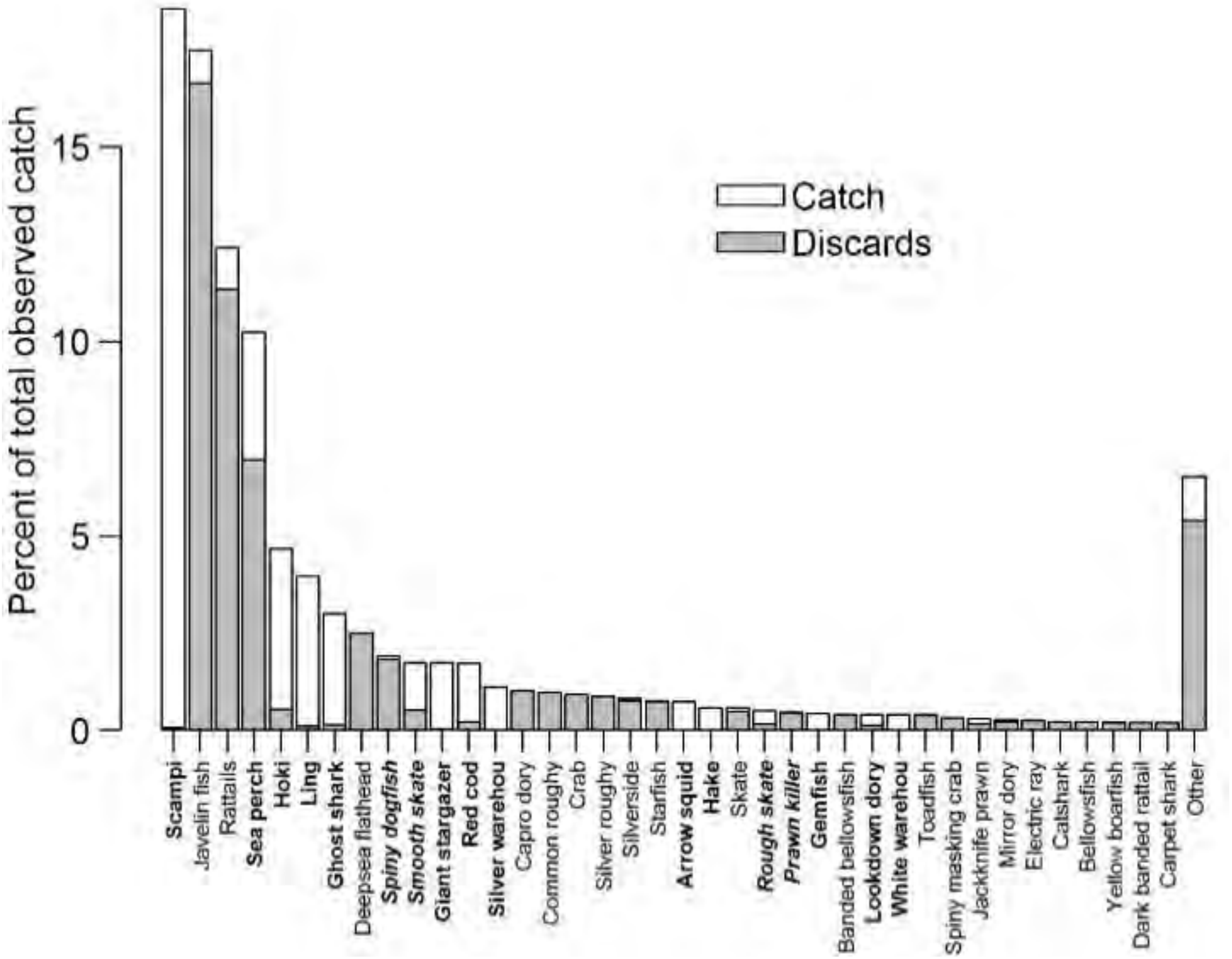

Figure 9.46: Percentage of the total catch contributed by the main bycatch species (those representing 0.2% or more of the total catch; white) in the observed portion of the scampi fishery, 2001–02 to 2015–16, and the percentage discarded (grey; Anderson & Edwards 2018). The ‘Other’ category is the sum of all other bycatch species (fish and invertebrates) representing less than 0.2% of the total catch. QMS species are shown in bold, Schedule 6 species in italics.

Total annual bycatch since 2002–03 ranged from about 2400 t to 5600 t and, although highly variable in the early part of the period, showed no significant trend over time (Figure 9.47). Annual bycatch has overall been a relatively even mixture of QMS and non-QMS species, with invertebrate species accounting for only about 7% of the total bycatch for the whole period. Rattails (javelinfish and all other species combined) accounted for 45–95% of the annual non-QMS bycatch.

Total annual discards ranged from about 940 t in 2003–04 to about 4100 t in 2004–05 and, although quite variable from year to year, there was no significant trend in overall discard levels over time (Figure 9.48). Discards were dominated by non-QMS species (overall about 67%), followed by QMS species (24%), and invertebrates (9%). Rattail species accounted for about 75% of the non-QMS discards and about 50% of all discards.

TRENDS IN BYCATCH BY SPECIES FROM THE SCAMPI TRAWL FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1990–91 to 2016–17. The following conclusions were made:

- The most caught bycatch species were javelinfish (*Lepidorhynchus denticulatus*, JAV), unspecified rattails (Macrouridae, RAT), and sea perch (*Helicolenus* spp., SPE).
- Of the 332 bycatch species examined, 34 showed a significant decrease in catch over time and 58 showed a significant increase in catch.
- The species showing the greatest declines were skates (Rajidae and Arhynchobatidae, SKA), bluenose (*Hyperoglyphe antarctica*, BNS), and hāpuku and bass (*Polyprion oxygeneios* & *P. americanus*, HPB) (Figure 9.49).
- The species showing the greatest increases were geometric star (*Psilaster acuminatus*, PSI), smooth deepsea anemones (Actinostolidae, ACS), and Garrick's masking crab (*Leptomithrax garricki*, GMC) (Figure 9.49).

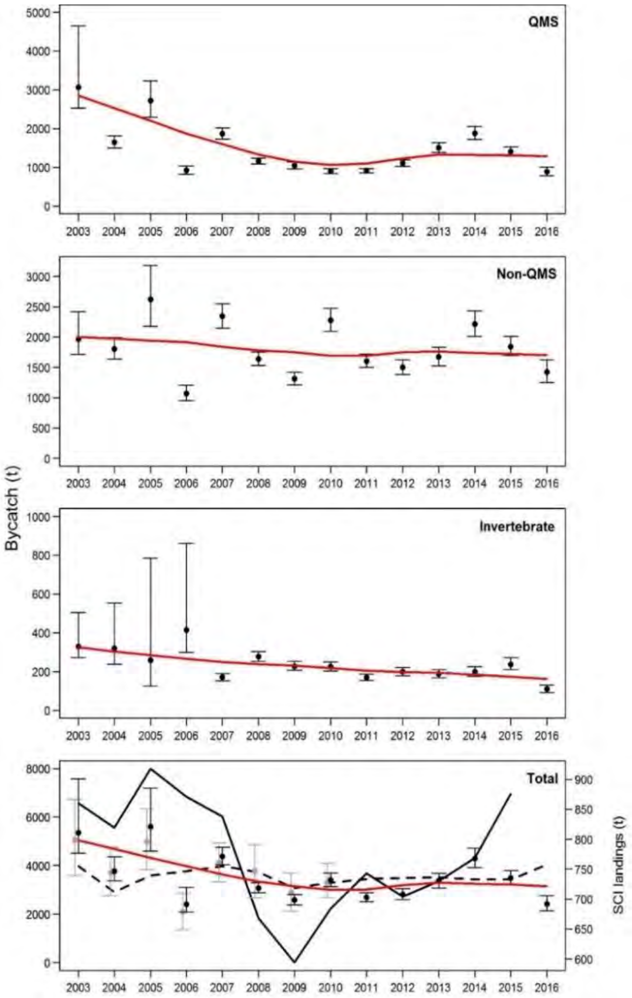

Figure 9.47: Annual estimates of bycatch in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16 (Anderson & Edwards 2018). Also shown (in grey) are estimates of Total bycatch calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted regression to annual bycatch. In the bottom panel the solid black line shows the total annual reported landings of scampi and the dashed line shows annual effort (scaled to have mean equal to that of total bycatch).

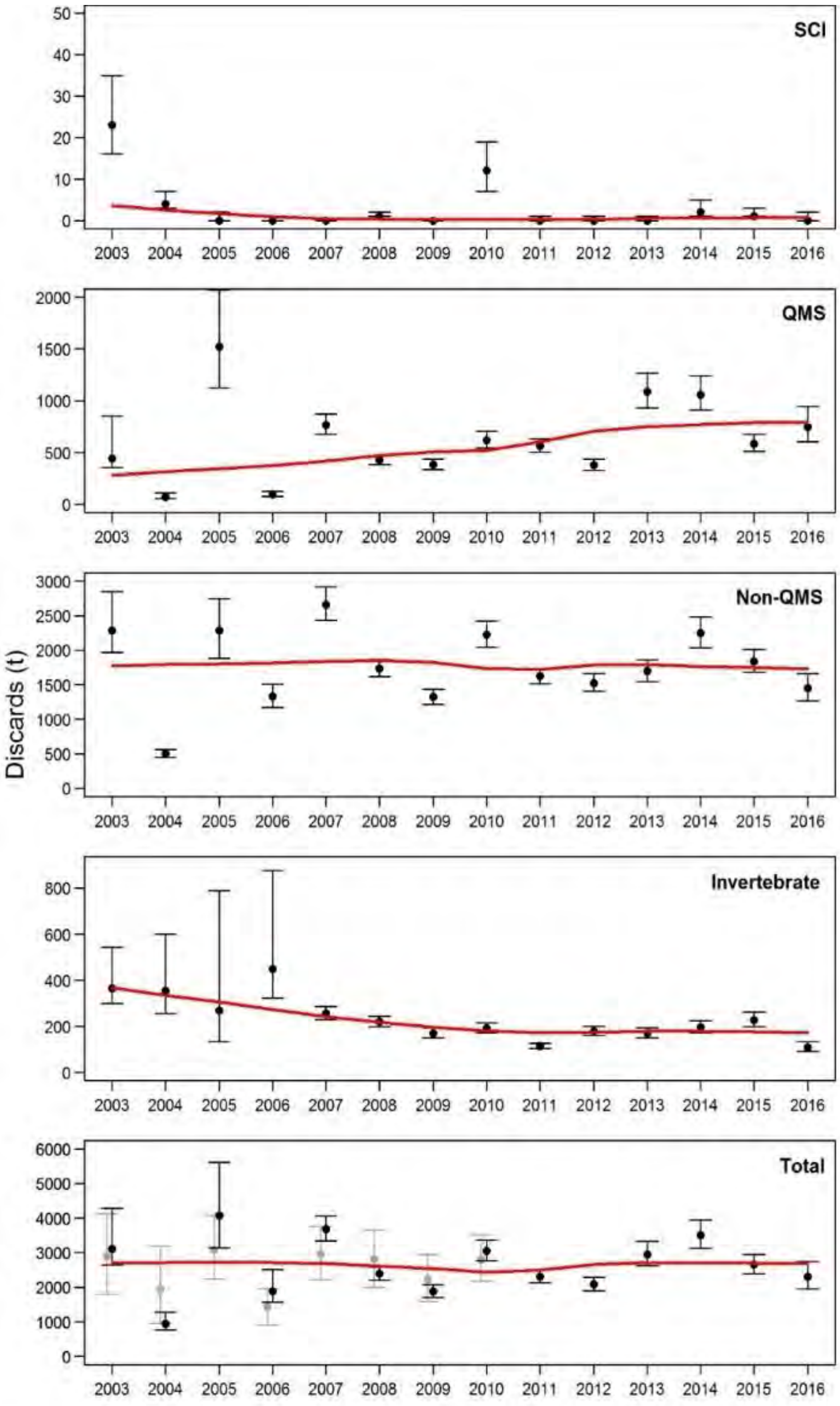

Figure 9.48: Annual estimates of discards in the scampi trawl fishery, for QMS species, non-QMS species, invertebrates (INV), and overall for 2002–03 to 2015–16. Also shown (in grey) are estimates of Total discards calculated for 2002–03 to 2009–10 (Anderson 2012). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted regression to annual discards.

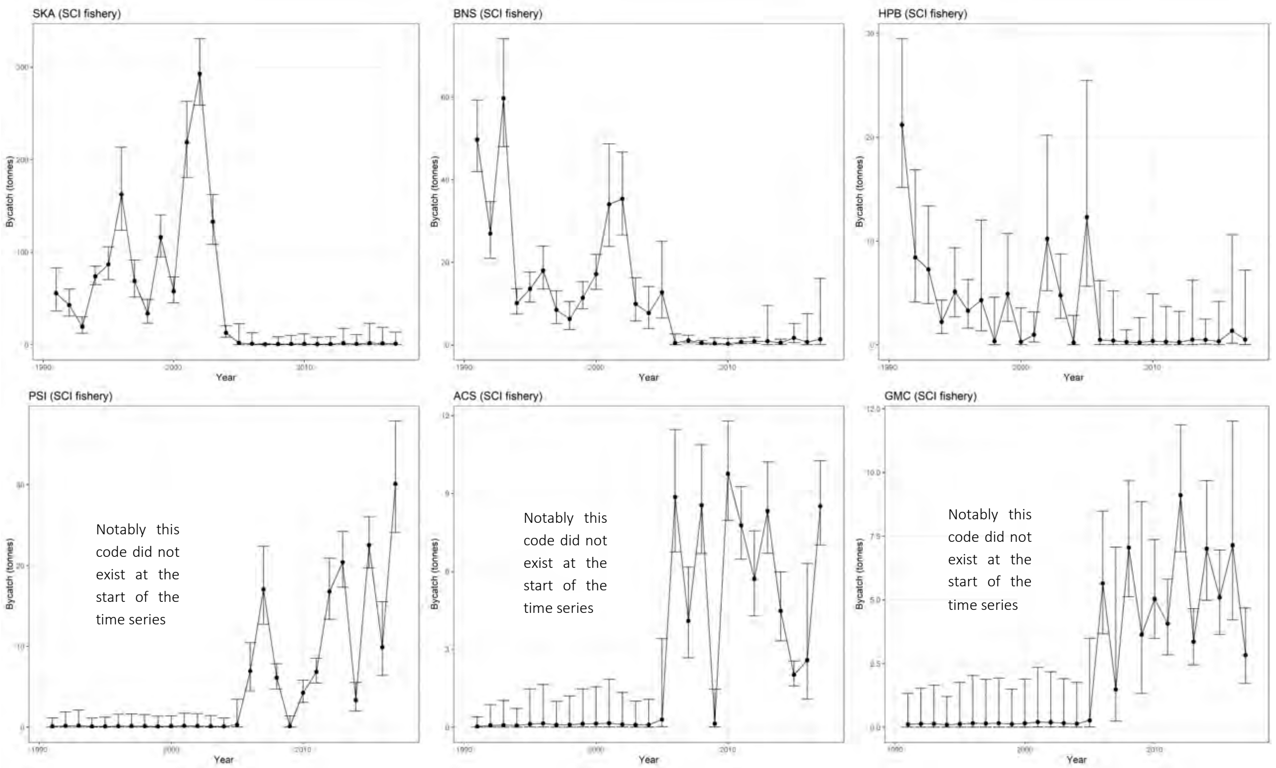

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Figure 9.49: Annual bycatch estimates in the target scampi trawl fishery for the species which have the most bycatch between 1990–91 and 2016–17, with 95% CIs, in descending order of total catch. See text above or http://marlin.niwa.co.nz for species code definitions. Note: the scale changes on the y-axis between plots

1.5.8 LING LONGLINE FISHERY

The first analysis of bycatch and discards in this fishery covered the period from 1990–91 to 1997–98 (Anderson et al. 2000), and a later analysis extended this to 2005–06 (Anderson 2008). The analysis was further updated in 2014, re-estimating annual bycatch and discards for all fishing years up to 2011–12, within the standard species categories (QMS, non-QMS, and Invertebrate) (Anderson 2014a). The most recent analysis (Finucci et al. 2020) updated estimates to 2017–18, using the statistical model method first introduced for the hoki/hake/ling trawl fishery (Anderson et al. 2019).

The estimator in the statistical model was based on observed catches per set, with vessel type (autoline /hand bait), standard area, and fishing year as covariates. Estimated parameters were then applied to effort totals for the residual (non-observed) fraction of the fishery to build estimates of total bycatch and discards for the target fishery as a whole..

Between 2002–03 and 2017–18 about one third of the vessels operating in this fishery were observed (39 vessels). These observed vessels included the main operators in the fishery (including the larger autoliners) and accounted for up to 69% of the annual fishing effort (in terms of catch). Coverage in the fishery was low (less than 10%) for the period between 2010–11 and 2015–16.

Ling made up to 65% of the total estimated catch from all observed sets targeting ling between 2002–03 and 2017–18, and spiny dogfish (much of which was discarded[3]) about a further 15% (Figure 9.50). About half of the remaining 20% of the catch included the commercial species ribaldo (*Mora moro*, 3.3%), rough skates (*Zearaja nasuta*, 2.7%), smooth skates (*Dipturus innominatus,* 1.5%), and sea perch (*Helicolenus* spp., 1.4%). Altogether, about 93% of the observed catch comprised QMS species. Over 230 species or species groups were identified by observers, the majority

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[3] Spiny dogfish can legally be discarded under Schedule 6 of the Fisheries Act 1996. They were not introduced into the QMS (and Schedule 6) until 2004, strongly reducing the estimated QMS catch for 2002–03.

being non-commercial species caught in low numbers, especially Chondrichthyans, often unspecified but including shovelnose spiny dogfish (*Deania calcea*), *Etmopterus* species, and seal sharks (*Dalatias licha*). Notably, given the fishing method, a large weight of echinoderms, especially starfish (of which over 29 t were observed caught during the period), anemones, sponges, crustaceans, and molluscs were also regularly recorded by observers.

Total annual bycatch estimates for 2002–03 to 2017–18 were 1408–4724 t, compared with approximate target species catches in the same period of 2500–5700 t. Bycatch weights showed little trend over the whole period, but increased notably after 2014–15 (Figure 9.51).

Total annual discard estimates for 2002–03 to 2017–18 were 188–2440 t, with lower values between 2010–11 and 2015–16 but no trend overall (Figure 9.52). In most years, between about 30% and 85% of these discarded fish were QMS species, mainly spiny dogfish, the remainder being non-QMS, generally non-commercial, species. Ling were discarded in small amounts (about 10–150 t per year), generally being attributable to fish being lost on retrieval or predated by marine mammals and birds.

.TRENDS IN NON-TARGET CATCH BYCATCH BY SPECIES FROM THE LING BOTTOM LONGLINE FISHERY

Finucci et al. (2019) estimated the level of individual fish and invertebrate species bycatch in each fishing year from 1992–93 to 2016–17. The following conclusions were made:

- The most commonly caught bycatch species were spiny dogfish (*Squalus acanthias*, SPD), ribaldo (*Mora moro*, RIB), and smooth skate (*Dipturus innominatus*, SSK).
- Of the 131 bycatch species examined, 1 had a significant decrease in catch over time and 34 had a signficant increase in catch.
- The species showing the greatest decline were unspecified skates (SKA), Antarctic rock cod (Nototheniidae, NOT), and lanternsharks (*Etmopterus* spp., ETM) (Figure 9.53).
- The species showing the greatest increase were the hairy conger (*Bassanago hirsutus*, HCO), hoki (HOK), and swollenhead conger (*Bassanago bulbiceps*, SCO) (Figure 9.53).

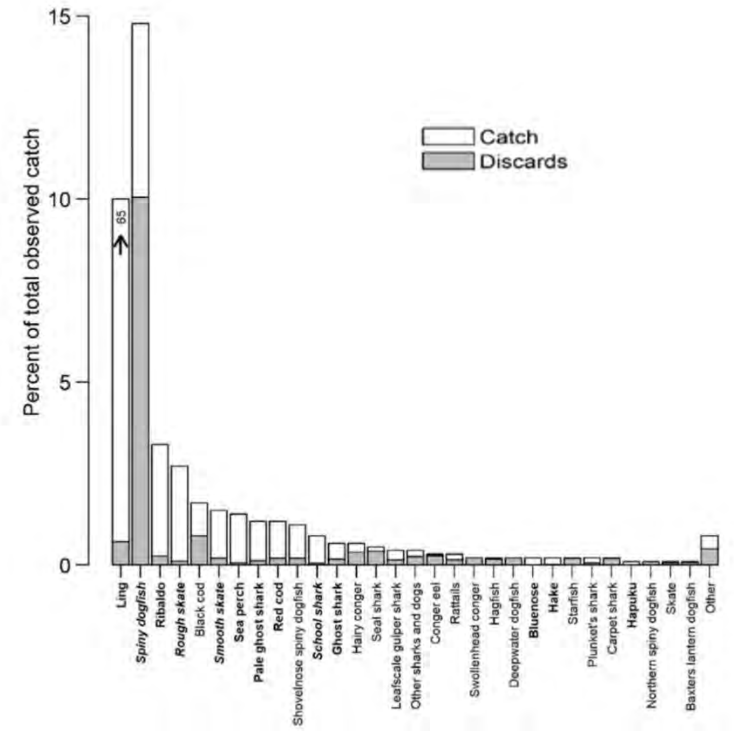

Figure 9.50: Percentage of the total catch contributed by the main bycatch species (those representing 0.02% or more of the total catch; white) in the observed portion of the ling longline fishery, 2002–03 to 2017–18 and the percentage discarded (grey; Finucci et al. 2020). QMS species are shown in bold, Schedule 6 species in italics

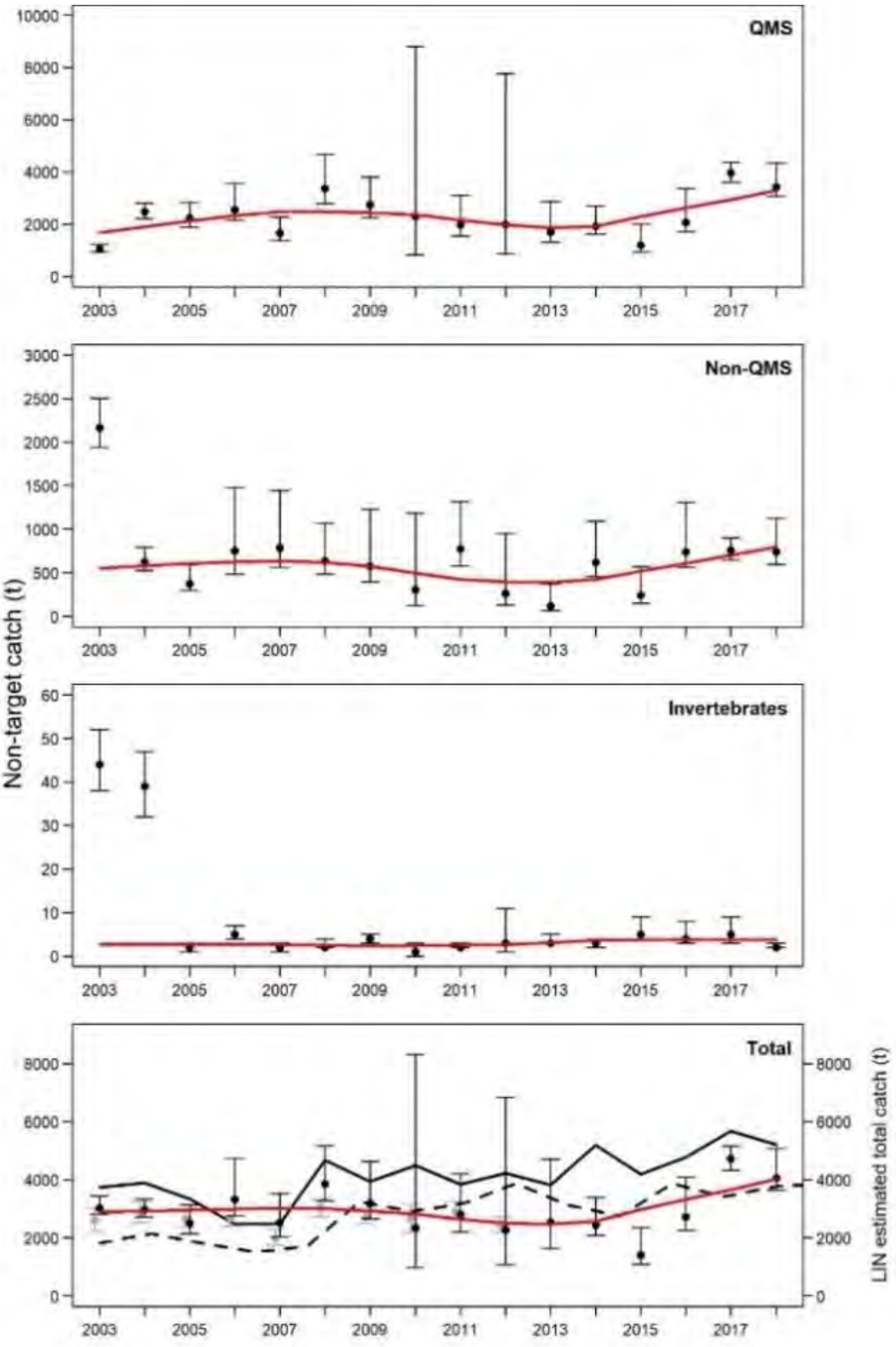

Figure 9.51: Annual estimates of non-target catch in the target ling longline fishery, by species category, for 2002–03 to 2017–18 (black dots). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual non-target catch. Bottom panel shows estimates (grey confidence intervals) of total bycatch calculated up to 2011–12 from Anderson (2014a), solid black line shows the total annual catch of ling, and the dashed black line shows annual effort (number of sets), scaled to have the mean equal to that of total bycatch.

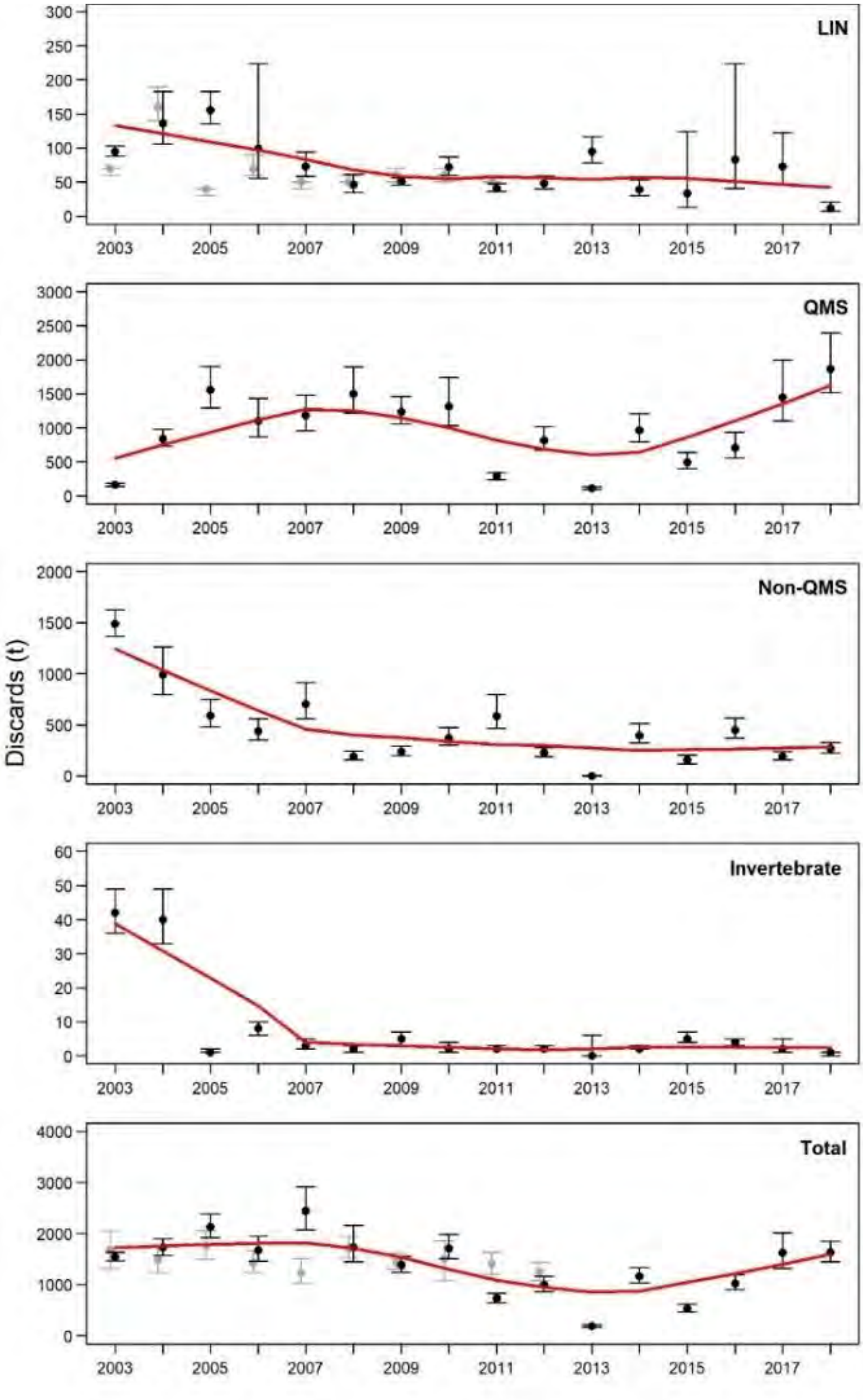

Figure 9.52: Annual estimates of discards in the target ling longline fishery, by species category, for 2002–03 to 2017–18 (black dots). Error bars indicate 95% confidence intervals. The red lines show the fit of a locally weighted polynomial regression to annual discards. Also shown (grey confidence intervals, bottom panel) are earlier estimates of target species and total discards calculated for up to 2012–13 by Anderson (2014a).

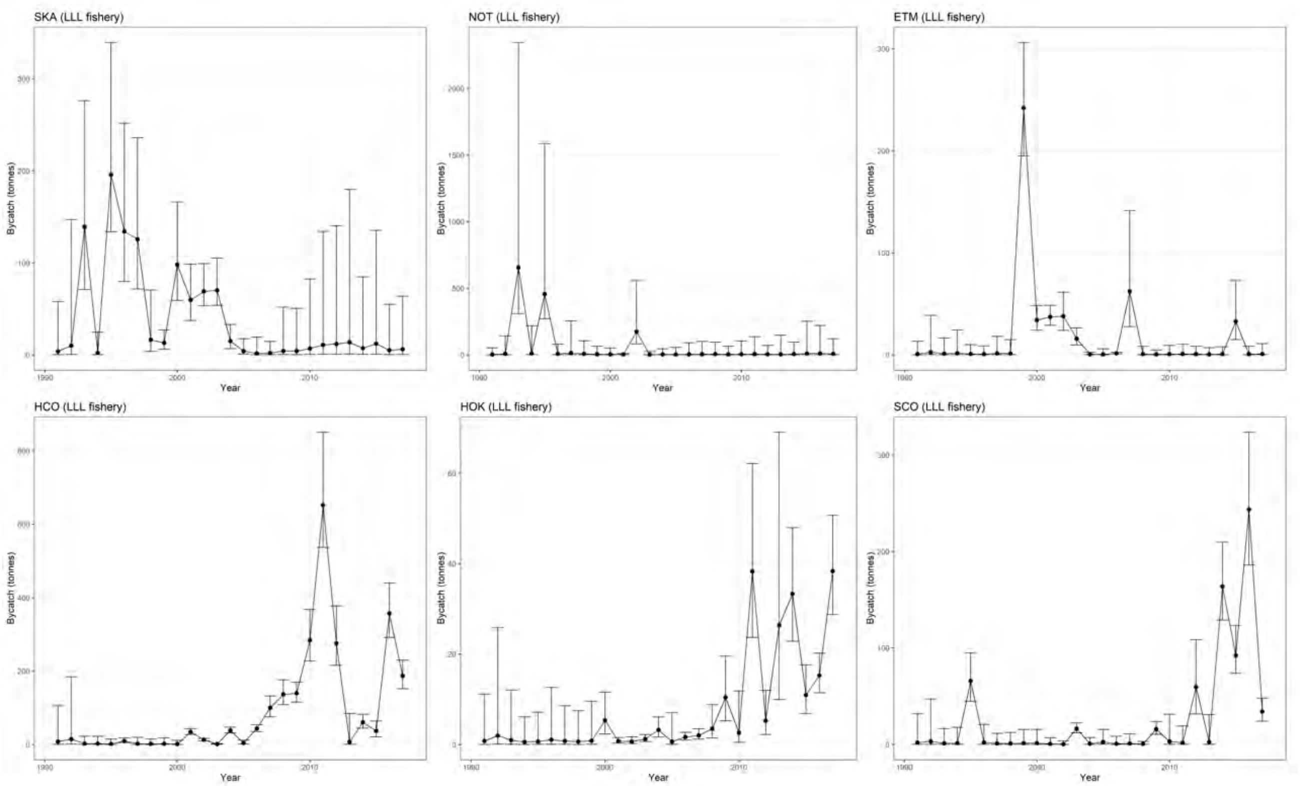

Figure 9.53: Annual non-target catch estimates in the ling longline fishery for the species which had the greatest decrease (top) and greatest increase (bottom) between 1992–93 and 2016–17 (Finucci et al. 2019). Some apparent changes in non-target catch may be due to improvements in observer identifications (see Section 9.3). See text above for species codes.

1.5.9 TUNA LONGLINE FISHERY

The New Zealand tuna longline fishery was dominated by foreign licensed vessels during the 1980s, then comprised chartered Japanese vessels and New Zealand domestic vessels from 1993–94 to 2014–15. The domestic fishing fleet has dominated the fishery since 1993–94 (Figure 9.54), and all FCVs became FOVs after 2016.

The Japanese charter fleet mainly targeted southern bluefin tuna off the west coast South Island (WCSI), and domestic vessels targeted southern bluefin tuna (*Thunnus maccoyi)* and bigeye tuna (*T. obesus*) and the fishery was concentrated off the east coast of the North Island (ECNI) with some fishing for southern bluefin tuna off the WCSI.

A detailed analysis of fish bycatch in tuna longline fisheries covered the 2006−07 to 2009−10 fishing years (Griggs & Baird 2013), and for 2010–11 to 2014–15 (Griggs et al. 2018). During 2010–11 to 2014–15, 137 492 fish and invertebrates from at least 60 species or species groups were observed. Most species were rarely observed, with only 37 species (or species groups) exceeding 100 observations between 1988–89 and 2014–15. The most commonly observed species over all years were blue shark (*Prionace glauca*), Ray's bream (*Brama brama*), and albacore tuna (*T. alalunga)*, these three making up nearly 70% of the catch by numbers. Blue shark and Ray's bream were the most abundant and second most abundant species during 2010–11 to 2014−15 (Table 9.2). Other important bycatch species were albacore, porbeagle shark, lancetfish, dealfish, offshore dogfish, swordfish, moonfish, bigscale pomfret, mako shark, and oilfish. The catch composition varied with fleet and area fished.

QMS bycatch species caught were blue shark, mako shark, porbeagle shark, school shark, moonfish, Ray's bream, and swordfish. Swordfish was also sometimes targeted.

Most blue, porbeagle, mako, and school sharks were processed in some way, either being finned or retained for their flesh, but there were significant fleet differences. Blue sharks were mainly just finned. Since October 2014, shark finning[4] has been banned in New Zealand waters. Most by-caught sharks were discarded in 2014–15, except for some mako and school sharks retained for their flesh.

Most albacore, swordfish, yellowfin tuna, moonfish, and Ray's bream were retained. Most bigscale pomfret and rudderfish were discarded, whereas butterfly tuna, escolar, and oilfish were often retained, with some year and fleet differences. Almost all offshore dogfish, dealfish, and lancetfish were discarded

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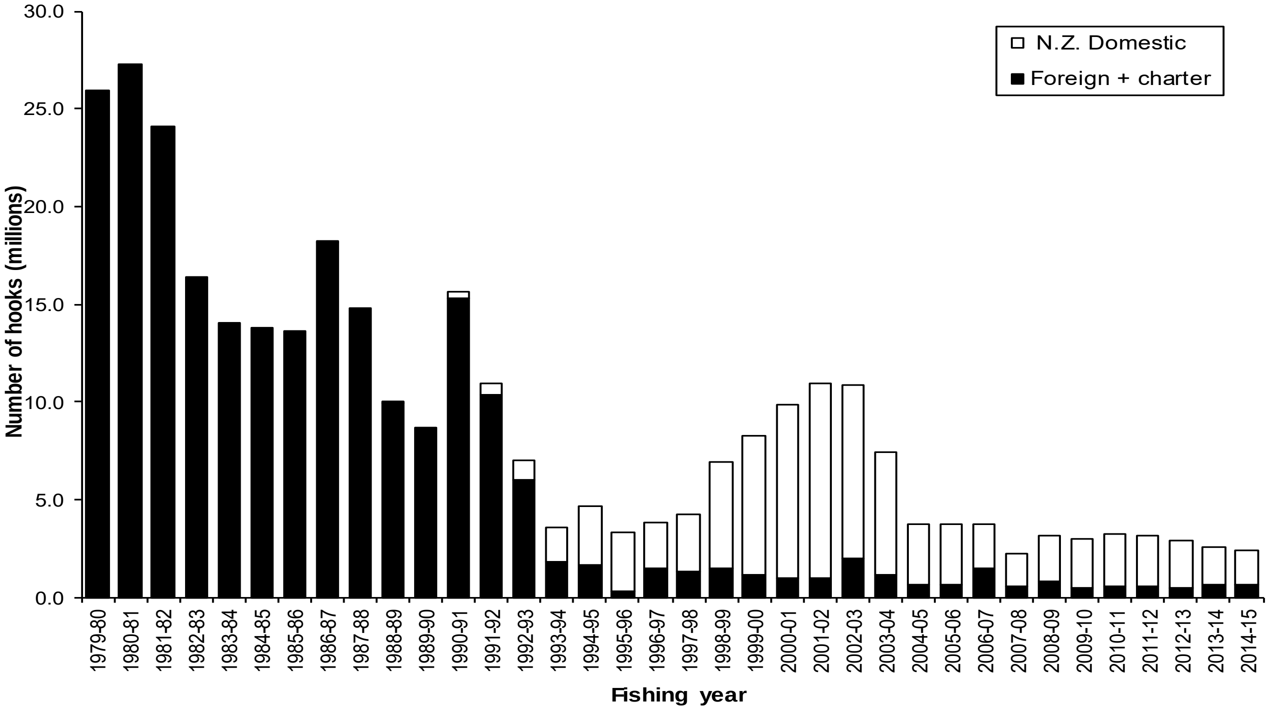

Figure 9.54: Effort (hooks set) in the tuna longline fishery. Black bars are Foreign and Charter vessels, white bars are New Zealand domestic vessels (Griggs et al. 2018).

Table 9.2: Numbers of fish reported by observers between 2010–11 and 2014–15, and the total observed catch since 1988–89. The top 30 species captured are ranked in descending order of abundance since 1988–89 (Griggs et al. 2018) [continued on next page].

`	Species	Scientific name	2010–11 to 2014–15	Total number
1	Blue shark	*Prionace glauca*	57 912	240 540
2	Ray's bream	*Brama brama*	26 427	124 632
3	Albacore tuna	*Thunnus alalunga*	9 707	111 023
4	Southern bluefin tuna	*Thunnus maccoyii*	19 149	62 440
5	Porbeagle shark	*Lamna nasus*	3 058	22 069
6	Lancetfish	*Alepisaurus ferox & A. brevirostris*	5 256	19 639
7	Dealfish	*Trachipterus trachypterus*	1 761	18 946
8	Deepwater dogfish	Squaliformes	2 459	11 571
9	Swordfish	*Xiphias gladius*	2 868	11 154
10	Moonfish	*Lampris guttatus*	1 070	10 204
11	Big scale pomfret	*Taractichthys longipinnis*	361	8 179
12	Mako shark	*Isurus oxyrinchus*	1 660	7 822
13	Oilfish	*Ruvettus pretiosus*	256	7 798
14	Escolar	*Lepidocybium flavobrunneum*	895	5 317

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[4] Shark finning is the act of removing fins from sharks and discarding the rest of the shark back into the ocean.

`	Species	Scientific name	2010–11 to 2014–15	Total number
15	Rudderfish	*Centrolophus niger*	370	5 277
16	Bigeye tuna	*Thunnus obesus*	663	5 053
17	Butterfly tuna	*Gasterochisma melampus*	510	4 979
18	School shark	*Galeorhinus galeus*	157	3 777
19	Sunfish	*Mola mola*	746	3 501
20	Yellowfin tuna	*Thunnus albacares*	29	3 371
21	Pelagic stingray	*Pteroplatytrygon violacea*	475	2 873
22	Hoki	*Macruronus novaezelandiae*	20	2 041
23	Thresher shark	*Alopias vulpinus*	100	1 500
24	Skipjack tuna	*Katsuwonus pelamis*	50	1 201
25	Dolphinfish	*Coryphaena hippurus*	192	800
26	Flathead pomfret	*Taractes asper*	106	622
27	Striped marlin	*Tetrapturus audax*	39	507
28	Black barracouta	*Nesiarchus nasutus*	84	470
29	Barracouta	*Thyrsites atun*	3	360
30	Pacific bluefin tuna	*Thunnus orientalis*	42	264

1.5.10 ALBACORE TUNA TROLL FISHERY

This fishery is carried out by small domestic vessels fishing over the summer months mainly off the west coasts of the North Island and South Island, especially WCSI.

Observers began to go to sea on troll vessels in 2007. The first two years were a trial period with one trip observed in each year. Targets were set in 2009. Coverage was 0.5–1.5% of days fished for the 2009–10 to 2012–13 fishing years.

Albacore comprised 94.4% of the observed catch over the past seven years, followed by Ray’s bream (2.7%), skipjack tuna (1.7%), and small numbers (less than 1%) of a few other species (Table 9.3).

Observer coverage on troll vessels was discontinued after 2012–13 because it was considered to not be representative enough of the fishery for length monitoring, which is carried out by port sampling.

Table 9.3: Species composition of observed albacore troll catches, 2006–07 to 2012–13.

Species	Scientific name	Number of fish caught							
		2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	Total of 7 years
Albacore tuna	*Thunnus alalunga*	1 684	1 776	1 755	5 403	4 905	2 772	3 881	22 176
Ray’s bream	*Brama brama*		18	12	537	35	7	15	624
Skipjack tuna	*Katsuwonus pelamis*	1	2	26	20	359	2		410
Barracouta	*Thyrsites atun*			1		24	13	23	61
Kahawai	*Arripis trutta*			6		3	14	14	37
Kingfish	*Seriola lalandi*			2	4	4			10
Dolphinfish	*Coryphaena hippurus*				1				1
Mako shark	*Isurus oxyrinchus*						1	1	2
Unidentified		2			174				176

1.6 INDICATORS AND TRENDS

A standard measure that can be used to characterise a fishery is the level of annual discards as a fraction of the catch of the target species. The most recent estimates (mean of last four years) are provided in Table 9.5 for those fisheries where the necessary data were available. The largest mean discard fraction comes from the scampi trawl fishery where 3.8 kg of bycatch is discarded for every kilogram of scampi caught, and the smallest discard fractions are seen in the oreo, jack mackerel, and southern blue whiting fisheries (0.01 kg).

Some general trends were identified in some fisheries, especially those examined in recent Fisheries New Zealand projects where the determination of trends in the rates and levels of bycatch over time was an explicit objective (Table 9.6).

Comparison of estimates of total bycatch over time from all the offshore trawl fisheries (Figure 9.55) shows the substantial contribution from the large combined hoki/hake/ling/silver warehou/white warehou trawl fishery (2018–19 hoki total TACC of 115 000 t) even though the relative rate of *discards* from these fisheries is low (see Table 9.5). This figure also shows the relatively large bycatch from the scampi fishery (2018–19 scampi total TACC of 1312 t) and the arrow squid fishery (2018–19 arrow squid total TACC of 82 120 t).

Finucci et al. (2019) analysed temporal (1990–91 to 2016–17) bycatch trends for individual species or species groups for seven offshore trawl fisherieries and one bottom longline (ling) fishery. Bycatch regression slope coefficients assessed for each species and fishery showed a consistent increase (in six or more of the eight fisheries) for pale ghost shark (*Hydrolagus bemisi*), rough skate (*Zearaja nasuta*), leafscale gulpher shark (*Centrophorus squamosus*), and Baxter's dogfish (*Etmopterus granulosus*); and consistent decline for skates (Rajidae and Arhynchobatidae), dark ghost shark (*Hydrolagus novaezealandiae*), unidentified sharks and rattails, and bluenose (*Hyperoglyphe antarctica*). Some of the trends may be attributable to changes in reporting behaviour, e.g., increased reporting of specific skates and reduced use of the generic skate category. It seems likely that a bycatch decline for well-known species such as bluenose may represent a change in availability, abundance, or distribution of that species.

Table 9.5: Utilisation rates (kilograms of discards per kilogram of target species catch) and annual target catch (tonnes). The numbers are the most recent estimate (mean of the most recent four years available) from referenced reports.

Fishery	Utilisation rate (discards/target species catch)	Mean annual target species catch (tonnes)
Arrow squid trawl	0.12	21 991
Ling longline	0.24	4 957
Hoki/hake/ling trawl	0.04	155 705
Jack mackerel trawl	0.01	30 058
Southern blue whiting trawl	0.01	25 132
Orange roughy trawl	0.04	7 185
Oreo trawl	0.01	10 203
Scampi trawl	3.83	767

Table 9.6: Trends in non-protected species bycatch from recent Fisheries New Zealand projects where trend determination was an objective.

Fishery	Trends	Fishery	Trends
Arrow squid trawl	Linear regressions of annual bycatch estimates since 2002–03 indicated decreasing bycatch over time (negative slopes) in each of the major species categories examined (i.e., QMS species, non-QMS species, Invertebrate species, and all species combined). These trends were statistically significant ($p < 0.01$) in each case. Linear regressions of annual discard estimates since 2002–03 also indicated decreasing levels over time in each catch category excluding the target species. These declines were statistically significant for non-QMS species, invertebrate species, and all species combined.	Orange roughy trawl	Increased non-QMS species bycatch quantities between the mid-1990s and mid-2000s were shown to strongly correlate with an overall increase in mean trawl length in the fishery resulting from increased effort away from undersea features (Anderson 2009a). Linear regressions indicated significantly decreasing levels of both bycatch and discards since 2001–02 for several species categories: QMS species, non-QMS species, invertebrates, morid cods, sharks, slickheads, and all species combined.
Jack mackerel trawl	Linear regressions showed decreasing non-target catch of QMS species and stable trends of non-target catch for non-QMS and invertebrate species. The trends for QMS and non-QMS species were statistically significant. Linear regressions showed increasing discards of QMS species which was statistically significant. All other categories showed no trend over time. Discarding has consisted mostly of Schedule 6 species (spiny dogfish and, since 2015–16, an increasing trend of kingfish).	Ling longline	Linear regression modelling of annual bycatch estimates since 2002–03 indicated increasing bycatch over time for QMS species and decreasing bycatch of non-QMS fish and non-QMS invertebrates, but these trends were not significant. Discards of ling significantly decreased, however, and non-significant decreases of non-QMS fish and non-QMS invertebrate were also observed.
Scampi trawl	Linear regressions of annual bycatch estimates since 2002–03 indicated decreasing bycatch over time (negative slopes) in each of the major species categories examined (i.e., QMS species, non-QMS species, invertebrate species, and all species combined). None of these trends were statistically significant ($p < 0.01$). Linear regressions of annual discard estimates since 2002–03 indicated decreasing levels over time for the target species and in the invertebrate species category and increasing levels in the QMS and non-QMS species categories, and for all species combined. However, none of these trends were statistically significant ($p < 0.01$).	Hoki, hake, ling trawl	Linear regressions of annual bycatch estimates since 1990–91 indicated that bycatch in the QMS species, invertebrate species, and all species combined categories generally increased over time, and these trends were statistically significant for QMS species and invertebrate species; but for the largest category, non-QMS species, there was a (non-significant) decline in bycatch over time.
Oreo trawl	Linear regressions indicated significantly decreasing levels of both bycatch and discards since 2001–02 for non-QMS species, invertebrates, morid cods, rattails, and all species combined.		

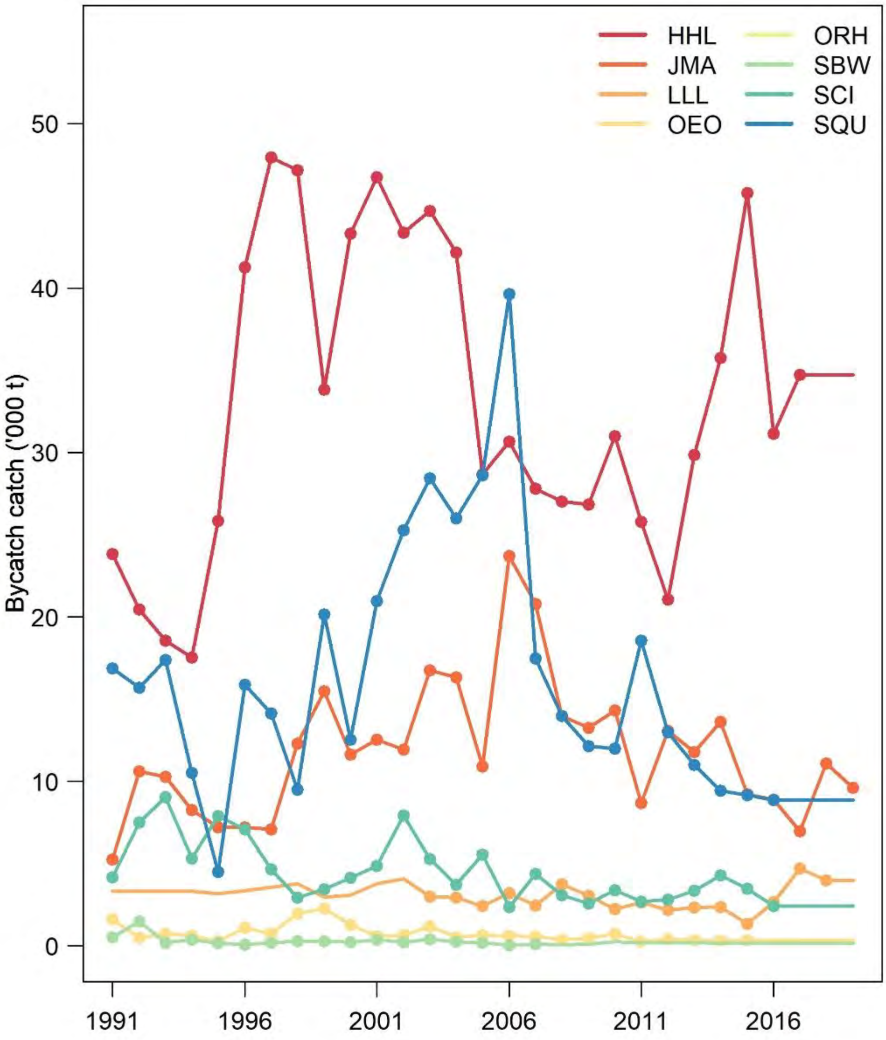

Figure 9.55: Comparison of total estimated bycatch for all the offshore trawl fisheries 1990–91 to 2016–17, and to 2017–18 for the ling longline fishery. Dots are model based estimates; no dots assume bycatch unchanged from previous year or, for the most recent assessment (for ling longline), indicate the years updated. For species codes see Table 9.1, HHL is hoki/hake/ling mixed trawl fishery.

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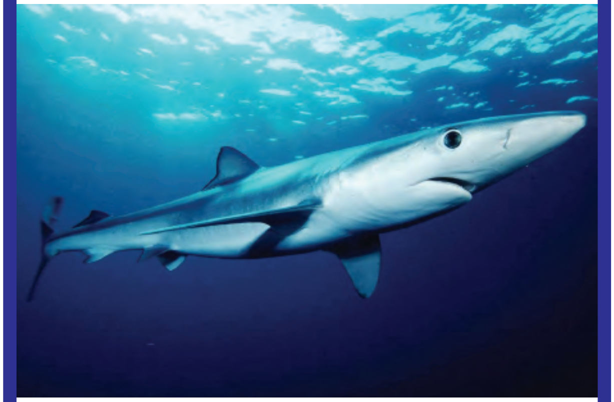
Chapter 10: Chondrichthyans (sharks, rays and chimaeras) - Technical Summary

Fishing threats to sharks



Non-target and incidental catches

1. THE ISSUE IN BRIEF

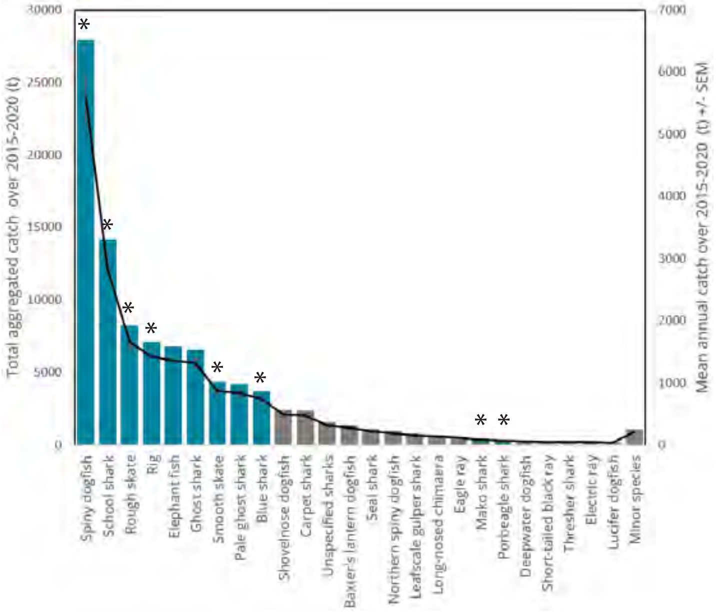
- Chondrichthyans (sharks, rays, and chimaeras) usually are long-lived, have low fecundity, and some are slow growing
- These characteristics make them potentially more vulnerable to adverse fishing effects
- 77 species of chondrichthyans have been recorded in fisheries landings, 11 of these are managed under the Quota Management System (QMS), and 7 are protected in NZ
- International conventions, national policy, and legislation govern the management of chondrichthyans in NZ

2. CHONDRICHTHYAN DIVERSITY

- There are 113 species of chondrichthyans reported in NZ waters (and 3 ray species in the Ross Sea, Antarctica), comprising of 12 chimaeras, 26 rays, and 74 sharks
- A high percentage (30%) of these chondrichthyans are only found in NZ waters (e.g. dark ghost shark)
- Chondrichthyans such as the whale shark or the manta ray are typical of warmer waters and are rare in NZ, but are occasionally seen in the northern part of NZ waters during summer (see 5.)

3. CHONDRICHTHYAN FISHERIES

- Several chondrichthyan species are of commercial interest for fisheries in NZ, and 11 species (accounting for up to 90% of total chondrichthyan catch) are managed through the QMS
- Fisheries data analyses indicate that QMS chondrichthyan stocks in NZ are either stable or increasing, with few local exceptions (e.g. pale ghost shark)



- Dogfish, porbeagle, blue shark, and other, non-QMS, species are often discarded. Some species can be discarded dead or alive (and are counted against the annual catch entitlement, i.e. Schedule 6 discard) whereas others can be released alive (and thus in most cases are not counted against the annual catch entitlement, i.e. Schedule 6 release)

Reported total catches (landing, discards, and live releases) for chondrichthyan species, aggregated across 2015-2020 (mean indicated with a solid line and on the right vertical axis). Spiny dogfish, school shark, and rough skate (all in the QMS) had the highest catch tonnage. Schedule 6 species are marked with *

4. RISK ASSESSMENT

- The 2013 NZ National Plan Of Action for the Conservation and Management of Sharks established a risk-based approach to prioritising management actions
- A qualitative risk assessment was run in 2014, gathering expert knowledge on the risk to all chondrichthyans from commercial fishing in New Zealand
- The risk-assessment process was repeated in 2017 with similar methods, incorporating updated knowledge and including 50 species prioritised by the first assessment, with the goal to further select species for additional research, more quantitative future assessments, and management

- The main outcomes of the risk assessment were:
  - A general consensus that there is a low risk of unsustainable fishing impacts for most species
  - Species that are caught in small quantities are data poor, and consequently there is low confidence in their risk scores
  - The highest estimated risks were for some QMS and deepwater shark species, whereas protected species (i.e. basking shark, spinetail devilray and white pointer shark) had lower risk scores

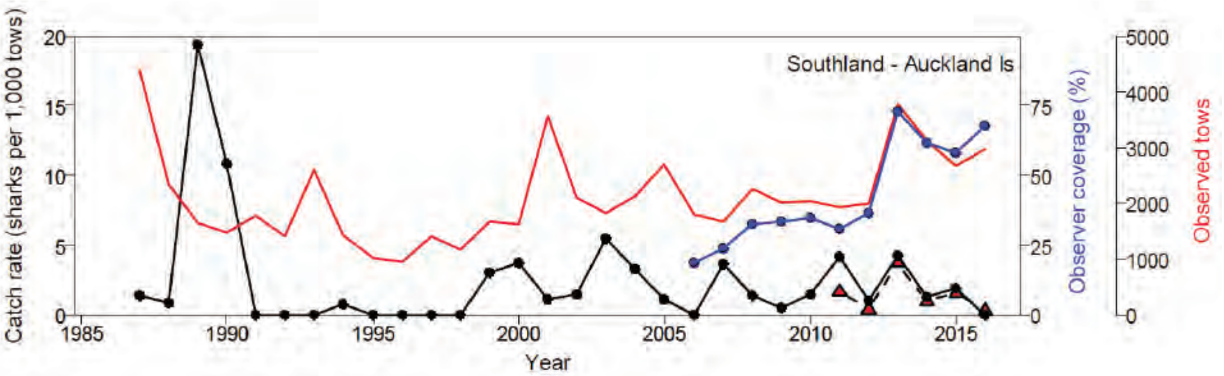
- An updated semi-quantitative risk assessment for selected species is in development

5. PROTECTED SPECIES

- The Wildlife Act 1953 protects seven species of chondrichthyans:
  - Basking shark (since 2007)
  - White pointer shark (since 2010)
  - Whale shark (since 2010)
  - Deepwater nurse shark (since 2010)
  - Oceanic whitetip shark (since 2013)
  - Spine-tailed devil ray (since 2010)
  - Manta ray (since 2010)

- It is illegal to hunt or kill these species within NZ waters, as well as trade in teeth, jaws, and fins.
  However, although it is not illegal to incidentally capture or injure these species, it is mandatory to report captures and interactions

- Despite some uncertainties, analyses suggest that basking shark capture indicators are stable in recent years
- Monitoring incidental captures should be continued to ensure sustainability of fisheries



Basking shark capture rate index for the Southland-Auckland Is. area (Francis 2017). Incidental captures per unit of effort in fisheries (black lines), as well as coverage by fisheries observers (coloured lines)

6. ONGOING RESEARCH

- The trialling of semi-quantitative risk assessments for chondrichthyan populations is underway
- A characterisation study of protected chondrichthyan species captures in commercial, recreational and customary fisheries is nearing completion

10 CHONDRICHTHYANS (SHARKS, RAYS, AND CHIMAERAS)

Status of chapter	This chapter has had minor updates for AEBAR 2021.
Scope of chapter	This chapter outlines the relevant biology of New Zealand chondrichthyans, the nature of any fishing interactions, the management approach, and trends in key indicators of fishing effects. This chapter covers Quota Management System (QMS), non-QMS, and protected sharks.
Area	The New Zealand EEZ and Territorial Sea.
Focal localities	This differs depending upon the species or fishery examined.
Key issues	Sustainability of fisheries mortalities, with a focus on non-QMS species
Fisheries New Zealand research (current)	ENV2018-06 Improved distribution information for higher risk non-QMS shark species.
Other government research (current)	DOC CSP research: DOC19302 Updated analysis of spine-tailed devil ray post release survival.
University research	Biological and behavioural information is currently known from research at Waikato, Otago, Victoria, and Auckland universities on a variety of species.
Related issues/chapters	See the Non-protected species (fish and invertebrates) bycatch chapter. More detail is provided for QMS species in the stock assessment plenary (Fisheries New Zealand 2021a, Fisheries New Zealand 2021b).

10.1 CONTEXT

Chondrichthyans (cartilaginous fishes) comprise all fish species (except lampreys and hagfish) that lack true bone in their skeletons (as do teleost fishes), specifically sharks, rays, skates, and chimaeras. In New Zealand, seven chondrichthyans are totally protected under the Wildlife Act (1953). The impacts of fishing on chondrichthyans are managed under the Fisheries Act (1996), with eleven species subject to the Quota Management System (QMS) and two species (hammerhead shark, *Sphyrna zygaena*, and sharpnose sevengill shark, *Heptranchias perlo*) prohibited as target species (although they might still be taken as non-target catch). The management policy framework is contained in Fisheries Plans developed for Deepwater, Highly Migratory, and Inshore fisheries (see Chapter 1 for fuller descriptions and web links).

New Zealand has international obligations to collaborate with other countries in the assessment and management of shared and migratory chondrichthyan stocks. New Zealand participates in a number of Regional Fisheries Management Organisations that have some responsibility for chondrichthyans, including Western and Central Pacific Fisheries Commission (which manages tuna fisheries and the associated species), Commission for the Conservation of Southern Bluefin Tuna, Commission for the Conservation of Antarctic Marine Living Resources (toothfish), and the South Pacific Regional Fisheries Management Organisation (SPRFMO; multiple non-Highly Migratory Species). New Zealand is also a signatory to conventions that play a role in the management of some species, including the Convention on International Trade in Endangered Species of Wild Fauna and Flora and the Convention on the Conservation of Migratory Species of Wild Animals.

To address global concerns about the management of chondrichthyans[1], the Food and Agriculture Organisation of the United Nations (FAO) developed an International Plan of Action for the Conservation and Management of Sharks (IPOA) [2]. The IPOA builds upon the FAO Code of Conduct for Responsible Fisheries and was endorsed by the FAO Council

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[1] In the IPOA and in the NPOA–Sharks, 'sharks' are defined to include all chondrichthyans, viz. sharks, rays and chimaeras. However, in this chapter, we use the terms chondrichthyans, sharks, rays, chimaeras in their strict sense to avoid confusion. Skates are a type of ray and are grouped with rays.

[2] FAO International Plan of Action for Conservation and Management of Sharks, http://www.fao.org/ipoa-sharks/en.

in June 1999 and subsequently adopted by the November 1999 FAO Conference. The overarching objective of the IPOA is: "to ensure the conservation and management of sharks and their long-term sustainable use". To achieve this goal the IPOA suggests that each member state of FAO that regularly catches sharks, either as target or incidental catch, should develop a National Plan of Action for the Conservation and Management of Sharks (NPOA-Sharks).

New Zealand developed an NPOA–Sharks which came into effect in October 2008 (Ministry of Fisheries 2008), and this was superseded by the NPOA-Sharks 2013 (Ministry for Primary Industries 2013). The NPOA uses "sharks" to include all chondrichthyans. The purpose of the NPOA-Sharks 2013 is:

> *"To maintain the biodiversity and the long-term viability of all New Zealand shark populations by recognising their role in marine ecosystems, ensuring that any utilisation of sharks is sustainable, and that New Zealand receives positive recognition internationally for its efforts in shark conservation and management."*

To do so, it identifies goals and five-year objectives in the following key areas:

- Biodiversity and long-term viability of shark populations;
- Utilisation, waste reduction, and the elimination of shark finning[3];
- Domestic (national) engagement and partnerships;
- Non-fishing threats;
- International engagement;
- Research and information.

It is a comprehensive plan with the objective of improving our knowledge of shark populations and their interactions with fisheries and assist us to base conservation and management actions on an assessment of risks. The NPOA-Sharks 2013 is expected to be updated in 2022.

The NPOA-Sharks 2013 applies to all chondrichthyans that are found within New Zealand's Exclusive Economic Zone (EEZ) and Territorial Sea (New Zealand waters within 12 nautical miles of the coast), migratory species that frequent New Zealand fisheries waters, and species taken by New Zealand-flagged vessels fishing on the High Seas (including the Ross Sea, Antarctica). Appendix 19.8.1 provides a list of all 116 known New Zealand and Ross Sea chondrichthyans, along with their management class and International Union for Conservation of Nature (IUCN) and Department of Conservation (DOC) threat classes.

10.2 BIOLOGY

The population dynamics of chondrichthyans differ markedly from those of most bony fishes. Their reproductive strategy produces a small number of well-developed young, rather than spawning large numbers of undeveloped eggs as do most bony fishes. Chondrichthyans either lay large yolky eggs on the seabed or give birth to live young, but in both reproductive modes the number of young produced annually is usually in single digits or in the low tens. A few species may produce more than 100 young per litter (e.g., the blue shark, *Prionace glauca*, has up to 135 young; Last & Stevens 2009), but even in these more fecund species, large litter sizes are exceptional and the average number of young per female is much lower (30–40 in the blue shark; Last & Stevens 2009). Gestation periods and reproductive cycles last 10 months to two years in many species and may be as high as three years (e.g., school shark, *Galeorhinus galeus*; mako shark, *Isurus oxyrinchus*; Mollet et al. 2000, Walker 2005). Fecundity may increase with the size of females (e.g., rig, *Mustelus lenticulatus*, and school shark; Francis & Mace 1980, Walker 2005), so if human activities reduce the average size of females in a population (as often happens in fisheries) the reproductive output may decline faster than the rate of population decline. These characteristics mean that chondrichthyans have a much closer, potentially almost linear, relationship between population size and recruitment. They also have limited capacity for density-dependent compensation that might boost reproductive output at low population sizes, e.g., through increased growth and reproductive rates.

Many cartilaginous fishes are also slow growing, further reducing their capacity for recovering from population declines. Many species have ages at maturity greater than

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[3] Shark finning is the act of removing fins from sharks and discarding the rest of the shark back into the ocean.

10 years and longevities in excess of 20 years, although some are faster growing and are therefore more productive (e.g., rig, Francis & Ó Maolagáin 2000). The combination of low reproductive rate and low growth rate makes chondrichthyans particularly vulnerable to overfishing (Camhi et al. 1998, Smith et al. 1998, Dulvy et al. 2003, 2017, Pikitch et al. 2008, Simpfendorfer & Kyne 2009).

Recent studies have been carried out to estimate the age, growth, length at maturity, and size and age composition of the catch of three main pelagic sharks taken as bycatch in surface longline fisheries (blue; porbeagle, *Lamna nasus*; and mako sharks) (Francis 2015, Francis 2016b, Francis & Ó Maolagáin 2016). Similar studies have also been carried out on a group of small inshore and deepwater sharks and rays (Francis et al. 2018a, 2018b). Age estimation was equivocal for some of these species (especially blue shark), and no ageing validation was carried out for any of the species, so our knowledge of growth and age-related parameters (maturity, longevity) remains uncertain. Nevertheless, vertebral band counts of common electric ray (*Torpedo fairchildi*), blind electric ray (*Typhlonarke aysoni*), and carpet shark (*Cephaloscyllium isabella*) indicated that these species were relatively fast growing and therefore productive, whereas band counts of dorsal fin spines in several deepwater sharks (Owston's dogfish, *Centroscymnus owstonii*; longnose velvet dogfish, *Centroselachus crepidater*; and Plunket's shark, *Scymnodon plunketi*) supported Australian studies indicating that they are slow-growing and long-lived.

Satellite tags have been deployed on porbeagle, mako, and hammerhead sharks to identify their movement patterns, migratory behavior, habitat requirements, and use of the water column (Francis et al. 2015b, Francis 2016a, M. Francis unpublished data). Porbeagle sharks make seasonal north-south migrations and are vertical migrators, spending daytime at depths of 200–600 m and night-time at 50–100 m. Mako sharks may remain resident in coastal waters for several months, but periodically (mainly in winter) migrate northwards to subtropical and tropical waters. Juvenile hammerheads inhabit coastal waters for their first few years of life. Satellite tagging has also been used on white pointer sharks (*Carcharodon carcharias*, also known as great white shark) and spinetail devil rays (*Mobula japonica)* to investigate movement and post-capture survival (see e.g., Bonfil et al. 2010, Duffy et al. 2012, Francis & Jones 2017).

Biological parameters (e.g., growth, longevity, reproduction, stock identity) were reviewed for Pacific pelagic sharks, providing a readily available summary of inputs for subsequent population modelling (Clarke et al. 2015).

Anthropogenic threats other than fishing were reviewed for rig populations (Jones et al. 2015). Kaipara Harbour is the most important rig nursery area in the country, and it is significantly impacted by agricultural activities, with the southern area especially vulnerable to the effects of future urbanisation. Raglan and Kawhia harbours may also represent significant nursery areas that are impacted mainly by agricultural activities rather than urbanisation.

Hernández et al. (2015) did a genetic study of South Pacific school shark and found evidence of mixing between New Zealand and Australia (which is consistent with previous tagging results), but genetic separation of the Australasian population from the Chilean population.

10.3 GLOBAL UNDERSTANDING OF FISHERIES INTERACTIONS

There are numerous examples worldwide of chondrichthyan stocks collapsing under fishing pressure, and until recently little attention has been focused on their management. This situation reflects the generally low importance of chondrichthyans in terms of quantity and value in commercial catches, and the consequent low research and management priority accorded to them. An important driver in the increase in global shark fishing mortality, over the last two decades, was growing demand for shark and ray meat and fins, and other products such as mobulid gill plates, cartilage, skin, or liver oil (Bräutigam et al. 2015). Although there is evidence of a decline in the shark fin trade since 2011 (Dent & Clarke 2015), many chondrichthyan populations are now believed to be severely depleted. There is also widespread public opposition to shark 'finning', in which only the fins are kept and the rest of the shark is discarded at sea, because of concerns about sustainability, wastage, and finning of live sharks. In New Zealand, live shark finning is an offence under the Animal Welfare Act 1999) and shark finning, as defined, was banned in October 2014. The results of this ban are now apparent, with high release rates for some species (e.g., blue and porbeagle sharks, see Table 10.3).

Chondrichthyans are caught in nearly all parts of the world, ranging from tropical to polar waters, and from estuaries and shallow coastal waters to the deepest areas fished. Chondrichthyans are caught by most fishing methods, although trawling, netting, and longlining are the most significant ones. Reported global landings of chondrichthyans increased steadily up to almost 900 000 t in the early 2000s but have subsequently declined to a level ~11% lower (Worm et al. 2013, Davidson et al. 2015; Figure 10.1). However unreported catches are undoubtedly substantial so the true extent of chondrichthyan catches remains unclear (Bonfil 1994, Camhi et al. 1998, Clarke et al. 2006, Worm et al. 2013). Estimates of the mortality rates of chondrichthyans at the time they are hauled to a fishing vessel are available for some species (e.g., Francis et al. 1999a, Campana et al. 2009, Griggs & Baird 2013), and increasing attention is being devoted to estimating the survival of sharks released alive by fishers (Moyes et al. 2006, Campana et al. 2009, Musyl et al. 2011, Hutchinson et al. 2013, Ellis et al. 2017). The post-release mortality of mako and silky (*Carcharhinus falciformis*) sharks released from tuna longlines in New Zealand, Fiji, New Caledonia, and Marshall Islands (Clarke et al. 2017) and the mortality of spinetail devil rays released from purse seiners fishing for skipjack tuna in New Zealand (Francis & Jones 2017) have been recently investigated

There is ample evidence that globally many chondrichthyan populations are now overfished and that fishing effort is still expanding in habitats containing some of the most vulnerable species, especially deepwater chondrichthyans (Kyne & Simpfendorfer 2007, Simpfendorfer & Kyne 2009, Rice & Harley 2012a, 2012b). Management measures have been implemented by many countries, particularly for targeted species, and Regional Fisheries Management Organizations are paying greater attention to the need to manage species that occur in international waters or waters that straddle national waters. Efforts are also focusing on reducing shark finning, particularly in fisheries catching pelagic sharks, by requiring fins to be attached to sharks at the point of landing, or to comprise no more than 5% of the landing by weight. It is not clear that this requirement has been effective in reducing catches (Clarke et al. 2012, Worm et al. 2013)

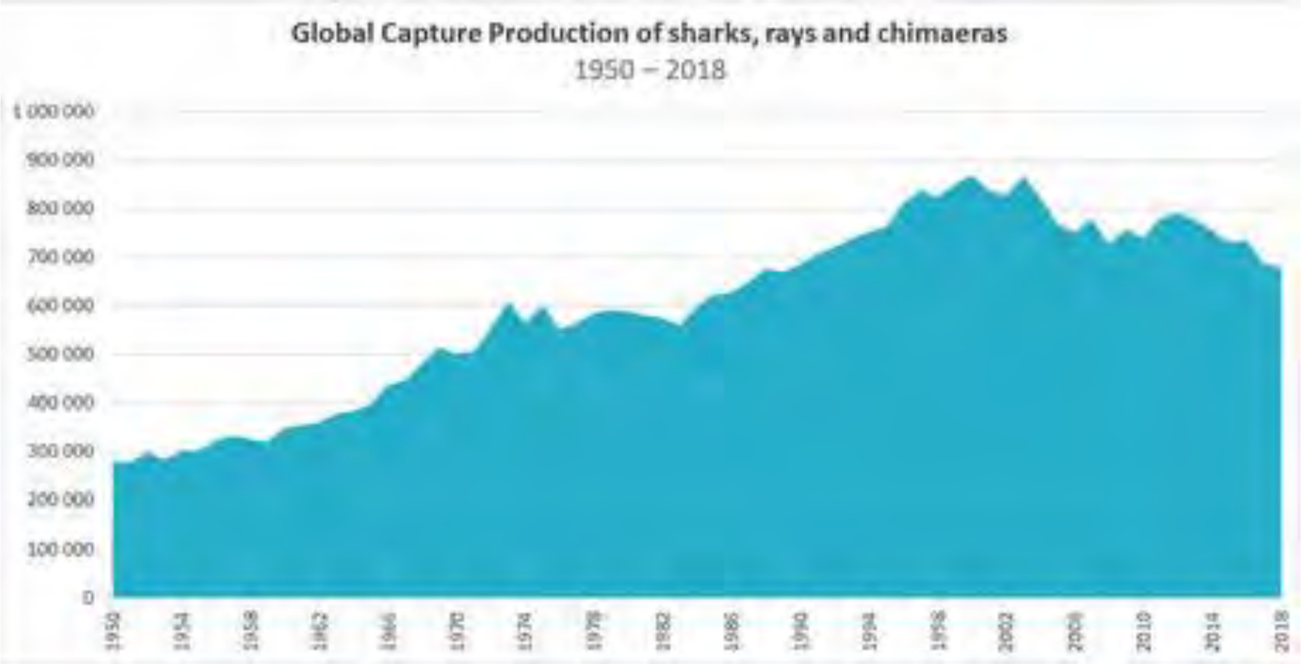

Figure 10.1: Global catches of chondrichthyans reported to FAO (1950–2018). *Source*: FAO 2020. Fishery and Aquaculture Statistics. Global capture production 1950-2018 (FishstatJ

10.4 STATE OF KNOWLEDGE OF FISHERIES INTERACTIONS IN NEW ZEALAND

A total of 112 chondrichthyans are known from New Zealand waters (and 3 from the Ross Sea) (Appendix 19.8.1), however that number is expected to change with the continuation of taxonomic studies on deepwater species. Of these species, 12 are chimaeras, 29 are skates and rays (3 from the Ross Sea), and 74 are sharks. Many New Zealand species also occur elsewhere in the world (some have worldwide distributions), but a high percentage (30%) are considered endemic to New Zealand. The high percentage of endemic species makes New Zealand's fauna unique and distinctive. New Zealand's chondrichthyan fauna is small compared with that in Australia, which has more than 322 species (Last & Stevens 2009), and that partly reflects New Zealand's lack of tropical environments.

The largest threat to chondrichthyan populations is from fishing activities, although other potential impacts include underwater noise, dredging, sonar surveys, electromagnetic fields generated by power stations and undersea cables, loss of habitat, eutrophication and sedimentation, entrapment by aquaculture facilities, shark ecotourism, and the indirect effects of climate change (Francis & Lyon 2013, Jones et al. 2015). More than 70 of New Zealand's chondrichthyan species are caught (deliberately or incidentally) by fishers (Ministry for Primary Industries 2013). Eleven chondrichthyans are managed under the QMS (Fisheries New Zealand 2021a, Fisheries New Zealand 2021b), seven are protected (Francis & Lyon 2012), two cannot be targeted, and the remainder are Non-QMS species (Appendix 19.8.1). Due to reporting requirements, commercial landings of chondrichthyans are well known, but less is known about recreational and customary catches.

A nationwide survey from 1 October 2011 to 30 September 2012 provided estimates of recreational chondrichthyan catches (Wynne-Jones et al. 2014) and was updated in 2017–18 (Wynne-Jones et al. 2019); the 2017–18 results are reported in Table 10.1. The majority of the recreational catch is from inshore QMS species, although mako shark inhabits both inshore and offshore regions. 'Stingray' is likely to include more than one species and 'sand shark' is likely to refer mainly to rig or school shark. Mako sharks are also targeted or caught as bycatch in the gamefish fishery, so estimates for mako are potentially underestimates because the survey was not designed to sample gamefishers. Estimates in tonnes are only available for rig and spiny dogfish (*Squalus acanthias*), and these constituted > 4.0% and > 0.4%, respectively, of the reported commercial landings in the same year for those species. All subsequent data reported in this chapter are from the commercial fisheries.

Commercial catches of chondrichthyan species during the five-year period 2015−16 to 2019−20 are shown in Table 10.2 and Figure 10.2. Spiny dogfish produced by far the greatest catches, followed by school shark, rough skate (*Zearaja nasuta*), rig, dark ghost shark (*Hydrolagus novaezealandiae*), and elephantfish (*Callorhinchus milii*). Blue shark, pale ghost shark (*Hydrolagus bemisi*), and smooth skate (*Dipturus innominatus*) formed a second tier. The remaining species had relatively low catches (less than about 400 t per year on average). In implementing the objectives of the NPOA Sharks 2013, Fisheries New Zealand has successfully worked with the fishing industry to increase the use of species-specific codes and decrease the use of generic codes (Unspecified sharks (OSD) and deepwater dogfish (DWD)). OSD was once a significant category (402 t/year), but the use of this code has declined by one-third since the late 2000s. DWD has become a minor category, declining by over 70% since the late 2000s. This indicates that fishers have become better at reporting their shark catch to the species level.

Among QMS species, the share of reported discards in 2015–16 to 2019–20 was highest for porbeagle and ghost sharks, but still significantly lower than those of non-QMS species (e.g., nearly all carpet shark, stingrays, electric ray, and three-quarters of northern spiny dogfish) (Table 10.3). Under Schedule 6 of the Fisheries Act, eight chondrichthyan species can be legally returned to the sea if alive, and from 2006−07 such releases were not required to be balanced against Annual Catch Entitlement (ACE) (Table 10.4). Spiny dogfish may also be discarded dead, but they must be balanced against a fisher's ACE whether dead or alive. The conditions of Schedule 6 releases have been amended for mako, porbeagle, and blue sharks: from 1 October 2014, fishers have been allowed to return these three species to the sea both alive and dead, although the status must be reported accurately. Those returned to the sea dead are counted against a fisher's ACE and the total allowable catch limit for that species. Schedule 6 returns in 2015–16 to 2019–20 were a large proportion of the catch of spiny dogfish and porbeagle, mako, and blue sharks (69%, 47%, 59%, and 85%, respectively, Table 10.3). The survival rate of discarded and released sharks is generally unknown and probably varies with species, fishing method, handling, and other factors.

10.4.1 QMS SPECIES

The eleven chondrichthyans managed under the QMS are given in Table 10.4 with their Total Allowable Commercial Catches (TACCs) and 2016−17 landings. Landings of all but one species (elephantfish) were below the TACCs.

QMS chondrichthyans are treated in detail in Fisheries New Zealand's annual Fisheries Assessment Plenary reports (Fisheries New Zealand 2021a, Fisheries New Zealand 2021b) and that material is not repeated here. Quantitative stock assessments have been attempted for only three chondrichthyan stocks (rig in SPO 3 and SPO 7, and elephantfish in ELE 3) but none were approved by the working groups and ultimately the status of all these stocks is now estimated from trends in standardised CPUE analyses and trawl surveys.

A summary of the status of the stocks of QMS chondrichthyans is given in Appendix 19.8.2 (based on Fisheries New Zealand Status of the Stocks 2021 data). Stock status has been estimated for six of the 11 QMS chondrichthyans, and 20 of the 45 non-nominal stocks (non-nominal stocks are fish stocks that have a demonstrated significant commercial or non-commercial potential). Other stocks are not assessed because an appropriate quantitative analysis to ascertain stock status relative to a target or limit has not been undertaken, or because such an analysis was not definitive, generally because of insufficient or inadequate data. None of the stocks assessed were considered to be below the 'soft limit' (below this level, a fish stock is considered to be overfished and needs to be actively rebuilt, for example by reducing the total allowable catch). Two elephantfish stocks and one rig stock were considered to be 'about as likely as not' to be in an ‘overfished’ state (reflected also in Table 10.5); the remainder of the stocks were considered likely to be in a favourable state.

Quantitative risk assessments have recently been conducted for the Southern Hemisphere porbeagle shark stock, and the Pacific Ocean bigeye thresher shark (*Alopias superciliosus*) stock. Both assessments incorporated New Zealand data and can be considered applicable to the New Zealand portion of those stocks (Fu et al. 2016, Hoyle et al. 2017). For bigeye thresher sharks, total fishing mortalities from pelagic longline fisheries in the Pacific since 2000 were generally low (less than 5%) but exceeded the maximum impact sustainable threshold in some years. For porbeagle shark, the risk assessment indicated low fishing mortality rates in the three regions comprising the assessment area (eastern Atlantic Ocean, Indian Ocean, and western Pacific Ocean), and low risk from commercial pelagic longline fisheries to porbeagle shark over the entire Southern Hemisphere

Table 10.1: Recreational harvest estimates for New Zealand chondrichthyan species for the 2017–18 fishing year. Mean fish weights are only available for two species, otherwise only the counts are shown. Mgmt class = Management class, QMS is shown, all others are Non-QMS and non-protected species; CV = Coefficient of variation of the estimate to the left. Reproduced in part from Wynne-Jones et al. (2019).

Species	Mgmt class	Fishers (n)	Events (n)	Harvest (n)	CV	Mean weight (kg)	Harvest (t)	CV
Rig	QMS	153	235	35 369	0.15	1.59	56.24	0.15
School Shark	QMS	42	50	6 826	0.26	–	–	–
Spiny Dogfish	QMS	54	68	13 985	0.23	1.53	21.42	0.23
Stingray		22	24	2 841	0.25	–	–	–
Elephantfish	QMS	22	33	3 047	0.31	–	–	–
Sand Shark		6	7	701	0.51	–	–	–
Hammerhead Shark		8	10	1 158	0.46	–	–	–
Bronze Whaler Shark		2	2	203	0.73	–	–	–
Mako Shark	QMS	3	4	1 048	0.77	–	–	–
Carpet Shark		6	6	422	0.46	–	–	–

Table 10.2: Reported total catches (tonnes, including discards and live releases) for chondrichthyan species from 2015–16 to 2019–20, arranged in descending order of total catch and rounded to the nearest tonne. Only species with more than 5 t of aggregated catch (over the 5 years) are included. The management class (Mgmt) is also shown. Source: Fisheries New Zealand catch-effort database. Note: Catches of QMS species differ from landings in Table 10.4 because they include discards and releases and are derived from a different source. Protected species are reported by number, not weight, and are not included in this table.

Species	Code	Mgmt	2015–16	2016–17	2017–18	2018–19	2019–20	Total	Mean
Spiny dogfish	SPD	QMS	4 523	6 270	6 603	5 635	4 904	27 933	5 587
School shark	SCH	QMS	2 934	2 851	3 027	2 732	2 626	14 171	2 834
Rough skate	RSK	QMS	1 627	1 914	1 783	1 533	1 436	8 294	1 659
Rig	SPO	QMS	1 427	1 437	1 481	1 384	1 412	7 140	1 428
Elephantfish	ELE	QMS	1 389	1 324	1 340	1 354	1 369	6 775	1 355
Ghost shark	GSH	QMS	1 369	1 322	1 433	1 424	1 047	6 596	1 319
Smooth skate	SSK	QMS	753	910	994	870	828	4 355	871
Pale ghost shark	GSP	QMS	712	898	1 016	886	685	4 197	839
Blue shark	BWS	QMS	707	619	778	871	732	3 708	742
Shovelnose dogfish	SND	Non-QMS	540	441	545	527	373	2 426	485
Carpet shark	CAR	Non-QMS	388	474	458	478	573	2 372	474
Unspecified Sharks &	OSD	Non-QMS	305	295	300	366	337	1 603	321
Baxter's lantern dogfish	ETB	Non-QMS	259	314	319	297	203	1 392	278
Seal shark	BSH	Non-QMS	254	245	216	182	220	1 117	223
Northern spiny dogfish	NSD	Non-QMS	135	174	211	232	223	975	195
Leafscale gulper shark	CSQ	Non-QMS	178	125	194	160	106	763	153
Long-nosed chimaera	LCH	Non-QMS	134	139	161	139	120	692	138
Eagle ray	EGR	Non-QMS	116	134	122	110	132	614	123
Mako shark	MAK	QMS	150	80	84	74	64	451	90
Porbeagle shark	POS	QMS	94	62	100	71	43	370	74
Unspecified Deepwater	DWD	Non-QMS	60	73	77	47	29	287	57
Short-tailed black ray	BRA	Non-QMS	44	67	67	52	48	279	56
Thresher shark	THR	Non-QMS	57	50	57	55	49	268	54
Electric ray	ERA	Non-QMS	42	53	51	54	45	245	49
Lucifer dogfish	ETL	Non-QMS	24	36	53	56	38	206	41
Longnose velvet dogfish	CYP	Non-QMS	21	26	36	31	25	140	28
Broadnose sevengill shark	SEV	Non-QMS	21	23	20	28	32	125	25
Whiptail ray	WRA	Non-QMS	21	27	35	20	18	121	24
Stingray (Unspecified)	STR	Non-QMS	6	13	26	25	34	104	21
Plunket's shark	PLS	Non-QMS	10	16	17	20	21	84	17
Hammerhead shark	HHS	Non-QMS	15	14	19	14	18	80	16
Bronze whaler shark	BWH	Non-QMS	16	18	14	14	17	79	16
Slender smooth-hound	SSH	Non-QMS	29	13	16	12	9	79	16
Sixgill shark	HEX	Non-QMS	9	7	6	8	6	37	7
Purple chimaera	CHG	Non-QMS	5	13	10	3	3	33	7
Skate, Other	OSK	Non-QMS	8	4	8	6	7	32	6
Chimaera spp.	CHI	Non-QMS	7	7	12	3	1	30	6
Smooth skin dogfish	CYO	Non-QMS	8	4	9	2	3	26	5
Prickly dogfish	PDG	Non-QMS	2	3	4	4	4	17	3
Widenosed chimaera	RCH	Non-QMS	1	1	4	1	7	13	3
Deepwater spiny skate	DSK	Non-QMS	1	4	6	2	0	13	3
Electric ray	BER	Non-QMS	2	1	3	2	2	9	2
Chimaera, purple	CHP	Non-QMS	1	1	2	2	2	8	2
Long-tailed skate	LSK	Non-QMS	1	0	0	0	6	8	2
Rays	RAY	Non-QMS	1	1	0	1	4	7	1
Portuguese dogfish	CYL	Non-QMS	0	1	1	1	3	6	1
Pelagic stingray	DAS	Non-QMS	0	0	1	0	4	6	1
Longnosed deepsea skate	PSK	Non-QMS	1	2	1	1	1	6	1
Deepsea sharks	CEN	Non-QMS	–	–	–	1	5	6	3
Hydrolagus spp.	HYD	Non-QMS	3	–	2	–	0	5	2

Table 10.3: Percentages of chondrichthyan species landed, discarded and released alive from 2015–16 to 2019–20. Only the top 25 species, in terms of total catch (over the 5 years), are included here. Protected species are not included (all are required to be discarded or released). Discards here are defined as non-QMS species that are not kept or were QMS species that were authorized as a discard by an onboard observer; Released is defined here as fish below a legal size or sharks returned dead or near-dead that do not meet Schedule 6 criteria; Schedule 6 are a special type of release where specific stocks which may be returned to the sea in accordance with the requirements laid out in the Fisheries Act (1996).

Species Name	Code	Mgmt class	Landings%	Discards%	Released%	Schedule 6%	Losses%
Spiny dogfish	SPD	QMS	31	0	0	69	0
School shark	SCH	QMS	99	1	0	0	0
Rough skate	RSK	QMS	92	1	0	7	0
Rig	SPO	QMS	98	0	0	1	0
Elephant fish	ELE	QMS	100	0	0	0	0
Ghost shark	GSH	QMS	95	4	0	0	0
Smooth skate	SSK	QMS	82	1	0	17	0
Pale ghost shark	GSP	QMS	98	1	0	0	0
Blue shark	BWS	QMS	0	0	14	85	1
Shovelnose dogfish	SND	Non-QMS	64	36	0	0	0
Carpet shark	CAR	Non-QMS	3	97	0	0	0
Unspecified Sharks & Dogfish	OSD	Non-QMS	44	56	0	0	0
Baxter's lantern dogfish	ETB	Non-QMS	71	29	0	0	0
Seal shark	BSH	Non-QMS	49	51	0	0	0
Northern spiny dogfish	NSD	Non-QMS	12	87	0	1	0
Leafscale gulper shark	CSQ	Non-QMS	67	33	0	0	0
Long-nosed chimaera	LCH	Non-QMS	81	19	0	0	0
Eagle ray	EGR	Non-QMS	38	62	0	0	1
Mako shark	MAK	QMS	10	0	29	59	1
Porbeagle shark	POS	QMS	3	2	47	47	0
Unspecified Deepwater Dogfish	DWD	Non-QMS	84	16	0	0	0
Short-tailed black ray	BRA	Non-QMS	1	98	0	0	1
Thresher shark	THR	Non-QMS	9	90	0	0	0
Electric ray	ERA	Non-QMS	8	92	0	0	0
Lucifer dogfish	ETL	Non-QMS	69	31	0	0	0

Table 10.4: TACCs and 2019–20 estimated landings (tonnes) of the eleven chondrichthyans managed under the QMS. Also shown are the year of entry of each species into the QMS and year of addition to Schedule 6 of the Fisheries Act that allows release of fish into the sea. Source: Fisheries Assessment Plenaries (Ministry for Primary Industries 2019). Note: Landings differ from the catches in previous tables because the latter include discards and releases.

Species Name	Species Code	TACC (t) 2019–20	Landings (t) 2019–20	Entry into QMS	Addition to Schedule 6
Spiny dogfish	SPD	12 660	4 670	2004	2004
School shark	SCH	3 436	2 613	1986	2013
Rough skate	RSK	1 986	1 326	2003	2003
Rig	SPO	2 018	1 376	1986	2012
Dark ghost shark	GSH	3 025	1 016	1998	
Elephant fish	ELE	1 464	1 359	1986	
Pale ghost shark	GSP	1 780	681	1999	
Smooth skate	SSK	849	652	2003	2003
Blue shark	BWS	1 860	112	2004	2004
Mako shark	MAK	200	30	2004	2004
Porbeagle shark	POS	110	25	2004	2004

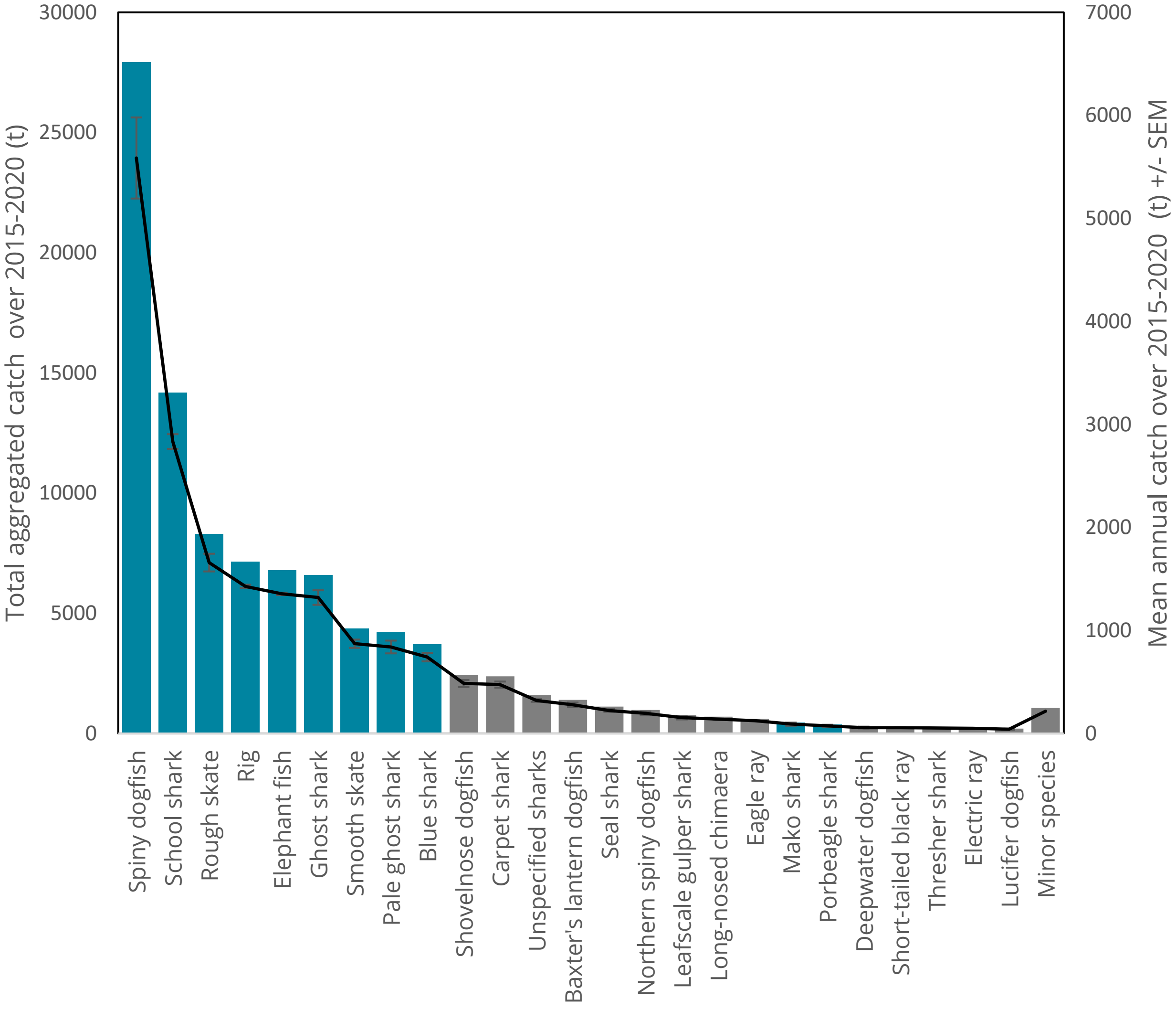

Figure 10.2: Reported total catches (landings, discards, and releases) for chondrichthyan species aggregated across 2015−16 to 2019−20. The average annual catches (solid black line) are shown on the right axis. Source: Fisheries New Zealand catch-effort database. Turquoise bars indicate QMS species and grey bars indicate non-QMS species

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10.4.2 PROTECTED SPECIES

Seven chondrichthyans are currently protected in New Zealand fisheries waters: white pointer shark was protected in 2007; spinetail devil ray, manta ray (*Mobula birostris*), whale shark (*Rhincodon typus*), deepwater nurse shark (*Odontaspis ferox*), and basking shark (*Cetorhinus maximus*) in 2010; and oceanic whitetip shark (*Carcharhinus longimanus*) in 2013.

Data from Fisheries New Zealand observers, along with fisher-reported data, are used to estimate protected species captures. Observer coverage has been reasonably high over the last decade (2008–09 to 2018–19) or longer in some larger fisheries (e.g., trawl fisheries for hoki (15.3–38.6% per year) and orange roughy (11.5–44.1% per year), and in the southern bluefin target longline fishery at 20.8–56.6% per year). Some trawl fisheries around southern New Zealand and skipjack tuna purse seine fisheries in northern New Zealand also had sufficient coverage over the last decade (southern blue whiting (25.2–100.0% per year), squid (12.9–87.1% per year), and purse seine tuna (13.8–29.1% per year 2005–06 to 2015–16), providing credible information on captures of basking sharks and white pointer sharks (in trawls) and of spinetail devil rays (in purse seine nets). However, observer coverage has not always been representative of the spatial and temporal distribution of these fisheries. Inshore fisheries have received only sparse observer coverage. These fisheries potentially have unobserved and unrecorded mortality of some protected species, like basking shark, white pointer shark, and deepwater nurse shark, which are all occasionally present in inshore areas.

In recent years, fishers have been reporting increasing numbers of protected species bycatch on a mandatory dedicated reporting form, and fisher reports of basking shark and white pointer shark captures have exceeded those reported by observers (Francis 2017a, 2017b).

PROTECTED SPECIES GENETICS

Francis & Ritchie (2016) reviewed the available genetic information for the seven protected chondrichthyan species. They established a repository for genetic samples of protected fish species, conducted a stock-take of completed, current, and planned genetic analyses for these species internationally, and provided recommendations on the most appropriate methods of furthering genetic analyses to inform management of New Zealand's protected fish species in relation to fisheries bycatch. The tissue repository contains samples from white pointer shark (N=102) and basking shark (N=56), but few or no samples from the other protected species.

BASKING SHARK

Since 2010, incidental capture of protected species has been reported on Non-Fish/Protected Species Catch Returns (NF/PSCR). A total of 85 basking sharks were reported as incidental captures in fisheries from 2010–11 to 2020–21, mostly by bottom and midwater trawls targeting arrow squid, hake, hoki, or silver warehou. Of these 89 that were reported, 39 were also reported by observers. Approximately 64.0% (55 individuals) were already dead when captured and the remaining released alive. It is presumed that few of the individuals returned to the sea alive are likely to survive post release. Captured basking shark individuals are predominantly adult males. Adult female captures are relatively rare and captures of juveniles were never recorded.

Finucci et al. (2021a, b) found that areas with a high vertical flux of organic particulate matter at the seabed, indicative of higher levels of primary production, had high predicted habitat suitability index scores for basking sharks. Such areas had strong overlap with the main historic capture locations: the east coast South Island off Banks Peninsula (FMA 3), the west coast South Island between Westport and Hokitika, Puysegur (FMA 7), the shelf edge south and east of Stewart Island (FMA 5) and the Snares Islands, and around the Auckland Islands (FMA 6). Captures (and sightings) of basking sharks have also occurred around the North Island but were relatively uncommon (Francis & Duffy 2002, Francis & Sutton 2012, Francis 2017b).

Most basking shark capture records came from large offshore trawlers in a variety of target fisheries with both midwater and bottom trawls. Basking sharks are also caught in set nets (Francis & Duffy 2002) but have rarely been reported by fishers since 2010 (Francis 2017b). The observer coverage of the set net fisheries has been traditionally low, even if it was increased in the last decade, so the set net bycatch cannot be easily quantified. Basking sharks are rarely entangled in surface longlines (Francis & Duffy 2002).

Francis & Sutton (2012) found a highly significant association between the numbers of basking sharks caught and vessel nationality in each of the three main fishery areas. This was due to relatively large numbers of sharks being caught by Japanese-owned trawlers in the late 1980s and early 1990s. Other operational fleet variables and environmental variables examined were not correlated with shark catch rates. Reasons for the high catch rates by Japanese trawlers are unknown but may relate to targeting of the sharks for their liver oil and fins, or a relatively high abundance of sharks in the late 1980s and early 1990s (Francis & Sutton 2012).

WHITE POINTER SHARK

Between the 2008-09 and 2020-21 fishing years, a total of 133 white pointer sharks have been reported as caught by commercial fishers, mostly in set nets, bottom trawls, and bottom long lines targetting a variety of fisheries species. Of these 133 that were reported, 17 were also recorded by fisheries observers. About 29% (39 individuals) of these were reported dead at capture, but the remainder were released alive and, overall, in good condition (12 were reported as injured and 82 were reported as uninjured). The survival of white pointer sharks after live release is unknown (Francis 2017a).

White pointer shark captures were reported from throughout mainland New Zealand and as far south as the Auckland Islands, but not from around the other outlying islands (Francis & Lyon 2012, Francis 2017a). Despite their small share of effort (only 20% of the overall length of nets set), and limited geographical extent, three regions (Great Exhibition Bay, Taranaki, and Foveaux Strait) accounted for

89% of the 36 white pointer sharks reported caught by set net vessels (Francis 2017a).

Tagging and genetic studies have shown that New Zealand and eastern Australian white pointer sharks comprise a single stock (Duffy et al. 2012, Francis et al. 2015a). A close-kin genetics study which included New Zealand white pointer shark tissue samples estimated that the total population size of this stock of was 5460 (uncertainty range 2909–12 802) including 750 adults (uncertainty range 470 to 1030) (Bruce et al. 2018, Hillary et al. 2018). The trend in abundance of adult white pointer sharks in this population was not significantly different from zero (i.e., no trend and thus a stable population where births = deaths, on average) (Bruce et al. 2018).

WHALE SHARK

Whale sharks are typically only seen in north-eastern North Island waters during summer and are rare (Duffy 2002), making captures exceptional events.

An individual was reportedly caught by a coastal trawler off South Canterbury in the late 1970s (as communicated to C. Duffy by a fisher, as reported by Duffy 2005). Another whale shark was captured (and released alive) in March 2020 by a purse seine vessel off Great Barrier Island. No captures of whale sharks have been reported by fisheries observers in New Zealand waters.

DEEPWATER NURSE SHARK (SMALLTOOTH SANDTIGER SHARK)

Deepwater nurse sharks (ODO) have been reported frequently by fishers and observers from along the edge of the continental shelf between Otago Peninsula and south of the Snares Islands (Francis & Lyon 2012). Clusters of records are also available from the Chatham Islands, and off Banks Peninsula and Farewell Spit. However, the southern limit of the known distribution of deepwater nurse sharks in New Zealand is a line from Cape Kidnappers in Hawke Bay to Cape Egmont. Given that most of the records are from south of that range, and that many ODO weights were implausibly small, most records of this species are erroneous, probably because of an incorrect species code. Plausible commercial and observer database records of deepwater nurse shark captures include three from FMA 2 and one from the Louisville Seamount Chain, from depths > 800 m (Francis & Lyon 2012). In the most recent fishing year, 2020–21, a total of seven individuals were reported captured by commercial fishers, two from bottom trawl gear and five from rock lobster pots. This is in addition to the individual caught in a bottom trawl in 2015, west of the northern part of the North Island (where five had previously been reported captured by observers in 2012; Fisheries New Zealand). Only one was reported dead at capture, with the remaining seven released alive uninjured, of these eight individuals reported incidentally caught by commercial fishers.

There are other published records of deepwater nurse sharks being caught in set nets off New Plymouth (Stewart 1997, Fergusson et al. 2008), trawl in Hawke Bay, and by the NIWA research trawl vessel *Tangaroa* on the Norfolk Ridge (Garrick 1974, Stewart 1997, Fergusson et al. 2008), confirming that the species is occasionally caught in northern waters. Duffy (2005) cited anecdotal information that deepwater nurse sharks were ‘not uncommon’ bycatch in a set net fishery operating around White Island and Volkner Rocks in the eastern Bay of Plenty but noted that this fishery had ceased. Duffy (2005) and Fergusson et al. (2008) also reported the capture of deepwater nurse sharks from the same location for display at Kelly Tarlton’s Sea Life Aquarium from the mid-1980s to the early 2000s, but all the sharks died and the practice was discontinued.

SPINETAIL DEVIL RAY AND MANTA RAY

Spinetail devil rays and manta rays (Family: Mobulidae) occur during summer months, mainly in north-eastern North Island waters but also infrequently in the Taranaki Bight and north-western South Island (Duffy & Abbott 2003, Francis & Jones 2016). Historically, most mobulid rays reported caught in commercial fisheries were likely to have been spinetail devil rays (Paulin et al. 1982). However, it is possible that manta rays are occasionally caught in purse seine nets along the north-east coast of North Island, although observer coverage between 2005 and 2014 in FMA 1 skipjack tuna (0–31.8% per year) and mackerel purse seine fisheries (0–25.8% per year) has not reported any captures so far.

Between the 2010–11 and 2020–21 fishing years, a total of 501 mobulid ray incidental captures were reported by fishers, mostly in skipjack tuna purse seine fisheries, 92 of which were also reported by fisheries observers (Fisheries New Zealand). The vast majority of these (97.4%, 488) were released alive. In tagging experiments, 4 out of 7 rays (57%) released from purse seine nets with satellite tags, eventually died (Francis & Jones 2016), suggesting a high post-release mortality rate. However, a more recent tagging study has shown a reduced mortality rate (Francis & Jones 2019), suggesting that mortality could be mitigated through improved release practices. Some released rays have travelled north to Vanuatu and south of Fiji, indicating that they make a seasonal migration between New Zealand and the tropics.

Most commercial and observer records of mobulid rays were from off the northern North Island in FMAs 1 and 9, and most records came from purse seine vessels (Francis & Lyon 2012, Jones & Francis 2012, Francis & Jones 2016). Most observer records were from the edge of the continental shelf between the Bay of Islands and Great Barrier Island. Commercial purse seine records are available from the eastern Bay of Plenty, and there are a few commercial and observer records from the North Taranaki Bight. Three devil rays have been reported caught on surface longlines, mainly near the 1000 m depth contour. Observer and commercial records were not available before 2001–02, although devil ray bycatch in purse seine catches was documented between 1975 and 1981 by Paulin et al. (1982). All observed devil rays were returned to the sea by fishers. The three rays caught on surface longlines were alive when retrieved, but the life status of rays caught in purse seine nets was not recorded. Over the four fishing years 2010–11 to 2013–14, 153 spinetail devil rays were reported on Non-Fish/Protected Species Catch Returns. At an average weight of about 125 kg per ray (observer estimated weights), this represents about 19.1 t of total catch, or about 4.8 t per year.

OCEANIC WHITETIP SHARK

The oceanic whitetip shark is a tropical species that enters northern New Zealand waters only in summer, and possibly only in summers that are warmer than normal (Francis et al. 1999b). Only 19 observer and two commercial fishery records are known (one of which occurred in both datasets) (Francis & Lyon 2014). All records came from surface longlines set in the Kermadec Fisheries Management Area or off the north-eastern coast of North Island. Most (84%) of the observed sharks were alive when hauled to the vessel, and about half were processed in some way with the remainder being discarded (those captures pre-dated protection of the species in 2013). Given the low commercial reporting rate (1 out of 19 observed sharks) and the low observer coverage of domestic surface longlines, the interaction of the surface longline fisheries with oceanic whitetips is considered substantially underestimated (Ministry for Primary Industries 2012, Francis & Lyon 2014).

10.4.3 NON-QMS SPECIES

More than 50 species of Non-QMS chondrichthyans are known to be caught by fishers in New Zealand waters, but records of non-QMS chondrichthyans catches are not believed reliable (due to identification issues and a limited set of species required to be reported).

Inshore rays and sharks are caught by a variety of fishing methods. Closures of some inshore waters to set netting and trawling to protect Hector's and Māui dolphin on the north-west coast of North Island and around much of South Island may have benefitted shark and ray species that occur there. However, most of these species are highly vulnerable to trawl, set net, and bottom longline and have nurseries in shallow coastal waters and harbours that are still fished by set nets and longline and, to a lesser extent, trawls. Little is known about the fishery interactions of these species (but for an analysis of hammerhead shark captures see Francis 2010). Similarly, there is little information on the biological productivity of most of the species, but many (all the rays and thresher shark) have very low reproductive output (a few young per year) and are therefore highly susceptible to overfishing.

Deepwater chondrichthyans are caught incidentally in deepwater trawl tows, some species in considerable quantities (Blackwell 2010). Seven species of squaloid deepwater sharks — shovelnose dogfish, Baxter's dogfish, lucifer dogfish, Owston's dogfish, longnose velvet dogfish, leafscale gulper shark, and seal shark — commonly occur over the middle and lower continental slope in depths greater than 600 m. Shovelnose dogfish has a wider distribution, and it also occurs on the upper and middle slope (400–600 m in depth). These seven shark species are commonly taken as bycatch in the middle depths and deepwater fisheries for hoki, orange roughy, and oreos. They are either discarded at sea, processed for their fins (prior to the finning ban of 2014) or livers (Blackwell 2010).

Historical data are available from the Fisheries New Zealand Observer Programme (Figure 10.3), but a critical issue when using observer data to monitor the abundance and biology of deepwater sharks is species identification: many deepwater sharks are superficially similar and difficult to distinguish, so they have often been mis-identified or lumped under generic codes by commercial fishers. With the recent availability of good species identification guides (McMillan et al. 2011a, 2011b, 2011c, 2019), observers and fishers now have the tools to identify deepwater sharks accurately to species level. To test current observer identification accuracy, McMillan et al. (2018) compared observer identifications of six species of deepwater sharks with NIWA identifications made using images and DNA analyses of tissue samples collected from the same specimens. Of 331 observer identifications, 302 (91%) were confirmed by NIWA; most of the incorrect identifications came from a single observer. It was concluded that observer identification of deepwater sharks is generally accurate. Fisheries observer data were used in a recent study (ENV2018-06) to validate modelled distributions of seven species of deepwater sharks derived from 1982–2018 trawl survey data (Finucci et al. 2021a). This study identified some oceanographical hotspots for these species but did not find significant trends in abundance and spatial distribution.

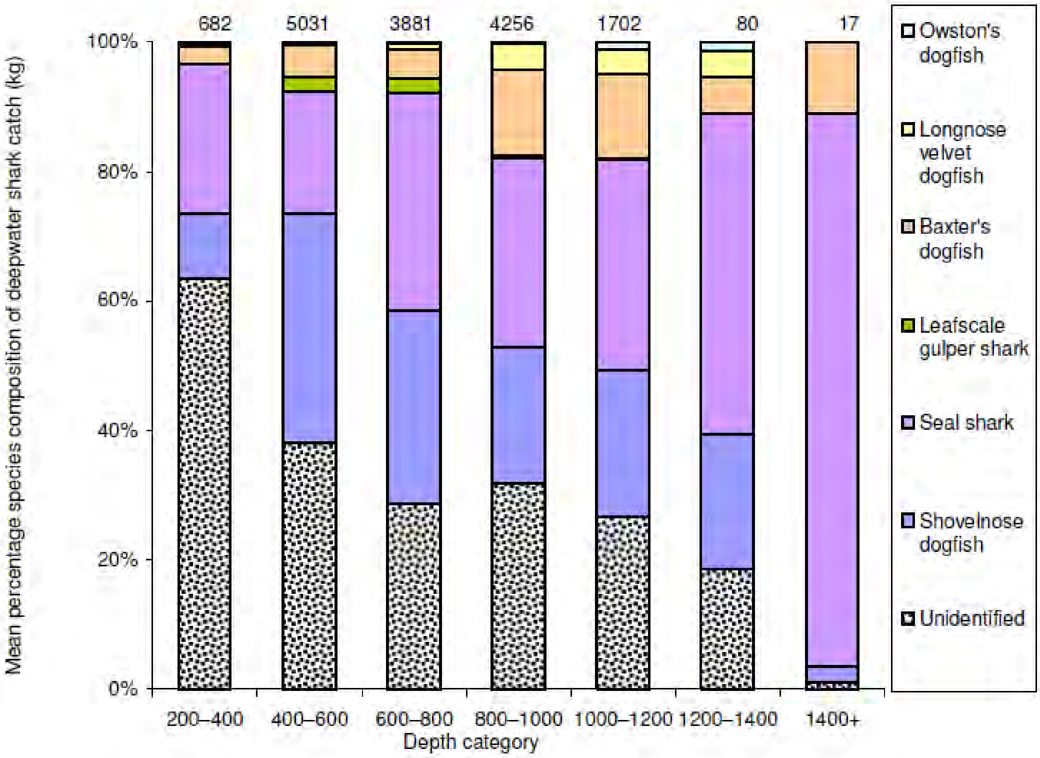


Figure 10.3: Mean catch composition of deepwater chondrichthyans reported from the Observer Programme database, all years 2001–02 to 2005–06, by major depth (m) category (number of observations shown above bars). Source: Blackwell (2010)

Some species that are not caught or reported in quantities sufficient to be included in Table 10.2 may also be vulnerable to overfishing. These include endemic species with limited geographic and/or depth ranges that overlap in space with the operations of deepwater trawlers, for example Dawson's catshark (Francis 2006) and some of the rarer deepwater skates and chimaeras. Their low catch weights probably reflect their rarity.

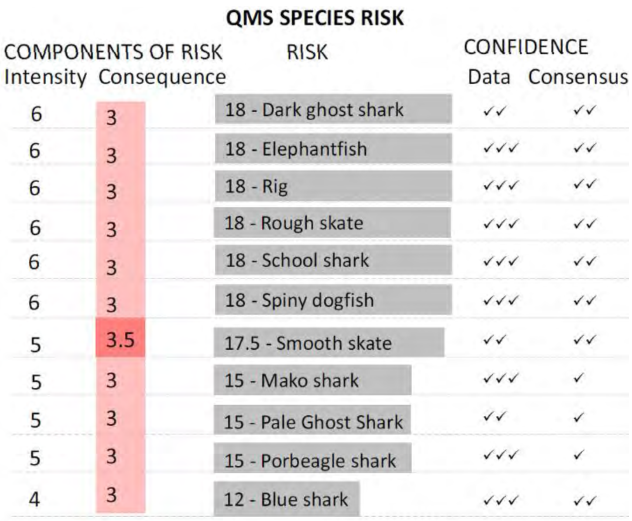
RISK ASSESSMENT AND THREATENED SPECIES CLASSIFICATION

One of the objectives of the 2013 NPOA-Sharks was to establish a risk-based approach to prioritising management actions. MPI hosted a workshop in November 2014 that produced a qualitative (Level 1) risk assessment (RA) for all New Zealand chondrichthyan taxa (except for species with uncertain taxonomy) from commercial fishing (Ford et al. 2015). This was updated in 2017 (Ford et al. 2018).

The qualitative RA used a modified Scale Intensity Consequence Analysis approach. Before the workshops, data on catches, effort, distribution, abundance, and biological productivity were collated for all species and summarised to inform the RA. An expert panel then scored the relative risk to each taxon from commercial fishing, based on fishing information from the last five years, on an EEZ-wide scale. This process scored intensity and consequence of the fishery to the shark taxa, and the rationales for the scores were documented. These intensity and consequence scores were then multiplied together to get a total risk score (Ford et al. 2018). Results were reported within the three management classes of chondrichthyans – QMS (figure 10.4), Non-QMS (figure 10.5), and protected species.

However, out of the 50 taxa considered in detail, the panel had low confidence in the risk scores for three of 11 QMS species, 26 of 36 non-QMS taxa, and all three protected species. However, the panel considered that the available information did not indicate that commercial fishing is currently causing, or in the near future could cause, serious unsustainable impacts to any sharks, rays, or chimaera population examined.

Six QMS species attained the equal highest total risk score (dark ghost shark, elephantfish, rig, rough skate, school shark, and spiny dogfish), highlighting the need for further assessments on these species.

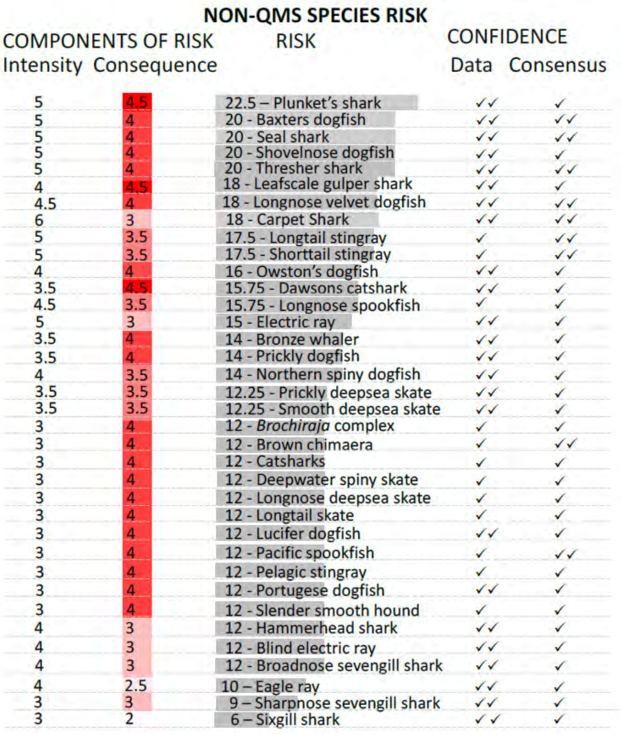


QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
6	3	18 - Dark ghost shark	✓✓	✓✓
6	3	18 - Elephantfish	✓✓✓	✓✓
6	3	18 - Rig	✓✓✓	✓✓
6	3	18 - Rough skate	✓✓✓	✓✓
6	3	18 - School shark	✓✓✓	✓✓
6	3	18 - Spiny dogfish	✓✓✓	✓✓
5	3.5	17.5 - Smooth skate	✓✓	✓✓
5	3	15 - Mako shark	✓✓✓	✓
5	3	15 - Pale Ghost Shark	✓✓	✓
5	3	15 - Porbeagle shark	✓✓✓	✓
4	3	12 - Blue shark	✓✓✓	✓✓

Figure 10.4: QMS species risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact. For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus (two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented in descending order of consequences and then alphabetically. From Ford et al. (2018).

Plunket's shark was the highest scoring Non-QMS species and basking shark and spinetail devil ray were the highest scoring protected species (Appendix 19.11.3).



NON-QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
5	4.5	22.5 – Plunket's shark	✓✓	✓
5	4	20 - Baxters dogfish	✓✓	✓✓
5	4	20 - Seal shark	✓✓	✓✓
5	4	20 - Shovelnose dogfish	✓✓	✓
5	4	20 - Thresher shark	✓✓	✓✓
4	4.5	18 - Leafscale gulper shark	✓✓	✓
4.5	4	18 - Longnose velvet dogfish	✓✓	✓✓
6	3	18 - Carpet Shark	✓✓	✓✓
5	3.5	17.5 - Longtail stingray	✓	✓✓
5	3.5	17.5 - Shorttail stingray	✓	✓✓
4	4	16 - Owston's dogfish	✓✓	✓
3.5	4.5	15.75 - Dawsons catshark	✓✓	✓
4.5	3.5	15.75 - Longnose spookfish	✓	✓
5	3	15 - Electric ray	✓✓	✓
3.5	4	14 - Bronze whaler	✓✓	✓
3.5	4	14 - Prickly dogfish	✓✓	✓
4	3.5	14 - Northern spiny dogfish	✓✓	✓
3.5	3.5	12.25 - Prickly deepsea skate	✓✓	✓
3.5	3.5	12.25 - Smooth deepsea skate	✓✓	✓
3	4	12 - *Brochiraja* complex	✓	✓
3	4	12 - Brown chimaera	✓	✓✓
3	4	12 - Catsharks	✓	✓
3	4	12 - Deepwater spiny skate	✓	✓
3	4	12 - Longnose deepsea skate	✓	✓
3	4	12 - Longtail skate	✓	✓
3	4	12 - Lucifer dogfish	✓✓	✓
3	4	12 - Pacific spookfish	✓	✓✓
3	4	12 - Pelagic stingray	✓	✓
3	4	12 - Portugese dogfish	✓✓	✓
3	4	12 - Slender smooth hound	✓	✓
4	3	12 - Hammerhead shark	✓✓	✓
4	3	12 - Blind electric ray	✓✓	✓
4	3	12 - Broadnose sevengill shark	✓✓	✓
4	2.5	10 – Eagle ray	✓✓	✓
3	3	9 – Sharpnose sevengill shark	✓✓	✓
3	2	6 – Sixgill shark	✓✓	✓

Figure 10.5: Non-QMS Species Risk scores. See previous figure for detailed legend. From Ford et al. (2018).

The risk assessment was designed to help prioritise actions to conserve chondrichthyans and is most useful to prioritise future quantitative assessments, noting that protected species are also given priority under the NPOA-Sharks 2013. The panel made several recommendations for high-risk or protected species regarding potential research options. These included better use of existing data, data grooming, or analysis to improve inputs to assessment scores, improved taxonomy and training to underpin identification of sharks, and collection of more biological information to increase understanding of productivity (especially the ability of a taxon to withstand and to recover from fishing impacts). The RA panel also stressed that, particularly where abundance indices are lacking, the consequence scale was more relevant to risk than the total risk score which was often dominated by the level of intensity (masking differences in potential consequence). Taxa with high consequence scores have low productivity or presumed low productivity. For such taxa, more information may improve the scores or our confidence in them, but in the interim a more precautionary approach to management was recommended by the panel (Ford et al. 2018).

Recently, there have been two updated assessments of the threatened species status of New Zealand chondrichthyans. In 2016, the Department of Conservation carried out a review of their threatened species classification (Duffy et al. 2018). The conservation status of two taxa has worsened: white pointer shark was assessed as Nationally Endangered (previously assessed as Gradual Decline) and basking shark moved to Nationally Vulnerable (from Gradual Decline). Conversely, four taxa had an improved conservation status (Galapagos shark, Kermadec smooth-hound, sixgill shark, and southern sleeper shark). In 2017, the IUCN Redlist categories of New Zealand chondrichthyans were reviewed and a number of changes made; however, the results have not yet been finalised or published.

Australian researchers (with New Zealand input) carried out an ecological risk assessment for deepwater chondrichthyans in the South Pacific Ocean (SPRFMO Convention Area) and the South Indian Ocean (Georgeson et al. 2020). Productivity-Susceptibility Analysis (PSA) and Sustainability Assessment for Fishing Effects (SAFE) methods were compared to assess the potential vulnerability of deepwater chondrichthyans to demersal trawls, midwater trawls, demersal longlines, and demersal set nets. The main outcomes highlighted probable misclassifications in the PSA relative vulnerability rankings, and the value of applying more quantitative tools, such as SAFE, when adequate data are available. *Squalus fernandezianus*, *Deania calcea*, and *Gollum attenuates* were among the species with highest risk (particularly from demersal longlines) in the South Pacific Ocean.

QMS SPECIES

Standardised CPUE analyses have been carried out to monitor trends in the relative abundance of some stocks of 6 of the 11 QMS chondrichthyans species (rig, school shark, elephantfish, blue shark, porbeagle shark, and mako shark) . (Table 10.5). Rig in QMA 2, school shark in QMAs 3 and 5, elephantfish in QMA 5, and pale ghost shark in FMA 4 are declining; all other stocks that are monitored are stable or increasing in recent years.

For blue, porbeagle, and mako sharks, other abundance indicators have been developed in addition to standardised CPUE. They include high-CPUE (the proportion of half-degree rectangles having unstandardised CPUE greater than a specified threshold); proportion-zeroes (the proportion of half-degree rectangles having zero reported catches in a fishing year); geometric mean index (the geometric mean of the species abundances in catches; proportion of males in the catch; and median lengths of males and females (Francis et al. 2014). A previous analysis covering the period 2005−13 suggested positive trends for all three species (Francis et al. 2014, Francis & Large 2017), but the latest analysis (Francis & Finucci 2019) suggested a declining trend in 2017−18. Since the previous indicators were published in 2014, there have been major changes in the surface longline fishery: airline restrictions on the export of shark fins in 2014, a ban on shark finning in 2015, and the addition of porbeagle shark to CITES Appendix II in 2014. As a result, most pelagic sharks are now discarded dead or released alive. Furthermore, chartered Japanese longliners ceased fishing in New Zealand waters in 2015, thus terminating an important time series of indicators. These factors combine to make interpretation of the stock status of the three species (and its trends) problematic.

Trawl survey relative abundance indices are used to monitor the populations of rig, school shark, spiny dogfish, elephantfish, rough and smooth skates, and pale and dark ghost sharks (Table 10.5). For 20 out of 21 species/FMA combinations, abundance is stable or increasing in recent years; however pale ghost shark in FMA 4 has shown a downward trend

Table 10.5: Trends in abundance of QMS species monitored by standardised CPUE analysis and trawl surveys. Changes in trends through time are indicated by forward slashes. Blanks, none or unreliable. Source: Fisheries New Zealand (2021a), Fisheries New Zealand (2021b) unless otherwise indicated. 'Recent years' refers to the last five years but may be longer for long time series. Time series that have not been updated in the last five years are not included.

CPUE indices

Species	Species code	QMA1	QMA2	QMA3	QMA4	QMA5	QMA6	QMA7	QMA8	Source
Rig	SPO	Nil	Down	Up				Nil		
School shark	SCH	Up/Nil		Down	Nil	Down		Nil	Nil	
Elephantfish	ELE			Nil		Down		Up		Francis & Finucci (2019)
Blue shark	BWS	Up/Down								Francis & Finucci (2019)

Table 5 [continued]

CPUE indices										
Species	Species code	QMA1	QMA2	QMA3	QMA4	QMA5	QMA6	QMA7	QMA8	Source
Porbeagle shark	POS	Up/Down								Francis & Finucci (2019)
Mako shark	MAK	Up/Down								
Trawl survey indices				FMA3	FMA4	FMA5	FMA6	FMA7		
Rig	SPO			Nil				Nil		
School shark	SCH			Nil				Nil		
Spiny dogfish	SPD			Nil	Nil			Nil		
Elephantfish	ELE			Nil				Up		
Rough skate	RSK			Up/Nil	Nil			Up/Nil		
Smooth skate	SSK			Up/Nil	Down/Nil			Down/Up		
Dark ghost shark	GSH			Up	Nil	Nil		Nil		
Pale ghost shark	GSP				Down	Up				

Legend

🟩 Trend up in recent years

⬜ Stable in recent years

🟥 Trend down in recent years

PROTECTED SPECIES

Of the seven protected chondrichthyan species, only the basking shark has any form of population monitoring and that is limited to assessing trends in relative abundance from incidental captures. Observer-based unstandardised CPUE analyses of trawl catches in three trawl fisheries (East Coast South Island EC, West Coast South Island WC, and Southland–Auckland Islands SA) are shown in Figure 10.6 (Francis & Sutton 2012). Inter-annual variation was large, with peak observer records occurring in 1987–92, 1997–2000, and 2003–05 depending on the region. Some years had very low or zero CPUE. Francis & Smith (2010) used Bayesian predictive hierarchical models to estimate catches and catch rates in the three trawl fisheries from observer data between 1994–95 and 2007–08. The predicted strike rates showed no overall trend since 1994–95 in any of the three areas. A total of 95 shark captures were observed from 49 165 tows over the 14-year period, an overall unstandardised capture rate of 1.9 per 1000 tows. The overall predicted capture rate was 2.5 sharks per 1000 tows, with area-specific rates of 3.9 (EC), 2.0 (WC), and 1.9 (SA) per 1000 tows. The total predicted number of captures from 1987 to 2012 was 922 individuals with a CV of 19%. Predicted captures peaked in 1997–98 and then declined steadily to low numbers. Much of the recent decline in basking shark bycatch was probably attributable to a decline in fishing effort of about 50% between 2002–03 and 2006–08 in the three areas (Francis & Smith 2010). However, unstandardised catch rates from observer data were much higher in 1988–92 than at any time since. Those high rates in earlier periods may be attributable to specific targeting by Japanese vessels (Francis & Sutton 2012). Raw observer CPUE values were updated to the 2015–16 fishing year by Francis (2017b), confirming patterns previously seen: i.e., CPUE was zero or close to zero in EC and WC fisheries and fluctuated around low levels in the SA fishery (Figure 10.6).

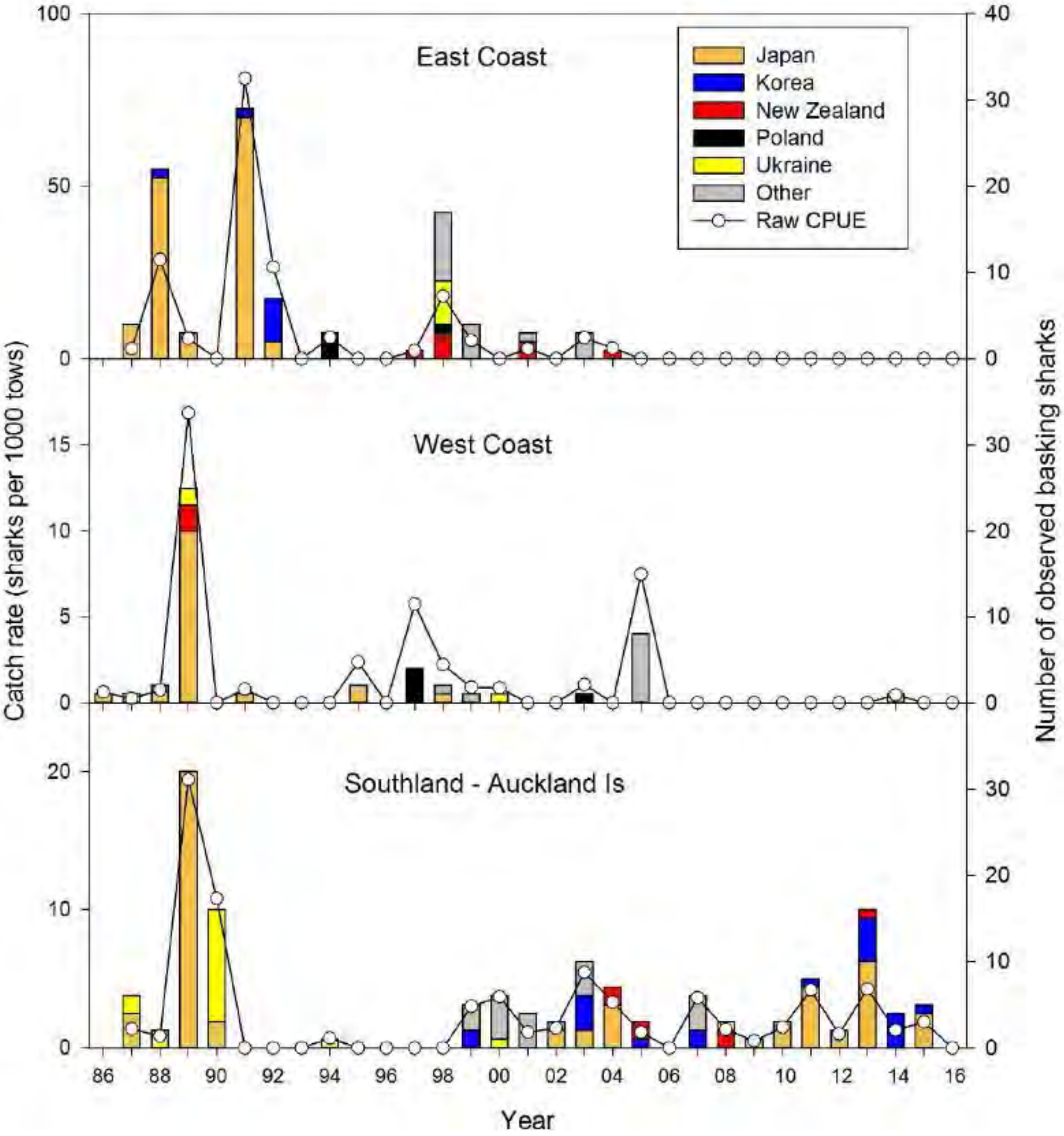


Figure 10.6: Basking shark catch rate indices (markers and lines) and number of shark observations by vessel flag (coloured bars) for three fishery areas. For raw CPUE indices, years are calendar years for West Coast and July–June years (labelled as the greater of the two years) for East Coast and Southland-Auckland Islands. Source: Francis 2017b

The very low (often zero) CPUE in EC and WC regions since 2006, and lack of large numbers and aggregations of basking sharks observed in Department of Conservation aerial surveys for dolphins around Banks Peninsula during the last decade (C. Duffy, DOC, pers. comm.), are cause for concern. There may not have been large aggregations of basking sharks in New Zealand waters since 1992. Whether such a long period without large aggregations is part of a long-term, natural cycle, or evidence of a decline in population abundance, cannot yet be determined (Francis & Smith 2010).

NON-QMS SPECIES

Indicator analyses have been carried out for eight shark and chimaera species: carpet shark, Baxter's dogfish, seal shark, longnose velvet dogfish, Plunket's shark, leafscale gulper shark, shovelnose dogfish, and longnose spookfish (Francis et al. 2016). The indicators calculated were relative biomass, median shark length, and proportion of male sharks (all from trawl surveys); and distribution (proportion of half-degree rectangles having raw catch per unit effort greater than a specified threshold), proportion of half-degree rectangles having zero reported catches in a fishing year, species composition, concentration (a measure of whether fishing effort focuses on or avoids areas of high shark abundance), and nominal and standardised CPUE (all from commercial catch-effort or observer data). Because of data limitations, or non-applicability of the method, only a subset of the indicators could be applied to each species, and then only to a few FMAs. None of the species showed clear and consistent evidence of recent declines in abundance. However, estimated trends were often uncertain, inconsistent among indicators, based on indicators that may be unreliable (e.g., trawl survey biomass estimates for species that are not well surveyed), and based on too few indicators (only trawl survey indicators were available for five out of eight species). For a number of species, one or more indicators showed signs of decline, and ongoing monitoring is recommended (Francis et al. 2016).

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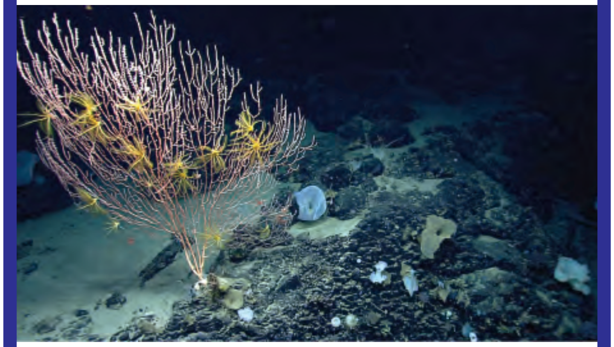
THEME 3: BENTHIC IMPACTS

Chapter 11: Benthic (seabed) impacts - Technical Summary

Disturbance of seabed habitats



Mapping bottom habitats and overlap with fisheries

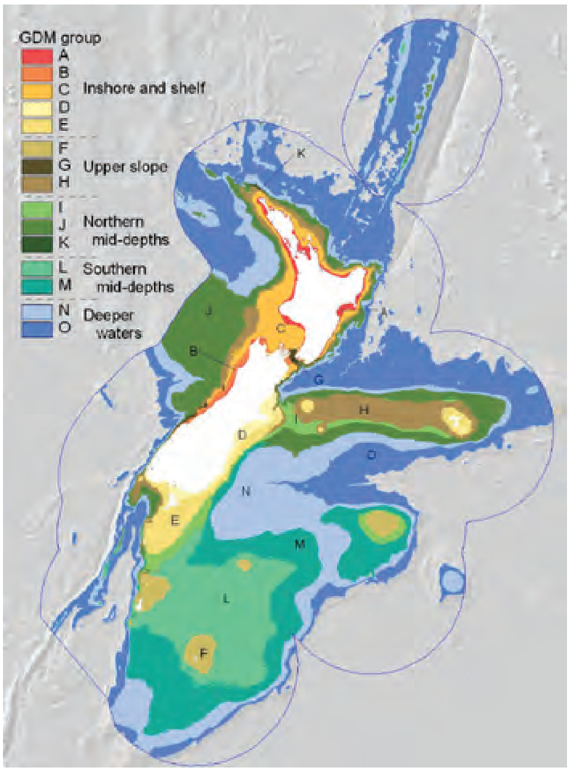
1. THE ISSUE IN BRIEF

- Biogenic habitats are important features of the seabed, because they support diverse communities
- Different types of fishing gear are dragged across the seabed to capture fish, crustaceans, or shellfish
- Contact with fishing gear can damage or kill benthic organisms, create a disturbance, and change the nature of the physical habitat
- Consequently, bottom fishing is monitored and restricted in some areas to manage (avoid, mitigate, remedy) adverse effects

2. BENTHIC IMPACTS AND RECOVERY

- Bottom fisheries can have extensive and long-lasting effects on seabed species and habitats
- Intensity of impacts depends on, among other factors, the type of fishing gear, the sensitivity of habitats or organisms to gear disturbance, and oceanographic currents
- There are challenges in estimating the effects of bottom fishing because multiple biological, environmental and anthropogenic factors need to be accounted for
- Recovery of habitats following disturbance by bottom fisheries in NZ has been studies in shallow and deep marine environments, with no recovery observed for some deep sea habitats after 14 years, and is the subject of ongoing research

3. MAPPING BOTTOM HABITATS



- Habitats are characterised by different physical conditions and assemblages of bottom dwelling species
- Remote imaging devices, cameras, and mechanical bottom samplers are used to identify the spatial distribution of species/habitats on the seafloor
- Modelling approaches are used to predict the distributions of benthic species, communities, and habitats at large spatial scales, based on the relationships between observed distributions of species and gradients in environmental variables

The map on the left represents the model-predicted distribution of different seafloor habitats in areas shallower than 3000 m (Benthic-Optimized Marine Environment Classification, habitats classified with letters from A to O)

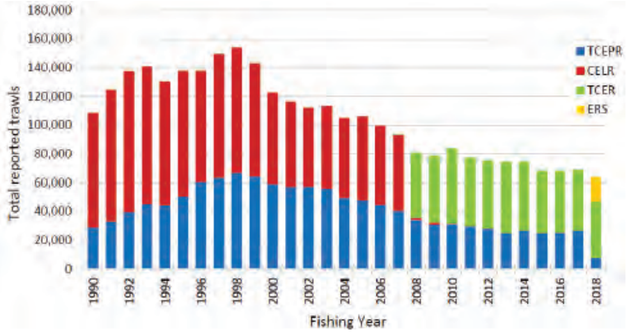
4. EXTENT AND INTENSITY OF SEABED CONTACT

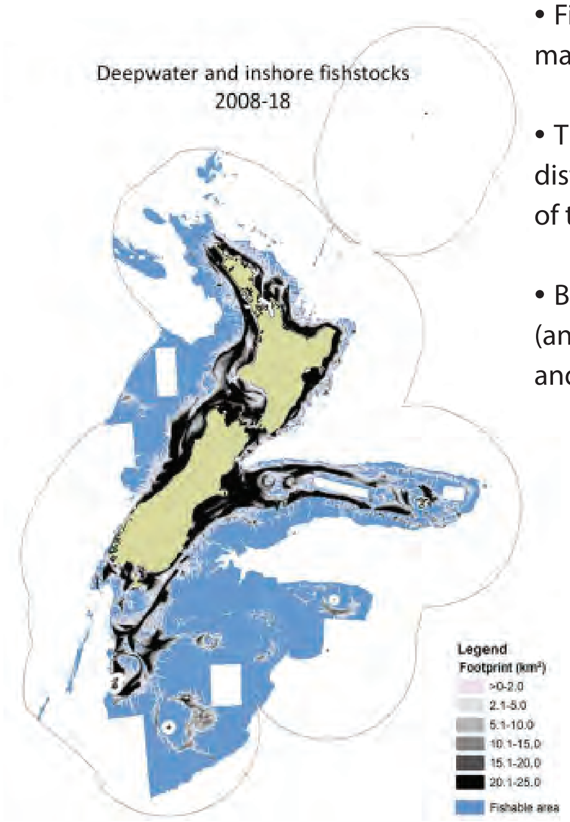
- Fisher-reported data, detailing where fishing occurs, are used to map the distribution of bottom fishing in NZ

- The extent and intensity of fisheries can be compared with the distribution of seabed habitats, and examined over different periods of time, to identify trends in fisheries overlap and impacts

- Between 2008 and 2018 bottom trawl fisheries contacted 7.2% (annual average of 2.2%) of the seabed in the NZ Territorial Sea (TS) and Exclusive Economic Zone (EEZ)

The figure below shows that the total number of trawl tows has been declining progressively since the late ’90s





Map of NZ cumulative bottom fisheries footprint 2008-2018 (in black)

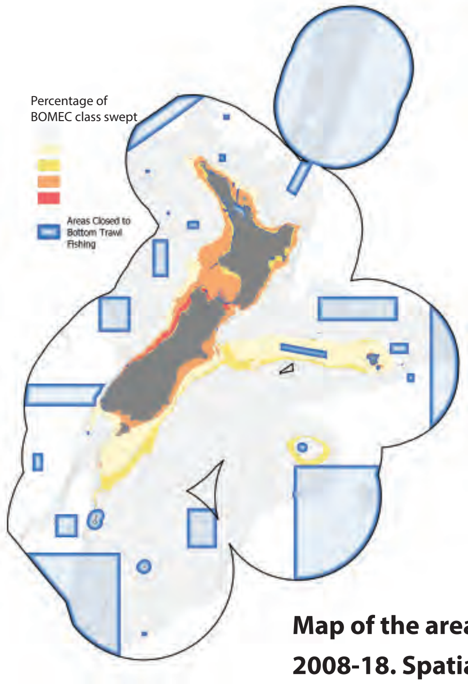
5. SPATIAL CONTROLS

- Limiting fisheries interactions with sensitive habitats is one of the management options to protect them

- The ‘trawlable’ depth zone (<1600 m) covers about 40% of the NZ EEZ area. 15% of this area is closed to fishing within 100 m from the seabed (Benthic Protection Areas, BPAs)

- Currently 19.6% of the NZ TS and 30.2% of the NZ EEZ area is closed to bottom fishing by dfferent spatial control measures. Dedicated BPAs and sea mount closures protect 7.6% of the NZ TS and 29.8% of the NZ EEZ

- Different sensitivities of animals and seabed habitats to fishing pressure should be taken into account when managing fishing



Map of the area swept by trawl fishing gear (%) for each of the BOMEC classes in 2008-18. Spatial protection areas are represented as blue shaded zones

6. ONGOING RESEARCH

- Monitoring seabed recovery after trawling closures in offshore (Graveyard) and inshore (Spirits bay) areas
- Improving predictive modeling to map seabed habitats and develop decision support tools for management
- Refining methods to estimate the extent and effects of bottom fishing, as well as recovery rates

11 BENTHIC (SEABED) IMPACTS

Status of chapter	This chapter has been partially updated for AEBAR 2021.
Scope of chapter	This chapter outlines the main effects of mobile bottom (or demersal) fishing gear on seabed habitats and communities. All trawl gears contacting the seabed and shellfish dredges are included. Danish seine and more or less static methods like bottom longline and potting are excluded in this version, as are fisheries outside the Exclusive Economic Zone (EEZ).
Area	All of the New Zealand Territorial Sea (TS) and EEZ. There will be some relevance for out-of-zone bottom trawl fisheries.
Focal localities	Areas that are fished more frequently and habitats that are more sensitive to disturbance are likely to be most affected; areas that are closed to bottom impacting methods will not be directly affected. Bottom trawling offshore is most intense on the western flanks and to the south-west of the Chatham Rise, the edge of the Stewart-Snares shelf, south-east of the Auckland Islands Shelf, and off the north-west coast of the South Island. In coastal waters shallower than 250 m, bottom trawling is most intense along the east coast of North Island, south of East Cape, and in Tasman Bay and Golden Bay. Shellfish dredges probably have the greatest effect but their footprint is much smaller than that of bottom trawl fisheries and generally in shallow waters in specific areas (e.g., Foveaux Strait).
Key issues	Habitat modification, potential loss of biodiversity, potential loss of benthic productivity, potential modification of important breeding or juvenile fish habitat leading to reduced fish recruitment.
Emerging issues	Potential for effects on habitats of particular significance to fisheries management (HPSFM). Opportunities presented by better spatial information from finer scale reporting of fishing locations (electronic reporting) and from vessel tracking data. Cumulative effects and interactions with other stressors (including existing effects, especially in the coastal zone, and climate change).
Fisheries New Zealand research (current)	BEN2019-04 *A spatially explicit benthic impact assessment for inshore and deepwater fisheries in New Zealand*; BEN2019-05 *Towards the development of a spatial decision support tool for managing the impacts of bottom fishing on in-zone, particularly vulnerable or sensitive habitats*; BEN2020-01 *Extent and intensity of seabed contact by mobile bottom fishing in the New Zealand Territorial Sea and Exclusive Economic Zone*; BEN2020-07 *Extent and intensity of trawl effort on or near underwater topographic features in New Zealand's Exclusive Economic Zone*; ZBD2020-06 *Exploring options for balancing fishing and habitat protection and recovery in the Hauraki Gulf*; ZBD2020-09 *Cumulative effects of stressors on scallops and scallop habitats in the Marlborough Sounds;* ZBD2020-11 *Review of land-based effects on coastal fisheries and kaimoana and their habitats*; BEN2021-01 *Characterisation of benthic habitats in four inshore areas and their potential as HPSFM*; BEN2021-03 *Taxonomic identification of benthic invertebrate samples*; BEN2021-05 *Habitat degradation impacts on blue cod populations*; ZBD2021-02 *Habitat degradation impacts on fishery productivity in northern harbours*.
New Zealand government research (current)	MBIE programme: Sustainable Seas COIX1515 Sustainable Seas Ko Nga Moana Whakauka.
Related chapters/issues	Biodiversity, habitats of particular significance for fisheries management (HPSFM), marine environmental monitoring, marine mining/sand extraction, land-based effects.

11.1 CONTEXT

For the purpose of this document, the term 'mobile bottom fishing methods' includes all types of trawl gear that are used in contact with the seabed as well as shellfish dredges of various designs and Danish seine nets. Relative to the information about trawls and dredges there is little information available about the distribution and effects of Danish seining, so Danish seining is not considered in detail. The benthic effects of other methods of catching fish on or near the seabed that do not involve deliberately towing or dragging fishing gear across the seabed are thought to be considerably less than those of the mobile methods (although they are not always negligible) and these methods are not considered in this document.

Trawls and dredges are used to catch a relatively high proportion of commercial landings in New Zealand and such methods can represent the only effective and economic way of catching some species. However, the resulting disturbance to seabed habitats and communities may have consequences for biodiversity and ecosystem services, including fisheries and other secondary production. The guiding sections of the Fisheries Act 1996 for managing the effects of fishing, including benthic effects, are s.8(2)(b), which specifies that 'ensuring sustainability' (s.8(1)) includes 'avoiding, remedying, or mitigating any adverse effects of fishing on the aquatic environment' and s.9, which specifies a principle that 'biological diversity of the aquatic environment should be maintained'. Also potentially relevant is the principle in s.9 that 'habitat of particular significance for fisheries management should be protected' (see the chapter on Habitats of Particular Significance for Fisheries Management for more details).

One approach to managing the effects of mobile bottom fishing methods is through the use of spatial controls. A wide variety of such controls apply in New Zealand waters (Figure 11.1). Some of these controls were introduced specifically to manage the effects of trawling, shellfish dredging, and Danish seining in areas or habitats considered sensitive to such disturbance (e.g., the bryozoan beds off Separation Point, between Golden Bay and Tasman Bay, and the sponge-dominated fauna to the north of Spirits and Tom Bowling bays in the far north). Other closures exist for other reasons but have the effect of protecting certain areas of seabed from disturbance by mobile bottom fishing methods. These include no-take marine reserves, marine farms, pipeline and power cable exclusion zones, and areas set aside to protect marine mammals (e.g., see Figure 11.2 for areas where trawling is prohibited, Figure 11.3 for areas where gear and seasonal restrictions apply, and Figure 11.4 for areas related to marine reserves and marine farms). Marine reserves provide marine protection in a range of habitats within the Territorial Sea. Although marine reserves provide a higher level of protection by prohibiting all extractive activities, most tend to be small. New Zealand's marine reserves protect about 9.4% of New Zealand's Territorial Sea; however, most of this is in two marine reserves in the territorial seas around offshore island groups in the far north and far south of New Zealand's EEZ (Helson et al. 2010). Until 2000, most closures that had the effect of protecting areas of seabed from disturbance by trawling and dredging were in the Territorial Sea.

In the Exclusive Economic Zone, 18 seamount closures were established in 2001 to protect representative underwater topographic features from bottom trawling and dredging (Brodie & Clark 2003; see Figure 11.1). These areas included 25 features, including 12 large seamounts more than 1000 m high, covering 2% (81 000 km^2) of the EEZ. The seamount areas are closed to all types of trawling and dredging. In 2006, members of the fishing industry proposed the closure of about 31% of the EEZ to bottom trawling and dredging in Benthic Protection Areas (BPAs), including the existing seamount closures. The design criteria for the BPAs were they should be large, relatively unfished, have simple boundaries, and be broadly representative of the marine environment. After a consultation process, a substantially revised package of BPAs (including three additional areas totaling 13 887 km^2, 10 additional active hydrothermal vents, and 35 topographic features) that complemented the existing seamount closures was implemented by regulation in 2007 (Helson et al. 2010; Figure 11.1). BPAs cover about 1.1 million km^2 (31%) of New Zealand's EEZ and are closed to trawling on or close to the bottom. Midwater trawling well off the bottom is permitted in the BPAs if two observers are on board and an approved net monitoring system is used. Much of the seabed within BPAs is below trawlable depth (maximum trawlable depth is about 1600 m) and all are outside the Territorial Sea. In combination, the seamount closures and the BPAs include: 46% of underwater topographic features (a term that includes underwater hills, knolls, and seamounts); 55% of seamounts over 1000 m high; and 88% of known active hydrothermal vents.

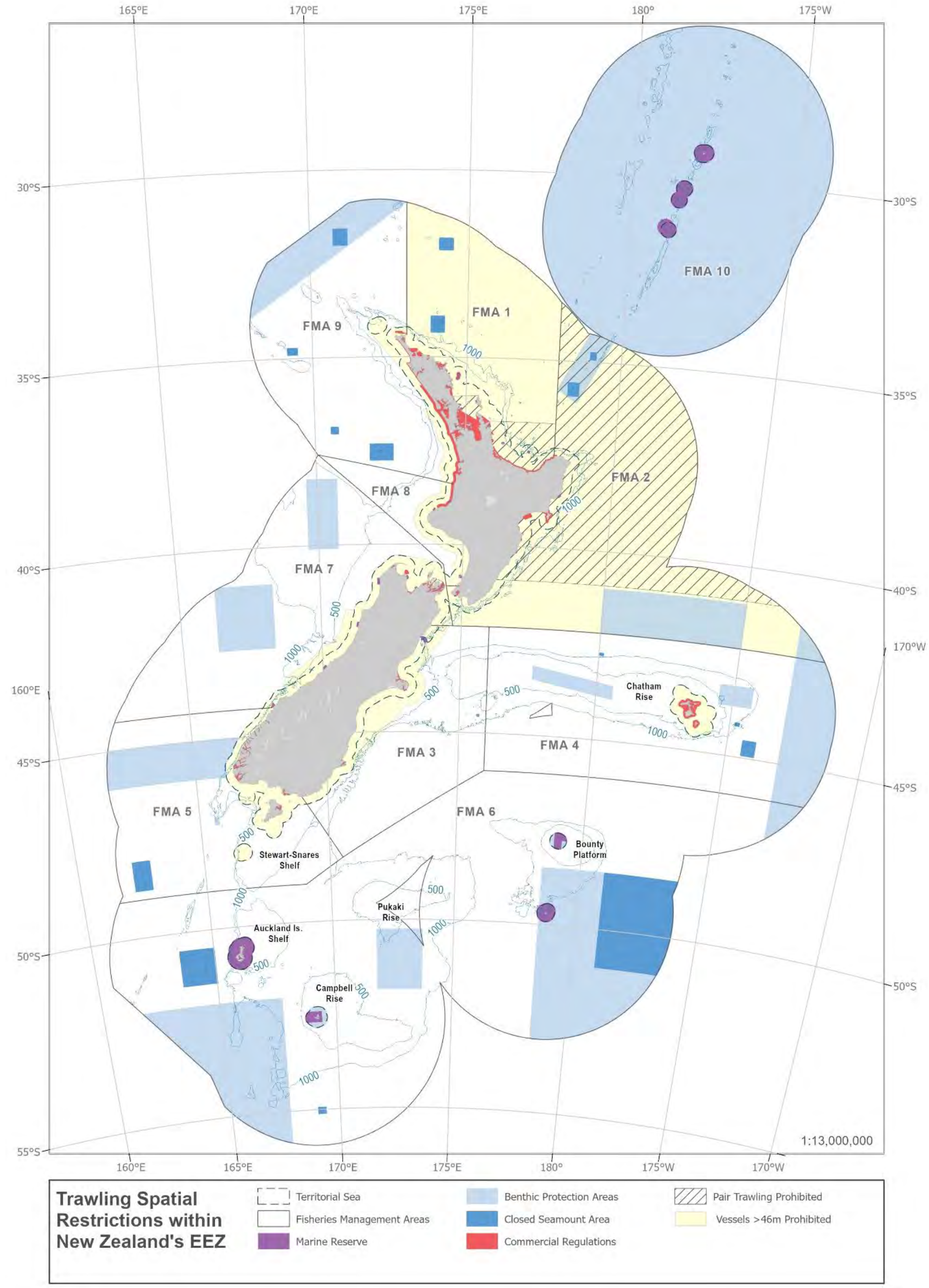


Figure 11.1: Map of the major spatial restrictions to trawling and Fisheries Management Areas (FMAs) within the outer boundary of the New Zealand EEZ. Vessels longer than 28 m may not trawl within the TS and additional restrictions are specified in the Fisheries (Auckland Kermadecs Commercial Fishing) Regulations 1986, the Fisheries (Central Area Commercial Fishing) Regulations 1986, the Fisheries (Challenger Area Commercial Fishing) Regulations 1986, the Fisheries (South East Area Commercial Fishing) Regulations 1986, and the Fisheries (Southland and Sub-Antarctic Areas Commercial Fishing) Regulations 1991. For more details of BPAs, see Helson et al. (2010). Map represents restrictions as at September 2021.

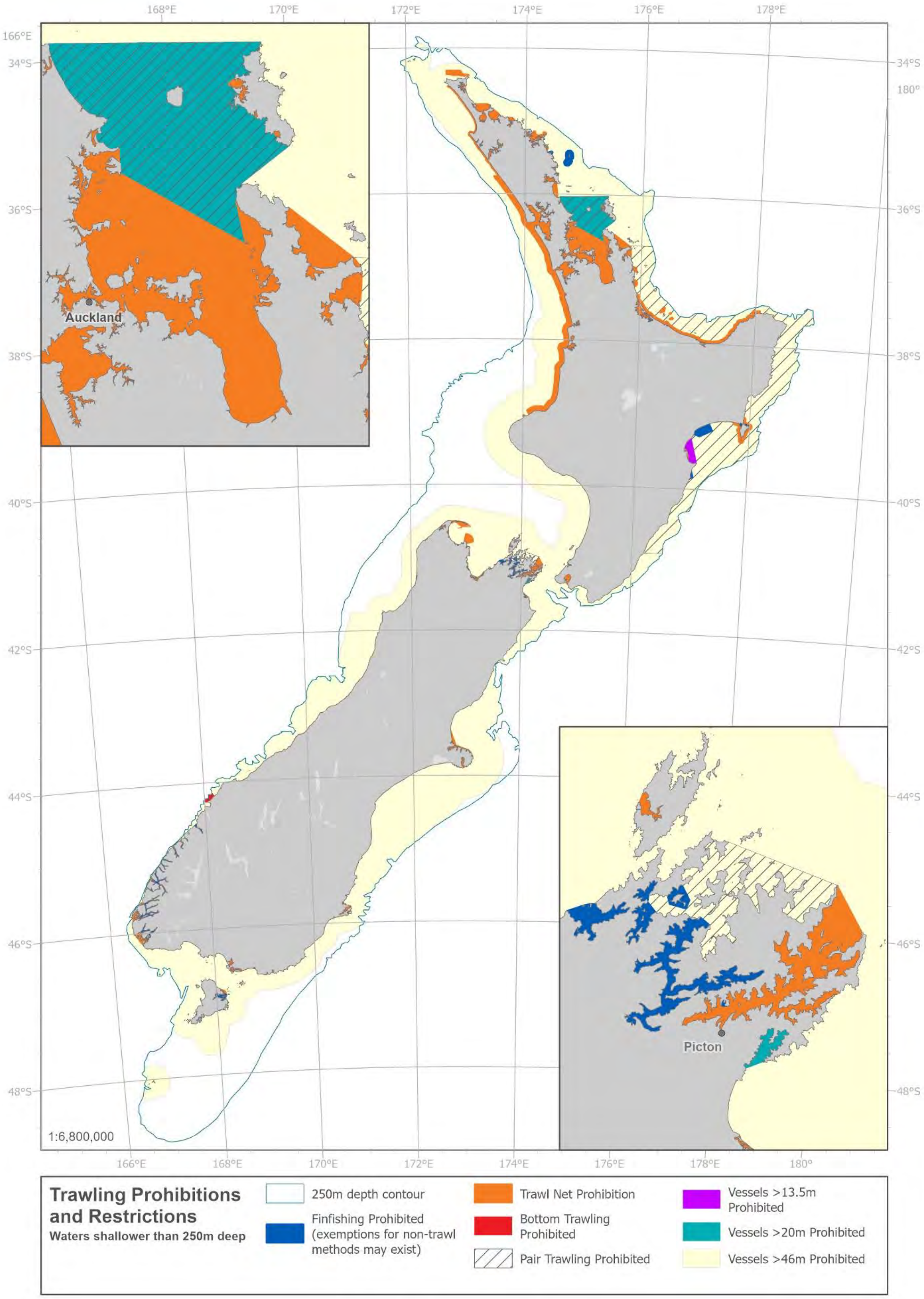


Figure 11.2: Areas showing where trawling is prohibited and other relevant restrictions apply in waters shallower than 250 m depth. Map represents prohibitions and restrictions as at September 2021.

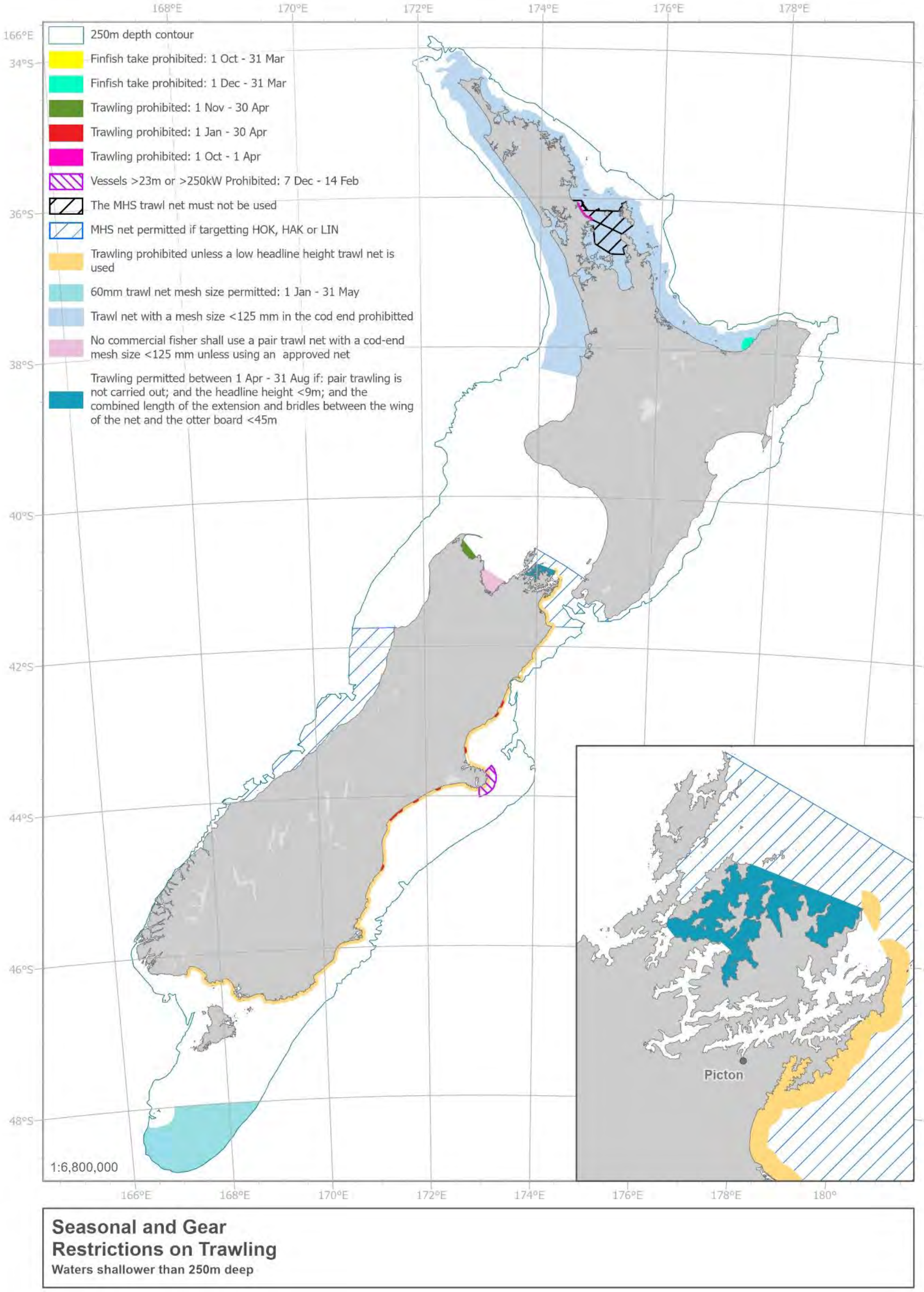


Figure 11.3: Areas where gear and seasonal restrictions apply to the use of trawl gear, in waters shallower than 250 m depth. Map represents restrictions as at September 2021.

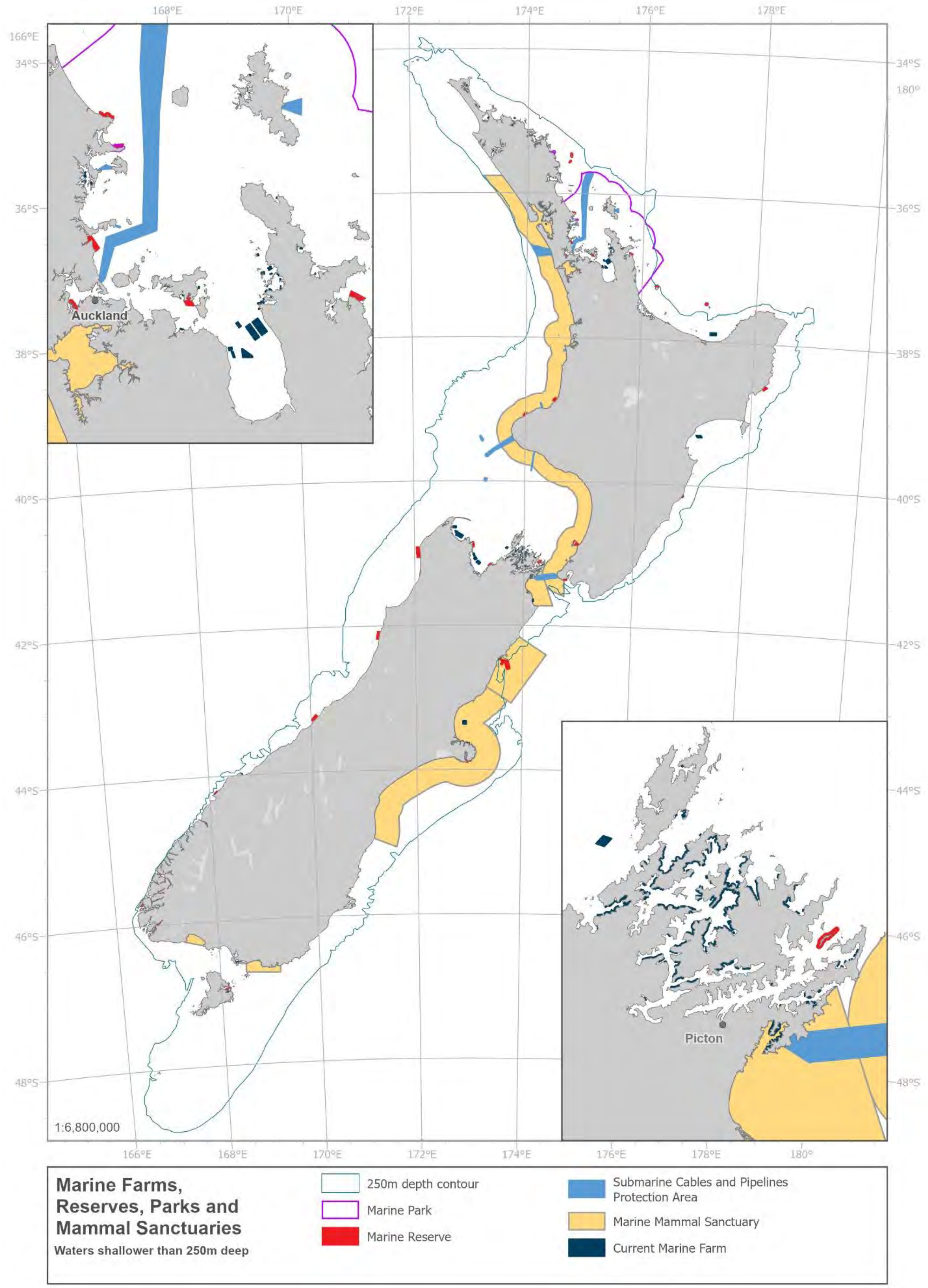


Figure 11.4: Locations of marine reserves and marine farms, marine mammal sanctuaries, marine parks, and cable and pipeline zones in waters shallower than 250 m depth. Map represents locations as at September 2021.

11.1 GLOBAL UNDERSTANDING

Concerns about the use of towed fishing gear on benthic habitats were first raised by fishermen in the fourteenth century in the UK (Lokkeborg 2005). They were worried about the capture of juvenile fish and the detrimental effects on food sources for harvestable fish. Despite this long history of concern, it is really only in the last 30 years that research efforts have focused strongly on the effects of mobile bottom fishing methods on benthic (seabed) communities, biodiversity, and production. This activity, combined with controversy around fishing effects, has spawned numerous reviews that seek to summarise or synthesise the information (Jones 1992, Dayton et al. 1995, Jennings & Kaiser 1998, Watling & Norse 1998, Lindeboom & de Groot 1998, Auster & Langton 1999, Hall 1999, ICES 2000a, 2000b, Kaiser & de Groot 2000, National Marine Fisheries Service 2002, National Research Council 2002, Dayton et al. 2002, Thrush & Dayton 2002, Lokkeborg 2005, Barnes & Thomas 2005, Clark & Koslow 2007, Clark et al. 2016).

Benthic habitats provide shelter and refuge for juvenile fish and the associated fauna can be the prey of demersal fish species. Towed fishing gears (particularly trawl doors) affect benthic habitats and organisms but the level of effect will depend on the type of trawl doors and ground gear used and the way the gear is rigged, and the physical and biological characteristics of the seabed habitats in the fishing grounds. The effects are difficult to assess because of the complexity of benthic communities and their temporal and spatial variability, and interpretation can also be complicated by environmental gradients or change. For reasons of accessibility, cost, and tractability, most research on seabed disturbance caused by human activities worldwide has been carried out in coastal systems, and our understanding of the effects of physical disturbance on the highly diverse communities of the deep sea has developed only recently.

The reviews above broadly indicate that numerical abundance of many invertebrates decline (sometimes substantially) after mining, trawling, or other major disturbance. Trawling and dredging can re-suspend sediment and can, depending on sediment and local currents, alter sediment characteristics. Physical effects include furrows and berms from trawl doors, furrows from the bobbins and rock hoppers, and sediment re-sorting, but the magnitude of these effects depends on sediment type, currents, and wave action (if any). Bottom trawling can also alter natural sediment fluxes and reduce organic carbon turnover (Pusceddu et al. 2014), the depth of the oxic layer in sediments (Churchill 1989, Warnken et al. 2003, Bradshaw et al. 2012), and the shape of the upper continental slope (Puig et al. 2012), reducing morphological complexity and benthic habitat heterogeneity. The mixing of sediments and overlying water can alter the chemical makeup of the sediment and have considerable effects in deep, stable waters (Rumohr 1998). Chemical release from the sediment can also be changed, as shown for phosphate in the North Sea (ICES 1992, noting lower fluxes were observed after trawling events).

Trawling can alter benthic communities, reduce total biomass of benthic species, and increase predation by scavengers. Sites subject to greater natural disturbance are generally thought to be less susceptible to change from bottom contact fishing (but see Schratzberger et al. 2009 who concluded that common anthropogenic disturbances differ fundamentally from natural disturbance). There has been less work on the effects of other methods of catching demersal fish or crustaceans that do not involve deliberately towing or dragging fishing gear across the seabed, but some of these methods can have non-negligible effects (e.g., Sharp et al. 2009, Williams et al. 2011).

Studies of recovery dynamics are rarer still, but a return to pre-disturbance levels after bottom-contacting fishing can take up to several years, even in some sites subject to considerable natural disturbance (see Kaiser et al. 2006 for a summary). In shallow regions with mobile sediments, the effects are generally difficult to detect and recovery can be rapid (e.g., Jennings et al. 2005). Examining epifauna, Lambert et al. (2014) estimated recovery from scallop dredging to take from less than 1 year to over 10 years, depending on functional group, with faster recovery in areas with faster tidal currents, and large-bodied species recovering faster when conspecifics were abundant locally. Hard-bottom fauna is predicted to recover most slowly and Williams et al. (2010) concluded that hard-bottom fauna on Australasian seamounts did not show signs of recovery within 5–10 years. Recovery rate is typically correlated with the spatial extent of a disturbance event (e.g., Hall 1994, Kaiser et al. 2003; see also Figure 11.5) and the effects of some ‘catastrophic’ natural disturbance events, such as large-scale marine mudslides, can be detected for hundreds of years, even for taxa thought to be robust to physical disturbance such as nematodes (Hinz et al. 2008).

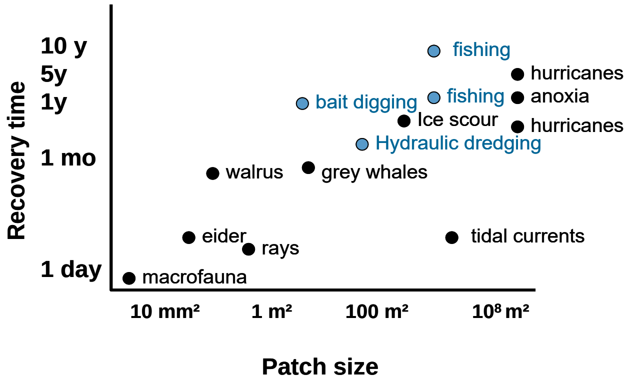


Figure 11.5: General relation between the spatial extent of disturbance events and the time taken to recover from such events in marine systems (after Kaiser et al. 2003). Blue dots signal human impacts, including fishing in habitats of different abilities to recover, and black dots signal natural disturbance.

Rice (2006) summarised the findings of five major reviews of the effects of mobile bottom-contacting fishing gears on benthic species, communities, and habitats. In this 'review of reviews' Rice (2006) summarised the findings of the multiple working groups that contributed to the reviews as follows:

Rice's (2006) conclusions about the effects on habitats of mobile bottom fishing gears were that they can:

- Damage or reduce structural biota (all reviews, strong evidence or support).
- Damage or reduce habitat complexity (all reviews, variable evidence or support).
- Reduce or remove major habitat features such as boulders (some reviews, strong evidence or support).
- Alter seafloor structure (some reviews, conflicting evidence for benefits or harm).

Other emergent conclusions on habitat effects included:

- There is a gradient of effects, with greatest effects on hard, complex bottoms and least effect on sandy bottoms (all reviews, strong support, with qualifications).
- There is a gradient of effects, with greatest effects on low energy environments and least (often negligible) effect on high-energy environments (all reviews, strong support).
- Trawls and mobile dredges are the most damaging of the gears considered (three of the reviews considered other gears; all drew this conclusion, often with qualifications).

Mobile bottom gears affect benthic species and communities in that they:

- Can change the relative abundance of species (all reviews, strong evidence or support).
- Can decrease the abundance of long-lived species with low turnover rates (all reviews, moderate to strong evidence or support).
- Can increase the abundance of short-lived species with high turnover rates (all reviews, moderate to occasionally strong evidence or support).
- Affect populations of surface-living species more often and to greater extents than populations of burrowing species (all reviews, weak to occasionally strong evidence or support).
- Have lesser effects in high-energy or frequent natural disturbance environments than in low energy environments where natural disturbances are uncommon (four reviews (the other did not address the factor), strong evidence or support).
- Affect populations of structurally fragile species more often and to greater extents than populations of 'robust' species (all reviews, variable evidence and support).
- Temporarily increase the abundance of scavengers in areas where bottom trawls have been used (three reviews, variable support or evidence, all argue for the effects being transient).
- Increase the rates of nutrient cycling or sedimentation in areas where bottom trawls have been used (two reviews, mixed views on magnitude of effects and conditions under which they occur).

Considerations in the application or adoption of mitigation measures:

- The effect of mobile fishing gears on benthic habitats and communities is not uniform. It depends on:
  - The features of the seafloor habitats, including the natural disturbance regime (all reviews, strong evidence or support);
  - The species present (all reviews, strong evidence or support, though not mentioned by National Marine Fisheries Service panel);
  - The type of gear used and methods of deployment (all reviews, moderate to strong evidence or support);
  - The history of human activities, particularly past fishing, in the area of concern (all reviews, strong evidence or support).

- Recovery time from trawl-induced disturbance can take from days to centuries, and depends on the same factors as listed above (all reviews, strong evidence or support).
- Given the above considerations, the effect of mobile bottom gears has a monotonic relationship with fishing effort, and the greatest effects are caused by the first few fishing events (all reviews, moderate to strong evidence or support).
- Application of mitigation measures requires case specific analyses and planning; there are no universally appropriate fixes (three reviews, moderate to strong evidence or support. The issue of implementing mitigation was not addressed in the FAO review. It was also stressed in the US National Academy of Sciences review and discussed in the ICES review that extensive local data are not necessary for such case-specific planning. The effects of mobile bottom gears on seafloor habitats and communities are consistent enough with well-established ecological theory, and across studies, that cautious extrapolation of information across sites is legitimate).

Rice (2006) concluded *"These overall conclusions on impacts and mitigation measures, and recommendations for management action form a coherent and consistent whole. They are relevant to the general circumstances likely to be encountered in temperate, sub-boreal, and boreal seas on coastal shelves and slopes, and probably areas … beyond the continental shelves. They allow use of all relevant information that can be made available on a case by case basis, but also guide approaches to management in areas where there is little site-specific information."*

Since Rice's (2006) paper, Kaiser et al. (2006) published a meta-analysis of 101 separate manipulative experiments that confirms many of Rice's findings. Shellfish dredges have the greatest effect of the various mobile bottom fishing gears, biogenic habitats are the most sensitive to such disturbance (especially for attached fauna on hard substrates), and unconsolidated, coarse sediments (e.g., sands) are the least sensitive. Kaiser et al. (2006) concluded that recovery from disturbance events can take months to years, depending on the combination of fishing method and benthic habitat type. This meta-analysis of manipulative experiments was an important development, reinforcing the inferences drawn from multiple mensurative observations at much larger scale ('fisheries scale') in New Zealand (e.g., Thrush et al. 1998, Cryer et al. 2002) and overseas (e.g., Craeymeersch et al. 2000, McConnaughey et al. 2000, Bradshaw et al. 2002, Blyth et al. 2004, Tillin et al. 2006, Hiddink et al. 2006). This is a powerful combination that implies substantial generality of the findings.

The international literature is, therefore, clear that bottom (demersal) trawling and shellfish dredging are likely to have largely predictable and sometimes substantial effects on benthic community structure and function. The positive or negative consequences for ecosystem processes such as production have been addressed in several studies (e.g., Jennings et al. 2001, Reiss et al. 2009, Hiddink et al. 2011). It has been mooted that frequent disturbance should lead to the dominance of smaller species with faster life histories and that, because smaller species are more productive than larger ones, system productivity and production should increase under trawling disturbance. However, when this proposition has been tested, it has not been supported by data in real fishing situations (e.g., Hermsen et al. 2003, Reiss et al. 2009) and, where overall productivity has been assessed, it decreases with increasing trawling disturbance.

For example, Veale et al. (2000) examined spatial patterns in the scallop fishing grounds in the Irish Sea and found that total abundance, biomass, and secondary production (including that of most individual taxa examined) decreased significantly with increasing fishing effort. Echinoids, cnidarians, prosobranch molluscs, and crustaceans contributed most to the differences. Jennings et al. (2001) showed that, in the North Sea, trawling led to significant decreases in infaunal biomass and production in some areas even though production per unit biomass rose with increased trawling disturbance. The expected increase in relative production did not compensate for the loss of total production that resulted from the depletion of large-bodied species and individuals. Hermsen et al. (2003) found that mobile fishing gear disturbance had a conspicuous effect on benthic megafaunal production on Georges Bank, and cessation of such fishing led to a marked increase in benthic megafaunal production, dominated by scallops and urchins. Hiddink et al. (2006) estimated that more than half of the southern North Sea was trawled sufficiently frequently to depress benthic biomass by 10% or more, and that 27% was in a state where benthic production was depressed by 10% or more. They estimated that recovery from this situation would take 2.5–6 years or more once fishing effort had been eliminated. They further estimated that fishing reduced benthic biomass and production by 56% and 21%, respectively, compared with an unfished situation. Reiss et al. (2009) found that, although sediment composition was the most important driver of benthic community structure in their North Sea study area, the intensity of fishing effort was also important and reductions in the secondary production of the infaunal community could be detected even within this heavily fished region.

The types of models developed by Hiddink et al. (2006, 2011; but see also Ellis & Pantus 2001 and Dichmont et al. 2008) can be used to assess the likely performance of different management approaches or levels of fishing intensity. Such management strategy evaluation (MSE) methods involve specifying management objectives, performance measures, a suite of alternative management strategies, and evaluating these alternatives using simulation (Sainsbury et al. 2000). For instance, the early study by Ellis & Pantus (2001) assessed the effect of trawling on marine benthic communities by combining an implementation of the spatial and temporal behaviour of the local fishing fleet with realistic ranges for the removal and recovery of benthic organisms. The model was used to compare the outcomes of two radically different management approaches, spatial closures and reductions in fishing effort.

From a New Zealand perspective, Mormede & Dunn (2013) developed a simple spatially explicit population model as a tool to assist ecological risk assessments, and Lundquist et al. (2010, 2013) used a more sophisticated spatially explicit landscape mosaic model with variable connectivity between patches to assess the implications of different spatial and temporal patterns of disturbance in the model landscape. They found that the scale of the disturbance regime (which could be trawling or any other physical disturbance) and the dispersal processes interact, and that the scales of these processes greatly influenced changes in the structure and diversity of the model community, and that recovery across the mosaic depended strongly on dispersal. System stability also decreased as dispersal distance decreased. Patterns of abundance of different species groups observed across gradients of fishing pressure were in general agreement with model predictions.

11.2 STATE OF KNOWLEDGE IN NEW ZEALAND

To understand the effects of mobile bottom fishing methods on benthic habitats, it is necessary to have knowledge of:

- the distribution of such habitats,
- the extent to which mobile bottom fishing methods are used in each habitat (the overlap),
- the consequences of any such disturbance (potentially in conjunction with other disturbances or stressors), and
- the nature and speed of recovery from the disturbance.

These components are discussed below.

11.2.1 DISTRIBUTION OF HABITATS

Mapping of benthic habitats at the large scales inherent in fisheries management is expensive and time-consuming so the New Zealand government commissioned an environmental classification to provide a spatial framework that subdivided the TS and EEZ into areas having similar environmental and biological character. The Marine Environment Classification (MEC) was launched in 2005 (Snelder et al. 2004, 2005, 2006) using available physical and chemical predictors, because environmental pattern was thought to be a reasonable surrogate for biological pattern. The authors suggested that the MEC provided managers with a useful spatial framework for broad-scale management, but cautioned that the full utility and limitations would become clear only as the MEC was applied to real issues. They described the MEC as a tool to organise data, analyses, and ideas, and as only one component of the information that would be employed in any analysis. The 20-class version (Figure 11.6) has been the most widely cited, although additional classification levels provide more detail that is significantly correlated with biological layers. The 2005 MEC was not optimised for any specific ecosystem component but was ‘tuned’ against data for demersal fish, phytoplankton, and benthic invertebrates. It performed least well as a classification of benthic invertebrates and, at the 20-class level, grouped most of the Chatham Rise and Challenger Plateau into a single class. Although separation of these two areas was evident as the MEC was driven to larger numbers of classes, their inclusion within a single class in the 20-class classification was considered counter-intuitive because their productivity and fisheries are known to be very different.

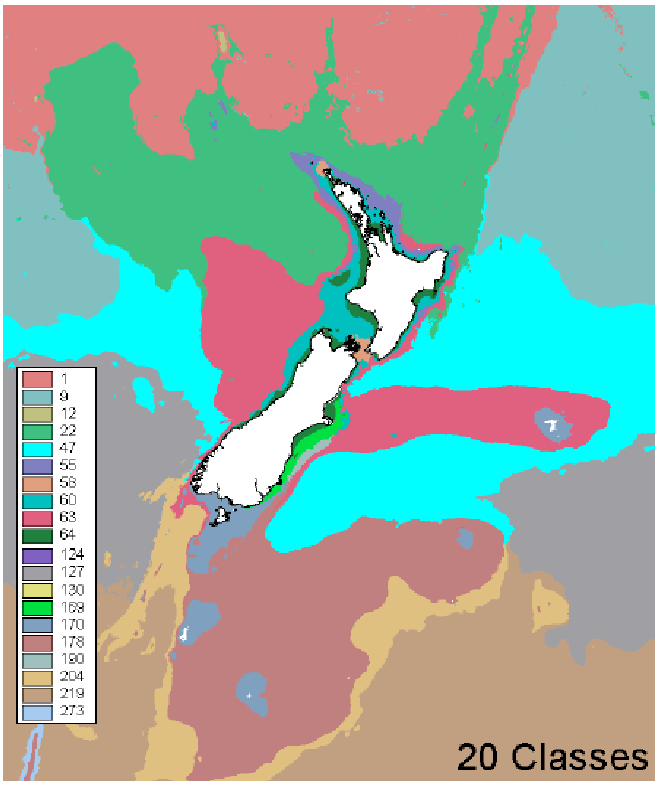


Figure 11.6: The 20-class version of the 2005 general purpose Marine Environment Classification (MEC, from Snelder et al. 2005).

This disquiet with the predictions of the original MEC for benthic habitat classes led to the development of alternatives that might perform better for benthic systems. First of these was a classification optimised for demersal fish (Leathwick et al. 2006). Several variants of this classification outperformed the original MEC for demersal fish, particularly at lower levels of classification detail and it was adopted by the Ministry for the Environment for their indicators related to bottom trawling and their 2010 Environmental Snapshot where the trawl footprint is compared with putative habitats (Ministry for the Environment 2010).

Based partly on this experience, the Ministry of Fisheries commissioned a Benthic-Optimised Marine Environment Classification, BOMEC (Leathwick et al. 2012). Many more physical, chemical, and biological data layers were available for the development and tuning of this classification than for the 2005 MEC. Especially relevant for benthic invertebrates was the inclusion of a layer for sediment grain size (notably absent from the MEC). Generalised Dissimilarity Modelling (GDM; Ferrier et al. 2002, 2007, Leathwick et al. 2011) was used to define the classification because this approach is well suited to the sparse and unevenly distributed biological data available. The BOMEC classes (15-class level version shown in Figure 11.7) were strongly driven by depth, temperature, and salinity into five major groups: inshore and shelf, upper slope, northern mid-depths, southern mid-depths, and deeper waters (generally beyond the fishing footprint, down to 3000 m, the limit of the analysis). Waters deeper than 3000 m could be considered an additional class. The 15-class BOMEC levels were used in conjunction with a broad sediment type classification and broad depth bands to identify 112 benthic habitats shallower than 250 m (Figure 11.8) (Baird et al. 2015).

Testing by Bowden et al. (2011) indicated that the BOMEC out-performed the original MEC at predicting benthic habitat classes on and around the Chatham Rise, but that none of the available classifications were very good at predicting the abundance and composition of benthic invertebrates at the fine scale of the sampling undertaken (tens of metres to kilometres). This, in conjunction with the findings of Leathwick et al. (2006), reinforces the role of environmental classifications as broad-scale predictors of general patterns at broad scale (tens to hundreds of kilometres) when more specific biological information is not available.

The Department of Conservation recently commissioned a Seafloor Community Classification (SCC, Stephenson et al. 2021). Like the previous classifications, the SCC is a numerical classification, and it provides advances on both the MEC and BOMEC because substantially more biological and environmental data were used. Using Gradient Forest (GF) models, occurrence records for demersal fish, benthic invertebrates, macroalgae, and reef fish informed the transformation of 33 environmental variables to represent spatial patterns of taxa compositional turnover. The overall predicted spatial patterns of compositional turnover was classified into 75 groups and uncertainty estimates of compositional turnover for each of the groups were produced. The SCC groups were differentiated primarily according to oceanographic conditions such as depth and bottom temperature at broad scales. More pronounced environmental differences between groups were evident at shallower depths where variation in more localised environmental conditions such as productivity, seafloor topography, seabed disturbance, and tidal currents were important differentiating factors. An associated publication was produced providing detailed descriptions of the 75 SCC groups (Petersen et al. 2020).

Where broad-scale classification methods are not applicable, other approaches have been taken. The trawl fisheries for orange roughy, oreos, and cardinalfish take place to a large extent on seamounts or other features (Clark & O'Driscoll 2003, O'Driscoll & Clark 2005). These features are often geographically small and, in common with other localised habitats such as vents, seeps, and sponge beds, do not appear on broad-scale habitat maps (e.g., at EEZ scale) and cannot realistically be predicted by broad-scale environmental classifications. Many features have been extensively mapped in recent years (e.g., Clark et al. in review), and seamount classifications based on biologically-referenced physical and environmental 'proxies' have also been developed, in New Zealand waters by Rowden et al. (2005) and globally by Clark et al. (2010a, 2010b). Davies & Guinotte (2011) developed a method of predicting the framework-forming (i.e., physically structuring) coldwater corals that are a focus for benthic biodiversity in deepwater systems.

MPI and MBIE funded a range of projects to inform the spatial management of the South Pacific Region and support the role of New Zealand in SPRFMO. The project 'Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific Region' is led by NIWA in collaboration with Victoria University of Wellington and the Marine Conservation Institute (USA). This has involved a major compilation of data on VME species distribution (including corals) throughout the SPRFMO area, as well as the EEZ. Modelling has been completed for the EEZ and general New Zealand region (Anderson et al. 2016a), as well as validation analyses of a wider regional model based on a survey of the Louisville Seamount Chain in 2014 (Anderson et al. 2016b). Work continues worldwide, including in New Zealand, on the development of sampling, analytical, and modelling techniques to provide cost-effective assessments of the distribution of marine habitats at a range of scales. Bowden et al. (2015) provide a desk top assessment of future options for monitoring deepwater benthic communities and conclude that photographic approaches sampling mega-epifauna are likely to be the most cost effective and relevant for detecting ecological effects at the scale of deep sea fisheries. Such sampling could be added to existing surveys but would require dedicated time. Opportunistic sampling from trawl surveys or observer data cannot be relied upon to provide representative samples of the benthic community.

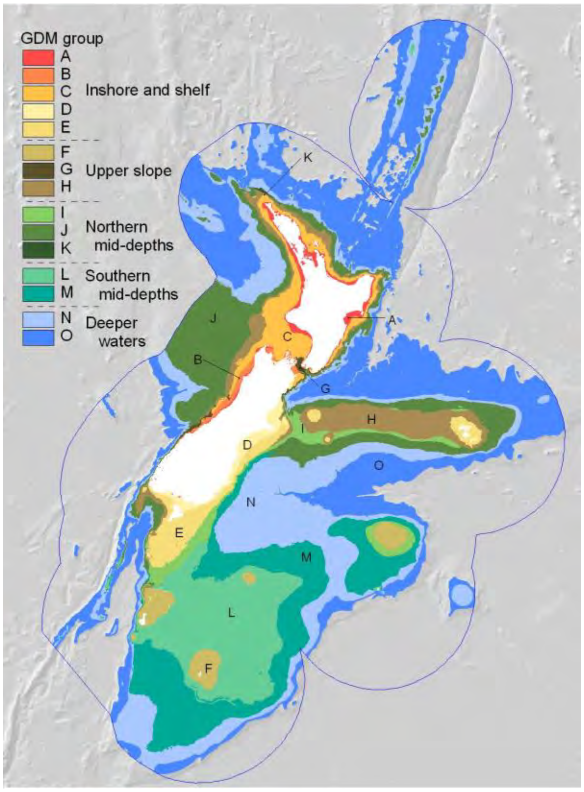


Figure 11.7: Map of the distribution of Benthic-Optimised Marine Environment Classification (BOMEC) classes defined by multivariate classification of environmental data transformed using results from GDM analyses of relationships between environment and species turnover averaged across eight taxonomic groups of benthic species. [From Leathwick et al. (2012).]

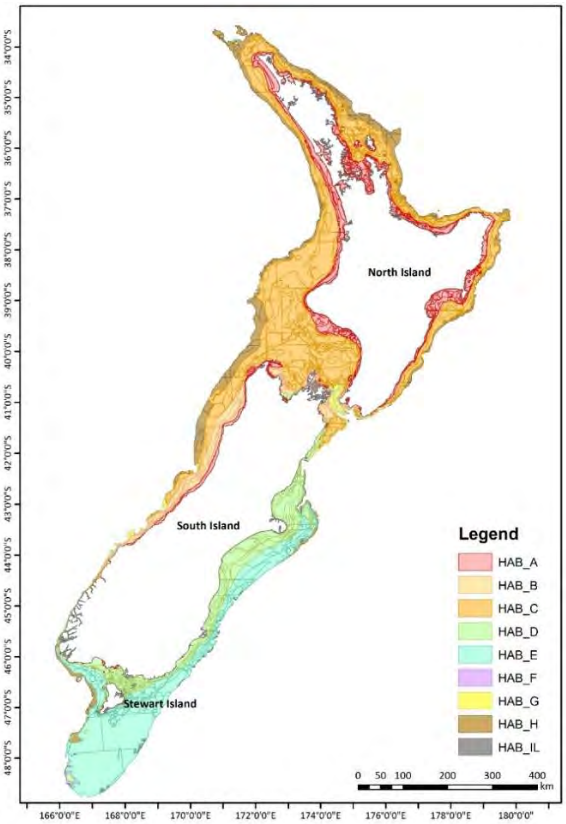


Figure 11.8: The broad habitat definitions based on the BOMEC classes, with divisions indicating areas of different sediment, depth zone, and statistical area in waters shallower than 250 m depth. [From Baird et al. (2015).]

11.2.2 DISTRIBUTION OF FISHING

Since 1989–90, mobile bottom fishing effort has been reported through one of four data collection methods each of which was introduced in different years (Table 11.2). Trawl Catch Effort and Processing Returns (TCEPRs) contain detailed spatial and other information for each trawl tow. These were introduced for vessels over 28 m that generally fished deeper waters within the New Zealand EEZ, with the first data collection year being 1 October 1989 to 30 September 1990. Since 2007–08, Trawl Catch and Effort Returns (TCERs) have been available for smaller, predominantly inshore trawlers. These include spatial and other information for each trawl tow but in less detail than on TCEPRs. Catch Effort and Landing Returns (CELRs) were used prior to 2007–08 and include only summarised information for each day's fishing, with very limited spatial resolution. This effort is not included in trawl footprint estimations due to the lack of tow-specific spatial data. Before 2007–08, the amount of overall trawl effort each year could be double that included in the trawl footprint estimation based on tow-level data reported on TCEPRs and TCERs (Baird et al. 2011).

Between 2007–08 and 2018–19, almost all trawling effort was reported on TCEPR or TCER forms. Electronic Reporting Systems (ERS) were introduced in 2018 for deepwater fishstocks and in 2019 for inshore fishstocks. This has provided similar data but at a finer resolution and, for inshore data, an endpoint for each tow. The change of data collection during the time series needs to be considered when interpreting trawl footprint estimations.

Trawl effort for all target species are included in the data shown in Figure 11.9. After the peak of over 140 000 reported trawl tows in 1996–97 and 1997–98 (Figure 11.9) when slightly over half of all tows were reported on TCEPRs, overall trawling effort declined to less than 60 000 tows per year by 2020–21, all of which are reported electronically (ERS).

Baird et al. (2002) and Baird et al. (2011) described the distribution and frequency of reported fishing by mobile bottom fishing gear (dredge, Danish seine, bottom trawl, bottom pair trawl, and midwater trawl in contact with the bottom) in New Zealand's TS and EEZ during the 1990s and up to 2004–05, respectively, for all commercial targets. This work was updated to 2011–12 by Black & Tilney (2017) for deepwater target species effort reported on TCEPRs. These reports showed that fishing was highly heterogeneous (spatially), but had considerable consistency among years; sites that were fished heavily in one year were likely to be fished heavily in other years. A similar but more detailed analysis was conducted for the Chatham Rise and subantarctic areas by Baird et al. (2006). Tows reported on TCEPRs were included in the main spatial analysis, but some additional analysis was possible using tows reported on CELRs. Until 2006–07, a substantial proportion of reported trawling effort was captured using CELRs. This was predominantly by vessels targeting inshore fish species but did also include some targeted effort for some 'deepwater' species. For instance, Cryer & Hartill (2002) estimated that, in the Bay of Plenty in the 1990s, 78%, 75%, and 39% of trawl tows targeting tarakihi, gemfish, and hoki, respectively, were reported on CELR forms.

Baird et al. (2015) analysed a combined data set of TCEPR and TCER data for the area shallower than 250 m for 2007–08 to 2011–12. Previous trawl footprint analyses (e.g., Baird et al. 2011, Black et al. 2013) underestimated trawl effort in inshore areas because tows recorded on CELR forms, which lack suitable spatial data, could not be included in the estimation of the trawl footprint.

Baird & Wood (2018) updated the three annual measures of fishing effort for deepwater fishstocks (1989–90 to 2015–16): the number of tows, the aggregate swept area (using assumed door spreads), and the coverage ('footprint') of the total trawl contact. Trawls were represented spatially as tracklines between the reported start and finish positions buffered by the assumed door spread to generate trawl polygons. The aggregate swept area for a year is the sum of the areas of the polygons and the 'footprint' is the estimated area of the seabed that is covered by the polygons overlaid. The estimated swept areas and footprint do not account for any modification that might occur alongside the trawl path as represented by the swept area polygon (e.g., by suspended sediments transported by currents away from the trawl track).

The 1989–90 to 2015–16 footprint (Baird & Wood 2018) contacted about 20% of the area shallower than 1600 m and 23% of the area that is open to bottom trawling and shallower than 1600 m.

Baird & Wood (2018) used reported tows on small topographic features that are a focus for orange roughy and cardinalfish fisheries by defining polygons for these tows as radii around the reported start position with the area swept estimated from the reported duration and speed of the tow. These short tows do not appear to contribute substantially to broad-scale plots at the scale of the EEZ, yet can represent intense fishing effort on particular, small seamount features (e.g., O'Driscoll & Clark 2005, Rowden et al. 2005).

Baird & Mules (2019) generated a trawl footprint from the bottom-contacting trawl effort for deepwater fishstocks for 2007–08 to 2016–17, as reported on TCERs and TCEPRs . This 10 year footprint contacted about 11% of the area shallower than 1600 m and 13% of the area that is open to bottom trawling and shallower than 1600 m.

Trawl footprint estimations for inshore (2008 fishing year onwards), deepwater (1990 fishing year onwards) and all stocks combined have recently been completed up until the 2018 fishing year. (Baird & Mules 2021a) and the 2019 fishing year (Baird & Mules 2021b).

The 1990–2019 dataset used to estimate the trawl footprint includes only the effort data collected on a tow-by-tow basis. During this period, the trawl footprint was based on TCEPRs (all fishing years) and TCERs (fishing years 2008 to 2019) only; thus, for the fishing years before 2008 the data predominantly represented deepwater offshore trawling. A substantial portion of trawl effort for the period 1990–2007 (approximatley 60% of effort was reported on CELRs, see Figure 11.9) is excluded from this estimation.

The 30-year dataset is useful for assessing the annual estimated deepwater trawl footprint as well as providing an estimated footprint of all trawl tows that have been reported on a tow-by-tow basis between 1990 and 2019. A combined footprint for all bottom-contacting trawl effort (based on tow-by-tow data) between 1990 and 2019 is shown in Figures 11.10.- (Baird & Mules 2021b). The intensity of effort (measured by the number of tows per 25-km^2 cell) for the combined inshore and deepwater fishstock data for 1990 and 2019 is shown in Figure 11.11. The most intensely contacted cells were close inshore and on some shelf edges. This 30-y period combined footprint contacted 11% of the seafloor area within New Zealand Territorial Sea and EEZ and 33% of the area shallower than 1600 m (fishable depths) and open to trawling (termed the fishable area).

The most comparable period to assess trends in the annual trawl footprint for all stocks combined is for 2008–2019 where deepwater fleet trawl contact is represented mainly by TCEPR and also TCER during 2008–2017 and mainly by the ERS data in 2018 and 2019, and the inshore component is represented by the TCER in all years except for a small amount of effort in 2018 and under 50% in 2019 when ERS data collection replaced TCERs. Over this 12-year period approximately 2% of the seafloor area within New Zealand Territorial Sea and EEZ and 6–7% of the fishable area was contacted by trawl gear each fishing year. The percentage coverage of the fishable area each year is about 3% for inshore fishstocks and 3–4% for deepwater fishstocks; note there is some overlap between the inshore and deepwater footprints (Baird & Mules 2021a, 2021b). The all stocks footprint decreased over the 12 years, with the lowest value estimated for 2019. These data reflect the decreasing amount of bottom-contacting trawl effort during these years; the numbers of tows in the TCEPR, TCER, and ERS data steadily dropped from 89 236 in 2010 to 66 039 in 2019. Over the12 years, the intensity of trawling within cells was steady despite the decrease in aggregate area, implying that the contact was more concentrated.

The deepwater footprint was updated in 2020 (Baird & Mules 2021b) for all years from 1989–90 (1990) up until 2018–19 (2019). Deepwater trawl fisheries have contacted 8.6% of the EEZ+TS and 25.3% of the fishable area between 1990 and 2019. Between 1990 and 2007 annual footprint estimates ranged from 1.2% and 2.0% of the EEZ+TS and 3.4% and 5.8% of the fishable area (peaks in 2002 and 2003); whereas, between 2008 and 2019, the annual footprint contacted 1.1–1.2% and 3.2–3.7% of the fishable area (lowest values in 2019). Across the 30-year time series for deepwater data, there was a steady increase in the footprint from under 50 000 km^2 in 1990 to a sustained period of contact during 1998 to 2003 (range 72 612 to 81 005 km^2), followed by a steady decrease to 43 481 km^2 in 2019, the lowest of the full 30-year time series, with declines seen for most deepwater targets and the swept area data reflecting the drop in effort (see Figure 11.12).

An updated trawl footprint for inshore fishstocks grouped by the inshore Fishery Management Areas, was completed for the period 2007–08 (2008) to 2018–19 (2019) (Baird & Mules 2021b). Inshore trawl fisheries have contacted 3.6% of the EEZ+TS and 10.7% of the fishable area between 2008 and 2019. The annual inshore footprint has decreased, from a peak of about 47 220 km^2 in 2010 to a nadir of 38 131 km^2 in 2019 (see Figure 11.12). This contact was equivalent to 0.9–1.1% of the EEZ+TS seafloor area, and 2.7–3.4% of the fishable area, with the lowest values from 2019.

Table 11.2: Attributes, usage, and resolution of spatial reporting required on Trawl Catch Effort and Processing Returns (TCEPRs), Trawl Catch and Effort Returns (TCERs), Catch Effort and Landing Returns (CELRs), and Electronic Reporting (ERS).

	Trawl catch and effort reporting forms			ERS
	CELR	TCEPR	TCER	
Year of introduction	1988–89	1989–90	2007–08	2017–
Vessels using	Trawlers not using TCER or TCEPR Shellfish dredgers	All trawlers >28 m Other vessels as directed Other vessels optional	All trawlers 6–28 m unless exempted	All trawlers >28 m. Phased in for all other vessels (from January–December 2019)
Trawl tow reporting	Daily summary, number of tows, gear, target, fishery area	Tow by tow, start and finish locations, speed, depth, target, gear, duration	Tow by tow, start location, speed, depth, target, gear, duration	Tow by tow, start and finish locations, speed, depth, target, gear, duration, lost gear, mitigation
Spatial resolution	Statistical reporting area (optionally lat/long)	1 minute (lat/long)	1 minute (lat/long)	4 decimal places of a degree (lat/long)

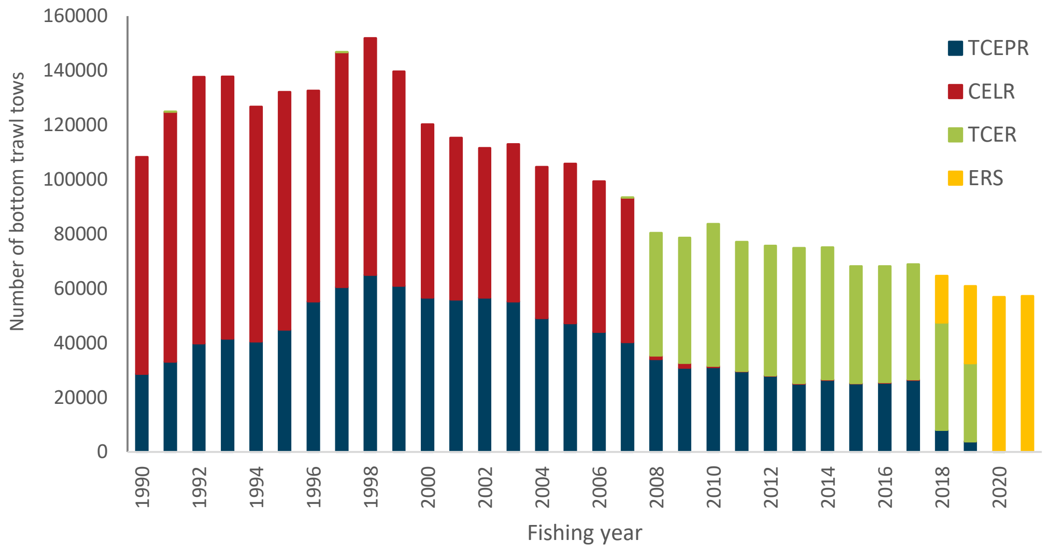


Figure 11.9: Total number of reported bottom trawl tows by fishing year and reporting form. Note: Effort reported on CELR forms is not included in the trawl footpint estimation.

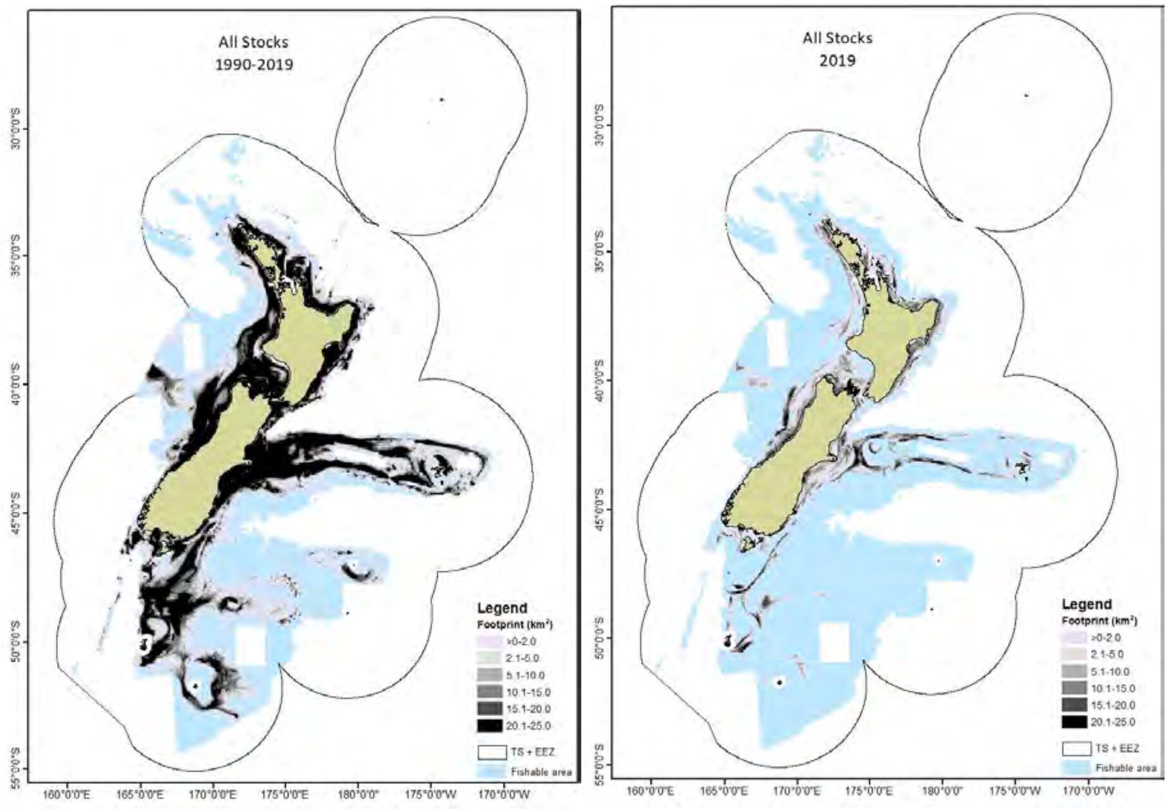


Figure 11.10: Distribution of the TCEPR and TCER all stocks footprint represented by 25-km² cells, 1990–2019 and 2019.

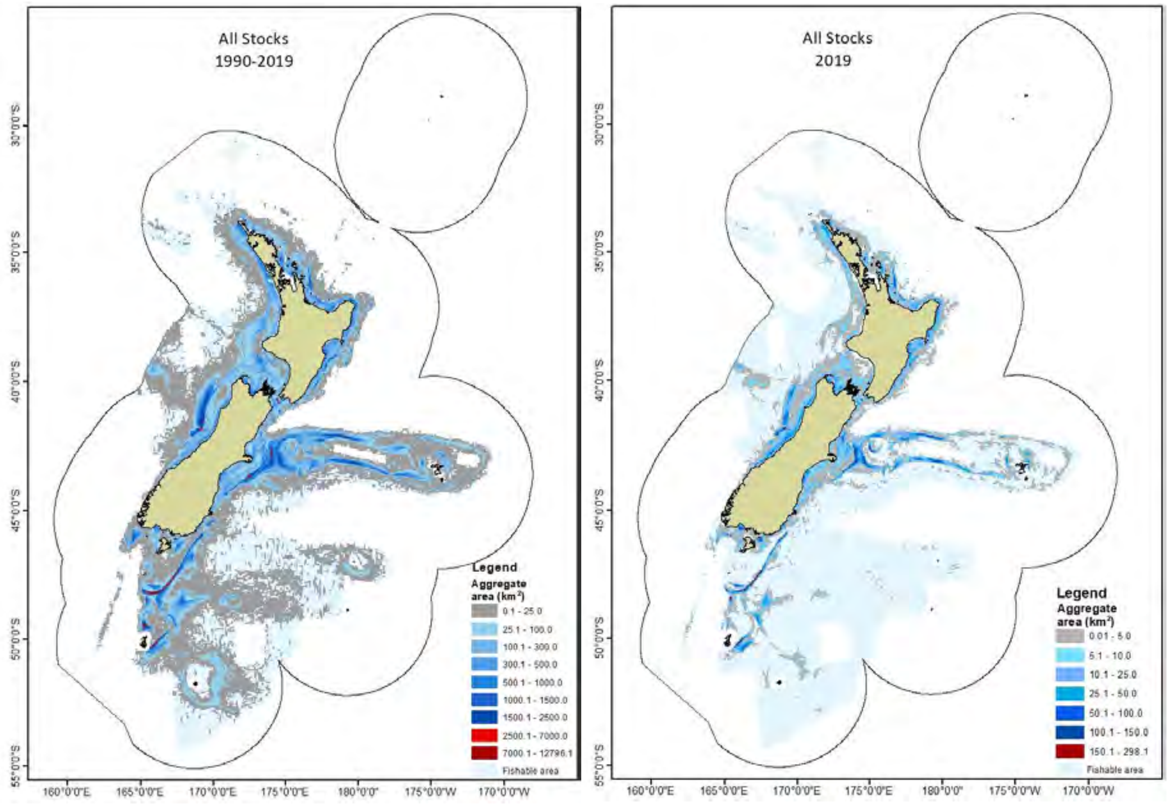


Figure 11.11: Distribution of the TCEPR and TCER all stocks aggregate area per 25-km² cell, during the combined fishing years 1990–2019 (left) and for 2019 (right).

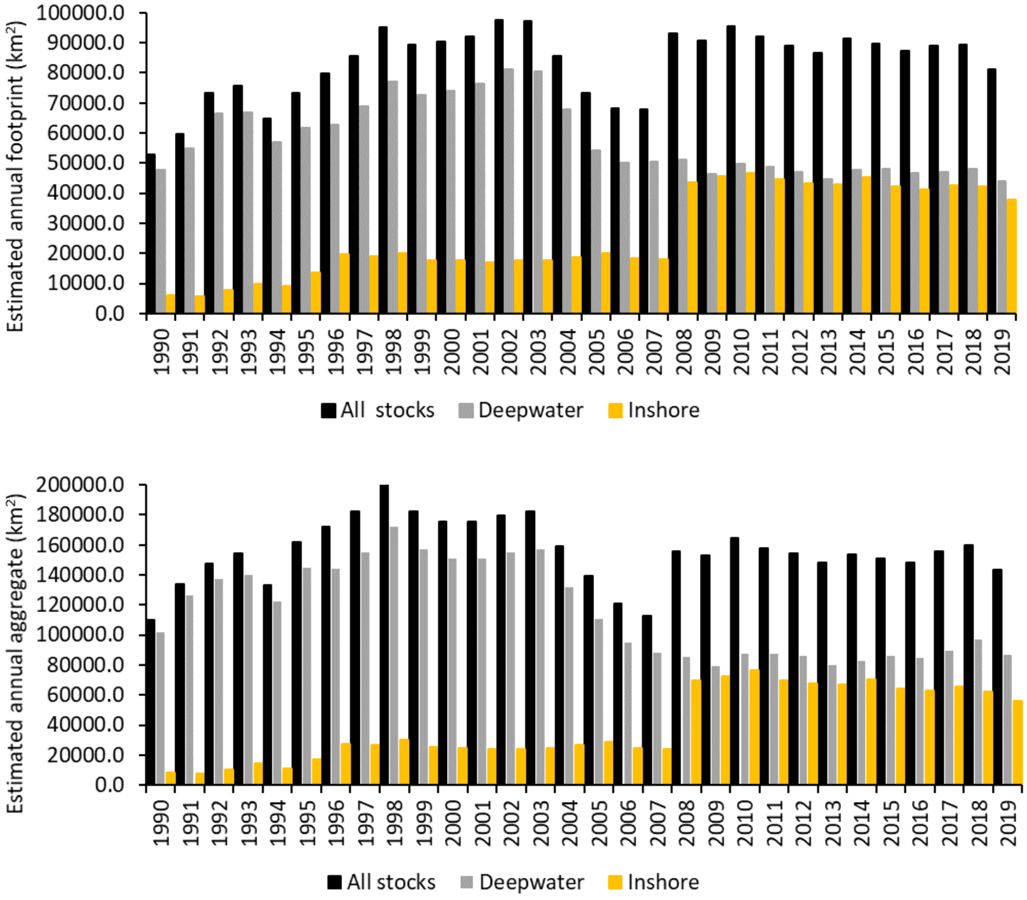


Figure 11.12: Annual estimated footprint and aggregate area for All stocks, Deepwater stocks, and Inshore stocks, 1990–2019. The data represent TCEPR (1990–2019), TCER (2008–19), and ERS (2018–2019) bottom-contacting effort. Note: TCER data collection was introduced in the 2008 fishing year and ERS data collection started in the 2018 fishing year. Effort reported on CELR forms is not included in the trawl footpint estimation. Thus, the comparable years for the all stocks footprint are for fishing years 2008 to 2019.

Dredging for shellfish (oysters and scallops) is conducted in a number of specific areas that have separate, smaller statistical reporting areas (Figure 11.13). Over the 30-year dataset, there were approximately 2 million scallop dredge tows in the four main scallop fisheries and over 1 million oyster dredge tows in the two dredge oyster fisheries. Up until 2019 data were collected on CELRs, usually at the spatial scale of a scallop or oyster fishery area and the data were summarised as the number of dredge tows. No estimates of the area swept by these dredges have been made, but the number of reported tows has declined markedly since the early 1990s with several of these fisheries being subject to multiple closures (Figure 11.14, Baird & Mules 2021b).

During the 2019 fishing year, ERS reporting was introduced in the Foveoux Strait oyster fishery (OYU 5) throughout the season. Effort was previously reported on CELR forms which collect daily effort data in the fishery-specific areas. The ERS data are collected on a 1-nm grid that overlays the fishery-specific areas (Figure 11.13), Effort data are collected at the level of a cell which effectively provides a cell-based number of tows, thereby allowing for a finer resolution for analysis than is provided by the larger fishery-specific areas. Baird & Mules (2021b) used the ERS data to develop a preliminary swept area estimation.

Our knowledge of the distribution of mobile bottom fishing effort within our TS and EEZ is, by international standards, very good; since 2007–08 we have had tow-by-tow reporting of almost all trawling with a spatial precision of about 1 n. mile. The distribution of dredge tows for shellfish has not been reported with such high precision for this period, but records kept by fishers in industry logbooks are often much more detailed than the Fisheries New Zealand statutory returns, and have sometimes been used to support spatial analyses that would not have been possible using the statutory returns (e.g., Tuck et al. 2006 for project

ZBD2005/15 on the Coromandel scallop fishery and Michael et al. 2006 for project ZBD2005/04 on the Foveaux Strait oyster fishery). The introduction of ERS in the Foveaux Strait oyster fishery in recent years has provided a means to report effort data at a finer spatial resolution and has made dredge footprint estimation possible (Baird & Mules 2021b). These studies indicate the value of records with higher spatial precision.

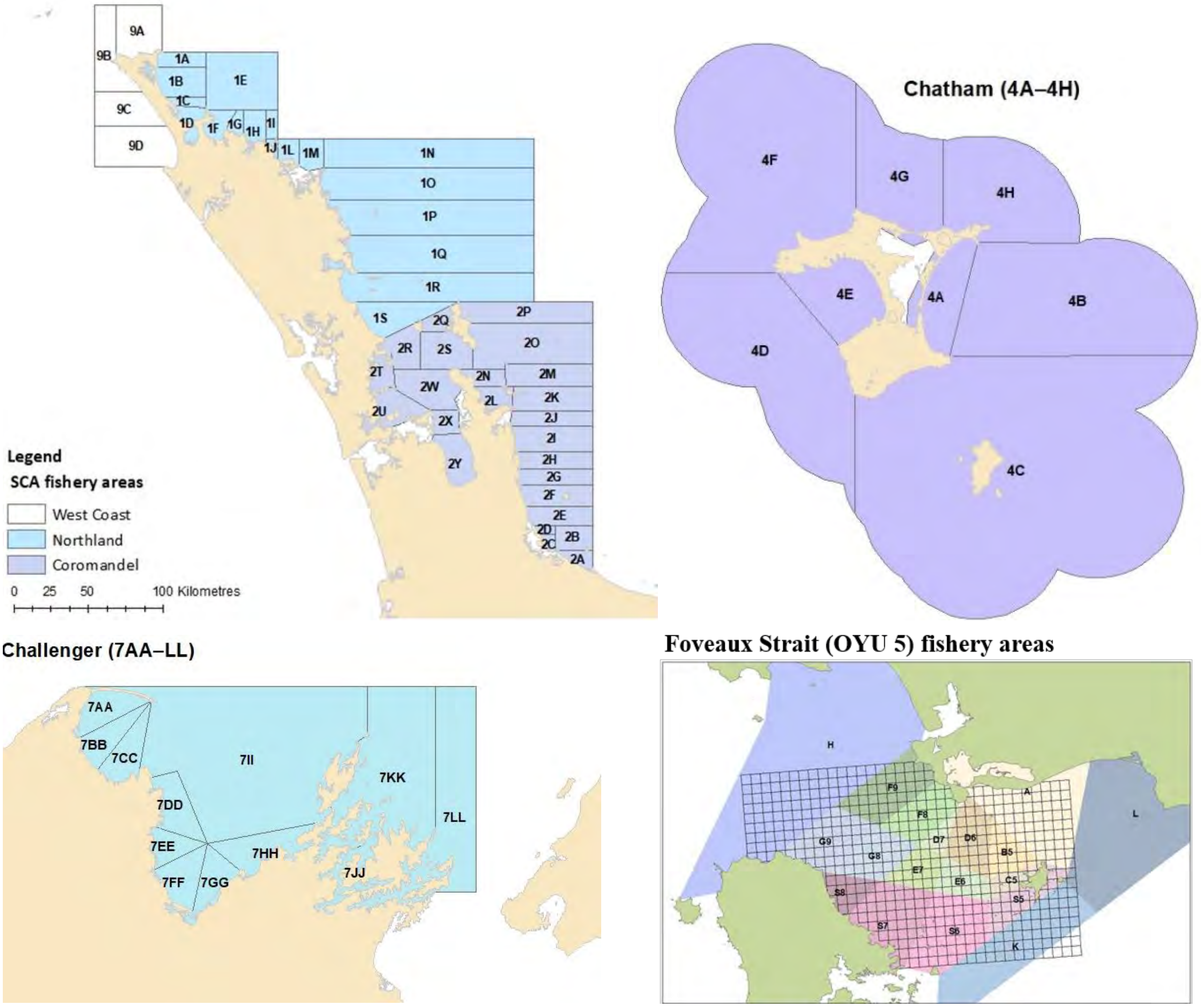


Figure 11.13: Maps taken from Baird & Mules (2021b) of statistical reporting areas for scallop and oyster fishery areas (scales differ). Note that these reporting areas are generally much smaller than the General Statistical Areas used for finfish reporting.

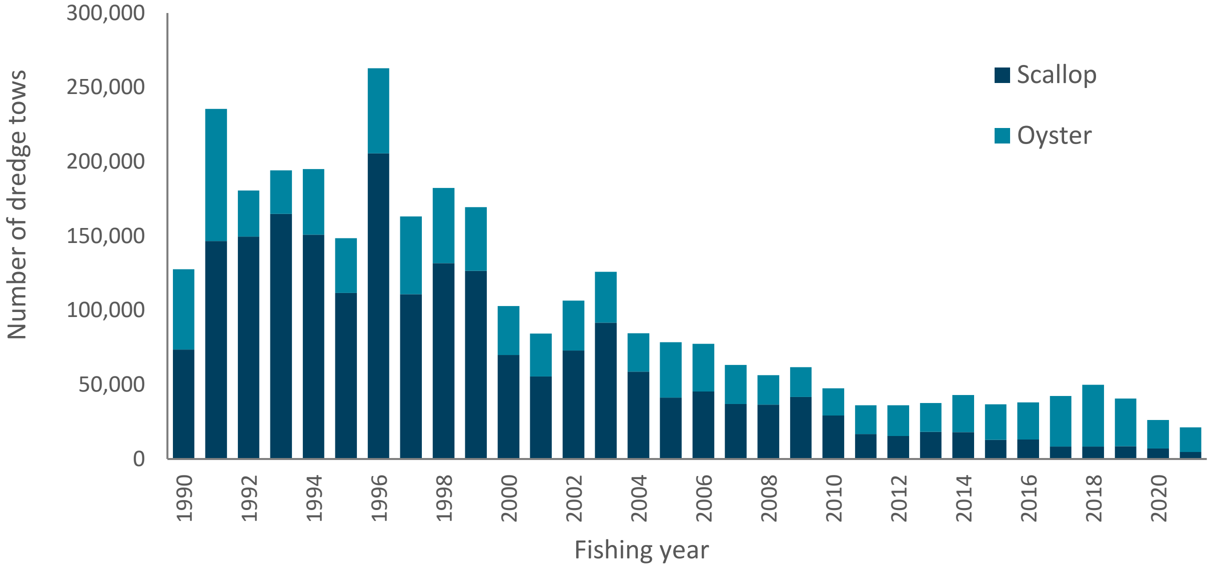


Figure 11.14: The number of dredge tows for scallop or oysters reported on Catch Effort and Landing Returns (CELR) for the 1989–90 (1990) to 2018–19 (2019) fishing years and via electronic reporting (ERS) from 2018-19 (2019) to 2020-21 (2021). [Note there was a transition to ERS during 2019]

11.2.3 OVERLAP OF FISHING AND PREDICTED HABITAT CLASSES

Tuck et al. (2014) reviewed a wide range of ecosystem indicators for deepwater fisheries and concluded that in relation to benthic impact of fishing, indices of fishing footprint and fishing intensity by habitat and gear or fishery were likely to be the most useful.

Baird & Wood (2012) provided the first assessment of the overlap of BOMEC classes and bottom trawling covering the period 1989-90 to 2004-05. Baird & Mules (2021a, 2021b) overlaid the 2007–08 to 2017–18 deepwater and inshore footprints on the 15-class BOMEC to estimate the proportion of each class that had been trawled (and reported on TCERs, TCEPRs, and ERS) in the 11 year period. They found that the size of each fishery footprint (deepwater and inshore, separately) and the proportion of each class trawled varied substantially between habitat class and fishery (Table 11.3). High percentages of the coastal BOMEC class areas are contacted by the inshore footprint; whereas, class O (the largest BOMEC class) has almost no reported fishing effort because it is mainly beyond trawlable depths. Conversely, class I is one of the smaller classes but has a larger trawl footprint (mainly deepwater) that overlaps 74% of the total class area. Two contrasting classes, together with their deepwater trawl footprints, are shown in Figure 11.15, based on analysis up to 2015–16 (Baird & Wood 2018).

Table 11.3: Estimated area of each BOMEC class (in depths of under 3000 m), the seafloor area of waters open to bottom trawling within each BOMEC class (in depths of under 1600m), the trawl footprint from TCER, TCEPR, and ERS deepwater and inshore fishstocks over the fishing years 2007–08 to 2017–18 (Baird & Mules 2021a), and the percentage overlap by the deepwater and inshore footprints. Note: previous versions have taken a longer prior time window (from 1989–90) for deepwater fisheries. Note there will be some overlap between the inshore and deepwater footprints in some inshore BOMEC classes.

BOMEC class	Total area (km^2)	Area open to bottom-fishing (km^2)	Deepwater footprint area (km^2)	Deepwater footprint area (% of total)	Deepwater footprint area (% area open to bottom fishing)	Inshore footprint area (km^2)	Inshore footprint area (% of total)	Inshore footprint area (% area open to bottom fishing)
A	27 557	19 764	493	1.8	2.5	16 109	56.6	81.5
B	12 420	11 984	3 359	27.0	28.0	10 814	89.0	90.2
C	89 710	87 914	22 419	25.0	25.5	51 256	58.1	58.3
D	27 268	25 786	1 983	7.3	7.7	20 443	74.1	79.3
E	60 990	60 211	12 656	20.8	21.0	14 037	22.7	23.3
F	38 608	30 931	3 607	9.3	11.7	0	–	–
G	6 342	6 033	2 475	39.0	41.0	4 067	69.2	67.4
H	138 550	129 323	38 856	28.0	30.0	19 218	14.0	14.9
I	52 224	51 910	26 946	51.6	51.9	676	1.3	1.3
J	311 361	277 138	32 234	10.4	11.6	4 756	1.5	1.7
K	1 290	1 290	0	–	–	36	2.8	2.8
L	198 577	175 509	23 950	12.1	13.6	6	<0.1	0.0
M	233 825	183 402	5 099	2.2	2.8	264	0.1	0.1
N	493 034	388 647	3 003	0.6	0.8	1 047	0.2	0.3
O	935 315	598 607	184	0.0	0.0	13	<0.1	0.0
Total	2 627 073	2 048 448	176 954	6.7	8.6	142 744	5.5	7.0

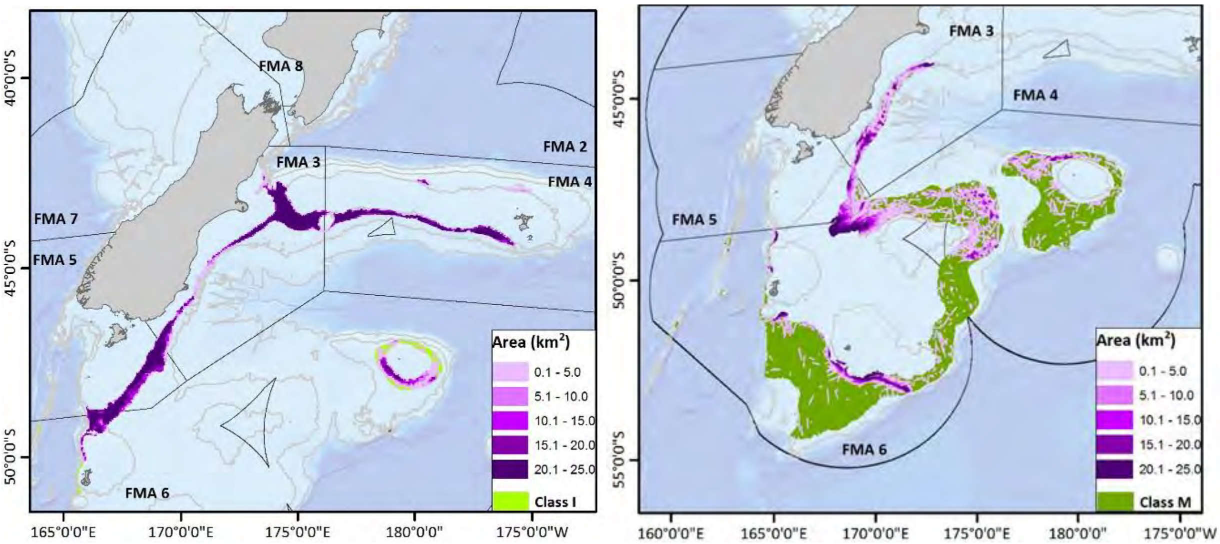


Figure 11.15: Maps created from Baird & Wood (2018) data showing BOMEC classes I (left) and M (right) overlaid with the footprint of deepwater Tier 1 and Tier 2 fishstock trawls on or near the seafloor reported on TCER and TCEPR forms between 1989-90 and 2015-16 for each 25-km^2 cell. Grey contour lines indicate depths of 500 m, 1000 m, and 1500 m.

11.2.4 STUDIES OF THE EFFECTS OF MOBILE BOTTOM FISHING METHODS IN NEW ZEALAND

The widespread nature of bottom trawling suggests that fishing is the main anthropogenic disturbance agent to the seabed throughout most of New Zealand's EEZ. Wind waves are certainly very widespread, but both field studies and modelling (Green et al. 1995) suggest that erosion of the seabed deeper than 50 m by waves occurs only very rarely in the New Zealand EEZ. Despite their widespread distribution at the surface, therefore, wind waves are not a dominant feature of the long-term disturbance regime throughout most of the EEZ. In some places, especially in the coastal zone and in areas close to headlands, straits, or islands, currents and tides may dominate the natural disturbance regime and a community adapted to this type of disturbance will have developed. However, over most of the EEZ between about 100 and 1000 m depths, especially in areas where there are few strong currents, fishing is probably the major broad-scale disturbance agent.

Several studies have been conducted since 1995 in New Zealand, focusing on the effects of various dredge and trawl fishing methods on a variety of different habitats in several geographical locations (Table 11.4). Despite the diversity of these studies, and their different depths, locations, and habitat types, the results are consistent with the global literature on the effects of mobile bottom fishing gear on benthic communities. Generally, there are decreases in the density and diversity of benthic communities and, especially, the density of large, structure-forming epifauna, and long-lived organisms along gradients of increasing fishing intensity. Large, emergent epifauna such as sponges and framework-forming corals that provide structured habitat for other fauna are particularly noted as being susceptible to disturbance by mobile bottom fishing methods (Cranfield et al. 1999, 2001, 2003, Cryer et al. 2000), especially on hard (non-sedimentary) seabeds (Clark & Rowden 2009, Clark et al. 2010a, 2010b, Williams et al. 2011). Even though large emergent fauna seem most susceptible, effects have also been shown in the sandy or silty sedimentary systems usually considered to be most resistant to disturbance (Thrush et al. 1995, 1998, Cryer et al. 2002). Also reflecting the international literature is a substantial variation in the extent to which individual New Zealand studies have shown clear effects. For instance, in Foveaux Strait, Cranfield et al. (1999, 2001, 2003) inferred substantial changes in the benthic system caused by over 130 years of oyster dredging, but Michael et al. (2006) did not support such conclusions in the same system. Subsequent review of these studies found much common ground but no overall consensus on the long-term effects of dredging on the benthic community of the strait.

These studies have focused predominantly on changes in patterns in biodiversity associated with trawling and/or dredging and less work has been done to assess changes in ecological process or to estimate the rate of recovery from fishing. Projects that have started on recovery rates are focused on relatively few habitats and primarily those that are known to be sensitive to physical disturbance, including by trawling or dredging (e.g., seamounts, project ENV2005/16, and areas of high current and natural biogenic structure, projects ENV9805, ENV2005/23, and BEN2009/02). Thus, the understanding of the consequences of fishing (or of ceasing to fish) for sustainability, biodiversity, ecological integrity and resilience, and fish stock productivity in the wide variety of New Zealand's benthic habitats remains incomplete. Reducing this uncertainty would allow the testing of the utility and likely long-term productivity of a variety of management strategies, and enable a move towards a regime that maximises value to the nation consistent with the MPI 'Our Strategy' document (http://www.mpi.govt.nz/about-mpi/our-strategy).

An expert-based assessment of 65 threats to 62 marine habitats from saltmarsh to the abyss (MacDiarmid et al. 2012) concluded that only 7 of the 20 most important threats to New Zealand marine habitats were directly related to human activities within the marine environment. The most important of these was bottom trawling (ranked third-equal most important), but invasive species, coastal engineering, and aquaculture were also ranked highly. However, the two top threats, five of the top six threats, and over half of the 26 top threats stemmed largely or completely from human activities external to the marine environment (the most important being ocean acidification, rising sea temperatures, and sedimentation resulting from changes in land use). The assessment suggested that the number and severity of threats to marine habitats declines with depth, particularly deeper than about 50 m. Shallow coastal habitats face up to 52 non-trivial threats whereas most deepwater habitats are threatened by fewer than five. Coastal and estuarine reef, sand, and mud habitats were considered to be the most threatened habitats whereas slope and deepwater habitats were among the least threatened.

Table 11.4: Summary of studies of the effects of bottom trawling and dredging in New Zealand waters (not updated). [Continued on next page]

Location	Approach	Key findings	References
Mercury Islands sandy sediments. Scallop dredge	Experimental	Density of common macrofauna at both sites decreased as a result of dredging at two contrasting sites; some populations were still significantly different from reference plots after three months.	Thrush et al. 1995
Hauraki Gulf various soft sediments. Bottom trawl and scallop dredge.	Observational, gradient analysis	Decreases in the density of echinoderms, longlived taxa, epifauna, especially large species, the total number of species and individuals, and the Shannon-Weiner diversity index with increasing fishing pressure (including trawl and scallop dredge). Increases in the density of deposit feeders, small opportunists, and the ratio of small to large heart urchins.	Thrush et al. 1998
Bay of Plenty continental slope. Scampi and other bottom trawls.	Observational, multiple gradient analyses	Depth and historical fishing activity (especially for scampi) at a site were the key drivers of community structure for large epifauna. The Shannon-Weiner diversity index generally decreased with increasing fishing activity and increased with depth. Many species were negatively correlated with fishing activity; fewer were positively correlated (including the target species, scampi).	Cryer et al. 1999 Cryer et al. 2002
Foveaux Strait, sedimentary and biogenic reef. Oyster dredge.	Observational, various	Interpretations of the authors differ. Cranfield et al.'s papers concluded that dredging biogenic reefs for their oysters damages their structure, removes epifauna, and exposes associated sediments to resuspension such that, by 1998, none of the original bryozoan reefs remained. Michael et al. concluded that there are no experimental estimates of the effect of dredging in the strait or on the cumulative effects of fishing or regeneration, that environmental drivers should be included in any assessment, and that the previous conclusions cannot be supported. The authors agree that biogenic bycatch in the fishery has declined over time in regularly fished areas, that there may have been a reduction in biogenic reefs in the strait since the 1970s, and that simple biogenic reefs appear able to regenerate in areas that are no longer fished (dominated by byssally attached mussels or reef-building bryozoans). There is no consensus that reefs in Foveaux Strait were (or were not) extensive or dominated by the bryozoan *Cinctopora*.	Cranfield et al. 1999, 2001, 2003 Michael et al. 2006
Spirits Bay, sedimentary and biogenic areas. Scallop dredge.	Observational, gradient analysis	In 1999, depth was found to be the most important explanatory variable for benthic community composition but a coarse index of dredge fishing intensity was more important than substrate type for many taxonomic groups. Sponges seemed most affected by scallop dredging, and samples taken in an area once rich in sponges had few species in 1999. This area had probably been intensively dredged for scallops. Analysis of historical samples of scallop survey bycatch showed a marked decline in sponge species richness between 1996 and 1998. In 2006, significant differences were identified between areas within which fishing was or was not allowed. Species contributing to these differences included those identified as being most vulnerable to the effects of fishing. These differences could not be attributed specifically to fishing because of interactions with environmental gradients and uncertainty over the history of fishing. No significant change between 1999 and 2006 was identified. In 2010, analysis of both epifaunal and infaunal community data identified change since 2006, and significant depth, habitat, and fishing effects. The combined fishing effects accounted for 15–30% of the total variance (about half of the explained variance). Individual species responses to fishing were examined, and those identified as most sensitive to fishing in this analysis had previously been categorised as sensitive on the basis of life history characteristics within the 2006 study.	Cryer et al. 2000 Tuck et al. 2010 Tuck & Hewitt 2013
Tasman Bay and Golden Bay. Bottom	Observational, gradient analysis	A gradient analysis was adopted to investigate the importance of the different factors affecting epifaunal and infaunal communities in Tasman Bay and Golden Bay. Fishing was consistently identified as an	Tuck et al. 2017

Location	Approach	Key findings	References
trawl, scallop and oyster dredge		important factor in explaining variance in community structure, with recent trawl and scallop effort being more important than other fishing terms. Important environmental variables included maximum current speed, maximum wave height, depth, % mud, and salinity. Fishing accounted for 31–50% of the explained variance in epifaunal and infaunal community composition, species richness, and Shannon-Weiner diversity. Overall, models explained 30–54% of variance, and additional spatial patterns identified in the analysis explained a further 5–16% of variance.	
South Canterbury Bight. Bottom trawl	Observational, gradient analysis	A gradient analysis was adopted to investigate the importance of the different factors affecting epifaunal and infaunal communities in the South Canterbury Bight. Both fishing effort and environmental variables were identified as being important in explaining the patterns in the community data observed, although fishing effort accounted for only a relatively small component of the overall variance (5 – 9%). The important environmental variables for both infaunal and epifaunal community analysis, included sediment grain size and organic carbon parameters, wave height parameters, chlorophyll-*a*, and distance from earthquake epicentres (although it must be remembered that this parameter was correlated with distance north). In addition to the weak (but mostly significant) fishing effects detected in relation to species based community and univariate measures, functional trait effects were also detected, with the predicted factor ceiling response identified for long-lived, sedentary, habitat-forming species, and a significant negative effect of fishing identified on this functional trait group.	Tuck et al. 2017
Graveyard complex 'seamounts', northern Chatham Rise. Orange roughy bottom trawl.	Observational, multiple analyses	From surveys in 2001 and 2006, substrate diversity and the amount of intact coral matrix were lower on fished seamounts. Conversely, the proportions of bedrock and coral rubble were higher. No change in the megafaunal assemblage consistent with recovery over 5–10 years on seamounts where trawling had ceased. Some taxa had significantly higher abundance in later surveys. This may be because of their resistance to the direct effects of trawling, their protection in natural refuges, or because these taxa represent the earliest stages of seamount recolonisation.	Clark & Rowden 2009, Clark et al. 2010a, 2010b Williams et al. 2010

11.2.5 STUDIES OF RECOVERY OF BENTHIC HABITATS IN NEW ZEALAND

Benthic community data from surveys conducted in 2006, 2010, and 2017 were analysed to investigate the effects of fishing in the Spirits Bay area, and recovery following the closure of areas within this region (under projects BEN2014-03 and ZBD2017-05). Multivariate and univariate analyses of epifaunal and infaunal community data from the Spirits Bay area consistently identified year, habitat, and depth effects, but scallop and trawl fishing were also retained in minimum adequate models (accounting for a median level of 20% of the total variance, and up to 50% of the explained variance), with effects still detectable 7–9 years after fishing in some analyses. The effects detected were independent of similarity measure, analysis approach, or data set used, and the effects of fishing were weaker in analyses of more recent survey data, where recent fishing effort was lower. Species sensitivities, categorised on the basis of morphology and life history characteristics, were consistent with species responses to fishing terms within the modelled analysis, and most of the most sensitive species were only found in areas with no recent fishing history.

Project BEN2014-02 provided additional support to work started under FRST-funded research on underwater topographic features from 1999. This programme, and its MBIE successor ('Vulnerable Deep-Sea Communities' (CO1X0906)) developed a fishing impact recovery comparison based on repeated towed camera surveys on six of the Graveyard Knolls on the northern flank of Chatham Rise. These knolls cover conditions where trawling has ceased, where trawling is still active, or knolls which have been untrawled. Surveys were carried out in 2001, 2006, 2009, and 2015, with support from MFish, NIWA, the cross-departmental Oceans Survey 20/20 programme, and in 2015 from MPI under BEN2014-02. Results from this time series have recently been published (Clark et al. 2019).

Univariate community metrics of biodiversity (abundance, species richness, diversity) were almost always higher for untrawled Ghoul and Gothic knolls than the other four. Multivariate community analyses of each knoll at each time-step showed a similar pattern, with the untrawled Gothic and Ghoul knolls having similar levels and patterns of community structure at one end of the ordination space, the persistently heavily trawled Graveyard seamount at the other end along with the previously heavily trawled and now closed Morgue knoll, and intermittently trawled knolls lying in-between (Figure 11.16). This ordination matches the gradient in commercial fishing effort. Community structure on Graveyard knoll was more consistent than on the other knolls, with persistently lower faunal richness, possibly due to a regular 're-setting' of the community by disturbance from trawling.

The time series of surveys indicates low resilience of benthic communities on the knolls to the effects of bottom trawling. There is no evidence that benthic communities on Morgue knoll are recovering following its closure to fishing in 2001. Intact scleractinian coral 'reef' is or was a 'climax habitat' on the Graveyard Knolls, and levels of this habitat on Morgue knoll remain much lower than those on the untrawled knolls (Figure 11.17).

Another linkage with the MBIE Vulnerable Deep-Sea Communities' project is an ongoing evaluation (under NIWA funding) of the relative vulnerability of benthic communities in several deep-sea habitats (e.g., seamounts, canyons, continental slope, hydrothermal vents, seeps) and their risk from bottom trawling. The importance of fishing effort as a factor influencing benthic invertebrate community composition has been examined for epi-megafauna from towed camera data (Bowden et al. 2016) and epibenthic sled tows (Rowden et al. 2016), and from multicorer samples for macro-infauna and meiofauna (Leduc et al. 2016, Rosli et al. 2016).

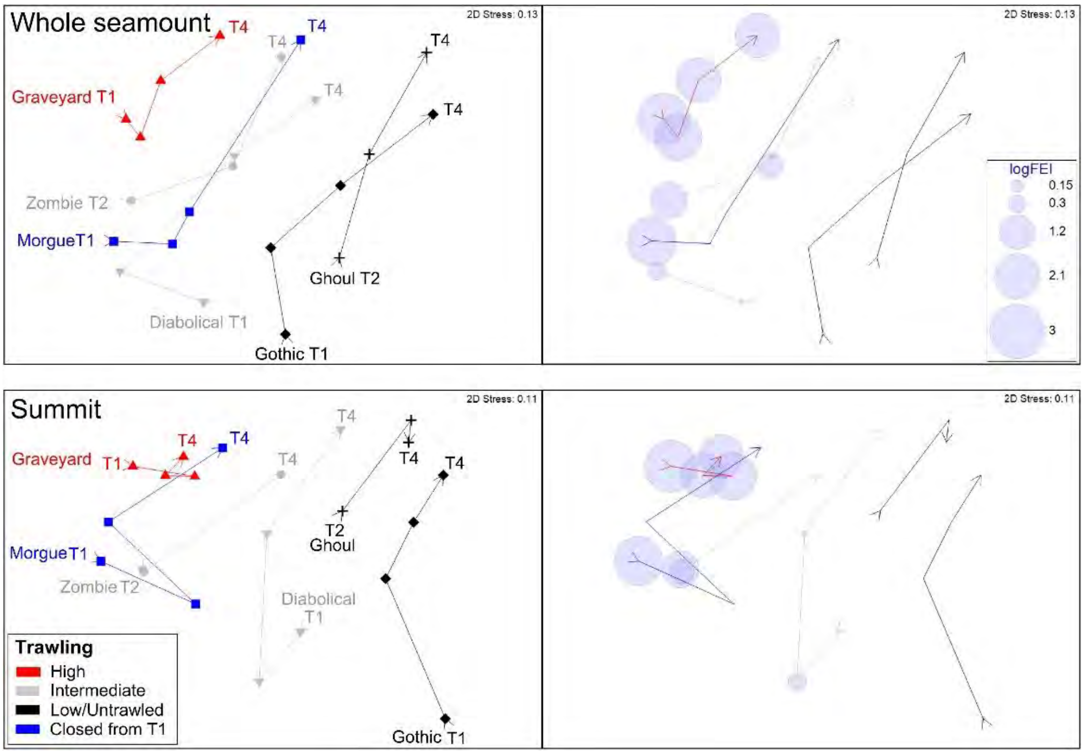


Figure 11.16: Ordinations (nMDS) illustrating benthic community similarities over time steps T1 to T4 and in relation to relative trawl history (see legend): Top panel, ordination using all images ('whole seamount'); bottom panel, only the summit sector. Right windows show trawling intensity as the Fishing Effects Index (FEI) superimposed as bubble plots for each seamount at each time step. [From Clark et al. (2019).]

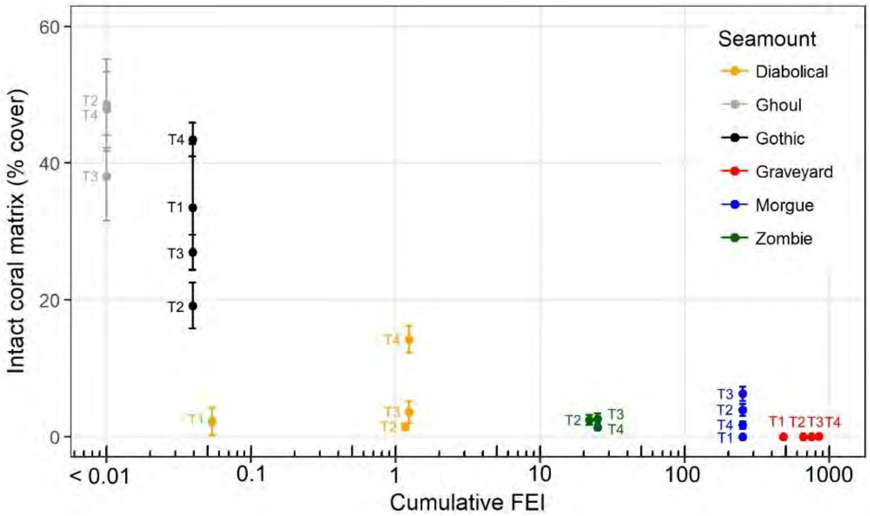


Figure 11.17: Intact coral matrix, as mean percent cover measured in individual seabed photographs, on six seamounts at each of four survey times (2001, 'T1'; 2006, 'T2'; 2009, 'T3'; and 2015, 'T4') in relation to cumulative trawling impact (Fishing Effects Index, FEI, all years up to and including the year of each survey). [From Clark et al. (2019).]

11.2.6 CURRENT RESEARCH

BEN2020-01 will provide spatial descriptions and annual monitoring of the seabed areas contacted by mobile bottom fishing gear (trawl and dredge) results for the 2007–08 to 2020–21 fishing years for all inshore fisheries and for the 1989–90 to 2020–21 fishing years for all deepwater fisheries. This will allow for the continued monitoring of trends in the extent and intensity of seabed contact by bottom fisheries in New Zealand. BEN2020-01 will also include the development of methods to estimate the spatial extent of seabed contact using Geospatial Position Reporting (GPR) data for the 2019–20 fishing year and a comparison of the estimate generated to that produced using fisher reported position data.

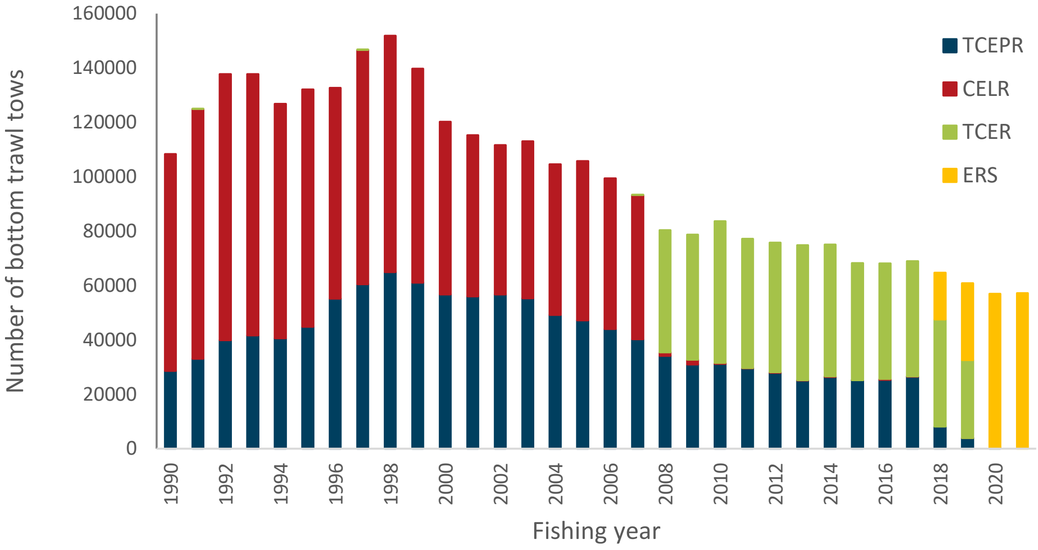
Project BEN2020-07 is nearing completion and will provide an update of all available information on UTFs in the New Zealand region and will assess the extent to which UTFs have been trawled annually by feature based fisheries within the New Zealand EEZ.

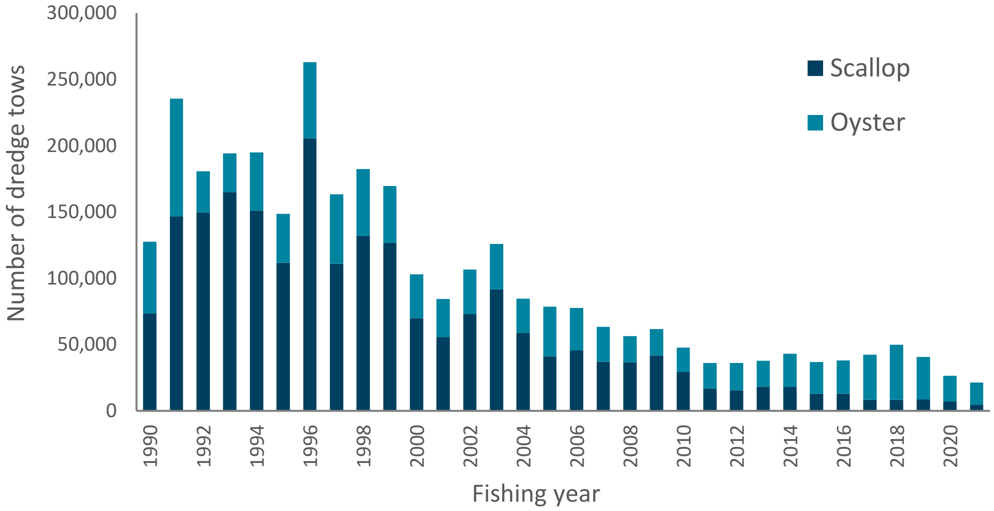
Various spatially explicit benthic impact approaches are being applied and evaluated in project BEN2019-04 to describe and quantify the likely nature and extent of impacts to benthic taxa or communities by mobile bottom fishing methods in New Zealand.

Two spatial planning projects are currently underway. Project BEN2019-05 was initiated in 2020 to develop the datasets and layers required for a spatial decision support tool to inform management of mobile bottom fishing in New Zealand's EEZ. Project ZBD2020-06 commenced in 2021 and seeks to assess spatial planning options to balance fishing activities (trawling and Danish seining) with the protection of benthic biodiversity and recovery of biogenic habitats within the Hauraki Gulf Marine Park. Both projects involved the collation and assessment of data by a research provider followed by a series of workshops with stakeholders to enable collaboration in the assessment of the spatial planning approach and outputs. The projects will rely on the use of the spatial decision support tool Zonation to test a range of scenarios.

11.3 INDICATORS AND TRENDS

<table>
<tr><td>Annual number of tows</td><td>2020–21 fishing year:
57 219 trawl tows
4 625 scallop dredge tows
16 642 oyster dredge tows</td></tr>
<tr><td>Trend in number of tows</td><td>Trawl and dredge effort stable or decreasing in recent years:





</td></tr>
</table>

Overlap of trawl footprint with BOMEC habitat classes for 1989-90 to 2018–19

BOMEC class	Area (km^2)	Footprint overlap (km^2)		% of total footprint		Footprint overlap (%)	
		1990–2019	2019	1990–2019	2019	1990–2019	2019
A	30 661.0	17 481.6	5 236.8	3.8	6.5	57.0	17.1
B	12 786.1	11 374.4	5 297.1	2.5	6.5	89.0	41.4
C	90 256.5	74 461.7	15 782.4	16.2	19.5	82.5	17.5
D	28 085.7	21 748.7	6 934.7	4.7	8.6	77.4	24.7
E	61 258.0	34 431.6	6 316.9	7.5	7.8	56.2	10.3
F	38 775.8	6 900.9	316.7	1.5	0.4	17.8	0.8
G	6 702.3	5 173.3	1 092.4	1.1	1.3	77.2	16.3
H	138 399.1	78 072.4	12 987.0	16.9	16.0	56.4	9.4
I	52 008.3	38 811.1	9 906.8	8.4	12.2	74.6	19.0
J	312 604.9	81 921.7	10 365.4	17.8	12.8	26.2	3.3
K	1 200.2	41.5	0.4	0.0	0.0	3.5	0.0
L	198 578.4	56 638.2	5 518.0	12.3	6.8	28.5	2.8
M	233 837.4	19 138.8	719.9	4.2	0.9	8.2	0.3
N	495 154.2	13 771.8	555.4	3.0	0.7	2.8	0.1
O	1 006 911.1	614.7	19.7	0.1	0.0	0.1	0.0
Total	2 707 219.0	460 627.2	81 054.9	100.0	100.0	17.0	3.0

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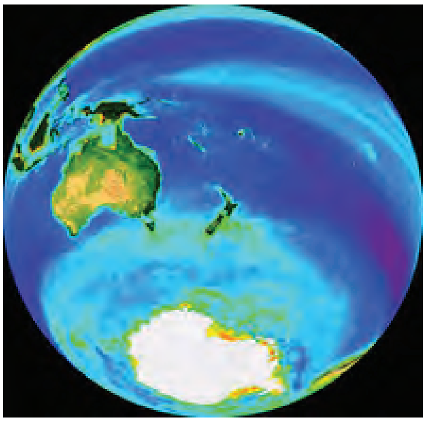
THEME 4: ECOSYSTEM EFFECTS

Chapter 12: NZ Climate and Oceanic Setting - Technical Summary

Ocean variability and change



Trends, extremes, decadal cycles

1. THE ISSUE IN BRIEF

- Climate and oceanographic variability, and long-term environmental changes, are of key relevance to the carrying capacity of the marine environment
- Understanding the trends and cycles observed in the ocean allows us to understand the links between observed fisheries patterns and drivers of biological processes
- It also allows for the exploration of likely future scenarios for New Zealand fisheries as climate change and global warming continue
- The cumulative effects of climate change and other anthropogenic stressors on the ocean (productivity, structure, and function) are likely to be significant, and increasingly seen in the next 20–30 years

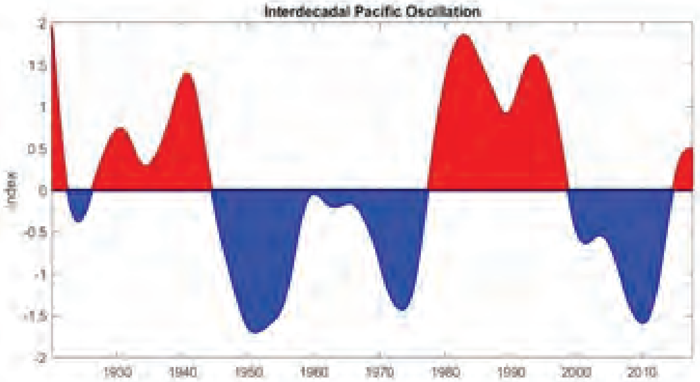
2. NEW ZEALAND'S SEAS ARE CHANGING

- The New Zealand Exclusive Economic Zone straddles ~30° of latitude and ~30° of longitude in the SW Pacific Ocean from the tropics to the Southern Ocean, with a large area, ~4 million square km, of jurisdiction
- Essentially, the direction of flow is from west to east across the Tasman Sea that has a relatively slow circulation, and then around northern and southern New Zealand to the more dynamic eastern side, bordering the Pacific Ocean
- Primary productivity is higher than most of Australasia, but lower than coastal upwelling systems around the rest of the world
- The greatest productivity is across the Chatham Rise associated with the Subtropical Front and mixing of water masses either side of the front
- Some long-term trends in the marine environment available at a national scale are incorporated in the Environmental Reporting system developed by MfE and Statistics NZ
- Like the rest of the world, our ocean is showing measurable effects of climate change and global warming. Sea temperatures are increasing, ocean acidification is increasing, storm frequencies are higher and more intense, and the knock-on effects to fish and biodiversity are evident in some areas. Extreme events such as marine heatwaves are likely to occur more often

3. INTER-DECADAL CYCLES SINCE 1900

- Positive and negative phases of the Interdecadal Pacific Oscillation index bring different ocean conditions to New Zealand waters. Under positive phases (red), La Niña conditions tend to prevail and reduced westerly flow results in less upwelling on the west coast, and warmer air and sea temperatures

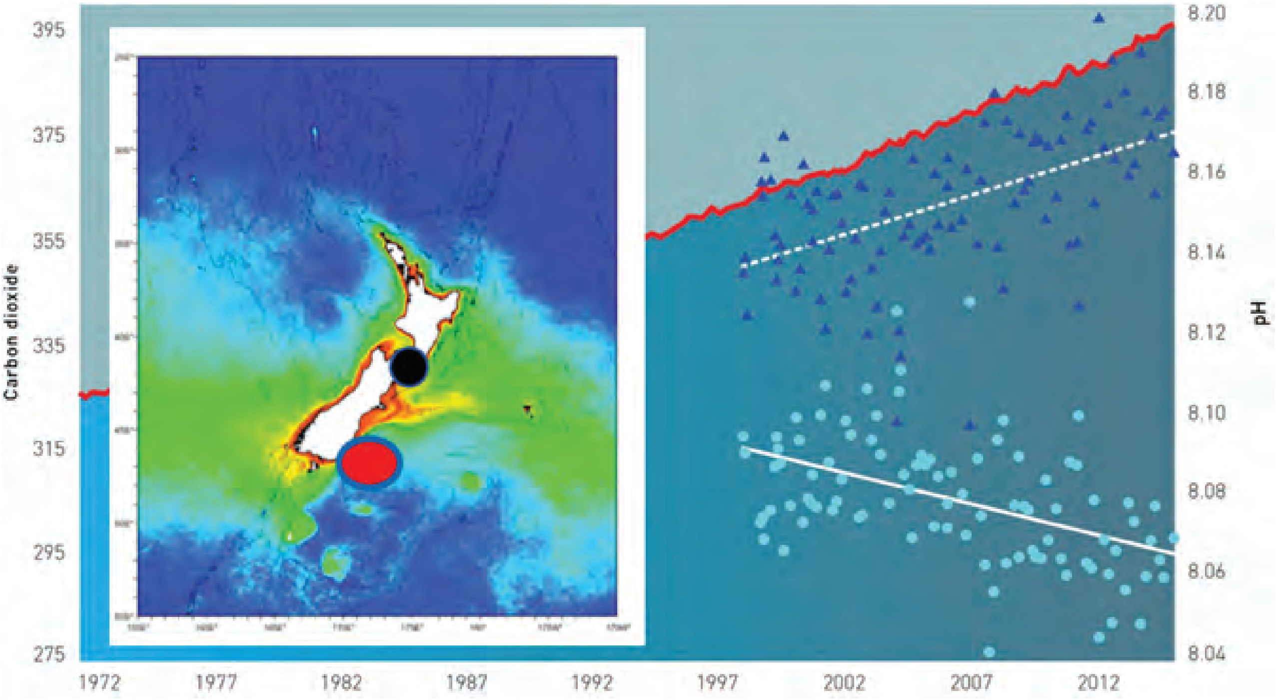
- Under negative IPO conditions (blue), El Niño conditions tend to prevail with increased westerly winds and higher upwelling. The abundance of some fish stocks reflects these cycles



4. CARBON DIOXIDE AND OCEAN ACIDIFICATION

- Levels of atmospheric CO_2, sea-surface dissolved CO_2, and pH in New Zealand offshore waters parallel similar trends reported in the northern hemisphere, i.e., increased acidification

- A large number of global studies has identified a range of direct and indirect effects of ocean acidification across a broad range of marine taxa, from bacteria to fish, and from coastal to deep sea ecosystems. Of particular concern in New Zealand are the potential effects on shellfish fisheries, aquaculture, and calcifying deep sea corals



Time series of atmospheric CO_2 (red line) at Baring Head (black dot), and surface water dissolved CO_2 (dark blue triangles and dashed trendline) and pH (light blue circles and solid trendline) near Otago (red dot)

- The highest projected regional warming occurs in the East Australian Current and in subantarctic waters to the south-east of NZ. Because of the present warming rate in the south-west Tasman Sea is similar to that measured for waters around NZ, this region may provide an analogue for future changes in NZ waters

- The depth of the surface mixed layer will decrease across much of the NZ EEZ area, except for some subantarctic water regions. The resulting increase in light exposure for plankton may be beneficial to productivity and food webs in subantarctic waters where nutrients are plentiful, but may be deleterious in warmer subtropical waters that are oligotrophic (i.e., low nutrient)

5. ONGOING RESEARCH

- Model projections for food supply to fisheries on the Chatham Rise are conflicting and need to be resolved. The Chatham Rise is identified as one of the more vulnerable areas in New Zealand, yet it may be one of the areas least affected by ocean acidification changes
- Risk assessments for fisheries under climate change scenarios are limited by short time series and limited water temperature data in particular. Methods of incorporating climate change indicators into stock assessment, and effects on species movements, are underway (see also Chapter 18)

12 NEW ZEALAND’S CLIMATE AND OCEANIC SETTING

Status of chapter	This chapter has been updated for AEBAR 2021.
Scope of chapter	This chapter provides context within which to consider interactions between the environment and the seafood sector. It provides an overview of primary productivity, oceanography, bentho-pelagic coupling, ocean acidification and oceanic climate trends in the Southwest Pacific region.
Area	New Zealand regional setting.
Focal localities	All New Zealand waters.
Key issues	• Climate and oceanographic variability and long-term changes are of relevance to resilience in fisheries and the broader marine environment. • Allows improved understanding of the links between observed patterns and drivers of biological processes. • Allows for testing of likely future climate scenarios. • New Zealand trends of increasing air and sea temperatures, as well as ocean acidification are consistent with observed global trends.
Emerging issues	• New Zealand’s oceanic climate is changing. • Causal mechanisms that link the dynamics of a variable marine environment to variations in biological productivity, particularly of fisheries and biodiversity, are not well understood in New Zealand or internationally, but are the subject of multiple studies. • Cumulative effects of ocean climate change and other anthropogenic stressors on aquatic ecosystems (productivity, structure and function) are likely to be high, and seen in the next 20–30 years. • Some long-term trends in the marine environment available at a national scale are incorporated in the new Environmental Reporting system being developed by MfE and Statistics New Zealand. • There is a growing recognition that stressors will act both individually and interactively, confounding predictions of the net effects of climate change. • Improved scenario setting and the need for risk evaluation. • The first regime shift in IPO since most fisheries monitoring began occurred in 2000, which is likely to result in fewer El Niño events for a 20–30 year period, which in turn is likely to impact fish productivity. • 2018 saw a very strong marine heat wave (MHW) in the Tasman Sea. MHWs would be expected to become more frequent in a warming world. • Sub-surface temperature data in fishable depths are a major gap in our current datasets. • Understanding how physical changes propagate through ecosystems and impact on fish populations is another gap.
Fisheries New Zealand research (current)	Nil
Government and other research	NIWA Coast & Oceans Centre; NIWA Climate and Atmosphere Centre; University of Otago-NIWA shelf carbonate geochemistry & bryozoans; Munida time-series transect; Physical Oceanography research; Geomarine Services-foraminiferal record of human impact; Regional Council monitoring programmes; Ministry for the Environment-Environmental reporting “Our marine Environment 2019” https://environment.govt.nz/publications/our-marine-environment-2019/; Department of Conservation the impacts of climate change on marine protected species. Relevant global climate programmes: Argo; Southern Ocean Observing System. CARIM (Coastal Acidification Rates and Implications to Management) and MOANA project (temperature data from fishing depths below the water surface). Dragonfly science: Fast-forward fish: resilience of exploited marine populations to a changing ocean.

Related chapters/issues	This chapter provides background environmental information relevant to all chapters, but particularly to chapters on Biodiversity; Trophic and ecosystem-level effects.

12.1 CONTEXT

This chapter was updated in August 2021 and summarises information on oceanography and ocean acidification around New Zealand, and climate trends in the Southwest Pacific region. This information provides context to understand the interactions between the environment and seafood productivity in the region. Climate and oceanographic conditions play an important role in driving the productivity of our oceans and the abundance and distribution of our fish stocks and fisheries. The most recent analyses of trends in climate and oceanographic variables relevant to fisheries management in New Zealand are given by Hurst et al. (2012), Dunn et al. (2009), and Cummings et al. (2021).

New Zealand is part of a large submerged continent (Figure 12.1). The Territorial Sea (TS, extending from mean low water shoreline to 12 nautical miles), Exclusive Economic Zone (EEZ, extending from 12 nautical miles to 200 nautical miles offshore), and the extended continental shelf (ECS) combine to produce one of the largest areas of marine jurisdiction in the world, an area of almost 6 million square kilometres (Figure 12.1). New Zealand waters straddle more than 25 degrees of latitude from warm, salty subtropical waters at 30° S to cooler, fresher subantarctic waters at 56° S, and 30 degrees of longitude from 161° E in the Tasman Sea to 171° W in the western Pacific Ocean. New Zealand's coastline, with its numerous embayments, is long, with estimates ranging from 15 000 to 18 000 km, depending on the measurement method (Gordon et al. 2010).

New Zealand lies across an active subduction zone in the western Pacific plate; tectonic activity and volcanism have resulted in diverse and varied seascapes within the EEZ. The undersea topography comprises a relatively narrow band of continental shelf down to 200 m water depth, extensive continental slope areas from 200 to 1000 m, extensive abyssal plains, submarine canyons and deep-sea trenches, ridge systems and numerous seamounts, and other underwater topographic features such as hills and knolls. There are three significant submarine plateaus: Challenger Plateau, Campbell Plateau, and Chatham Rise.

The physical oceanography of the deep seas around New Zealand has recently been reviewed by Chiswell et al. (2015). Measurements from platforms including satellites, drifting and profiling floats, moorings, and oceanographic voyages have provided a wealth of new observations over the last 30 years and analyses of these observations have substantially improved our understanding of the oceanography. Chiswell et al. (2015) summarise and integrate earlier research through a series of schematics of the ocean currents around New Zealand. Their surface currents are shown in Figure 12.2.

The Tasman Sea, west of New Zealand, is isolated from the South Pacific Gyre by the New Zealand landmass. The South Pacific Western Boundary Current, the East Australian Current (EAC), flows down the east coast of Australia before separating from the Australian land mass at about 31–32° S (e.g., Ridgway & Dunn 2003). Part of the separated flow crosses the Tasman Sea as the Tasman Front (Stanton 1981, Ridgway & Dunn 2003, Sutton & Bowen 2014) which has recently been reinterpreted as an eddy field because the variability dominates the mean flow in many places (Oke et al. 2019). The remaining flow continues south in the EAC extension. The Tasman Front then feeds the western boundary currents adjacent to New Zealand — the East Auckland Current (EAUC) between North Cape and East Cape and the East Cape Current (ECC) between East Cape and the northern side of Chatham Rise. Sutton & Bowen (2014) found that the Tasman Front is a weaker connection than previously thought between the EAC and EAUC, with the Tasman Front being shallower and transporting less water than the EAC and EAUC.

At the southern limit of the Tasman Sea is the Subtropical Front which separates warm, salty subtropical water to the north from cold, fresh subantarctic water to the south. The Subtropical Front passes south of Tasmania and approaches New Zealand at the latitude of Fiordland (Stanton & Ridgway 1988, Hamilton 2006). Around 165° E, the front diverts south across Macquarie Ridge where it has two clear branches (Smith et al. 2013), which continue onto Campbell Plateau where they merge to form the Southland Front along the Otago Coast (Chiswell 1996, Sutton 2003).

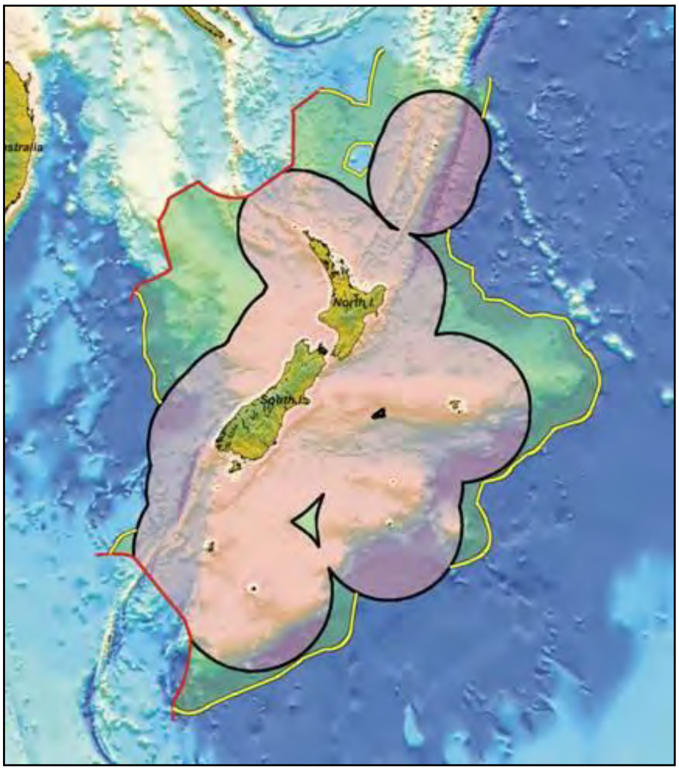


Figure 12.1: New Zealand land mass area 250 000 km^2; EEZ and territorial sea area (pink) 4 200 000 km^2; extended continental shelf extension area (light green) 1 700 000 km^2; total area of marine jurisdiction 5 900 000 km^2. The black line shows the boundary of the New Zealand EEZ, the yellow line indicates the extension to New Zealand's legal continental shelf (note: to date there is no agreement on the line definition with northern neighbours), and the red line the agreed Australia/New Zealand boundary under UNCLOS Article 76. Image courtesy of GNS.

The circulation in the central Tasman Sea, east of the influence of the EAC, and between the Tasman Front and Subtropical Front is thought to be relatively slow. Ridgway & Dunn (2003) showed eastward surface flow across the interior of the Tasman Sea sourced from the southernmost limit of the EAC, with the flow separating around Challenger Plateau and, ultimately, New Zealand. Reid's (1986) analysis indicates that a small anticlockwise gyre exists in the western Tasman Sea at 1000–2500 m depth. This gyre is centred at about 35° S, 155° E on the offshore side of the EAC and west of Challenger Plateau. All indications are that the eastern Tasman region overlying Challenger Plateau has weak flows.

In contrast, the east coast of both islands and Cook Strait have strong and variable currents. Along the north-east coast of the North Island there are two semi-permanent eddies that vary in size and strength, the North Cape and East Cape Eddies (Roemmich & Sutton 1998). The inshore sides of these eddies comprise the EAUC, which flows down the east coast of the North Island to East Cape. Most of the EAUC water continues south in the ECC, with the remainder being split between the East Cape Eddy and the Pacific Ocean. There are several eddies in the East Cape Current region, with the largest known as the Wairarapa Eddy. It sits between the North Island and the northern flanks of Chatham Rise (Chiswell 2005, Chiswell et al. 2015).

Along the south-eastern coast of the South Island there is a narrow band of warm, salty subtropical water on the shelf separated from offshore cold, fresh subantarctic water by the Southland Front (the local manifestation of the Subtropical Front). This front has an associated flow of mainly subantarctic water, called the Southland Current (Sutton 2003) which flows north to the southern flank of Chatham Rise. The Southland Current then turns east, flowing along the southern flank of Chatham Rise before turning south at the Chatham Islands and then east into the Pacific Ocean contributing to the South Pacific Current (Figure 12.2, Stramma et al. 1995).

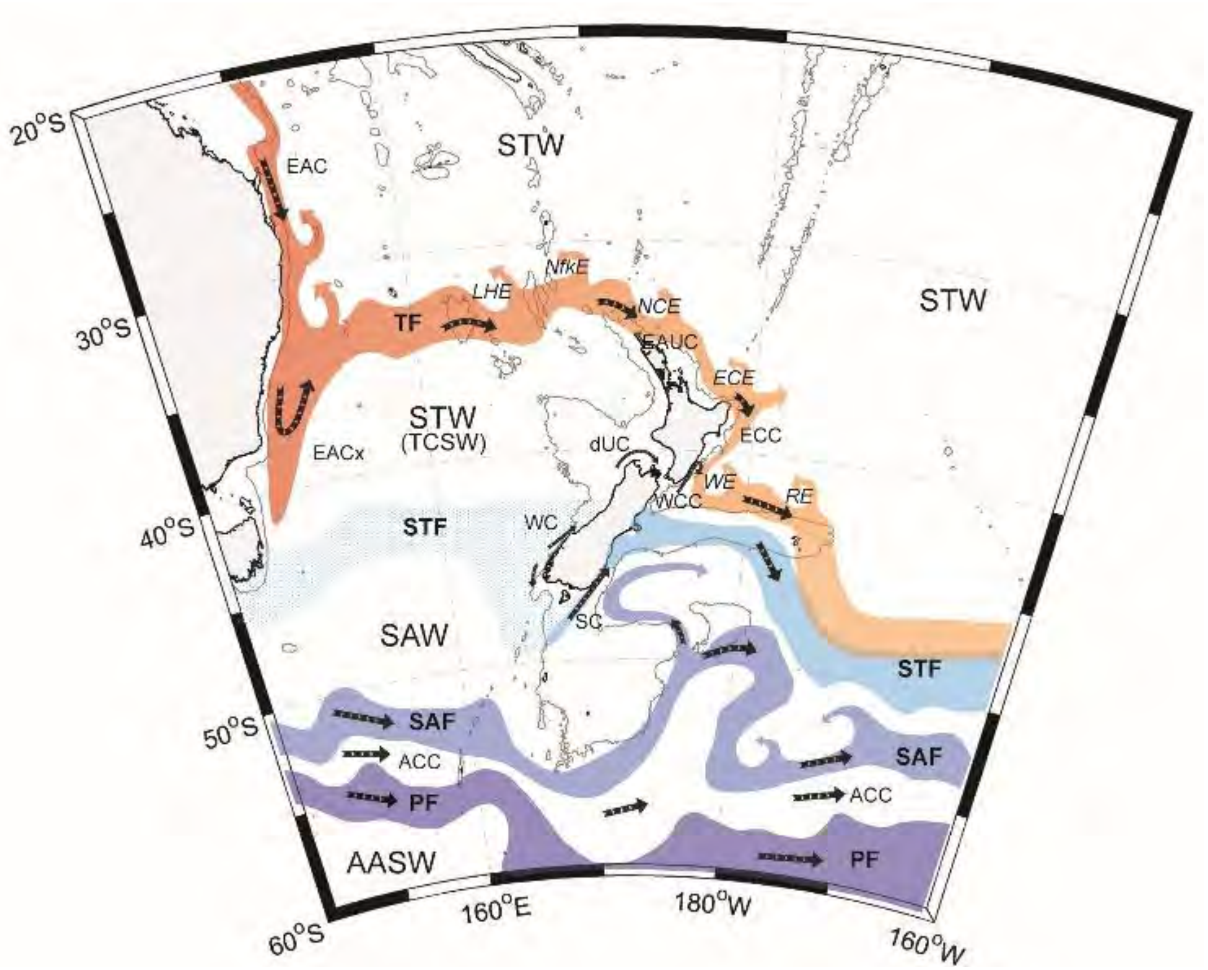


Figure 12.2: Schematic surface circulation around New Zealand based on drifter and hydrographic data. Regions of flow are shown as coloured streams. Colours reflect the temperature of the flows with red being warmest, and dark blue being coldest. The STF in the Tasman Sea is density compensated with little flow, as indicated by the shading. Water Masses are Subtropical Water (STW), Tasman Sea Central Water (TSCW), Subantarctic Water (SAW) and Antarctic Surface Water (AASW). Ocean fronts are Tasman Front (TF), Subtropical Front (STF), Subantarctic Front (SAF), and Polar Front (PF). Ocean currents are East Australia Current (EAC), East Australia Current extension (EACx), East Auckland Current (EAUC), East Cape Current (ECC), d'Urville Current (dUC), Wairarapa Coastal Current (WCC), Westland Current (WC), Southland Current (SC), and Antarctic Circumpolar Current (ACC). Eddies are Lord Howe Eddy (LHE), Norfolk Eddy (NfkE), North Cape Eddy (NCE), East Cape Eddy (ECE), Wairarapa Eddy (WE), and Rekohu Eddy (RE). Reproduced with permission from Chiswell et al. (2015).

Forcén-Vázquez (2015) showed that water from the Subtropical Front is found over a large fraction of Campbell Plateau and varies significantly from year to year. They also found that water from the Southern Ocean is mixed onto the plateau, giving it an oceanography distinct from the surrounding seas. These waters are well-mixed and are known to be iron limited (Boyd et al. 1999). It has been suggested that there are high transfer efficiencies between low and high trophic levels (Bradford-Grieve et al. 2003). Steering of currents along and around plateaus and ridges gives rise to higher ocean productivity than might be expected in the generally oligotrophic western Pacific Ocean (Figure 12.3).

The abundance and rate of growth of phytoplankton in the ocean is affected by the availability of macronutrients (such as nitrate, phosphate, silicate), micronutrients (such as iron), and light availability (which depends on cloudiness and water clarity). New Zealand phytoplankton abundance and net primary productivity levels are high compared with most of Australasia, but lower than most coastal upwelling systems around the world (Field et al. 1998; Murphy et al. 2001). Ocean colour satellites estimate phytoplankton abundance using the metric of the chlorophyll-a concentration in near-surface waters ('chl-a'). Satellite estimates of chl-a are used as a proxy for primary production by phytoplankton (i.e., the growth of phytoplankton creating organic matter). Satellite-based estimates of primary production require local validation which is ongoing (Pinkerton et al. 2019).

Chatham Rise has the highest chl-a levels in the New Zealand region, indicative of elevated productivity associated with the Subtropical Front and mixing between the different water masses (Figure 12.3). In the Subtropical Front region, macronutrient-limited subtropical waters (Bradford-Grieve et al. 1997) mix with high nutrient, low-chlorophyll (NHLC) iron-limited subantarctic waters (Boyd et al. 1999), resulting in increased primary productivity which is apparent in ocean colour remote sensing as elevated chl-a (Figure 12.4 left panel). Seamounts, seamount chains, and ridge structures can also provide additional localised areas of mixing and increased primary productivity.

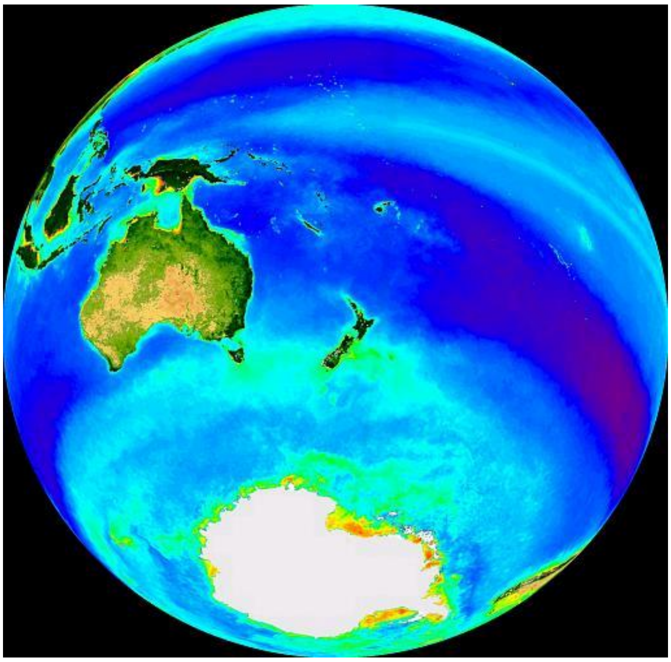


Figure 12.3: SeaWIFS image showing elevated chlorophyll a (green) near New Zealand. Image courtesy of NOAA.

Around the coast, turbulence, upwelling, resuspension of sediment, and land run-off also play key roles in determining primary productivity (Pinkerton et al. 2005, Chiswell et al. 2016). Care is needed in interpreting remotely-sensed ocean colour measurements in coastal waters because the presence of suspended sediment and dissolved coloured material can invalidate established methods for processing open-ocean satellite data. Novel approaches are now available for estimating chl-a in such ‘optically-complex’ coastal waters (e.g., Pinkerton et al. 2018), and so regionally-tuned coastal analyses are presented here separately from the ocean-scale analysis. Ocean and coastal processing steams will be merged in the future. Satellite observations show that chl-a in New Zealand coastal areas is elevated in areas where nutrients (especially nitrate) are brought into the coastal zone in rivers, and where upwelling of nutrient-rich deep water occurs because of wind and current patterns (Figure 12.4, right panel).

Elevated chl-a and primary production over the New Zealand shelf and slope underpin commercial shellfish and finfish fisheries from the shoreline to depths of about 1500 m. In the deeper ocean, primary productivity supports commercial deepwater fisheries (Aiken et al. 2004, Chassot et al. 2010). Primary productivity in surface waters also leads to a transfer of organic matter to the sea floor (e.g., Lutz et al. 2007) where it supports benthic ecosystems.

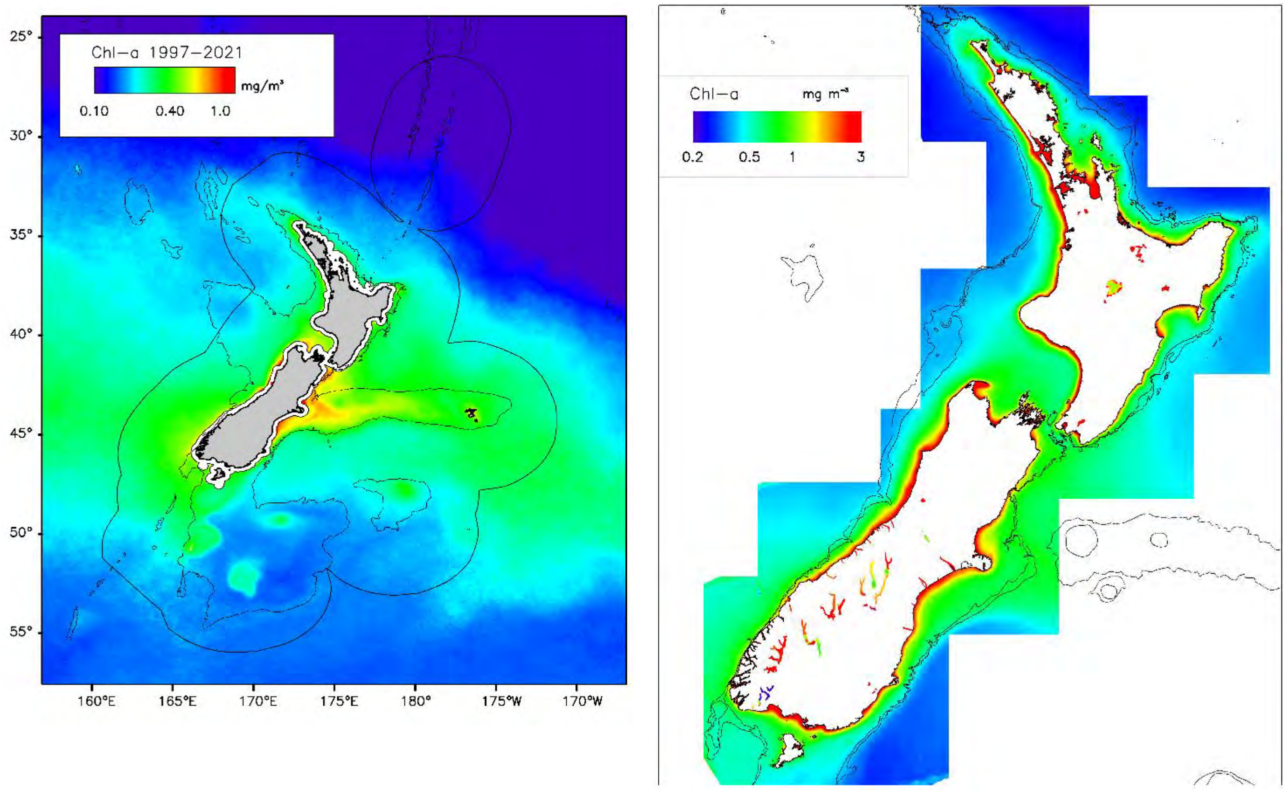


Figure 12.4: Mean annual concentration of chlorophyll-a (the ubiquitous phytoplankton pigment, chl-a) from satellite ocean colour observations [Left panel]. Ocean chl-a based on the merged dataset of SeaWiFS and MODIS-Aqua, which covers the period 1997–2021. The boundary of the New Zealand EEZ is shown and data in the territorial sea (12 n.mile from the coast) are excluded because of possible contamination by suspended sediment and river run-off. [Right panel]. Coastal chl-a based on quasi-analytic processing of MODIS-Aqua data, which covers the period 2002–2021. Images courtesy of NIWA; data used courtesy of NASA.

12.2 INDICATORS AND TRENDS

12.2.1 SEA TEMPERATURE

Sea Surface Temperature (SST), sea surface height (SSH), air temperature, and ocean temperature to 1000 m are all somewhat correlated over seasonal and interannual time scales (e.g., Sutton et al. 2005, Hurst et al. 2012, Sutton & Bowen 2019). New Zealand air temperatures have increased by about 1 °C since 1900 (Figure 12.5).

Although a linear trend has been fitted to the seven-station temperatures in Figure 12.5, the temperature changes are not uniform over time. For example, marked warming occurred through the periods 1940–60 and 1993–2000. Higher frequency variations can be related to fluctuations in the prevailing north-south airflow across New Zealand (Mullan et al. 2010). Temperatures are higher in years with stronger or more prevalent northerly winds and are lower in years with stronger or more prevalent southerly winds. This is as expected, since southerly winds transport cool air from over the Southern Ocean to New Zealand.

The unusually steep warming in the 1940–60 period is paralleled by an unusually large increase in northerly winds during this same period (Mullan et al. 2010). On a longer timeframe, there has been a trend towards less northerly (more southerly) winds since about 1960 (Mullan et al. 2010). However, New Zealand temperatures have continued to increase over this time, albeit at a reduced rate compared with earlier in the twentieth century. This is consistent with a warming of the entire Southwest Pacific (Mullan et al. 2010).

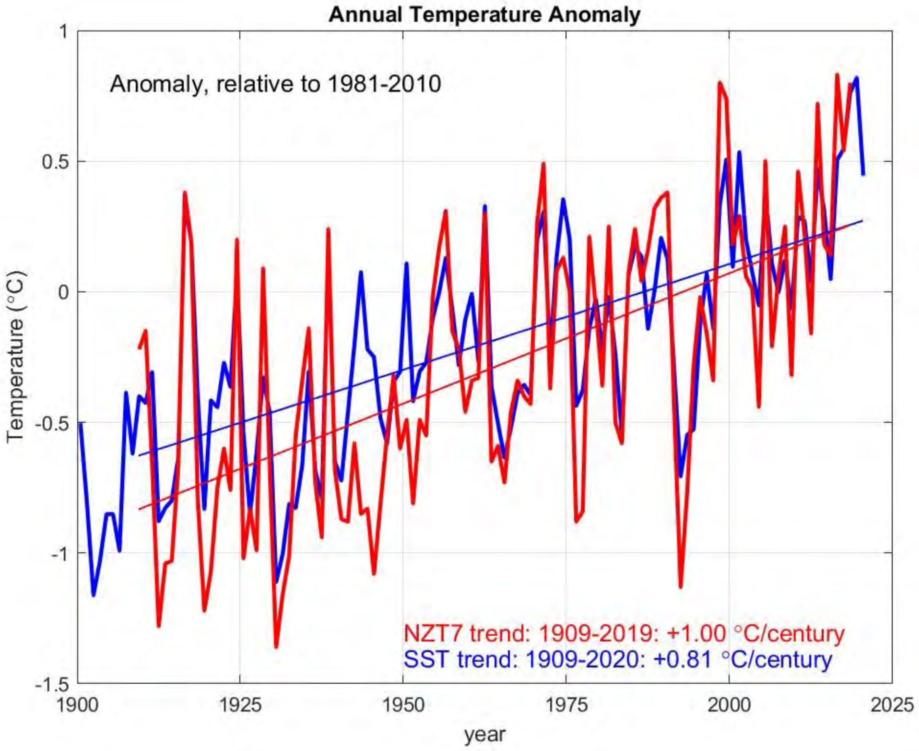


Figure 12.5: Annual time series in New Zealand. NOAA annual mean sea surface temperatures (blue line) averaged over the box outlined in black in Figure 12.6 together with NIWA's seven-station annual mean air temperature composite series (red line), expressed as anomalies relative to the 1981–2010 climatological average. Linear trends over the period 1909–2017, in °C/century, are noted under the graph. (NZT7 data from NIWA. NOAA data from ERSSTv5 (Huang et al. 2017). ERSST is Extended Reconstructed SST.)

Trends in sea surface temperature (SST) in the New Zealand region tend to be slightly smaller than trends in air temperature over land (Figure 12.5). Mullan et al. (2010) describe the pattern of warming in New Zealand as consistent with changes in sea surface temperature and prevailing winds. Their review shows enhanced rates of warming along the East Australian coast and to the east of the North Island, and much lower rates of warming south and east of the South Island (Figure 12.6).

Figure 12.7 shows SST trends since 1982 calculated from daily satellite measurements. These are at higher spatial and temporal resolution than Figure 12.6, providing more detail but span a shorter period. It is apparent that SSTs are increasing north of about 45° S and they are increasing more slowly, and decreasing in recent decades, east of Otago and south of New Zealand. This regional pattern of cooling (or only slow warming) to the south, and strong warming in the Tasman Sea and western Pacific Ocean can be related to increasing westerly winds and their effect on ocean circulation (Mullan et al. 2010, Roemmich et al. 2007, Roemmich et al. 2016). Thompson & Solomon (2002) discuss the increase in Southern Hemisphere westerlies and the relationship to global warming; Roemmich et al. (2007) and Roemmich et al. (2016) describe recent ocean circulation changes with the South Pacific subtropical gyre spinning up in response to the changing winds and Thompson et al. (2009) discuss the consequent effect on sea surface temperatures in the Tasman Sea.

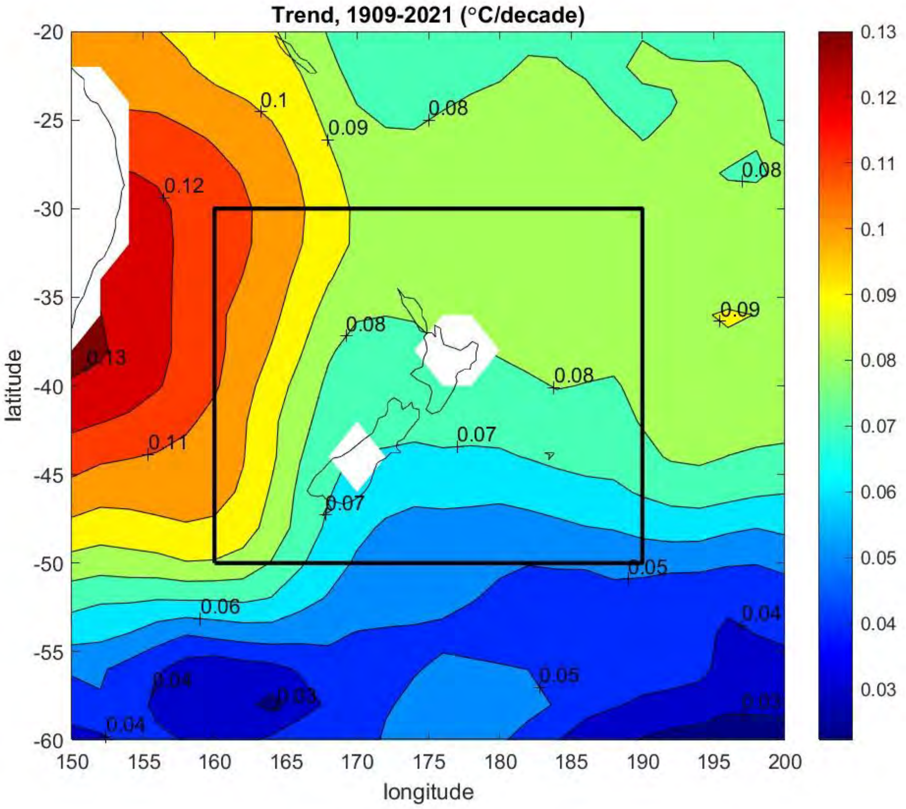


Figure 12.6: Trends in sea surface temperature, in °C/decade over the period 1909–2021, calculated from the NOAA_ERSST_v5 dataset (provided by NOAA's ESRL Physical Sciences Division, Boulder, Colorado, USA, from their web site at http://www.esrl.noaa.gov/psd). The data values are on a 2° latitude-longitude grid. The white areas are where the 2° grid points fall on land.

Comparing this updated analysis with a previous version based on measurements up until 2018 (Sutton & Bowen 2019) shows that the warming trend has generally increased. In particular, regions south of New Zealand which showed no significant trends over 1981–2018 show weak warming over 1981–2021. This acceleration is largely due to two extreme marine heatwaves that occurred in the 2017–18 and 2018–19 summers which are discussed later.

Figures 12.6 and 12.7 show some similarity but differ because trends are calculated over 1909 to 2021 in Figure 12.6 compared with 1981 to 2021 in Figure 12.7. Figure 12.6 is based on coarser resolution (2° lat/lon, monthly) ERSST that has been reconstructed from sometimes sparse ship measurements prior to the satellite era. Figure 12.7 is based on a consistent satellite dataset (AVHRR) with no changes in underlaying data or processing and 0.5° lat/lon, daily resolution.

Previous versions of this report have discussed coastal SSTs from a limited number of sites — notably the University of Auckland site at Leigh and Otago University site at Portobello. An analysis by Chiswell & Grant (2018) showed that satellite-based SSTs extracted for locations near the fixed coastal sites reproduced the variability seen in the direct coastal measurements very well. With that in mind, here we examine the SST records extracted from locations encircling the New Zealand coastline (Figure 12.8) to provide a picture of the complete coastal SST around New Zealand since 1981. This is an update of work by Sutton & Bowen (2019).

Pixels close to the coast were selected (Figure 12.8A) and annually-smoothed temperature anomaly time series from these locations are shown in Figure 12.8B with the coastline 'unwrapped' so that the y axis begins with Stewart Island before running northward up the east coast to North Cape and then southward along the west coast to complete the circuit at Stewart Island. Correlation length scales implicit in the daily OI-SST product are of the order of 100–150 km (Reynolds et al. 2007) or about half the distance between Banks Peninsula and Cook Strait, meaning that Figure 12.8B is effectively smoothed over this length scale. The OI-SST correlation time scales are much shorter than the annual smoothing and so will not impact Figure 12.8B. Strong coherence over the entire coastline is clear in Figure 12.8B, consistent with the finding of Bowen et al. (2017) that interannual changes are correlated over large areas.

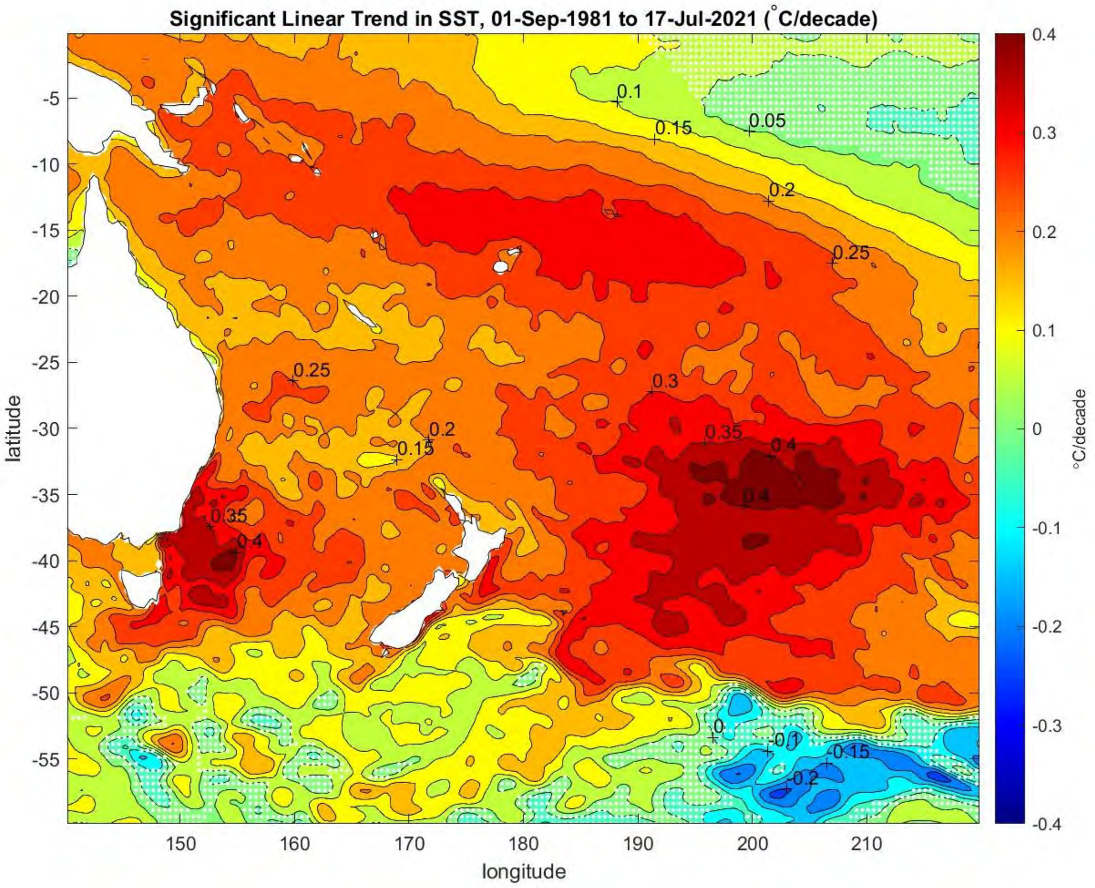


Figure 12.7: Trends in sea surface temperature, in °C/decade over the period 1982–2021. The data are from NOAA based on daily, interpolated satellite AVHRR measurements over a 0.25° grid (http://www.ncdc.noaa.gov/oisst, Reynolds et al. 2007). Areas where the trend is not statistically significant are shaded in white. Updated from Sutton & Bowen (2019).

The general pattern is that 1982–1983 is relatively cool, with a second cool period occurring in the 1990s. There was strong warming in the later 1990s (e.g., Sutton et al. 2005) with 1998 being the warmest year on record in the New Zealand air temperature seven-station record until it was surpassed in 2016 (https://www.niwa.co.nz/our-science/climate/information-and-resources/nz-temp-record/seven-station-series-temperature-data). The banding in Figure 12.8B indicates significant interannual variability. Within the large-scale pattern of variability, there are subtle changes around the coastline. The section of coast between Cook Strait and East Cape was anomalously cool at the start of the time series and shows more warm events through the 2000s, consistent with it being near the region of strongest trends. Conversely, the coast between East Cape and North Cape was not as cool in 1982–83 and does not show the warm events between 2001 and 2015, consistent with it being a local minimum in warming (Shears & Bowen 2017). The west coast varies largely in unison, which indicates that the eastern end of a repeat survey between Sydney and Wellington discussed later is representative of much of the eastern Tasman Sea. The summer of 2017–18 is anomalously warm around the entire coastline, but particularly so south of Banks Peninsula and Cape Egmont, with much of this signal resulting from the extreme marine heatwave that occurred through that summer. The weaker 2019–20 marine heatwave did not generate such a clear warm anomaly.

The ocean temperature down to 800 m between Sydney and Wellington has been sampled about four times per year since 1991 by CSIRO and Scripps Institution of Oceanography by dropping expendable bathythermographs (XBTs) from container ships (http://www-hrx.ucsd.edu/).

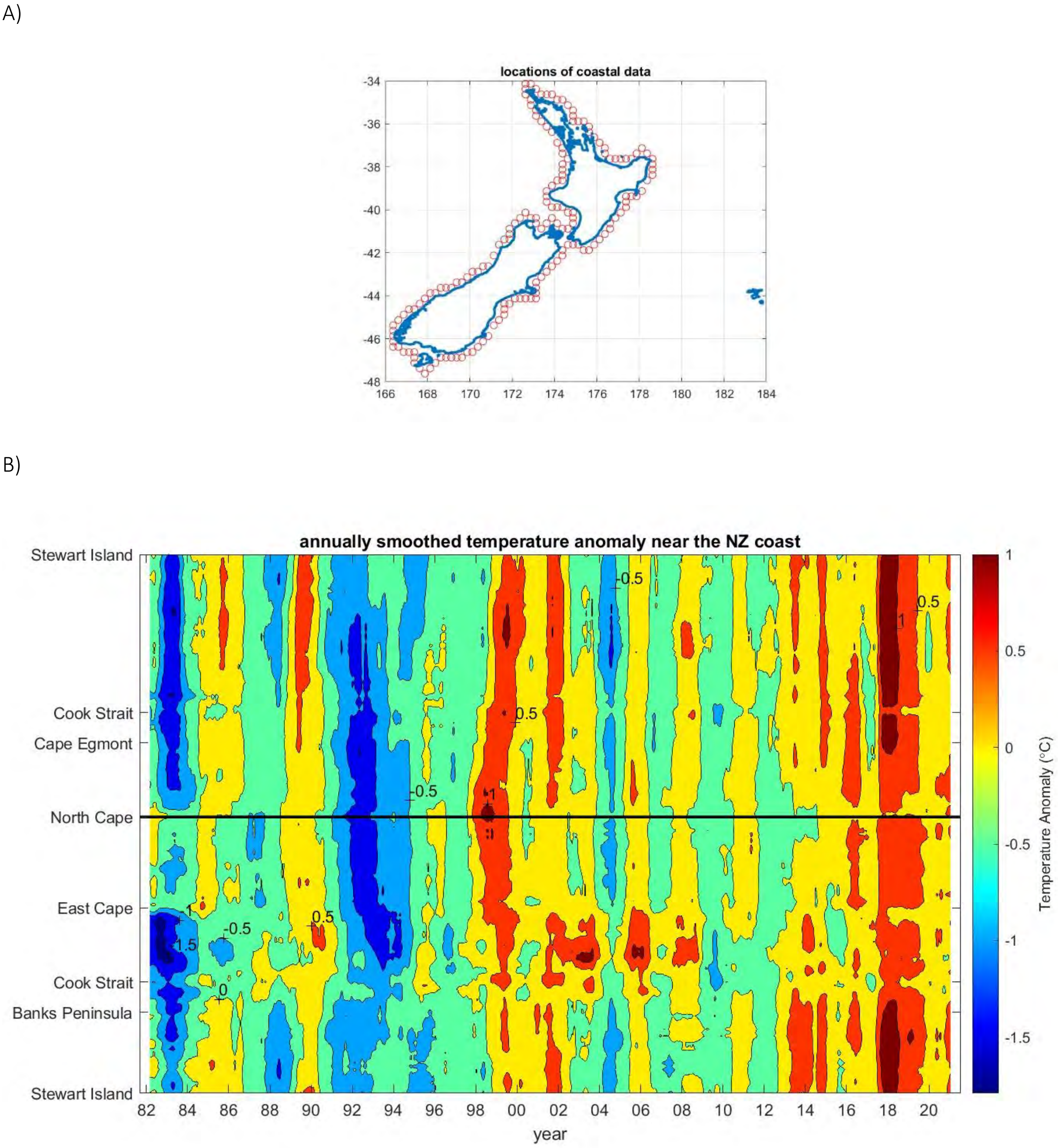


Figure 12.8: (A) The locations near the coast selected to build a near-coastal time series. (B) The time series of annually-smoothed temperature anomaly near the coast. The y axis begins at the southern-most location south of Stewart Island, then runs north along the east coast to North Cape (shown by the black line) before running south along the west coast. (Updated from Sutton & Bowen 2019.)

Analyses of the subsurface temperature field using these data include Sutton & Roemmich (2001) and Sutton et al. (2005). The temperature as a function of depth and time for the eastern portion (between 161.5° E and 172° E) of this section is shown in Figure 12.9. This eastern Tasman Sea section is close to New Zealand and has low oceanographic variability meaning that subtle inter-annual changes can be seen. The portion of the transect shown is along a fairly constant latitude and is therefore unaffected by latitudinal temperature and seasonal cycle variation. The lower panel shows the temperature averaged along the transect (with seasons removed) between the surface and 800 m and from 1991 to the most recent sampling. Unfortunately, this sampling was halted with the onset of COVID-19 so there are no newer data and a gap in the time series has been created.

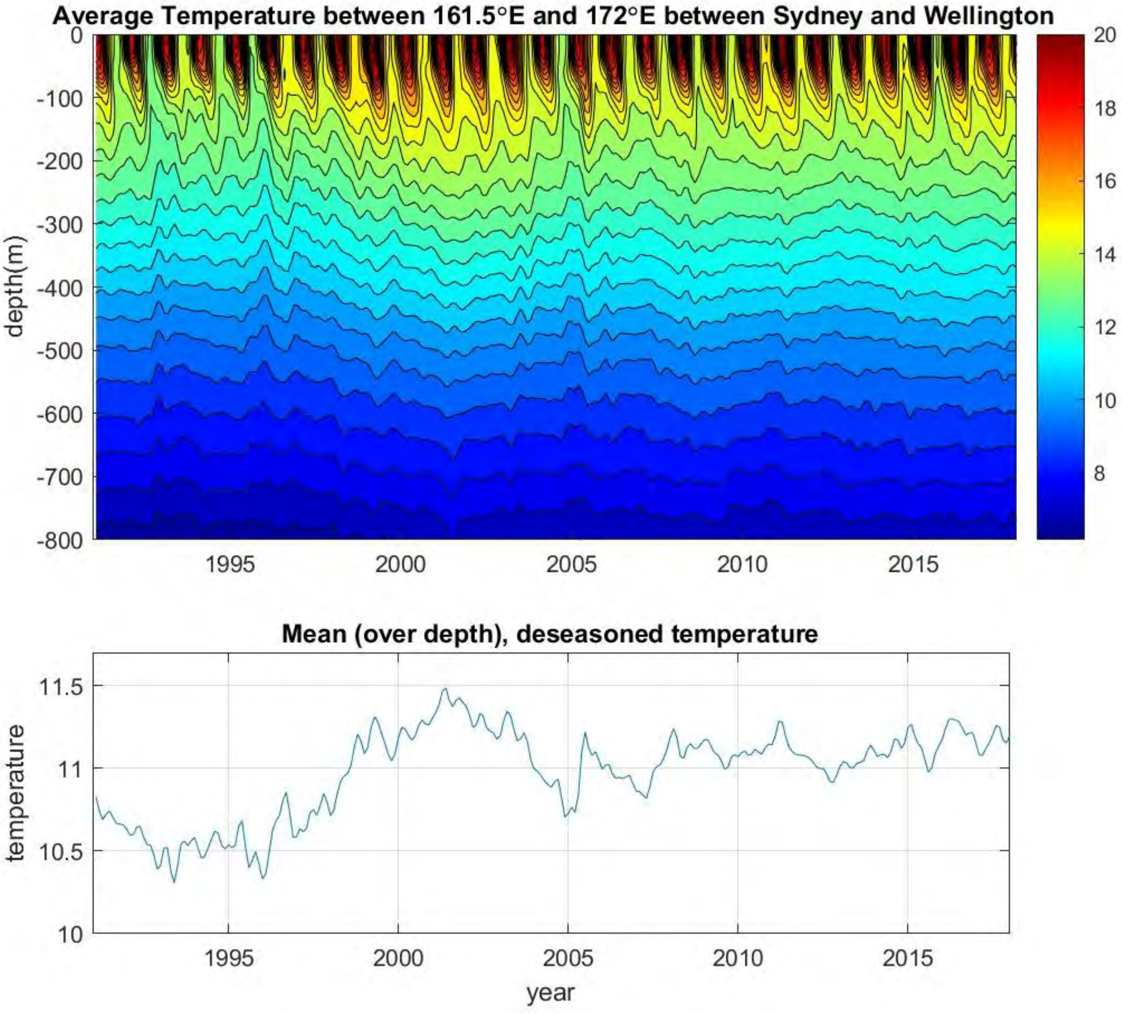


Figure 12.9: Top panel shows the eastern Tasman Sea ocean temperature along the Sydney to Wellington transect: 161.5° E to 172° E, 1991–2018. Coloured scale to the right is temperature (°C). Image updated from Sutton et al. 2005. Bottom panel shows the mean 0-800 m temperature with seasons removed.

The seasonal cycle is clearly visible in the upper 100–150 m. There is a marked warming signal that occurred through the late 1990s, apparent from the isotherms increasing in depth through that time period. This warming was significant in that it extended through the full 800 m of the measurements (effectively the full depth of the eastern Tasman Sea). It also began during an El Niño period when conditions would be expected to be relatively cool. It was thought to be linked to a large-scale warming event centred on 40° S that had hemispheric and perhaps global implications. This warming has been discussed by Sutton et al. (2005) who examined the local signals, Bowen et al. (2006) who studied the propagation of the signal into the New Zealand area, and Roemmich et al. (2007) who examined the broad-scale signal over the entire South Pacific Ocean. Roemmich et al. (2007) hypothesised that the ultimate forcing was an increase in high latitude westerly winds effectively speeding up the entire South Pacific gyre. An update of this analysis has confirmed that the gyre spin-up continued through to 2015 (Roemmich et al. 2016).

Other phenomena have led to periods of warming that are as yet not fully understood. Both stochastic environmental variability and predictable cycles of change influence the productivity and distribution of marine biota in our region.

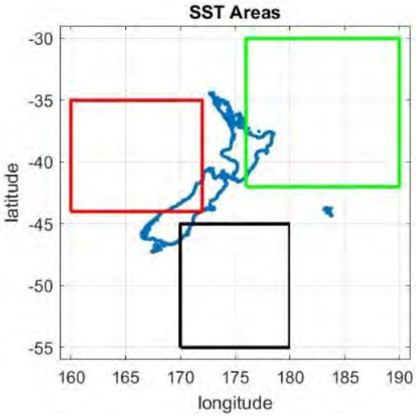
The 2017–18 and 2018–19 summers saw very warm SSTs in the Tasman Sea (Figure 12.10), with SST anomalies reaching 4–5 °C above normal in 2017–18 and 2.5 °C above normal in 2018–19. Studies indicate that these anomalous warm events were the result of prolonged periods of calm winds, in turn as a result of blocking atmospheric high pressure systems to the east of New Zealand (Salinger at al. 2018, Behrens et al. 2018, Salinger et al. 2020). Figure 12.10(C) shows the average vertical temperature anomaly in the upper 200 m of the eastern Tasman Sea since 2006 as measured by Argo floats. Looking at this 15-year time period, the decadal warming of ~0.3 °C as seen in the XBT analysis appears clearly. There are also upper (0–40 m) warm events, with weak events in the summers of 2007–08, 2010–11, and 2015–2016 and strong events in 2017–18 and 2018–19. The 2017–18 event stands out as the strongest. These warm events are typically associated with summers and are constrained to the upper 40 m or so of the water column — corresponding to the depth of the summer mixed layer. Cool events occur year-round and can extend to more than 100 m — corresponding to the depth of the winter mixed layer. An approximately 30 m deep layer of warm water formed in November 2017 and persisted until February 2018 before it was eroded at the surface by strong winds associated with Cyclone Fehi. The wind mixed the warm, upper water deeper into the water column as is evident in Figure 12.10(D). There was no significant change in the total heat content through this period because the anomaly was shallow, and there was also no discernible air-sea flux signature (Behrens et al. 2018). The similar, but weaker warm event in summer 2018–19 peaked at an anomaly of ~+2.5 °C. It was also mixed down into the water column by winds once the atmospheric blocking high broke down.

Longer-term occurrences of anomalously warm and cold events are shown in Figure 12.10(B). Regional average SST anomaly time series are shown for three areas (Figure 12.10(A)), along with the annually-smoothed time series and associated linear trends (updated from Sutton & Bowen 2019). Time periods where the daily anomalies are more than two standard deviations from the mean are shown in bold symbols. In all regions, the occurrences of warm and cool events are modulated by interannual variability, that is warm extreme events almost exclusively occur during warm periods and cool extreme events during cool periods, consistent with the results of Behrens et al. 2018. This modulation is clear during the early 1990s cool period in Subtropical Water (the Tasman and northeast regions), particularly so for the northeast, with the bulk of northeast cool events occurring between 1991 and 1995. Beyond this decadal modulation, there is no clear trend in the occurrence of warm or cool events in the northeast or southeast regions. There is a suggestion that the occurrence of warm events in the Tasman region is becoming more frequent, and cool events look to be becoming rarer, with only one Tasman cool event since 2008, as could be expected given the significant warming trend in the eastern Tasman Sea (Figure 12.7). The summer 2017–18 warm event stands out as being exceptional in the Tasman Sea and southeast areas.

Both cool and warm events occur more often in summer (Figure 12.10). In their study offshore of the west coast, South Island, Chiswell & O'Callaghan (2021) also found this to be the case, but also concluded that warm events showed more seasonal variability. This summer bias is probably a result of the summer mixed layer being shallower and therefore requiring less energy to drive a given temperature change. In addition, the 2017–18 and 2018–19 marine heatwaves resulted from periods of light winds (Salinger at al. 2018, Behrens et al. 2018, Salinger et al. 2020), conditions that are more likely to occur during summer.

Chiswell & O'Callaghan (2021) also noted that the SST warming is not constant across seasons. Their west coast, South Island 2002–18 SST trend was positive over most of the region and resulted from increasingly warmer summers and marine heatwaves rather than year-round warming.

(A)


SST Areas
-30
-35
-40
-45
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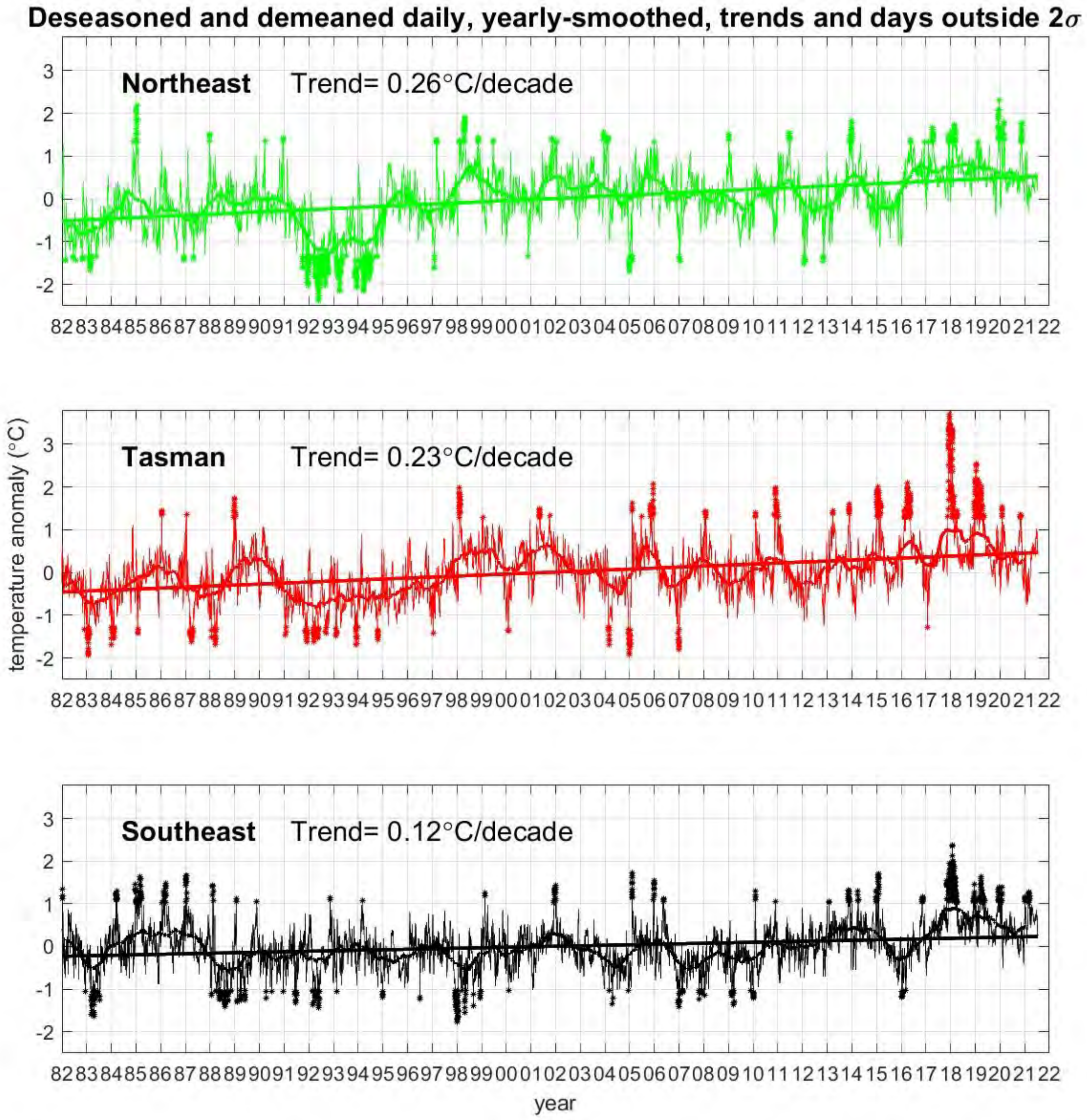
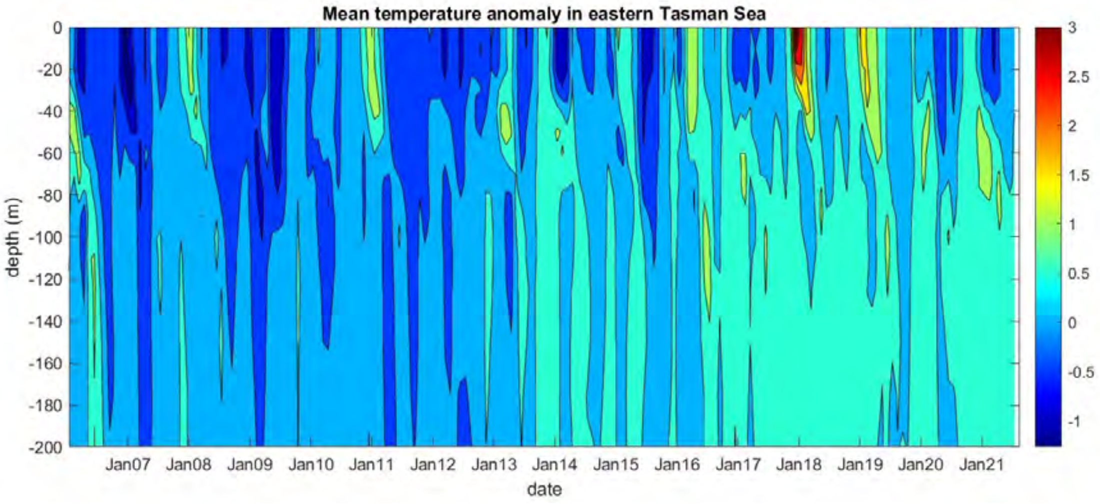
-55
latitude
160
165
170
175
180
185
190
longitude

(B)


Deseasoned and demeaned daily, yearly-smoothed, trends and days outside 2σ
Northeast
Trend= 0.26°C/decade
Tasman
Trend= 0.23°C/decade
Southeast
Trend= 0.12°C/decade
temperature anomaly (°C)
3
2
1
0
-1
-2
82 83 84 85 86 87 88 89 90 91 92 93 94 95 96 97 98 99 00 01 02 03 04 05 06 07 08 09 10 11 12 13 14 15 16 17 18 19 20 21 22
year

(C)



(D)

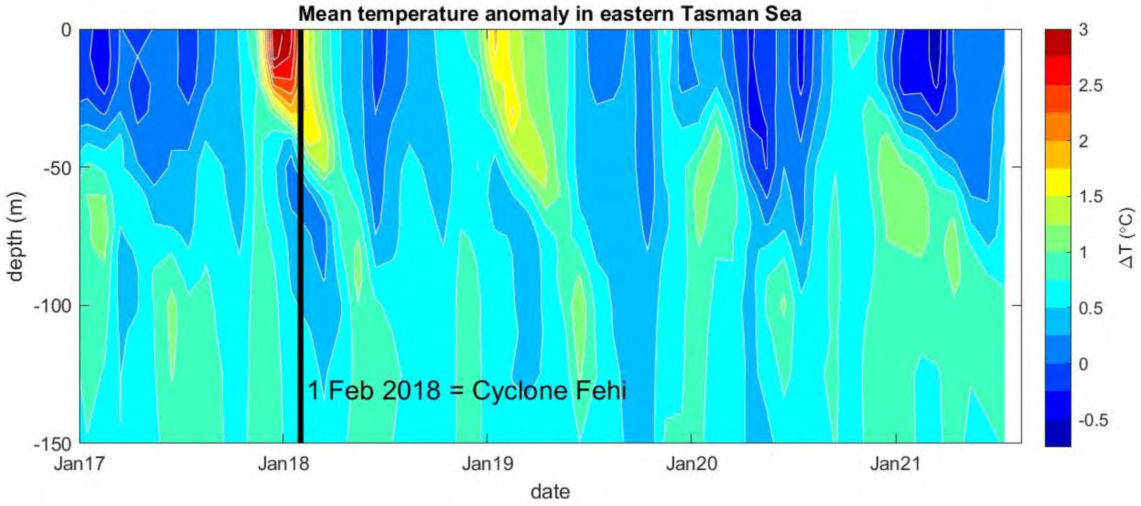


Figure 12-10: Marine heatwaves. (A) Three regions chosen to represent different regimes. (B) The mean satellite SST anomalies (with the seasonal data removed) for these regions, at daily resolution and annually-smoothed. The linear trends are also shown. Values more than two standard deviations from the mean are shown in bold symbols. (Modified from Sutton & Bowen 2019.) (C) and (D) The mean temperature anomaly in the eastern Tasman Sea as a function of depth and time from Argo data for the full Argo period (C) and since 2017 (D).

12.2.2 PRIMARY PRODUCTIVITY

Ocean primary productivity

A 23-year time series of observations of chl-a can be obtained for the ocean domain, by blending together data from overlapping satellite sensors operated by NASA (SeaWiFS and MODIS-Aqua), covering the period September 1997–August 2021. Linear trend analysis can be applied to the satellite data to estimate the long-term trends in chl-a and indicate likely changes in ocean productivity over this period (Pinkerton et al. 2019). Satellite data show that chl-a is increasing in some ocean areas around New Zealand, especially in the Subtropical Front east and west of New Zealand, and over Chatham Rise (Figure 12.11, upper). Negative trends in chl-a (indicative of decreasing ocean productivity) were found in Subtropical Water to the east of New Zealand (outside the EEZ) and around Northland and the northeast New Zealand continental slope (offshore of Hauraki Gulf, eastern Coromandel, and Bay of Plenty). Average rates of increase and decrease in chl-a in these areas was of the order of 1–2% per year which is small compared with interannual variability (~ ±20%). Most trends were not significant at the

0.95 confidence level because the time series is short and the variability in chl-a is large compared with the size of the trend.

Conventional wisdom is that upper ocean warming will result in stronger stratification and that will, in turn, impact primary productivity and chl-a. Two basic mechanisms act in opposition: 1) in nutrient-limited (oligotrophic) environments increased stratification will reduce primary productivity by reducing the availability of deep nutrients to the mixed layer; and 2) in light-limited environments, increased stratification will increase primary productivity by shoaling the depth of the phytoplankton. Chiswell & Sutton (2020) found that over the last two decades warming has reduced primary productivity in STW but has not over the STF or in SAW, consistent with this paradigm. The new analysis presented here is also consistent with this general understanding. However, the largest and most significant positive trends in chl-a anomalies over the entire 1997–2021 time series were found in the Subtropical Front both east and west of New Zealand landmass. The mechanisms leading to these observed strong increases in chl-a in the Subtropical Front are not well understood; trends in SST anomalies in these regions were not particularly high and there were no strong correlations between chl-a anomalies and SST anomalies.

Overall, the changes in ocean chl-a observed in the satellite data since 1997 show the effects of large-scale climate-environmental forcing in the New Zealand EEZ (Figure 12.11, lower panel). Negative chl-a anomalies (and likely reduced ocean primary productivity) through most of the EEZ over the period September 1997 to 2003 were followed by higher values in the middle of the time series (2004–2012), then lower values from 2012 to 2018, and finally a period of strong positive anomalies between 2018 and 2020. There was also a very large and unprecedented (in the satellite data record) EEZ-scale positive chl-a anomaly in November 2019.

Coastal primary productivity

Linear trend analysis can be carried out for the coastal data as for the ocean observations, but only for the period July 2002–June 2021 because SeaWiFS data are not suitable for this analysis. The coastal analysis shows areas with significant negative trends in coastal chl-a (likely indicative of decreasing primary productivity) along the east coast of Northland, around the Hauraki Gulf and Coromandel, Bay of Plenty, East Cape, Wairarapa coast, South Taranaki Bight, and offshore of Golden Bay and Tasman Bay (Figure 12.12, upper panel). Evidence of increasing coastal productivity over the last two decades was found in the Firth of Thames, Hawke Bay, around the Taranaki coast, and around most of the coastline of South Island. Changes in coastal primary productivity and chl-a will result from the interplay of oceanographic change at the continental shelf scale, and smaller-scale, localised effects due to the effects of land use on riverine input of nutrients and sediment. These patterns of change can hence give insights into the factors causing the changes in different areas.

The time series of change in chl-a in the territorial waters around North Island and South Island are shown in Figure 12.12, lower panels). There is evidence of a small decreasing trend in chl-a around North Island since 2002, but the magnitude of the trend is small compared with interannual variability. Around South Island, coastal chl-a has increased since late 2017, with some very large, positive anomalies in June 2020 and June 2021.

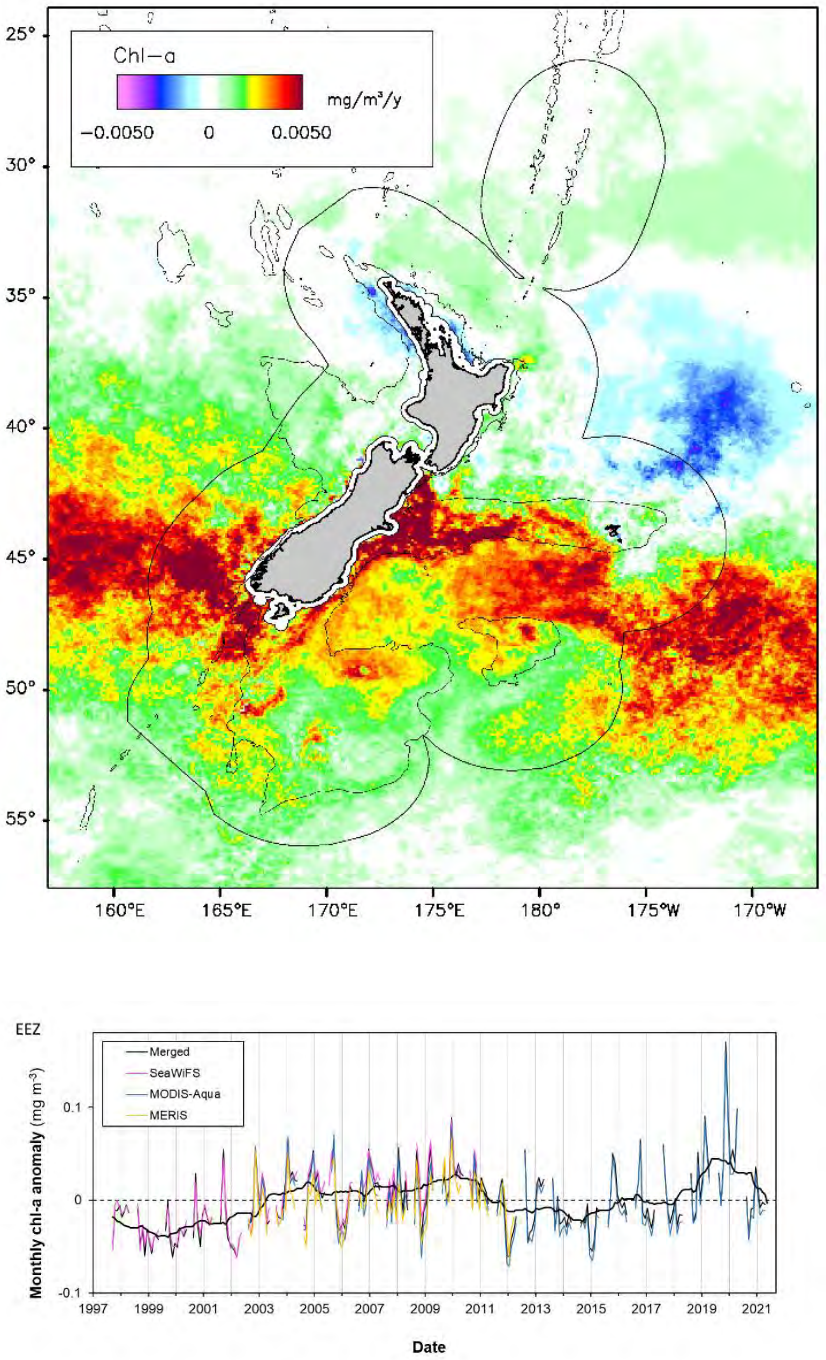


Figure 12.11: Trends (long-term change) in oceanic chlorophyll-a (the ubiquitous phytoplankton pigment) based on the merged satellite dataset of SeaWiFS and MODIS-Aqua, which covers the period 1997–2021. [Upper panel]: Spatial trends where red/orange shows increasing trends in chl-a and pink/purple shows decreasing trends. The boundary of the New Zealand EEZ is shown and data in the territorial sea (12 n.mile from the coast) are excluded because of possible contamination by suspended sediment and river run-off. [Lower panel]. Mean monthly anomalies (differences from mean monthly values) in chl-a averaged across the New Zealand Exclusive Economic Zone (EEZ). The monthly anomalies shown are the merged (SeaWiFS-MODIS-Aqua) dataset (thin black lines), SeaWiFS (pink), MODIS-Aqua (blue), and MERIS (orange). Monthly data are not shown where there was less than 80% data coverage for a given month. The thick black line is the merged dataset smoothed with a 4-year running mean, the vertical grey lines divide different years, and the dashed horizontal line shows zero anomaly. Images courtesy of NIWA; satellite data used courtesy of NASA and European Space Agency.

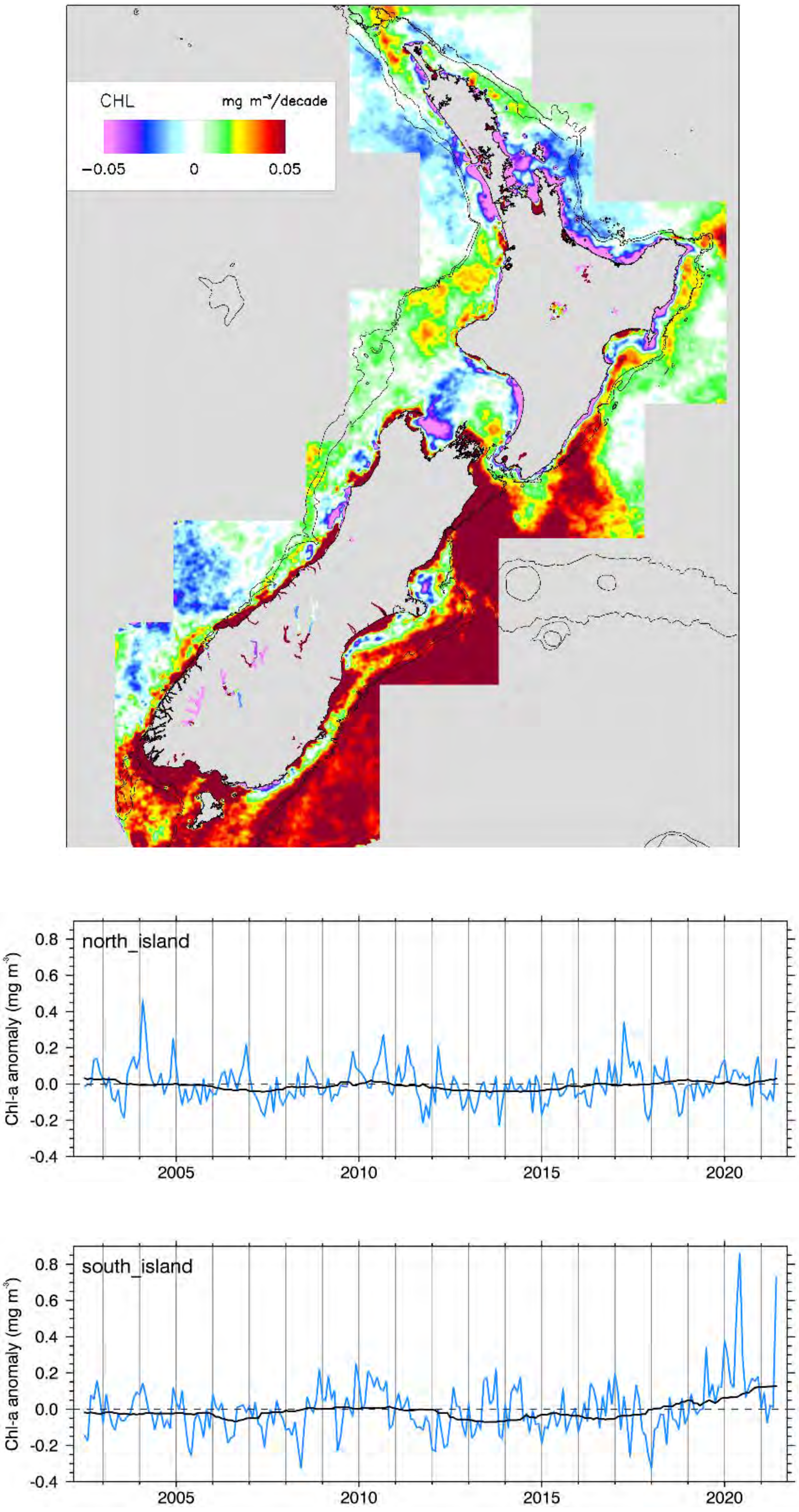


Figure 12.12: Coastal trends (long-term change) in chl-a based on the quasi-analytic processing of MODIS-Aqua data (see method of Pinkerton et al. 2018), which covers the period 2002–2021. [Top panel] Spatial trends where red/orange shows increasing trends in chl-a, and pink/purple, decreasing trends. The 250 m and 500 m depth contours are also shown. [Lower panels]: Mean monthly anomalies (differences from mean monthly values) of chl-a for the territorial waters (shore to 12 n-miles) for North Island and South Island. Thick black lines are smoothed with a 4-year running mean, and the vertical grey lines divide different years. The dashed horizontal lines show zero anomaly. Images courtesy of NIWA; satellite data used courtesy of NASA.

12.2.3 CLIMATE VARIABLES

The Interdecadal Pacific Oscillation (IPO) is a Pacific-wide reorganisation of the heat content of the upper ocean and represents large-scale, decadal temperature variability with changes in phase over 15–30-year time scales. In the past 100 years, phase changes occurred in 1926, 1943, 1978, and 1998 (Figure 12.13). The latest shift should result in New Zealand experiencing a period of reduced westerlies, with associated warmer air and sea temperatures and reduced upwelling on western coasts (Hurst et al. 2012).

The El Niño-Southern Oscillation (ENSO) cycle in the tropical Pacific has a strong influence on New Zealand. ENSO is described here by the Southern Oscillation Index (SOI), a measure of the difference in mean sea-level pressure between Tahiti and Darwin (Figure 12.14). When the SOI is strongly positive (persisting above +1), a La Niña event is taking place and New Zealand tends to experience more north-easterlies, reduced westerly winds, milder, more settled, warmer anticyclonic weather, and warmer sea temperatures (Hurst et al. 2012). When the SOI is strongly negative (persisting below -1), an El Niño event is taking place and New Zealand tends to experience increased westerly and south-westerly winds, cooler, less settled weather, and enhanced along shelf upwelling off the west coast South Island and north-east North Island (Shirtcliffe et al. 1990, Zeldis 2004, Chang & Mullan 2003, Sutton & Roemmich 2001). The SOI is available monthly from 1876 onwards (Mullan 1995).

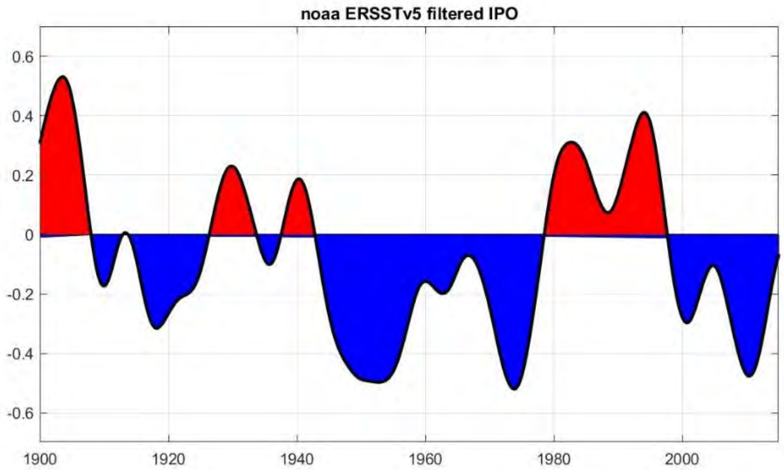


Figure 12.13: Smoothed index of the Interdecadal Pacific Oscillation (IPO) since 1900. (Filtered IPO Tripole Index of Henley et al. (2015): https://psl.noaa.gov/data/timeseries/IPOTPI/.)

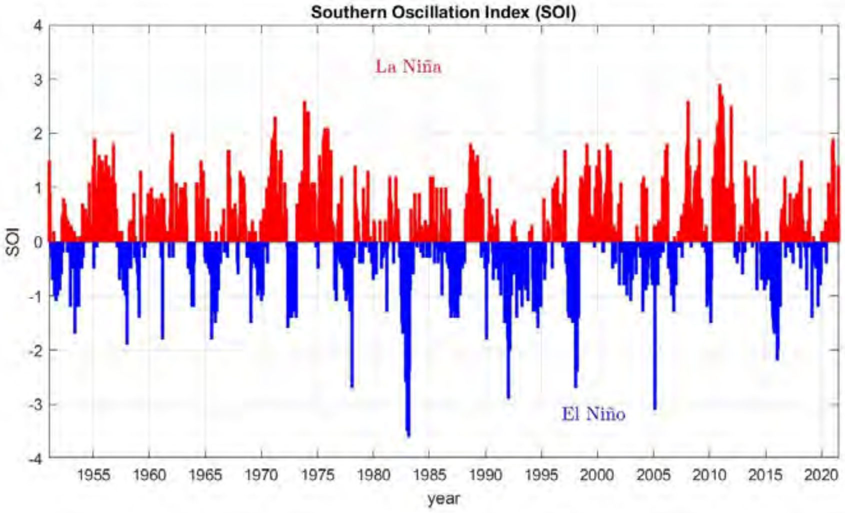


Figure 12.14: Southern Oscillation Index (SOI). Red indicates La Niña and a tendency for warmer temperatures around New Zealand, blue indicates El Niño and a tendency for cooler conditions around New Zealand. (https://www.ncdc.noaa.gov/teleconnections/enso/indicators/soi/ and https://www.cpc.ncep.noaa.gov/data/indices/soi)

12.2.4 WATER CHEMISTRY: OCEAN ACIDIFICATION

The increase in atmospheric carbon dioxide (CO_2) since the industrial revolution has been paralleled by an increase in CO_2 concentrations in the upper ocean (Sabine et al. 2004). The ocean is estimated to hold about 25% of all anthropogenic CO_2 released since 1750 (Le Quéré et al. 2018), with the anthropogenic CO_2 signal apparent to an average depth of about 1000 m and reaching depths of 3000 m in deep water formation regions (Sabine et al. 2004). The Southern Ocean is the primary route for the entry of anthropogenic CO_2, accounting for about 40% of current oceanic uptake (DeVries 2014).

Carbon dioxide absorbed by seawater reacts with the water and carbonate ions to form bicarbonate ions and releases hydrogen ions in the process (Figure 12.15). This reaction raises the acidity and lowers the pH of seawater, in addition to decreasing the amount of carbonate in the seawater. Since the beginning of the industrial era, average surface ocean pH has decreased by 0.1 units, with a further decrease of up to 0.4 units expected by the end of the century (Rhein et al. 2013). The pH scale is logarithmic, so a decrease of 0.4 in pH corresponds to a 150% increase in hydrogen ion concentration. Both the predicted pH in 2100 and the rate of change in pH are outside the range experienced by the oceans for at least half a million years.

Recent projections for the open ocean around New Zealand indicate a mean pH decline to 7.935–7.985 by 2050, depending upon the emission scenario. Under a more favourable pathway, RCP4.5, which anticipates zero emissions by mid-century, pH around New Zealand will fall to a mean of 7.95 by 2100; conversely, the worst-case scenario of continued emissions, RCP8.5, indicates that ocean pH will have fallen by about 0.33 to about 7.77 by 2100 (Law et al. 2018b). The latter projection is equivalent to an increase in hydrogen ion concentration (acidity) of 116%. The drop in pH shows little spatial variability in New Zealand waters because it is primarily driven by atmospheric CO_2.

The Munida Time Series has maintained bimonthly measurements of surface CO_2 and pH at a station in subantarctic waters off the Otago shelf for 20 years and is the longest running ocean acidification time series station in the Southern Hemisphere (Law et al. 2018a). The Munida Time Series data (available from https://marinedata.niwa.co.nz/nzoa-on-map/) show a pH decline in subantarctic surface waters since 1998, consistent with that expected from equilibrium with atmospheric CO_2 (Figure 12.16, Law et al. 2018a). Dissolved pCO_2 (partial pressure of CO_2) at the Munida station shows a statistically significant increase of 1.3 µatm yr^{-1} for 1998–2012 (Bates et al. 2014), that is consistent with long-term changes observed at other global time series sites. Surface pH in the open ocean has been determined at seven long-term time series stations, six of which are in the northern hemisphere (Bates et al. 2014, Figure 12.17). All the time series records show long-term trends of increasing pCO_2 and decreasing pH. The significant seasonal variability apparent in the subantarctic water time series is due to biological uptake of dissolved inorganic carbon and seasonal temperature changes (Brix et al. 2013).

On a global basis, the open ocean shows relatively low spatial and temporal variability in pH relative to coastal waters where pH may vary by up to 1 unit in response to precipitation events and biological activity. A New Zealand coastal ocean acidification observing network (NZOA-ON, https://marinedata.niwa.co.nz/nzoa-on/) has been operating for ten years to establish baseline conditions against which to assess future changes in pH. The network maintains pH time series for 15 coastal stations around New Zealand and includes a range of coastal morphotypes and conditions. A variety of stakeholders, including government agencies, councils, industry, and citizens, provide alkalinity and DIC samples on a fortnightly basis for analysis at a centralised facility (NIWA/University of Otago Research Centre for Oceanography). Examples of pH variability at three of the NZ-OAON sites are shown in Figure 12.16. The seasonal pH cycle is controlled by the temperature at most sites, with a maximum pH in late winter and minimum in late summer-autumn, as for Wellington Harbour in Figure 12.18. Local factors also influence pH, as at the Chatham Islands where phytoplankton blooms along the Subtropical Front result in periods of elevated pH, and in Jackson Bay (west coast, South Island) where freshwater input causes sharp pH declines. Freshwater input also supplies excess nutrients and organic matter in some regions, which enhance acidification of coastal waters. An example of this is the Firth of Thames, where seasonal surveys of surface water have shown a pH decline to 7.9 in autumn in the innermost firth (Zeldis & Swaney 2018; K. Currie, NIWA, pers. comm.), with subsequent monitoring with SeaFET sensors detecting pH events of less than 7.7 (J. Zeldis, NIWA, pers. comm.). This pH is lower than that projected for the surface open ocean around New Zealand by the year 2100 (Law et al. 2018b), indicating that some coastal waters already experience periods of intense acidification.

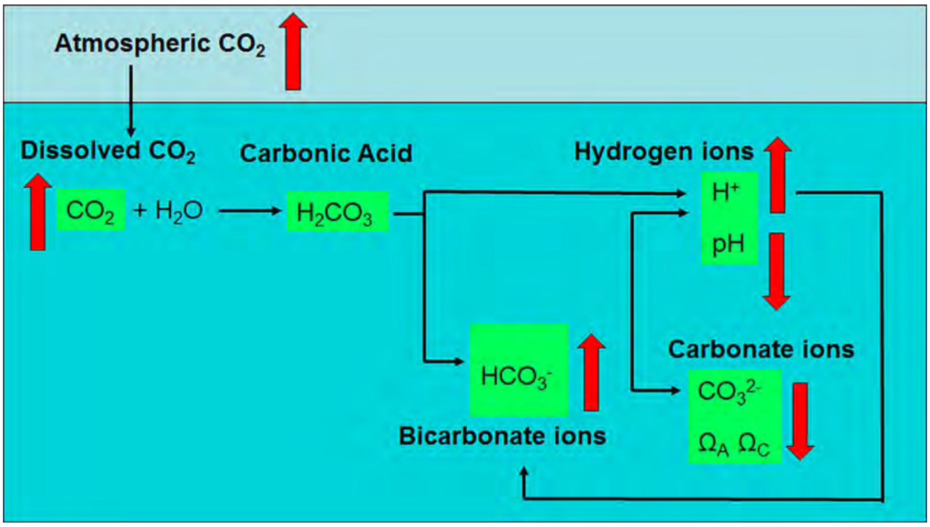


Figure 12.15: Conceptual diagram showing the reactions and chemical species of the marine carbonate system (black arrows), with the direction of the vertical red arrows indicating the net change in response to increasing atmospheric CO_2, and resulting in ocean acidification (Law et al. 2018a).

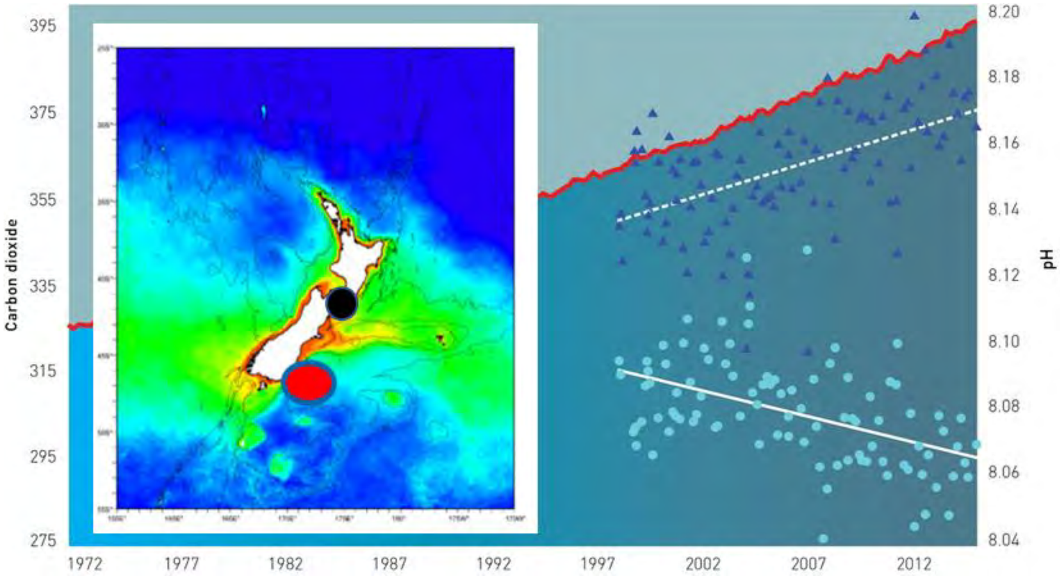


Figure 12.16: Time series of atmospheric CO_2 at Baring Head (location: black dot, Wellington; data: red line, Brailsford et al. 2012), and surface water pCO_2 (data: dark blue triangles) and pH (data: light blue circles) at the subantarctic water station (location: red dot, on the Munida transect off Dunedin) (Law et al. 2018a).

Biological implications of ocean acidification result from increasing hydrogen ion concentration (decreasing pH), decreasing carbonate availability, and increasing dissolved pCO_2. There are many international studies that have identified a range of direct and indirect effects of ocean acidification across a broad range of marine phylogenetic groups, from bacteria to whales and a number of ecosystems from coastal to deep sea. A particular concern regarding ocean acidification is that the reduction in carbonate availability may potentially impact organisms that produce shells or body structures of calcium carbonate, resulting in a weakening of shell integrity, redistribution of an organism's metabolic activity, and increased physiological stress.

Organisms likely to be affected by ocean acidification include those at the base of the food chain (protozoa, plankton), coralline algae, rhodoliths, shallow and deepwater corals, echinoderms, molluscs (possibly including cephalopods, e.g., squids), and high-activity pelagic fish (see Feely et al. 2004 and references therein, Orr et al. 2005, Langer et al. 2006). Early life stages are particularly vulnerable in calcifying (carbonate-forming) species across many phyla (e.g., Byrne et al. 2013, Bylenga et al. 2015, Mu et al. 2015). Ocean acidification is a threat to deep-sea habitats such as seamounts, which can support structural reef-like habitat composed of stony corals (Tracey et al. 2011). A shoaling carbonate saturation horizon could push such biogenic structures to the tops of seamounts, or cause widespread die-back (e.g., Thresher et al. 2012). This has important implications for the structure and function of benthic communities. Recent research on New Zealand cold-water corals identified potential changes in physiology when maintained under lower pH for one year (Gammon et al. 2018). Changes in cold-water corals may indirectly impact commercial fisheries for deepwater species such as orange roughy (Clark 1999).

In surface and shallow waters some phytoplankton and seagrasses may benefit from an increase in dissolved pCO_2 due to increased photosynthesis. Direct effects of acidification on the physiology and development of fish have also been investigated. Adverse effects on physiology development (e.g., Franke & Clemmesen 2011) and behaviour modification (Munday et al. 2014) have been documented. Such studies highlight the potential for increasing acidification to impact larval growth and development, with implications for survival and recruitment of both forage fish and fish harvested commercially. However, recent studies showed no adverse effects of low pH on yellowtail kingfish (Munday et al. 2015), with warming having a greater impact on their behaviour (Watson et al. 2018). Parsons et al. (2020) attempted to predict the yield of Hauraki Gulf snapper under climate change based on the physiological impacts of warming and acidification. They found it was difficult to translate the physiological effects into population level impacts, summarising that the direct effects of climate change on snapper are likely to: 1) be largely determined by impacts on the larval life stage; 2) contain either positive or negative effects; and 3) incorporate an effect on larval life stages that more or less translates into a similar magnitude effect at the population level. Their most negative scenario suggested a 29% decrease in fishery yield, and the most optimistic scenario suggested a 44% increase.

An assessment of the current knowledge, and vulnerability, of different New Zealand biological groups, species, and ecosystems to ocean acidification has been published, as summarised in Table 12.1 (Law et al. 2018a). The general findings of the synthesis are that a) calcifying organisms (those with carbonate shells) are vulnerable; b) early-life history stages (eggs and larvae) are more vulnerable than later life history stages and so are a potential bottleneck to species survival; c) more research is required to determine the response of larger marine fauna (crustacea, fish, and higher trophic levels) to ocean acidification (see Table 12.1), the interaction of ocean acidification with other climate stressors and its effect on ecosystem interactions, and the potential for adaptation to ocean acidification.

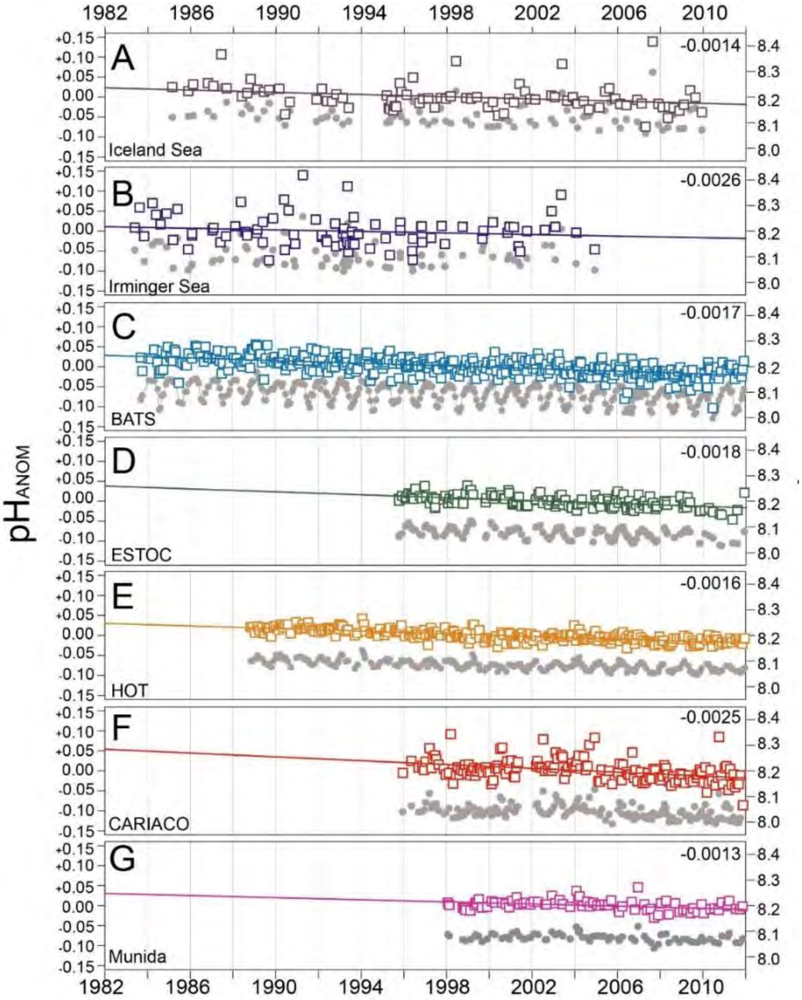


Figure 12.17: Time series of surface ocean pH (grey symbols, right hand axis) and pH anomaly (coloured symbols, left hand axis) at seven time series sites, including the Munida Time Series in New Zealand. Trends (pH change yr^{-1}) are given in the top right of each panel. (A) Iceland Sea, North Atlantic Ocean (purple); (B) Irminger Sea, North Atlantic Ocean (blue); (C) BATS, North Atlantic Ocean (cyan); (D) ESTOC, North Atlantic Ocean (green); (E) HOT, North Pacific Ocean (orange); (F) CARIACO, North Atlantic Ocean (red); and (G) MUNIDA, South Pacific (pink). Image directly sourced from Bates et al. (2014).

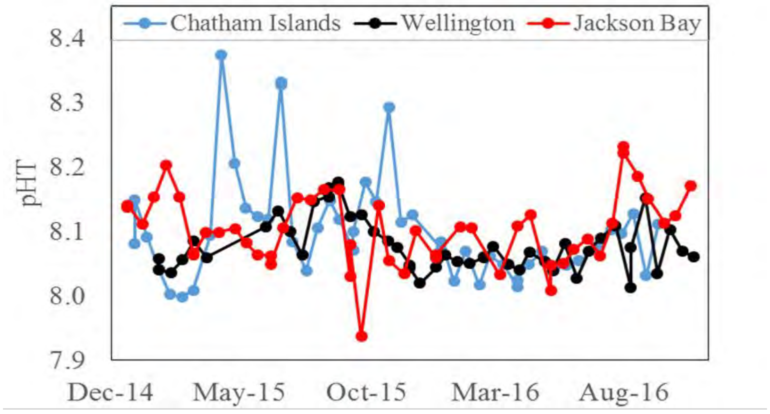
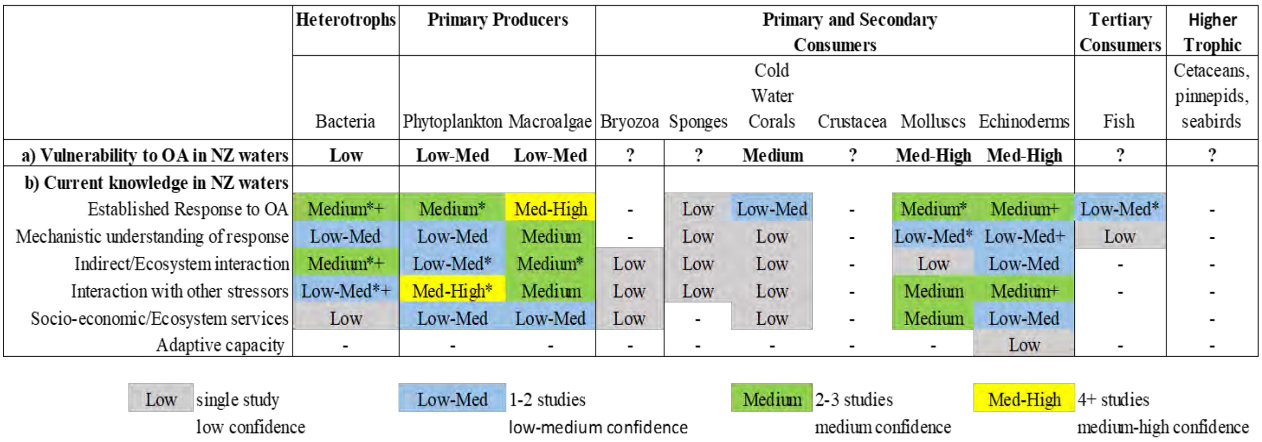


Figure 12.18: pH time series at three coastal stations: Chatham Islands (blue squares and line), Wellington (black circles and line), and Jackson Bay (red triangles and line) in the NZOA-ON (Law et al. 2018a).

Table 12.1: Qualitative assessment of a) the vulnerability, and b) the current state of knowledge of different aspects of ocean acidification research for the major biotic groups in New Zealand waters (Law et al. 2018a). Vulnerability is classified on a Low to High scale, with "?" indicating where vulnerability is currently unknown. Current knowledge is also classified on a Low to High scale and based upon the number and results of published studies (see key), with '-' indicating that no studies have been carried out. Ongoing New Zealand studies are indicated by '+', with those in the CARIM project indicated by '*' (from Law et al. 2018a).



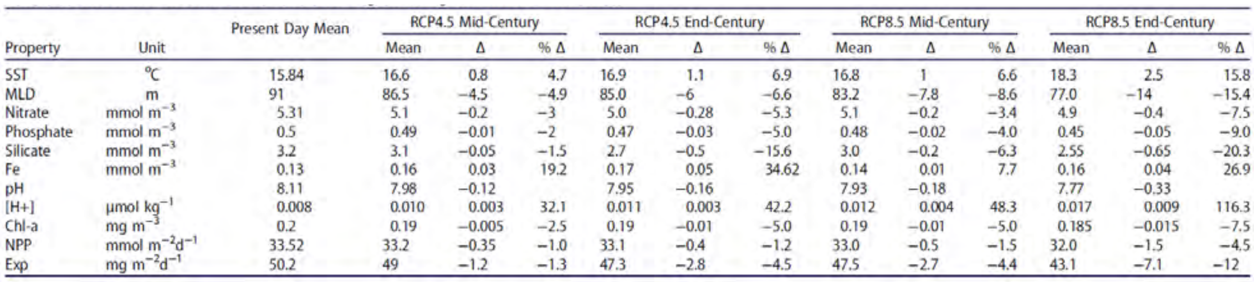
	Heterotrophs	**Primary Producers**		**Primary and Secondary Consumers**						**Tertiary Consumers**	**Higher Trophic**
	Bacteria	Phytoplankton	Macroalgae	Bryozoa	Sponges	Cold Water Corals	Crustacea	Molluscs	Echinoderms	Fish	Cetaceans, pinnepids, seabirds
a) Vulnerability to OA in NZ waters	**Low**	**Low-Med**	**Low-Med**	**?**	**?**	**Medium**	**?**	**Med-High**	**Med-High**	**?**	**?**
b) Current knowledge in NZ waters											
Established Response to OA	Medium*+	Medium*	Med-High	-	Low	Low-Med	-	Medium*	Medium+	Low-Med*	-
Mechanistic understanding of response	Low-Med	Low-Med	Medium	-	Low	Low	-	Low-Med*	Low-Med+	Low	-
Indirect/Ecosystem interaction	Medium*+	Low-Med*	Medium*	Low	Low	Low	-	Low	Low-Med	-	-
Interaction with other stressors	Low-Med*+	Med-High*	Medium	Low	Low	Low	-	Medium	Medium+	-	-
Socio-economic/Ecosystem services	Low	Low-Med	Low-Med	Low	-	Low	-	Medium	Low-Med		-
Adaptive capacity	-	-	-	-	-	-	-	-	Low	-	-

Low	single study low confidence
Low-Med	1-2 studies low-medium confidence
Medium	2-3 studies medium confidence
Med-High	4+ studies medium-high confidence

Projections for a range of climate-related variables in the surface ocean were generated by the *Climate Change: Impacts & Implications* project and published by Law et al. (2018b). The projections reflect the outputs of two Earth System Models (ESMs) identified as the 'best' models for simulating current conditions in the surface ocean around New Zealand from analysis by Rickard et al. (2016). Law et al. (2018b) provides the projected mean change for the middle and end of the 21st Century, for the RCP4.5 and 8.5 scenarios (as summarised in Table 12.2) and considers the spatial variation of change in waters around New Zealand. Results from this study include the following.

- Despite being the best models for New Zealand waters, the two ESMs provide different spatial distributions and magnitudes of change in most parameters (Figure 12.19).
- Projected Sea Surface Temperature increases of +1.6 to +2.95 °C by 2100 are comparable with the means projected for both Australian and global waters (Figure 12.19).
- Highest projected regional warming occurs in the East Australian Current and in subantarctic waters south of Chatham Rise. Because the present warming rate in the south-west Tasman Sea is similar to that projected for waters around New Zealand, this region may provide an analogue for future changes in New Zealand waters.

Table 12.2: Summary table of present-day and projected mean values (Δ = absolute change; % Δ = % change) for the middle and end of the 21st Century for all variables under RCP4.5 and 8.5 for the south west pacific region using the inner Earth System Models (R16) (from Law et al. 2018b).



		Present Day Mean	RCP4.5 Mid-Century			RCP4.5 End-Century			RCP8.5 Mid-Century			RCP8.5 End-Century		
Property	Unit		Mean	Δ	% Δ	Mean	Δ	% Δ	Mean	Δ	% Δ	Mean	Δ	% Δ
SST	°C	15.84	16.6	0.8	4.7	16.9	1.1	6.9	16.8	1	6.6	18.3	2.5	15.8
MLD	m	91	86.5	−4.5	−4.9	85.0	−6	−6.6	83.2	−7.8	−8.6	77.0	−14	−15.4
Nitrate	mmol m^{-3}	5.31	5.1	−0.2	−3	5.0	−0.28	−5.3	5.1	−0.2	−3.4	4.9	−0.4	−7.5
Phosphate	mmol m^{-3}	0.5	0.49	−0.01	−2	0.47	−0.03	−5.0	0.48	−0.02	−4.0	0.45	−0.05	−9.0
Silicate	mmol m^{-3}	3.2	3.1	−0.05	−1.5	2.7	−0.5	−15.6	3.0	−0.2	−6.3	2.55	−0.65	−20.3
Fe	mmol m^{-3}	0.13	0.16	0.03	19.2	0.17	0.05	34.62	0.14	0.01	7.7	0.16	0.04	26.9
pH		8.11	7.98	−0.12		7.95	−0.16		7.93	−0.18		7.77	−0.33	
[H+]	µmol kg^{-1}	0.008	0.010	0.003	32.1	0.011	0.003	42.2	0.012	0.004	48.3	0.017	0.009	116.3
Chl-a	mg m^{-3}	0.2	0.19	−0.005	−2.5	0.19	−0.01	−5.0	0.19	−0.01	−5.0	0.185	−0.015	−7.5
NPP	mmol $m^{-2}d^{-1}$	33.52	33.2	−0.35	−1.0	33.1	−0.4	−1.2	33.0	−0.5	−1.5	32.0	−1.5	−4.5
Exp	mg $m^{-2}d^{-1}$	50.2	49	−1.2	−1.3	47.3	−2.8	−4.5	47.5	−2.7	−4.4	43.1	−7.1	−12

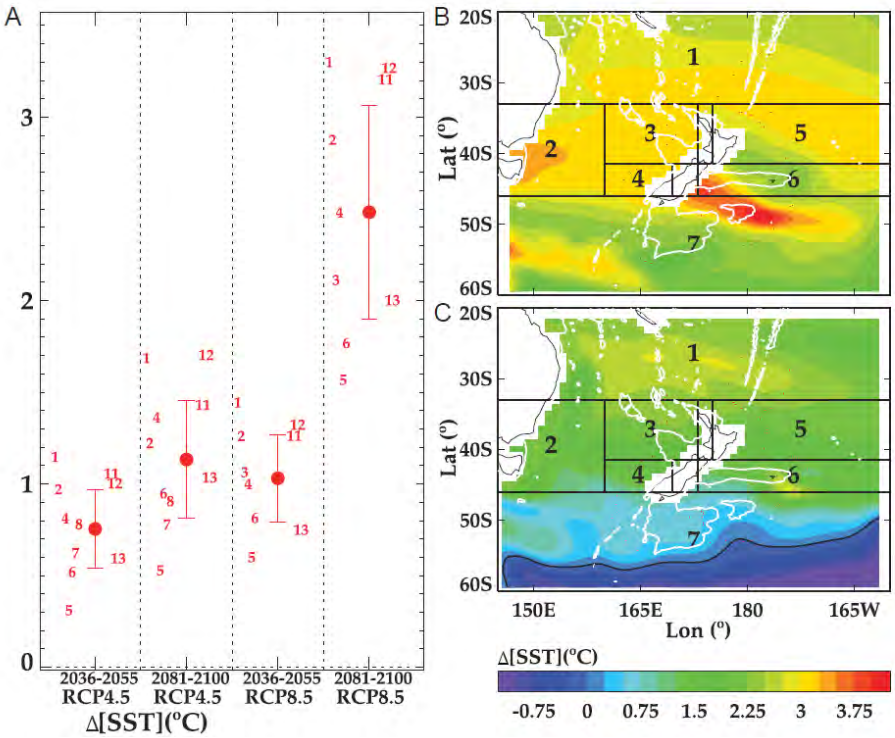


Figure 12.19: Projections of mean change in sea surface temperature (ΔSST, °C, ±1 standard deviation) for Mid and End-Century under RCP4.5 and 8.5 (delineated by vertical dashed lines) from the best ESM subset. The numbers indicate the mean for each individual ESM. The regional variation of ΔSST for the End-Century under RCP8.5 is shown for B, ESM2 and C, ESM5. The regional boxes are indicated by number, the white contours the 1000 m isobath, and the black contours indicate zero change (from Law et al. 2018b).

- The depth of the surface mixed layer will decrease across much of the New Zealand area, except for some subantarctic water regions. The resulting increase in light exposure for plankton may be beneficial to productivity and food webs in subantarctic waters where nutrients are plentiful but may be deleterious in warmer subtropical waters that are oligotrophic (i.e., low nutrient).
- Surface macronutrients (nitrate, phosphate, and silicate) will decline across the New Zealand region, although silicate may increase in subantarctic and polar waters south of New Zealand. The projected declines in nitrate and phosphate are greatest east of Chatham Rise and in subantarctic waters south of the rise. Conversely, dissolved concentrations of the micronutrient iron will increase in surface waters across much of the region.
- Surface chlorophyll-a (the pigment that provides an indicator of phytoplankton biomass) shows only minor changes across much of the region except for a band of frontal waters south-west and south-east of the South Island which shows the largest decline. The regional variations in primary productivity (see Figure 12.20) are greater, with decreases limited to warmer oligotrophic waters, and most of the region south of 40° S showing an increase in primary production.
- Projected decreases in vertical particle flux exports are 4.5% and 12% under RCP4.5 and 8.5, respectively, for the New Zealand region by 2100, which is within the range of projections for decreases in global export production under RCP8.5 of 7–18% (Bopp et al. 2013). One of the two models indicates that the decline will be greatest on Chatham Rise, suggesting a significant reduction in food supply for food webs and fisheries, whereas the second model suggests a potential increase in flux in this region.

The most vulnerable regions were identified where regional extremes occurred in three or more climate variables (see Figure 12.21). Both ESMs identified Chatham Rise as the most significant vulnerable area, with a number of parameters increasing or decreasing in this region. Polar waters south of New Zealand and subtropical waters north of 30° S were also identified as vulnerable.

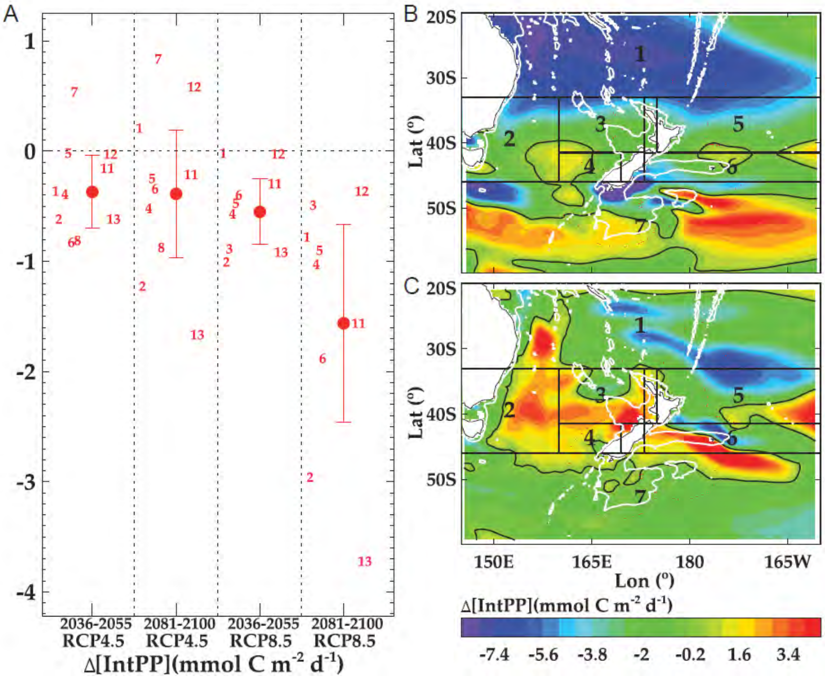


Figure 12.20: Projections of mean change in mean integrated primary production (ΔNPP, ±1 standard deviation) for Mid and End-Century under RCP4.5 and 8.5 (delineated by vertical dashed lines) from the best ESMs. Negative values indicate a decrease in rate and the horizontal dashed line indicates zero change. The numbers indicate the means for each ESM. Regional variation of the projected change in NPP for the End-Century under RCP8.5, using B, ESM2 and C, ESM5. The regional boxes are indicated by number, the white contour is the 1000 m isobath, and the black contours indicate zero change (from Law et al. 2018b).

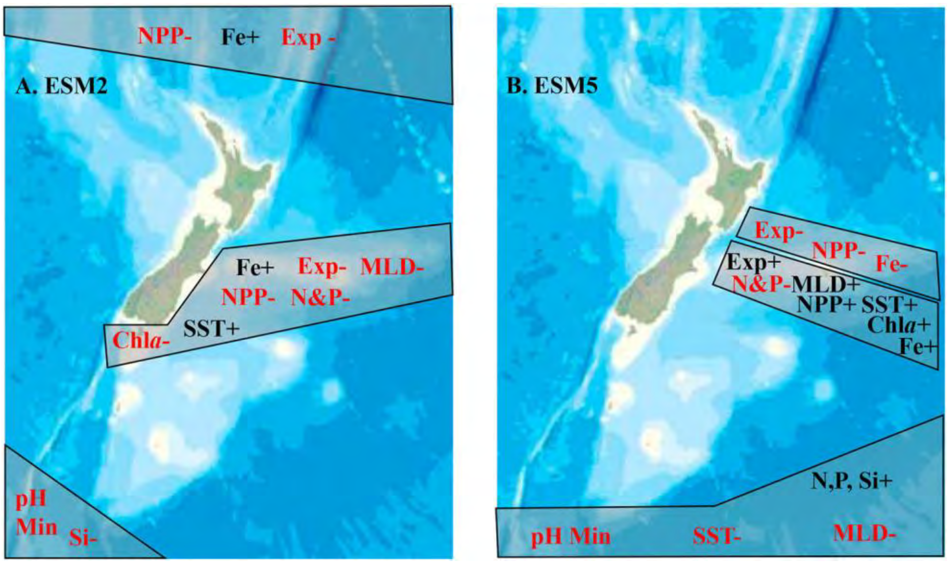


Figure 12.21: Regional extremes for climate-sensitive variables in the surface ocean around New Zealand projected for 2080–2100 using A) ESM2 and B) ESM5. The shaded areas represent potentially vulnerable regions, where two or more variables show significant change relative to the New Zealand mean. The change response is indicated by colour and sign, with a significant decrease indicated in red with a – symbol, and a significant increase in black with a + symbol. Key: SST: Sea Surface Temperature; MLD: Mixed Layer Depth; N&P: Nitrate and Phosphate; Si: Silicate; Fe: Dissolved Iron; Chl*a*: Chlorophyll-a; NPP: Integrated Primary Production; Exp: Particle flux; pH Min: lowest regional pH (from Law et al. 2018b).

12.3 OCEAN CLIMATE TRENDS AND NEW ZEALAND FISHERIES

Climate change could significantly affect New Zealand fisheries productivity, and both ocean warming and acidification may directly affect aquaculture, with indirect affects through changes in phytoplankton production also possible (Pinkerton 2017). Fishery and aquaculture industries are considering adaptation strategies, such as changing harvests and relocating farms (Pinkerton 2017). There is currently little information on how fish species and stocks around New Zealand will respond to climate change and ocean acidification. It has been postulated that distribution of habitat forming deep-sea corals may indirectly affect the future distribution of deep-sea fish that utilise these habitats (Law et al. 2018a). Cummings et al. (2021) summarised New Zealand-specific information on how changes in some of the climate variables might affect fisheries and aquaculture and noted a widespread lack of relevant data.

There have been recent quantitative projections for New Zealand fish species. Dunn et al. (2009) investigated variations in abundance and commercial catch rates of 56 New Zealand fish in relation to 20 climate indices over periods of 5–30 years; they found no consistent changes in abundance or catchability of "warmer-water" or "colder-water" species over this period. They did however find evidence of statistically-significant relationships between climate variables (e.g. SST, sea surface height, Southern Oscillation Index, and Kidson regimes (Kidson 2000)) and the productivity (i.e., biomass indices or year class strengths) of six species of New Zealand fish: macrourids *Coelorinchus aspercephalus*, *Coelorinchus bollonsi*, *Coelorinchus fasciatus*, and *Coelorhynchus oliverianus*; chimaeras *Hydrolagus novaezealandiae* and *Hydrolagus bemisi*; and ling *Genypterus blacodes*. Hurst et al. (2012) reported some general observations on recent trends in some of the key ocean climate indices that have been found to be correlated with a variety of biological processes among fish (including recruitment fluctuations, growth, distribution, productivity, and catch rates).

Recent studies have been undertaken to unravel fishing, climate change, and ocean acidification impacts (Cornwall & Eddy 2015, Christian & Holmes 2016, Morrongiello et al. 2021, Parsons et al. 2020, Neubauer et al. in prep., Dunn et al. in prep.).

A study of temperate New Zealand coastal ecosystems for the Wellington south coast suggested fishing may have a larger effect on trophic group biomasses and trophic structure than ocean acidification; the effects of ocean acidification were only large in the absence of fishing. The model results also suggested that ocean acidification may indirectly benefit certain species due to a reduction in predation (Cornwall & Eddy 2015). Under climate change scenarios, Christian & Holmes (2016) predicted that the available habitat for albacore tuna (*Thunnus alalunga*) in New Zealand waters would become much more extensive and would shift about two degrees latitude further south. A southern shift of oceanic tuna around New Zealand was also predicted by Cummings et al. (2021).

Recent tank experiments assessing the impact of elevated pCO_2 on yellowtail kingfish (*Seriola lalandi*) larvae demonstrated no effect on activity, growth, or survival but did result in reduced swimming speed and distance moved in response to startle stimuli and reduced resting metabolism at elevated pCO_2 (Laubenstein et al. 2018, Watson et al. 2018). Development of digestive organs in larval kingfish was not exacerbated by near-future ocean acidification and warming (Frommell et al. 2019). Similar experiments conducted on snapper (*Chrysophrys auratus*) suggested a range of both positive and negative effects of elevated pCO_2 on survival, distance moved from startle stimuli, aerobic scope, critical swimming speed, and hearing ability (Parsons et al. 2020). For juvenile snapper, elevated CO_2 had a greater effect on metabolic rates and swimming performance than heatwave conditions, which could reduce their overall performance and potentially have negative consequences for population recruitment (McMahon et al. 2020). A modelling assessment attempting to estimate the combined effect of elevated pCO_2 and temperature on New Zealand snapper populations was unable to determine if this effect would be positive or negative, with sensitivities suggesting both increases and decreases in fisheries yield (Parsons et al. 2020).

Morrongiello et al. (2021) showed from long-term growth chronologies of four commercially important fishes from New Zealand's coastal and shelf waters — demersal inshore species snapper and tarakihi (*Nemadactylus macropterus*) and deeper water species hoki (*Macruronus novaezelandiae*) and ling — that fishing and environmental factors can conflate to initially promote individual fish growth but then possibly heighten the sensitivity of stocks to environmental change. Regional-scale wind and temperature affected growth of tarakihi and snapper, whereas deepwater hoki and ling growth was sensitive to the Interdecadal Pacific Oscillation (IPO).

Black et al. (2021) showed that resilience of species to stressors and mortality decreases sharply with increasing depth and that temperature alone explains up to 30% of variation in growth for populations of deep-sea fishes including for several commercial species located in the Australasian region; e.g., orange roughy (*Hoplostethus atlanticus*), alfonsino (*Beryx splendens*), hoki, and various deep-sea oreo species (Zeiformes). The meta-analysis emphasised the need to accurately model growth rates of deeper-living fish to evaluate their resiliency to escalating anthropogenic disturbance.

Brooks (2020) used ecological niche modelling (Maxent) to predict current and future hoki distribution around New Zealand. The models were trained on catch data from the Fisheries New Zealand research trawl database and remote-sensed environmental data. Under less severe climate change scenarios, by 2100 hoki habitat was predicted to be lost from the southern Sub-Antarctic. Under more severe climate change scenarios, hoki habitat was predicted to contract to the south Chatham Rise and Subtropical Convergence Zone around southern New Zealand and be lost from off the west coast South Island. The main predictors of these changes were sea surface temperature and salinity.

Dunn & Jones (2013) suggested that John dory (*Zeus faber*) may have extended their distribution further south off the west coast of the South Island from around 2000 as a result of ocean warming. Langley (2020) estimated a larger increase in abundance of the warm-water snapper off the north and west coasts of the South Island at around the same time. Brooks (2020) also identified the west coast South Island as a focal area for potential species range shifts and that predicting change around the North Island was less certain because novel conditions would occur there in the future.

General observations on recent trends in some of the key ocean climate indices that have been found to be correlated with a variety of biological processes among fish (including recruitment fluctuations, growth, distribution, productivity and catch rates) include the following.

- The Interdecadal Pacific Oscillation (IPO): available from 1871; time scale 15–30 years. The IPO has been found to have been correlated with decadal changes ('regime shifts') in north-east Pacific ecosystems (e.g., Alaska salmon catches). In the New Zealand region, there is evidence of a regime shift into the negative phase of the IPO in about 1998. During the positive phase, from the late 1970s to 1998, New Zealand experienced periods of enhanced westerlies, with associated cooler air and sea temperatures and enhanced upwelling on western coasts. Opposite patterns are expected under a negative phase. For most New Zealand fisheries, monitoring of changes in populations began in the late 1970s, so there is little information on how New Zealand fish stocks might respond to these longer-term climatic fluctuations. Some of the recent changes in fish populations since the mid-1990s, for example, low western stock hoki recruitment indices (Francis 2009) and increases in some elasmobranch abundance indices (Dunn et al. 2009) may be shorter-term fluctuations that could be related to regional warming and longer-term monitoring is necessary to establish whether they might be related to longer-term ecosystem changes. The Southern Oscillation Index: available from 1876; best represented as smoothed values over at least 3–5 months. Causal relationships of correlations of SOI with fisheries processes are poorly understood but probably related in some way to one or more of the underlying ocean climate processes such as winds or temperatures. When the index is strongly negative, an El Niño event is taking place and New Zealand tends to experience increased westerly and south-westerly winds, cooler sea surface temperatures, and enhanced upwelling in some areas. Upwelling has been found to be related to increased nutrient flux and phytoplankton growth in areas such as the west coast South Island, Pelorus Sound, and north-east coast of the North Island (Willis et al. 2007, Zeldis et al. 2008). El Niño events are likely to occur on 3–7-year time scales and are likely to be less frequent during the negative phase of the IPO which began in about 1998. This is likely to impact positively on species that show stronger recruitment under increased temperature regimes (e.g., snapper; Francis 1993, 1994a, 1994b).

- Surface wind and pressure patterns: available from the 1940s. Variation in pressure patterns can be high over monthly and annual time scales and many of the indices are correlated with each other and with SOI and IPO indices (e.g., more zonal westerly winds, more frequent or regular cycles in southerlies in the positive IPO, 1977–2000). Correlations with biological processes in fish stocks may occur over short time scales (e.g., impact on fish catchability) as well as seasonal and annual scales (e.g., impact on recruitment success). Wind and pressure patterns have been found to be correlated with fish abundance indices for southern gemfish (*Rexea solandri*) (Renwick et al. 1998); hake, red cod (*Pseudophycis bachus*), and red gurnard (*Chelidonichthys kumu*) (Dunn et al. 2009); rock lobster (*Jasus edwardsii*) (Booth et al. 2000), and southern blue whiting (*Micromesistius australis*) (Willis et al. 2007, Hanchet & Renwick 1999). The mechanisms implied by the correlations are at best poorly understood; however, they motivate hypothesis testing into the relationship between wind and pressure patterns and fisheries.
- Temperature and sea surface height: available at least monthly over either long time scales (air temperatures from 1906) or relatively short time scales (ocean temperatures to 800 m since 1986 along two transects and more generally to 2000 m since 2006, satellite SST since 1981, SSH since 1993, and ocean temperatures to 2000 m since 2006). Ocean temperatures, SST, and SSH are all correlated with each other and smoothed air temperatures correlate well with SST in terms of inter-annual and seasonal variability; there are also some correlations of SST and SSH with surface wind and pressure patterns (Dunn et al. 2009). Sea surface temperature has been found to be correlated with relative fish abundance indices (derived from fisheries and/or trawl surveys) for elephantfish (*Callorhynchus milii*), southern gemfish, hoki, red cod, red gurnard, school shark (*Galeorhinus galeus*), snapper, stargazer (*Kathetostoma giganteum*), and tarakihi (Francis 1994a, 1994b, Renwick et al. 1998, Beentjes & Renwick 2001, Gilbert & Taylor 2001, Dunn et al. 2009). Air temperatures in New Zealand have increased since 1900 with most of the increase occurring since the mid-1940s. Increases from the late 1970s to 2000 may have been moderated by the positive phase of the IPO. Coastal SST records from 1954 at Portobello show a slight increase through the series. Satellite SST products indicate variable warming of 0–0.2 °C/decade since 1981 across almost the entire New Zealand region, with statistically insignificant changes in the Southern Ocean. Other time series (SSH, ocean temperature to 800 m) are comparatively short but show cycles of warmer and cooler periods on 1–6-year time scales. All air and ocean temperature series show the significant warming event during the late 1990s which has been followed by a relatively stable period. There have been two recent strong marine heatwaves.
- Ocean colour and upwelling: these are important time series because they potentially have a more direct link to biological processes in the ocean and are more easily incorporated into hypothesis testing. The ocean colour series starts in late 1997, so is not available for use to study changes that occurred before the late 1990s warming event. These indices also need to be analysed with respect to SST, SSH, and wind patterns at similar locations or on similar spatial scales. Preliminary series developed exhibit some important spatial differences and trends that may warrant further investigation in relation to fish abundance indices. Of note are increased chlorophyll indices off the west and south-west coasts of the South Island in spring/summer during the last 5–6 years and relatively low upwelling indices off the west coast South Island during winter in the late-1990s (Hurst et al. 2012). Chiswell & O'Callaghan (2021) found that there has been no significant decrease in the occurrence of upwelling events on the West Coast, South Island, but they did find that there has been general warming and that both warm and cold events have also showed warming.
- Currents: there are no general indices of trends or variability at present. Improvements in monitoring technology (e.g., satellite

observations of SSH; conductivity, temperature, and depth; acoustic Doppler current profile; Argo floats) have resulted in more information becoming available to study currents directly and enable the development of numerical models. Recent analyses of the currents along the eastern New Zealand margin (Fernandez et al. 2018) indicate that the currents are highly variable and that there is little coherence between water transports along the boundary. Furthermore, there are no discernible trends in the transports since the SSH measurements began in 1993. On the open ocean scale, there is considerable complexity in the New Zealand zone (e.g., frontal systems, eddy systems off the east coast). The coastal zone is further complicated by the effects of tides, winds, and freshwater (river) forcing. Nevertheless, the importance of current systems is starting to become more recognised and has been incorporated into analysis and modelling of fisheries processes and trends. Recent examples include the retention of rock lobster phyllosoma (mid-stage larvae) in eddy systems (Chiswell & Booth 2005, 2007), the apparent bounding of orange roughy nursery grounds by the presence of a cold-water front (Dunn et al. 2009), and the drift of toothfish eggs and larvae (Hanchet et al. 2008).

- Acidification: Recent model projections indicate a decrease in mean pH by about 0.33 to 7.77 by 2100. A 20-year New Zealand time series shows acidification of Sub Antarctic Water consistent with the increase in atmospheric CO_2. This timeseries is now complemented by a network of coastal stations that also highlight regional variation in pH. Some coastal waters, such as the Firth of Thames, already experience a seasonal decline in pH to 7.9, confirming that New Zealand coastal waters already experience greater acidification than open ocean waters (Law et al. 2018a). A review of the impacts of ocean acidification on New Zealand species and ecosystems (Law et al. 2018a) identifies that organisms with carbonate shells and skeletons are potentially most sensitive to a decrease in pH and that early life-history stages (eggs/larvae) may be particularly susceptible.
- Climate change was not specifically addressed as part of the report by Hurst et al. (2012), although the indices described are integral parts of monitoring the progress and impacts of climate change. As noted in the air temperature section, the slightly increasing trend in temperatures since the mid-1940s is likely to have been moderated by the positive phase of the IPO from the late 1970s to the late 1990s. With the shift to a negative phase of the IPO in 1998, it is likely that temperatures will increase more steeply. Continued monitoring of the ocean environment and response is critical. This includes not only the impacts on productivity, at all levels, but also increasing ocean acidification.

In conclusion, key ocean climate drivers in the New Zealand region for the last few decades have been:

- the significant warming event in the late 1990s;
- the shift to the negative phase of the IPO in about 1998, which is likely to result in fewer El Niño events for a 20–30-year period, i.e., fewer zonal westerly winds (already apparent compared with the 1980–2000 period) and increased temperatures — this is the first IPO change to occur since the start of most of our fisheries monitoring time series (the previous shift was in the late 1970s);
- global trends of increasing air and sea temperatures and ocean acidification; and
- recent strong marine heatwaves in summer 2017–18 and 2018–19.

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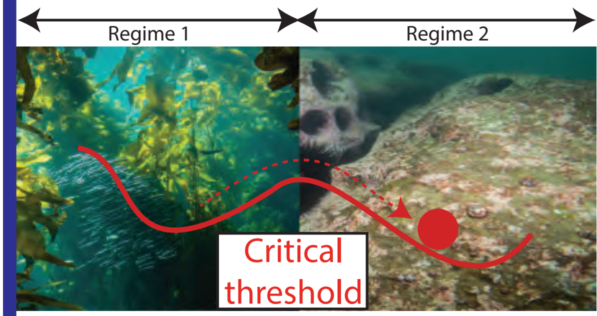
Chapter 13: Trophic and ecosystem-level effects - Technical Summary

Altered food webs, declining fish productivity



Baseline and regime shifts

1. THE ISSUE IN BRIEF

- Marine ecosystems are dynamic, and fluctuate naturally between different states
- Among other causes (e.g. climate change), fishing can create an imbalance by selectively removing species, which are prey and predators of other species in the food-web
- Ecosystems tend to correct any imbalance, but sometimes alterations to the food web functioning are long-lasting
- Understanding, evaluating, or predicting these changes (as well as separating natural and anthropogenic causes) is challenging, but progress has been made

2. DIFFERENT LEVELS OF EFFECTS

- Trophic and ecosystem-level effects can be ranked based on their magnitude and spatial scale
- At the local scale, effects of activities such as offal discard by fishing vessels can promote scavenging by seabirds, which can have a direct effect on the feeding behaviour of some species
- First order effects take place when the predator/prey balance is disrupted
- Second order effects influence many trophic levels, at larger scales, and are also known as trophic cascades
- Large-scale effects are called regime shifts, and can affect whole ecosystems, occasionally leading to a permanently altered ecological state

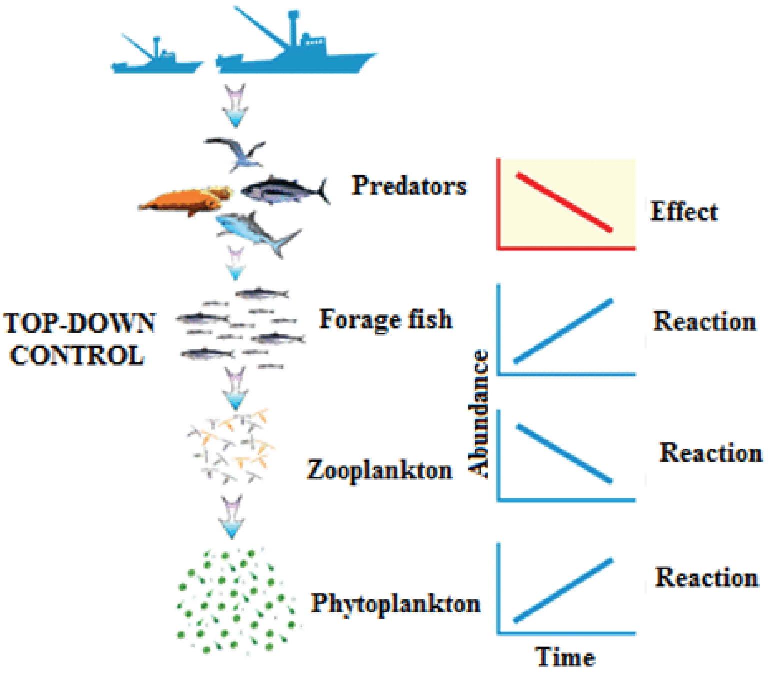
3. EXAMPLES OF FISHERIES-INDUCED CHANGES

Fishing pressure may alter fish populations
- demographic and genetic structure
- spatial distribution
- life-history traits (e.g. age at maturity)

as well as habitat structure and trophic structure of the food web, depending on fishing pressure and extent

Fishing typically affects food webs through:
- Top-down control (e.g. predator removal)
- Middle-out control (e.g. small demersal fish removal, thus altering nutrient transfer)

- Selective fishing pressures on key components of the food web are more likely to have significant effects
- Detection and scale of effects is a complex issue, and
- The use of multiple indicators can separate them from natural fluctuations



Schematic fisheries-induced top-down effects (from Cury et al. 2004)

13 TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

Status of chapter	This chapter has not been updated for AEBAR 2021. Each year, different chapters in the AEBAR are prioritised for update. This chapter was first published in AEBAR 2017 and, apart from minor editorial errors, has not been prioritised for update since then. It may therefore contain material that is potentially out of date or has been superseded in the scientific literature.
Scope of chapter	This chapter outlines the global and New Zealand understanding of trophic and ecosystem-level effects of fishing, with respect to types of effects, their causes, the types of ecosystems most likely to be affected, the spatial scales of effects, and indicators of trophic and ecosystem-level effects.
Area	All areas and fisheries.
Focal localities	Whole EEZ.
Key issues	Organisms in an ecosystem are linked by trophic (feeding) connections. Changes to one organism (by whatever means) can affect other organisms and sometimes large parts of the food web. Changes occurring across many trophic levels (ecosystem-level changes) can have implications for ecosystem resilience.
Emerging issues	Ecosystem approach to fisheries and how fishing interacts with other stressors of marine ecosystems.
MPI research (latest)	ZBD200505 *Long term change in New Zealand coastal ecosystems*; HMS2014-05 *Stable isotope analysis of highly migratory species to assess trophic linkages and spatial and temporal movement trends of HMS sharks*.
NZ government research (current)	• NIWA core funding - Coasts & Oceans centre: 'Ecosystem structure and function' and 'Marine Biological Resources'; Fisheries centre: 'Ecosystem effects of fishing'. • Climate Change Impacts and Implications (MBIE Contestable, http://ccii.org.nz). • Marine Futures (MBIE Contestable, http://www.niwa.co.nz/coasts-and-oceans/research-projects/marine-futures).
Related chapters/issues	• Effects of fishing on ecologically dependent species. • Benthic impacts of fishing (including habitats of particular significance for fisheries management). • Climate and oceanographic context of New Zealand fisheries (including effects of climate variability and change). • Land-based effects on fisheries. • Marine biodiversity. • Other work on fish stocks, marine mammals, seabirds, bycatch, etc.

13.1 CONTEXT

13.1.1 SCOPE OF CHAPTER

This chapter addresses trophic and ecosystem-level effects which may arise from fishing or from other drivers of change on marine ecosystems in the New Zealand region. 'Trophic effects' are changes to the structure and function of ecosystems occurring entirely or largely because of changes in the feeding of organisms within a food web. 'Ecosystem-level effects' are defined as changes occurring across several trophic levels.[1] An ecosystem is defined as a biological community of interacting organisms and their physical environment. The region of interest for the purposes of this chapter is the New Zealand marine exclusive economic zone (EEZ) and territorial waters, including coastal and offshore regions. The focus is on wild-caught fisheries rather than aquaculture.

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[1] 'Trophic level' is a measure of the position of an organism within a food web. Primary producers have trophic level 1, herbivores have trophic level 2, and carnivores have trophic levels between about 3 and 5 in aquatic systems (Lindeman 1942).

This chapter focuses on trophic and ecosystem-level effects that are relevant to the sustainability and environmental effects of New Zealand fisheries as set out in the relevant New Zealand legislation, current New Zealand government strategic/operational policies, and international best practice. Relevant legislation, policies and best practices are summarised in Chapter 1 (Sections 1.2 and 1.3). The relevance of these specifically to trophic and ecosystem-level effects include:

- The Fisheries Act 1996 requires that (a) associated or dependent species should be maintained above a level that ensures their long-term viability; (b) biological diversity of the aquatic environment should be maintained.
- MPI's Strategy 'Our Strategy': to grow the sustainable use of our natural resources.[2]
- FAO best practice requires the application of scientific methods and tools that go beyond the single-species approaches: '*Managers and decision-makers must now explicitly consider interactions in the ecosystem*' and scientific advice should include ecosystem considerations (FAO 2008).
- Marine Stewardship Council (MSC) Principle 2: '*Fishing operations should allow for the maintenance of the structure, productivity, function and diversity of the ecosystem (including habitat and associated dependent and ecologically related species) on which the fishery depends*.' (Marine Stewardship Council 2010). This only applies to those fisheries that are MSC certified.

Effects of fishing on target species are considered in the annual New Zealand Fisheries Assessment Plenary (Ministry for Primary Industries 2014) The Fisheries Assessment Plenary also includes consideration of the effects of fishing on the aquatic environment (under the 'environmental and ecosystem considerations' section for each stock). Effects of fishing all stocks on protected species, non-protected bycatch species, and on the benthos are given in other chapters of this AEBAR document. In particular, effects of fishing on seabirds and marine mammals which occur through trophic connections (e.g., fishing affecting the availability of prey for seabirds) are considered in Theme 1 of this report.

13.1.2 WHAT ARE TROPHIC AND ECOSYSTEM-LEVEL EFFECTS?

Trophic and ecosystem-level effects are changes to multiple parts of the food web. Such effects can occur in coastal or deepwater ecosystems and can involve a wide range of biological, chemical and physical processes. Because trophic and ecosystem-level effects occur over a range of different organisms and time/space scales, it is often difficult to be sure of the magnitude of the change or its underlying cause. This has led to much speculation and disagreement as to the mechanism or processes involved, and a corresponding high level of disagreement as to what management should have done to prevent it, or should do to respond to the change once it has occurred (Schiermeier 2004, Hilborn 2007, Murawski et al. 2007, Schiel 2013). Sometimes controlled experiments are conducted to see if trophic effects can be simulated, but low statistical power is a common problem of this kind of test (Schroeter et al. 1993). In general, international research on trophic and ecosystem-level effects is active and one where there are generally more hypotheses than well-accepted empirical demonstrations of the effects. It is probably useful to start with a few examples of some trophic and ecosystem-level effects.

As part of the widespread pattern of collapses of cod (*Gadus morhua*) populations in the North Atlantic in the late 1980s and the 1990s, cod biomass off the US East Coast dropped by a factor of five, from more than 150 000 metric tonnes (MT) to about 30 000 t (Mayo et al. 1998). With some slight lag, local stocks of the cod's favoured prey, Atlantic herring (*Clupea harengus*), increased over the same period 20-fold, to nearly 2 million t (NEFSC 1998). Elsewhere, on the opposite side of the Atlantic, a collapse of the cod resource in the Baltic Sea was followed by an eight fold increase in abundance of European sprat (*Sprattus sprattus*) – a major prey item for cod in that ecosystem (Köster et al. 2003b, Casini et al. 2008, 2009). In these cases, a reduction in the abundance of a piscine predator by fishing led to an increase in the prey species – a large-scale 'predation release' effect (see Section 13.1.3.1).

In New Zealand, observations in a number of northern marine reserves showed an increase in the abundance and

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[2] Ministry for Primary Industries. Our Strategy. http://www.mpi.govt.nz/about-mpi/our-strategy.

size of red rock lobsters and piscine predators of algal grazing invertebrates which coincided with a gradual decrease in urchin density and an increase in algal cover (Babcock et al. 1999, Shears & Babcock 2002, 2003, Salomon et al. 2008, Babcock et al. 2010). These changes, suggestive of a trophic cascade (see Section 13.1.3.2) are consistent with the results of ecosystem models of the role of rock lobsters in New Zealand rocky reef ecosystems, using both qualitative (Beaumont et al. 2009) and quantitative frameworks (Pinkerton et al. 2008, Eddy et al. 2014, Pinkerton 2012). Shears et al. (2008) found that the occurrence of this trophic cascade in northern New Zealand was likely to vary at local and regional scales in relation to abiotic factors. From a New Zealand-wide perspective, Schiel (2013) concludes that urchin predators play a role in the dynamics of kelp beds only in some northern localities, and that environmental and climatic influences, species' demographics, and catchment-derived sedimentation are generally more important.

13.1.3 TYPES OF TROPHIC AND ECOSYSTEM-LEVEL EFFECTS

13.1.3.1 FIRST ORDER TROPHIC EFFECTS: PREY AVAILABILITY AND PREDATION RELEASE

Changes to the abundance, size structure and functional type[3] of a species can affect both its predators and prey by trophic interactions (Pace et al. 1999). Increasing the abundance of a prey species may positively affect its predators (because they have to work less hard to find food) whereas reducing the abundance of a prey item may have a detrimental effect on the predators (by requiring them to hunt more intensively or by forcing a change in their diet); these are 'prey availability' or bottom-up effects (Trillmich et al. 1991, Jahncke et al. 2004). Alternatively, changing the abundance of a predator may affect the abundance of some or all of its prey by changing their natural mortality rates (a top-down effect; Northcote 1988). Decreasing the abundance of a predator (for example by fishing a predatory fish) may cause the abundance of some or all of its prey to increase (a 'predation release' effect; Casini et al. 2012). These effects act over one trophic link and are hence called 'first order' trophic effects.

13.1.3.2 TROPHIC CASCADES

Changes in the abundance of one species may go on to affect other species that are neither its predators nor its prey. This is a second-order trophic effect (occurring via an intermediate organism), often called a 'trophic cascade'. The awareness of trophic cascades arose originally from work in the marine intertidal zone, and lakes (Hrbácek et al. 1961, Shapiro et al. 1975, Paine 1980), but has since become the focus of considerable theoretical and empirical research in marine ecosystems (Carpenter et al. 1985, McQueen & Post 1988a, 1988b, Christoffersen et al. 1993, Pace et al. 1999, Frank et al. 2005, Borer et al. 2005, Daskalov et al. 2007, Möllmann et al. 2008, Casini et al. 2009, Schiel 2013). While the term trophic cascade was originally termed for top-down effects of predators, it is now usually defined as the propagation of indirect effects between nonadjacent trophic levels in a food chain or food web, whatever the direction of forcing (Gruner 2013). Thus, trophic cascades may also occur when changes in the populations of primary producers force changes at higher tropic levels (Beaugrand & Reid 2003, Bakun 2010). The potential for cascading effects of fishing in marine ecosystems is now thought to be as strong as or stronger than in freshwater ecosystems (Pace et al. 1999, ICES 2005, Borer et al. 2005).

A well-recognised example of a top-down cascade is the sea otter (*Enhydra lutris*), urchin (*Strongylocentrotus* spp.), kelp (*Macrocystis pyrifera* and other kelps) cascade in the north-east Pacific where hunting of sea otters in the eighteenth and nineteenth centuries allowed urchin populations to increase leading to over grazing of kelp beds (Szpak et al. 2013). Protection of sea otters and subsequent expansion or reintroduction of populations into its former range reversed this cascade (Estes & Palmisano 1974, Estes 1996, Estes & Duggins 1995). The generality of the sea otter-urchin-kelp cascade has been questioned; for example, based on experimental treatments, Carter et al. (2007) concluded that 'the sea otter-trophic cascade paradigm is not universally applicable across locations or habitat types.'

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[3] 'Functional type' refers to the collection of life history and ecological characteristics of an organism, including whether it is an herbivore, carnivore or omnivore, its feeding behaviour (including size of prey), location in the water column/benthos, and mobility.

Where ecosystems are subject to stressors acting on different parts of the system together, changes due to cascading trophic effects can be extensive. For example, using field data collected over a 33-year period, Casini et al. (2008, 2009) showed a four-level community-wide trophic cascade in the open waters of the Baltic Sea. The dramatic reduction of the cod (*Gadus morhua*) population directly affected its main prey, the zooplanktivorous sprat (*Sprattus sprattus*) and indirectly the summer biomass of zooplankton and phytoplankton. Changes to the stock size of cod also affected the type of ecosystem control at the level of zooplankton. The cod-dominated configuration was characterized by low sprat abundance and independence between zooplankton and sprat variations (zooplankton abundance was controlled by oceanographic forcing). An alternate sprat-dominated configuration also existed in which cod biomass was low and zooplankton were strongly controlled by sprat predation (Casini et al. 2009).

13.1.4 REGIME SHIFT AND INVASIVE SPECIES

An ecosystem can change to an alternative state if perturbations are greater than its resilience can accommodate – this transition is called a regime-shift (Aebischer et al. 1990, Estes & Duggins 1995, Beaugrand et al. 2002, Daskalov et al. 2007). Regime shifts can occur over large scales, affect many parts of the ecosystem and may be hard or slow to reverse ('hysteresis'). It has been suggested that ecosystem-level restructuring may maintain the system in its new state by means of negative feedbacks (Bakun 2006, Casini et al. 2009, Möllmann et al. 2009, Lindegren et al. 2010). Well-documented oceanographic-induced regime shifts in marine ecosystems have historically had substantial, long-lasting and typically (but not always) negative effects on fisheries. For example, during the 1980s, the North Sea experienced a change in hydro-climatic forcing that caused a rapid, temperature-driven ecosystem shift (Beaugrand & Ibanez 2004). In the North Sea the new dynamic regime after the late 1980s favoured jellyfish in the plankton and decapods and detritivores (echinoderms) in the benthos (Kirby et al. 2008, 2009). The cod stocks in the North Sea and central Baltic Sea collapsed simultaneously with the ecosystem changes caused by the large-scale oceanographic changes (Reid et al. 2003, Beaugrand 2004, Weijerman et al. 2005, Casini et al. 2008, Möllmann et al. 2008, Lindegren et al. 2010).

In another type of regime shift, there has been much recent debate as to whether in some regions, more intense, more frequent or more extensive blooms of jellyfish[4] are occurring in response to trophic and ecosystem-level changes in ocean ecosystems (Brodeur et al. 1999, 2002, Mills 2001, Lynam et al. 2006). In an example reported by Bakun & Weeks (2006), a massive ctenophore ('comb jelly') breakout in the early 1990s led to a nearly total collapse of fisheries in the Black Sea. The Black Sea ecosystems' historically dominant zooplanktivore, European anchovy (*Engraulis encrasicolus*), is a small, filter-feeding pelagic fish. In the late 1980s anchovy landings in the Black Sea increased to levels approaching 900 000 t per year. At their maximum, in 1988, the catch of anchovy represented more than 60% of the total fishery catches taken from the Black Sea. As a result of heavy fisheries exploitation, anchovy spawning biomass in the following year declined by more than 85%. Shiganova (1998) reports that in the year after this drastic reduction in anchovy biomass, zooplankton abundance increased markedly. It was at this point, probably due to the enhanced food source, that the biomass of the ctenophore *Mnemiopsi leidyi* (a gelatinous zooplanktivorous species) in the Black Sea increased to a billion tonnes.

Condon et al. (2013) assembled all available published and unpublished long-term time series on jellyfish abundance across the oceans (no data from the New Zealand region) and found evidence of an approximately 20-year oscillation in global jellyfish abundance. Although an overall global increase in jellyfish abundance over the whole observational period 1874–2011 could not be detected, there was a weak but significant overall increase in jellyfish abundance since 1970. Gibbons & Richardson (2013) note that it is clear that we currently do not know whether there are really global increases in jellyfish, but that a more relevant question is whether jellyfish abundances are increasing in areas that are particularly important for humans – i.e., the coastal zone and important fishing areas – because costs of jellyfish blooms in these areas can be considerable. Recent increases in jellyfish abundance may be linked to one or more of: (a) warmer seas that enhance production, feeding and growth rates of jellyfish (Purcell 2005); (b) overfishing of competitors of jellyfish (Daskalov et al. 2007); (c) increased supply of planktonic food for jellyfish associated with eutrophication of coastal waters

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[4] 'Jellyfish' is often taken to include Medusozoa, Ctenophora and Thaliacea (Condon et al. 2013) but should strictly be limited to Medusozoa and Ctenophora (Gibbons & Richardson 2013).

(Parsons & Lalli 2002); (d) the spread of hypoxia, to which jellyfish exhibit greater tolerance than most other metazoans (Vaquer-Sunyer & Duarte 2008, Purcell 2012); and (e) increase of artificial structures in coastal zones that may be habitats for jellyfish polyps (Duarte et al. 2012).

13.1.4.1 EFFECTS OF CLIMATE CHANGE

Internationally and domestically, there is increasing recognition of the potential impacts of climate change on fisheries (IPCC 2007a, 2007b, Valdes et al. 2009, Rice & Garcia 2011). A changing climate may:

- affect individual physiological and behavioural responses of organisms (or some life stages of organisms; Petitgas et al. 2013), which could lead to effects at the population level (Rijnsdorp et al. 2009, O'Connor et al. 2007, Perry et al. 2005);
- change species proportions in fish assemblages (Engelhard et al. 2011, Fulton 2011);
- lead to ocean acidification, which may affect lower food web structure and adversely impact calcifying organisms such as shellfish and corals (Fabry et al. 2008, Cooley & Doney 2009);
- increase climate variability (Collins 2000), which may increase the risk of regime shift (Mullan et al. 2001, Beaugrand 2004);
- change species ranges, which might destabilise species relationships that help maintain ecosystem processes (Rice & Garcia 2011);
- lead to phonological (timing patterns) mismatches of grazers and predators (Sydeman & Bograd 2009);
- lead to invasive species becoming a greater threat (ICES 2005).

The global scientific understanding of how a changing climate may affect marine ecosystems is largely hypothetical to date, but it seems likely that impacts of climate change are likely to be largely trophic or ecosystem-level effects in nature (reviews by Lehodey et al. 2006, Drinkwater et al. 2010, Bakun 2010, Portner & Peck 2010, Ottersen et al. 2010, Overland et al. 2010, Hollowed et al. 2013).

13.1.4.2 POTENTIAL FOR RECOVERY FOLLOWING OVER-DEPLETION

It is possible that trophic and system-level effects of fishing can affect the ability of fisheries to recover (rebuild) following over-exploitation, but this is disputed. Some scientists suggest that after a fisheries collapse the collapsed population often takes much longer to recover than expected based on known biological parameters, the previously observed carrying capacity of the habitat, and the fact that each adult female fish may spawn tens of thousands to millions of eggs (Hutchings 2000, Steele & Schumacher 2000). It is argued that something durable and significant can be done to the ecosystem during over-exploitation and that this inhibits recovery even if fishing mortality is reduced. For example, in the mid-1960s the sardine fishery in the northern Benguela collapsed from a high point of annual catches of about 1.5 million t (Boyer 1996). Meanwhile, the other major fishery resources of the region, hake (*Merlucius paradoxus* and *M. capensis*) and horse mackerel (*Trachurus trachurus capensis*) also fell to low abundance levels and have not recovered (Bakun & Weeks 2006). The suggestion is that sardines previously occupied the key central position in the ecosystem structure and that these exploitable species have now been largely replaced by a combination of 'jelly predators' and pelagic gobies in a stable, alternative ecosystem state (Boyer & Hampton 2001, Lynam et al. 2006, Bakun & Weeks 2006).

One hypothesis for how trophic effects can prevent stock recovery is the 'cultivation/depensation' mechanism (Köster & Möllmann 2000, Walters & Kitchell 2001). In this hypothesis, consider a species X whose adults predate a species Y, but whose recruits are predated by species Y. If adults of X are abundant they can create favourable conditions for their own offspring by reducing the abundance of Y and hence reducing mortality of their pre-recruits. If the abundance of adults of X is reduced by fishing, expansion of Y may prevent re-establishment of the former species by increasing predation on the recruits of X (Folke et al. 2004). A less theoretical example is that of Casini et al. (2008), based on a 33-year time series in the Baltic Sea, that showed the reduction of the cod population by fishing led to increases in abundances of sprat. Sprat, besides being preyed upon by cod, prey heavily on cod eggs and early larvae (Casini et al. 2004). Some authors have concluded that this predation, together with the likelihood that zooplanktivorous cod larvae may suffer food competition with the high sprat population, was probably a significant factor preventing the resurgence of that cod population (Jarre-Teichmann et al. 2002, Köster et al. 2003a, 2003b, Casini et al. 2009).

However, the prevalence of trophic or ecosystem-level effects slowing or stopping recovery after fisheries collapses is disputed. Cardinale & Svedäng (2011) studied the recent recovery of the eastern Baltic cod stock after more than 20 years of low biomass and productivity and concluded that the recovery was driven by a sudden reduction in fishing mortality and occurred in the absence of any exceptionally large year classes. The recovery of the cod stock during a 'cod-hostile' ecological regime is taken by Cardinale & Svedäng (2011) as indicative of fisheries (rather than climate or food web effects) being the main regulator of cod population dynamics in the Baltic Sea. Cardinale & Svedäng (2011) concluded that single species regulation still seems to be a well-functioning approach in handling natural resources, provided that it includes both temporal and spatial aspects of stock dynamics and fleet behaviour.

13.1.4.3 EFFECTS ON SCAVENGING SPECIES

Offal and discards from fishing vessels can be important sources of food for some marine species, and this constitutes a trophic perturbation to the ecosystem. In addition to scavenging of discards, fish are known to prey on biota damaged or revealed by recent trawling (Kaiser & Spencer 1994). This may include benthic prey items not normally available to the fish (Dunn et al. 2009a). Seabird diets (and ecological success) are also potentially affected by availability of offal and discards near the sea surface. Globally, populations of many scavenging seabirds have grown in recent years (e.g., Lloyd et al. 1991) and it is likely that some species have significantly benefitted from fishery discards (e.g., Furness & Barrett 1985, ICES 2005). However, population growth in scavenging seabirds can lead to displacement of other species because of limited suitable breeding habitat (Howes & Montevecchi 1993). For example, in Europe, many tern species have been displaced by larger gull species (Theissen 1986, Becker & Erdelen 1986). This has led in many instances to the culling of the large gulls in order to allow terns to return to their original nesting sites (Wanless 1988, Wanless et al. 1996).

13.2 WHAT CAUSES TROPHIC AND ECOSYSTEM-LEVEL EFFECTS?

As can be seen in the examples given so far, trophic and ecosystem-level effects in marine systems can be caused by a variety of factors, often acting simultaneously. These factors are often called stressors. Stress in this context refers to physical, chemical and biological constraints on the productivity of species, their interdependencies, and on the structure and function of the ecosystem. Stressors can act over various spatial scales (from local to basin-scale) and various time scales (from days to decadal). Stressors can be natural environmental factors or they may result from the activities of humans. Trophic and ecosystem-level effects can occur because of fishing, because of environmental factors entirely disconnected to fishing (especially related to climate variability/change) or by a combination of fishing and environmental variability/change acting together (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004, Schiel 2013). Trophic and system-level effects can also result from outbreaks of disease (Cobb & Castro 2006, Freeman & MacDiarmid 2009, Shields 2011), from the arrival of non-indigenous invasive species (Mead et al. 2013) and from eutrophication in estuarine ecosystems (Daskalov et al. 2007, Oguz & Gilbert 2007, Osterblom et al. 2007, Möllmann et al. 2008). Some of these causes of trophic and ecosystem-level effects are discussed further below.

13.2.1 ENVIRONMENTAL-DRIVEN CHANGE

Marine ecosystem are intimately linked to environmental (climate) forcing (Fasham et al. 2001, Schiermeier 2004, Frank et al. 2007, Mackinson et al. 2009). Variability of climate forcing of the ocean occurs on a wide range of time scales from seasonal periods, to 1–3 year oscillating but erratic periods, to decadal aperiodic variability at 5–50 years, to centennial and longer periods, and can include sudden, large-scale shifts in environmental forcing (Overland et al. 2010). Climate trends (such as due to global warming) are defined as changes that are not cyclical or seasonal and exist over a relatively long period (more than decadal).

There are many examples internationally of trophic and ecosystem-level effects occurring as a result of environmental change affecting the bottom of the food web (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004). For example, during the 1980s, the North Sea experienced a change in hydro-climatic forcing that caused a rapid, temperature-driven ecosystem shift (Beaugrand & Ibanez 2004). This change in sea surface temperature (SST) altered the plankton and negatively affected the recruitment of cod (Beaugrand & Reid 2003, Heath 2005). Changes in the North Sea plankton, following the ecosystem shift, included an increase in microalgae (Kirby et al. 2008), a change in the composition and abundance of zooplankton (Beaugrand et al. 2002), increases in the frequency of jellyfish (Kirby et al. 2009), increases in the abundance of decapod and echinoderm larvae, and a decrease in bivalve larvae (Kirby et al. 2008). Another example of bottom-up effects on upper-trophic-level marine predators is the abrupt decline in local primary and secondary production caused by El Niño/Southern Oscillation (ENSO) events in eastern Pacific boundary currents (Barber & Chavez 1983, Pearcy et al. 1985, Arcos et al. 2001, Hollowed et al. 2001). During these ENSO events, the production of small pelagic fishes can be drastically reduced (Barber & Chavez 1983, Rothschild 1994), and predatory fish, seabirds and pinnipeds, which are dependent on these small pelagic fish have been shown to shift their distributions, suffer reduced productivity, and have increased rates of mortality (Trillmich et al. 1991, Jahncke et al. 2004).

13.2.2 FISHERIES-DRIVEN CHANGE

To some degree, trophic effects will always arise as a consequence of fisheries. As well as reducing the overall abundance of fish, fishing usually reduces the average size of fish in harvested communities and can change the mix of species in a fish community (Pope & Knights 1982, Pope et al. 1987, Dayton et al. 1995). Fishing also has effects beyond changes to the abundance and population structure of target and bycatch species, including (a) the introduction of discarded bycatch/offal/bait into the ecosystem, (b) the alteration of fish behaviour (and potentially genetic make-up) as a result of fishing, and (c) the modification of the benthos by fishing gear. Fishing will certainly lead to changes (of greater or lesser magnitude) in predation pressure on prey species. Marine ecosystems seem to be remarkably resilient to even quite large trophic changes of this kind, but there are clearly limits to this resilience. Virtually all well-documented regime shifts seem to have been initiated from large-scale climate or oceanographic changes rather than excessive fishing pressure. In some cases however, ecosystem-level changes (regime shifts) have been demonstrated empirically to occur in very highly impacted (highly overfished/collapsed) systems as a result principally of trophic effects (Estes & Duggins 1995, Daskalov et al. 2007). For example, the round sardinella (*Sardinella aurita*) stock off West Africa collapsed in the 1970s following exceptionally high catches made possible by oceanographic changes (Bakun & Weeks 2006). This collapse resulted in a substantial and widespread outbreak of grey triggerfish (*Balistes capriscus*), which lasted through the 1970s and 1980s until the sardinella population rebuilt. At that point, grey triggerfish essentially disappeared from the ecosystem again. It seems possible that the juvenile triggerfish, being pelagic plankton feeders, took advantage of the collapse of the sardinella population to temporarily replace it as the dominant nektonic zooplanktivore of the ecosystem through one or more trophic effects. For example: (1) the sardinella collapse may have led to increased zooplanktonic food resources and hence accelerated the production rate of triggerfish; (2) the sardinella collapse may have promoted increased recruitment of triggerfish by reduced predation on their eggs and larvae (Bakun & Weeks 2006).

13.2.3 COMBINED EFFECTS OF FISHING AND ENVIRONMENTAL VARIABILITY/CHANGE

Although there have been few unequivocal empirical demonstrations of large-scale trophic and system-level effects arising solely from fishing, very many studies have pointed to the potential of fishing to lead to trophic and ecosystem-level effects in concert with other factors, such as environmental variability and change (e.g., Winder & Schindler 2004, Brierley & Kingsford 2009, Kirby et al. 2009, Perry et al. 2010). The effects of fishing that may lead to reduced ecosystem resilience (see Table 13.1 for definition of 'ecosystem resilience') include:

- **Alteration of demographic structure.** Size-selective removal truncates the population's age structure and lowers the buffering capacity of the population (its ability to withstand long periods of environmental conditions that are adverse for recruitment). This leads to the prediction that the relative importance of recruitment variability will be greater in exploited populations as has been observed in a comparison between exploited and unexploited fishes in the California Current Ecosystem (Hsieh et al. 2006).
- **Alteration of spatial structure.** The spatial structures of marine fish populations can encompass a wide range of configurations, including patchy populations, networks, and meta-populations (Kritzer & Sale 2004). Removal or curtailment of population spatial structure by fishing is likely to increase the sensitivity of the overall population to climate fluctuations at inter-annual to multi-decadal scales (e.g., Ottersen et al. 2006).

- **Alteration of life-history traits.** Perry et al. (2010) suggest that fishing would be likely to accelerate the response of populations to climate forcing by providing selective pressure to decrease growth rates and decrease age-at-maturity (Law 2000, de Roos et al. 2006).
- **Alteration of habitat structure.** Changes to benthic habitat by the direct effects of fishing may lead to a reduction in ecosystem resilience (Thrush & Dayton 2002).
- **Alteration of ecosystem trophic structure.** Theoretically, ecosystems under intense exploitation are likely to evolve towards stronger bottom-up control (Figure 13). Exploitation leads to a decrease in stock sizes of piscine predators, which may (a) reverse the control structure in top-down ecosystems to bottom-up control, and (b) amplify the control in already bottom-up controlled ecosystems. Multiple weak interactions and generalist predators may stabilise ecosystems by dampening oscillations caused by strongly interacting species (Shin & Cury 2001, Polunin & Pinnegar 2002, Rooney et al. 2006, McCann & Rooney 2009, Johnson et al. 2014) and by preferentially consuming competitively dominant prey species (Brose et al. 2005). Changes to trophic structure by fishing are hence predicted to increase ecosystem variability and reduce resilience (Jackson et al. 2001, Perry et al. 2010).

Theoretically therefore, fishing is predicted to strengthen the relation between oceanographic forcing and ecosystem variability and hence reduce ecosystem resilience. There are limited real-world, empirical examples of this. For example, the regime shifts of the North Sea and central Baltic Sea are considered to have been driven by the combined and synergistic effects of intense fishing and climate variability (Weijerman et al. 2005, Möllmann et al. 2009). Using a 47-year time series, Kirby & Beaugrand (2009) showed that the effects of temperature can be magnified by propagation through indirect pathways in the food web. This 'trophic amplification' can intensify the effect of environmental variability, potentially leading to a new stable or unstable ecosystem state (Scheffer & Carpenter 2003, Muradian 2001, Taylor 2002, Hsieh et al. 2005). Elsewhere, Ottersen et al. (2006) analysed the Arcto-Norwegian cod stock in the Barents Sea over the last 60 years and found evidence of a strengthening of the climate-cod recruitment link during the last decades

13.3 WHAT TYPES OF ECOSYSTEM ARE LIKELY TO BE MOST AFFECTED?

13.3.1 GLOBAL UNDERSTANDING

The scale and significance of trophic and ecosystem-level effects depend on the particular characteristics of the ecosystem as well as on the drivers of change (Pace et al. 1999, Brose et al. 2005, Pascual & Dunne 2006, Brander 2010, Jennings & Brander 2010). Ecosystems appear to be prime examples of complex adaptive systems (Levin 1998, 1999); ecosystems typically have non-linear dynamics, with thresholds (also called tipping-points) and positive and negative feedback loops (Hsieh et al. 2005). The complex behaviour of ecosystems over a wide range of time and space scales coupled with the myriad nature of stressors means that it is hard to forecast the response of ecosystems or establish quantitative estimates of tipping-points to guide management.

A number of multispecies or ecosystem models have been developed that can be used to investigate the potential for trophic and ecosystem-level effects in ecosystems (Plagányi 2007, Plagányi et al. 2014). These include Ecopath with EcoSim (EwE; Christensen & Walters 2004), Atlantis (Fulton et al. 2004, 2005), OSMOSE (Shin et al. 2004, Travers et al. 2009) and a range of models of intermediate complexity (MICE; Plagányi et al. 2014). Multispecies and ecosystem models can provide useful strategic insights for fishery and resource managers (Plagányi 2007, Fulton et al. 2005, Smith et al. 2011). However, there are often differences in model predictions about ecosystem consequences (or lack thereof) of fishing, especially in ecosystem-scale models, so model outputs need to be used cautiously for tactical decisions (Smith et al. 2011). MICE-models (where only part of the ecosystem is modelled) are likely to provide more robust guidance for tactical decision-making (Plagányi et al. 2014).

There have also been attempts to use knowledge of the structure of the food web to suggest types of behaviour and response to fishing and other changes as an alternative to dynamic ecosystem models (Ulanowicz & Puccia 1990, Libralato et al. 2006, Pinkerton & Bradford-Grieve 2014). Rice (2001) concluded that trophic and ecosystem-level effects of fishing depend on the overall type of ecosystem forcing structure. Three patterns of ecosystem forcing structure have been described: (a) top-down forced, (b) bottom-up forced, or (c) forced from the middle outwards or wasp-waisted (13.2). These patterns of ecosystem forcing have been the focus of hundreds of research articles. These three patterns should be considered as modes of forcing (rather like principal components); most real ecosystems will be a mixture of these types of forcing that may change over time (Rice 2001). Indeed, Pace et al. (1999) cautions that 'although there is some descriptive value in the use of top-down or bottom-up control, this motif also creates a false dichotomy.' Nevertheless, identifying dominant patterns of ecosystem behaviour may help to predict or explain the types of trophic and ecosystem-level behaviour resulting from the combined effects of fisheries harvesting, climate variability/change and other human activities (Rice 2001). For example, Pinsky et al. (2011) uncovered a high incidence of fisheries collapse among small, short-lived, middle trophic-level species of a type that are often the wasp-waist of the ecosystem. Even though short-lived species may recover quickly from excessive fishing mortality (Hutchings 2000), changes to them can have substantial impacts on the food web (Duffy 1983, Frederiksen et al. 2004, Crawford 2007)

Table 13.1: Ecosystem resilience.

Fishing can affect ecosystem resilience, the capacity of an ecosystem to absorb disturbance and reorganise while undergoing change so as to retain essentially the same function, structure, identity, and feedbacks (Pimm 1982, Holling 1973, Cohen et al. 1990, Walker et al. 2004). Three measures of ecosystem resilience have been identified:

- Does the ecosystem retain essentially the same function, structure, identity, and feedbacks after perturbation as before (Walker et al. 2004)?
- Do perturbations to one part of the ecosystem spread out and affect biota across many trophic levels or remain localised (i.e., are ecosystem-level changes likely)?
- How long does it take a food web to return to its original configuration when perturbed? Stable (resilient) food webs can absorb more perturbation without undergoing wholesale reorganisation, tend to have low tendency for ecosystem-level trophic cascades (food web perturbations remain local) and have short return times (Walker et al. 2004).

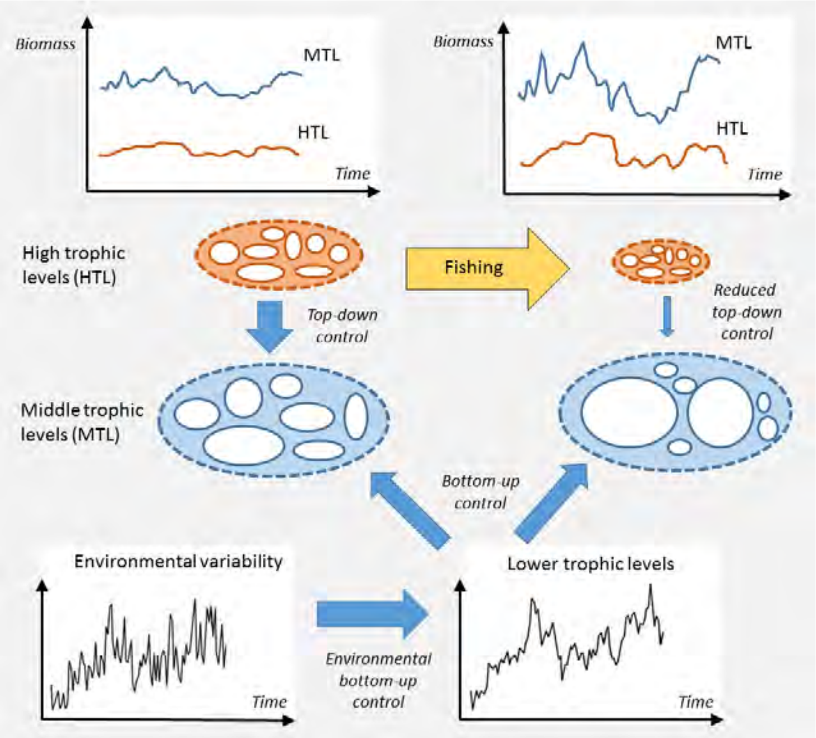

Figure 13.1: Schematic illustrating expected responses of unexploited and exploited marine ecosystems to climate forcing. Left side shows an unexploited ecosystem with multiple high trophic level (HTL) species that have relatively large abundances supported by several mid-trophic level (MTL) species, and how their aggregate biomasses vary through time (top left) in response to environmental variability acting on the lower food web. The right side illustrates how that same climate forcing is experienced by an ecosystem which has been exploited (top right graph). The abundances of the high trophic level species have decreased due to fishing, weakening the top-down control on the MTL. This is hypothesised to make the mid-trophic level groups less even causing their aggregate biomass to track the environmental forcing more closely (after Perry et al. 2010).

Table 13.2: Overall types of ecosystem forcing. [Continued on next page]

Bottom-up ecosystem forcing	If the ecosystem-level properties (i.e., across organisms at many trophic levels) respond strongly to changes in the environment (e.g., oceanography, water column structure), the ecosystem is said to show strong bottom-up forcing. There are many examples internationally of trophic and ecosystem-level effects occurring as a result of environmental changes at the bottom of the food web (Mackinson et al. 2009, Frank et al. 2007, Schiermeier 2004).
Top-down ecosystem forcing	An ecosystem is said to show strong top-down forcing if it responds strongly to changes in the abundance of top predators (seabirds, marine mammals, high trophic level fishes). Understanding of how predators shape marine ecosystems has arisen largely from experimental studies where the effect of predation is controlled either by removing predators or introducing them to the ecosystem under study, usually in the intertidal or nearshore subtidal zone (Hunt & McKinnell 2006 and references therein). In the open ocean, increases in prey populations upon the removal of their predators (e.g., by fisheries) have been taken as evidence of top-down limitation (e.g., Furness 2002, Worm & Myers 2003, Frank et al. 2005). Other evidence of top-down regulation in a marine ecosystem appears where predators are abundant at one site, but largely absent from a similar, nearby site. For example, Birt et al. (1987) found that small flatfish populations were depressed in a bay in Newfoundland that was frequented by cormorants compared to a bay that was located farther from the colony. In general, top-down ecosystem forcing is predicted to be stronger in aquatic than terrestrial ecosystems, and strongest in marine ecosystems where the predators are large and mobile with high metabolic rate, where prey species are long-lived, functional predator diversity is low, and predator intra-guild predation is weak or absent (Shurin et al. 2002, Borer et al. 2005, Heithaus et al. 2008).
Middle-out forced (wasp-waisted) ecosystem	Wasp-waist control of energy flow in marine ecosystems occurs when one or a very few species have a substantial influence on the flow of energy through the mid-trophic levels. The term has most frequently been applied to the role of small pelagic fishes that transfer energy from the plankton to larger predatory fish, seabirds and marine mammals (Rice 1995, Cury et al. 2000, 2004, Bakun 2004, 2006). Ecosystems with wasp-waist control are typically coastal, highly productive systems with relatively short food chains. However, waist-controlled ecosystems also include capelin in North Atlantic ecosystems (Lilly 1993, Bogstad & Mehl 1997, Leggett et al. 1984, Taggert & Leggett 1987, MacKenzie & Leggett 1991, Fossum 1992), krill in the Antarctic (Murphy et al. 1998) and, *Calanus* sp., when functioning as a 'gatekeeper' (sensu Steele 1998). When the species at the waist declines abruptly, predators often cannot compensate, at least fully, and suffer reduced growth, survivorship, and reproduction (Mehl & Sunnana 1991, Kjesbu et al. 1998, Dutil & Lambert 2000). Predators may control the wasp-waist when they are at intermediate population sizes (Bakun 2006). At other times, year-class strengths of species at the waist demonstrate strong, direct effects of environmental forcing. Wasp-waisted ecosystems typically follow from: (1) a food web containing a highly influential intermediate node that has a strong environmental signal in recruitment (Rice 2001) and/or (2) middle-trophic level fishery.

13.3.2 NEW ZEALAND

13.3.2.1 BOTTOM-UP FORCING

A New Zealand example of bottom-up forcing is the driver of mussel (*Perna canaliculus*) yield in Pelorus Sound in northern South Island. Though this example is from aquaculture, it is likely to also apply to wild mussels. Zeldis et al. (2008) correlated physical, chemical and biological data collected within a nine-year time series. Starting in early 1999, farm production in the sound declined by about 25% in terms of per-capita meat yield, followed by yield recovery through to 2002. These changes resulted in substantial economic impacts within the industry. Over-grazing by mussels (i.e., top-down effects on mussel food availability) did not explain the yield minimum. Instead, bottom-up (environmental) effects of nitrogen supply from oceanic and river sources drove the variation by affecting the abundance of seston[5] for the filter-feeding mussels. A subsequent study (Zeldis et al. 2013) provided quantitative models for Pelorus Sound mussel per-capita meat yield and elucidated the underlying oceanographic mechanisms. Yield was best predicted using biological variables, including the concentration of seston, based on measurements made next to the mussel farms, but it was also predictable using only physical variables that index large-scale environmental processes (Southern Oscillation Index, along-shelf winds, sea surface temperature and river flow).

13.3.2.2 TOP-DOWN FORCING

In moderately exposed coastal marine reserves in north-eastern New Zealand, predation by recovering populations of snapper (*Pagrus auratus*) and spiny lobsters (*Jasus edwardsii*) have gradually decreased the abundance of the grazing sea urchin (*Evechinus chloroticus*) and allowed turfing algae and kelp (*Ecklonia radiata*) to replace urchin grazed rock flats (Babcock et al. 1999, Shears & Babcock 2002, 2003). This is indicative of top-down forcing in the ecosystem. In adjacent areas which are heavily fished there are more urchins, and areas free of turfing algae and kelp are common (Shears et al. 2008). It seems that the occurrence of this trophic cascade varies at local and regional scales in relation to abiotic factors, implying some interplay with larger-scale bottom-up forcing (Shears et al. 2008).

A long-term study of changes to the ecosystem of the Hauraki Gulf region developed five balanced, quantitative models of the food web of the region (MPI project ZBD200505: Pinkerton 2012): (1) present day; (2) AD 1950, just prior to onset of industrial-scale fishing; (3) AD 1790, before European whaling and sealing; (4) AD 1500, early Maori settlement phase; (5) AD 1000, before human settlement in New Zealand. These models were used to estimate the strengths of trophic connections between different groups of organisms based on single-step and multiple step measures of trophic importance (Ulanowicz & Puccia 1990, Libralato et al. 2006). Before humans arrived in New Zealand, the models suggest that cetaceans and fur seals/sea lions were the most trophically important groups in the Hauraki Gulf ecosystem, implying the potential for strong top-down ecosystem control. With the extirpation[6] of seals/sea lions from the Hauraki Gulf ecosystem before the arrival of Europeans and the reduction in the abundance of cetaceans following European arrival, the trophic importance of these air-breathing predators drastically reduced. The trophic importance of other predators in the models of the Hauraki Gulf ecosystem also reduced over time as a result of human harvesting (rock lobsters and sharks especially) suggesting a transition to a more bottom-up controlled system.

13.3.2.3 MIDDLE-OUT (WASP-WAIST) FORCING

Research into deepwater ecosystems in the New Zealand EEZ is most advanced in the Chatham Rise region. Elevated primary production here is due to the convergence of subantarctic and subtropical water (Bradford-Grieve et al. 1997, Boyd et al. 1999, Murphy et al. 2001, Sutton 2001) and supports valuable deepwater fisheries, an unusually rich benthic ecosystem (Probert et al. 1996, McKnight & Probert 1997, Bowden 2011), and large seabird populations (Taylor 2000a, 2000b). Ecosystem modelling of the Chatham Rise food web has been underway since 2006, the most recent version being Pinkerton (2013) (Figure 13.2). Trophic impact matrices (Ulanowicz & Puccia 1990, Libralato et al. 2006) were calculated from the balanced model to investigate patterns of trophic interactions. Middle trophic level groups, especially small demersal fishes and mesozooplankton, had some of the highest trophic importances amongst consumers. Mesopelagic fishes, hoki, and arthropods (benthic prawns and shrimps) also had high trophic importances (Pinkerton 2013). These patterns of trophic importance were robust to uncertainties in the model parameterisation and balancing (Pinkerton 2014b). These results suggest some degree of middle-out control in the system, though the number and function diversity of these groups is higher than in other systems characterised in this way.

13.4 OVER WHAT SPATIAL SCALES DO TROPHIC AND ECOSYSTEM-LEVEL CHANGE OCCUR?

13.4.1 GLOBAL UNDERSTANDING

Delineating ecosystems is an important first step towards evaluating trophic and ecosystem-level effects of fishing. There are not usually clear spatial boundaries between

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[5] Organisms and non-living matter swimming or floating in a water body.

[6] Made locally extinct.

different ecosystems. Instead, different parts of ecosystems vary on different spatial scales; higher trophic-level organisms usually move over a greater spatial extent than lower trophic-level organisms. For example, some seabirds and marine mammals may move large distances seasonally and move between different ecosystems. In contrast, most phytoplankton, smaller zooplankton and most benthic invertebrates will live and die within a few kilometres. Some fish move long distances, but others remain in a small area all their lives (e.g., on a reef). Marine ecosystems should hence be viewed as an interlocking matrix of the life ranges of different organisms. As such, it is difficult to unambiguously separate different ecosystems but a number of approaches have been developed to do so. These include: (a) defining ecosystems on the basis of their physical properties, either using a priori thresholds (e.g., fixed depth ranges) or by multivariate clustering of physical properties (Snelder et al. 2005, Grant et al. 2006); (b) using maps of species occurrence to map biological assemblages (e.g., Leathwick et al. 2006); (c) relating community composition to environmental variables (e.g., generalised dissimilarity analysis; Ridgeway 2006, Leathwick et al. 2009) and using these relationships to extrapolate spatially.

13.4.2 NEW ZEALAND

The importance of spatial scale in the study of the ecosystem effects of fisheries has been recognised in New Zealand (e.g., Leathwick et al. 2006, 2009). In their assessment of the New Zealand hoki fishery for the Marine Stewardship Council (MSC), Akroyd & Pierre (2013) noted that there is currently no specific definition of 'regional effects' but MSC is working on adding clarity to the definition of regions and bioregions as part of the work on their current benthic impacts project in recognition that some areas are more vulnerable to impact than others.

A number of approaches have been developed in New Zealand to identify or describe ecosystem types:

- MacDiarmid et al. (2012) identified 62 distinct marine habitat types occurring within New Zealand's Territorial Sea and EEZ as part of an assessment of anthropogenic threats to New Zealand marine habitats. The approach taken by MacDiarmid et al. (2012) was to build on Halpern et al.'s (2007) list of marine habitats used in a global assessment of anthropogenic impacts on the global marine environment.
- New Zealand's Department of Conservation, jointly with MPI, have used a marine habitat classification system based on four depth intervals (intertidal, 0–30 m, 30–200 m, more than 200 m), seven substrate classes (mud, sand, gravel, undefined substrate, mixed sediment and rock, rock, and biogenic), and three exposure categories (exposed, moderate, sheltered). This habitat classification was used to define 58 habitats in the Territorial Sea alone in order to meet the needs of biodiversity conservation (DOC-MPI 2011).
- New Zealand Marine Environment Classification (MEC; Snelder et al. 2005). The MEC is a physically based classification, determined using multivariate clustering of several spatially explicit data layers that describe the physical environment (including depth, slope, orbital velocity at the sea floor, mean solar radiation, SST amplitude, SST gradient, winter SST, mean tidal current velocity). Large biological datasets were used to tune the classification so that the physically based classes maximised discrimination of variation in biological composition at various levels of classification detail. The classification was not optimised for a specific ecosystem component (e.g., fish communities or individual species) but sought to provide a general classification that had relevance to a broad range of biological groups. Depending on user requirements the MEC can provide two to 270 classes of classification.
- Leathwick et al. (2006) demonstrated how spatial analysis using boosted regression trees could provide distribution maps of over 100 species of demersal fish. Fish were chosen as there were good quality distributional data available from a series of scientific trawl surveys in deep waters. The overall approach used by Leathwick et al. (2006) was to fit statistical models relating the distributions of 122 fish species to a set of environmental variables, with the latter chosen for their functional relevance.
- A Benthic-optimised Marine Environment Classification (BOMEC; Leathwick et al. 2009) was developed specifically to identify New Zealand benthic bioregions that can be considered to be ecologically distinct to some degree. BOMEC was developed by combining data on the benthic community (made up of over 100 demersal fish species, and seven groups of invertebrates:

asteroids, bryozoans, foraminifera, octocorals, polychaetes, scleractinian corals, sponges), and environmental data including sediment type. A multivariate technique for fitting community compositions to environmental data, Generalised Dissimilarity Analysis, was used (Leathwick et al. 2009). BOMEC is restricted to depths less than 3000 m where reasonable amounts of scientific sampling have been conducted (Leathwick et al. 2009).

- The Ocean Survey 20/20 Chatham-Challenger biotic habitat classification (Hewitt et al. 2011) used benthic invertebrate and environmental data from the Chatham Rise and Challenger to delineate ecosystems in terms of their community and biogenic habitat associations.
- Sharp et al. (2007) summarised lessons learned from New Zealand's bioregionalisation experience for CCAMLR. The main conclusion was that bioregionalisations based on simple clustering of physical variables are likely to perform poorly in terms of separating assemblages of species (communities or ecosystems); measurements of the actual distributions and abundances of key organisms are needed to use physical environmental data to delineate bioregions effectively.

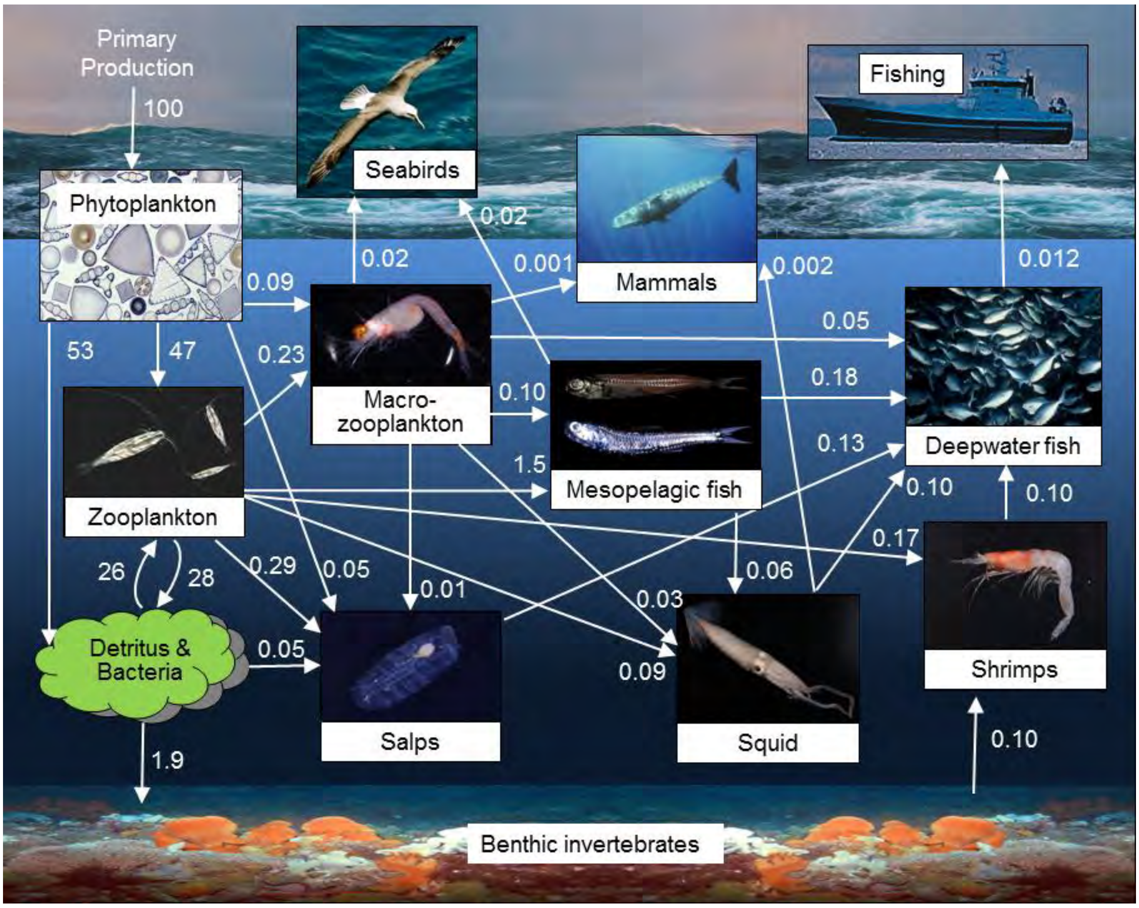

Figure 13.2: Simplified trophic model of the Chatham Rise, New Zealand (based on Pinkerton 2013). The growth of phytoplankton generates organic matter that is the fuel for the marine ecosystem. Figures show the annual flow of energy through unit area of the food web normalised to a net primary productivity (NPP) of100, based on an equilibrium mass-balance model (similar to Ecopath).

13.5 HOW CAN TROPHIC AND ECOSYSTEM-LEVEL EFFECTS BE DETECTED?

13.5.1 GLOBAL UNDERSTANDING

There has been increasing recognition over the last two decades that time series are essential to detect and potentially understand a trophic or ecosystem-level change in marine ecosystems. This has led to a high level of interest in the development and interpretation of indicators of the marine environment and its ecosystems. A huge number (more than 300) of marine ecosystem indicators are in use or proposed around the world (Cury et al. 2005, Rochet & Rice 2005, Rice 2003), with consensus that a suite of indicators is needed to monitor and understand the impact of human activities on marine ecosystems (Cury & Christensen 2005, Rice & Rochet 2005). Give the multi-trophic nature of ecosystem-level effects, indicators are needed that span the ecosystem, including primary producers, the microbial system, middle trophic levels, fish communities, the benthic community and top predators. A summary of some recommended indicators is given below.

13.5.1.1 MARINE PRIMARY PRODUCTION

The growth of phytoplankton in the upper layers of the ocean provides the vast majority of the energy that fuels marine ecosystems, and most fisheries, worldwide. Only in some (predominantly coastal) areas are other primary producers important: macroalgae (seaweed), seagrass, mangroves, epiphytes, autotrophic periphytes, microphytobenthos and chemosynthesisers. Light, temperature, and nutrient concentrations are major factors controlling net [7] primary production (NPP) by phytoplankton growth in the ocean (Parsons et al. 1977, Arrigo 2005). NPP can be measured accurately from ships (typically using radioactive carbon incubations), but because of the high spatial and temporal variability of NPP, ship-based sampling is not adequate for monitoring. Instead, remotely sensed data from sensors on Earth-observing satellites are typically used to estimate NPP. There are significant differences between different methods of estimating NPP from satellite data (Campbell et al. 2002). Often, the concentration of chlorophyll-a, the ubiquitous pigment in phytoplankton, is used as a proxy for phytoplankton biomass and NPP, because this can be measured remotely with better accuracy than NPP using ocean colour satellite sensors.

13.5.1.2 LOWER FOOD WEB (MICROBIAL SYSTEM)

Rice (2001) notes that processes that make large alterations to the allocation of production between the microbial loop, benthic detrital pathways and mesopelagic consumers may have much more impact on the dynamics of higher trophic levels than processes that alter NPP. More recently, Friedland et al. (2012) examined the relationships between NPP, fisheries yields, and parameters describing the transfer of organic matter through 52 large marine ecosystems and found that chlorophyll-a concentration, the particle-export ratio (p-ratio: the proportion of NPP exported from the surface layer of the ocean) and the ratio of mesozooplankton productivity to NPP (z-ratio) were all significantly related to fisheries yields. Stock & Dunne (2010) suggest that a warmer ocean will lead to lower z-ratio (less mesozooplankton for a given NPP) and Friedland et al. (2012) show that lower z-ratios correspond to lower fisheries yields at basin scales.

13.5.1.3 MIDDLE TROPHIC LEVELS

Small mesopelagic[8] and hyperbenthic[9] organisms are an important part of marine ecosystems. They act as the link between the microbial/planktonic system and larger predators such as seabirds, marine mammals, and larger fish. These ‘middle trophic level’ organisms are diverse, and include hard-bodied crustaceans (such as copepods, euphausiids, amphipods, prawns and shrimps), ‘jellies’ (such as jellyfish and salps), cephalopods (squids and octopods), and a range of small fishes (including juveniles of larger species) living in the water column (especially myctophids or lanternfishes) or near the seabed. These species are likely to be affected both by fishing, which may reduce top-down predation control, and by climate-driven changes in lower trophic food web components (Frank et al. 2007, Richardson 2008). Middle trophic level species have a key role in ocean ecology (e.g., Banse 1995, Marine Zooplankton Colloquium 2 2001, Smetacek et al. 2004, Pinkerton 2013). Studying these middle trophic level organisms is challenging: they are typically diverse, with varied and complex life histories, can be hard to capture, and have abundances that vary over a wide range of space and time scales. Consequently, the factors that affect their dynamics are generally poorly understood. Two methods have been used for monitoring middle trophic levels. First, in other parts of the world, long time series of measurements of the zooplankton community by the Continuous Plankton Recorder (CPR) has demonstrated change in marine ecosystem (Beaugrand et al. 2002, Aebischer et al. 1990, Reid et al. 1998, Beare & McKenzie 1999), and been recommended as an effective way of

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[7] ‘Net’ means after allowing for phytoplankton respiration.

[8] ‘Mesopelagic’: inhabiting the intermediate depths of the sea, between about 200 and 1000 m down.

[9] ‘Hyperbenthic’: ecologically associated with the seabed, but living for some time in the lower water column.

monitoring the state of pelagic ecosystems (Beaugrand 2005). Second, multifrequency acoustics have been used to monitor abundances of mesopelagics over extended time and space scales (McClatchie & Dunford 2003, O'Driscoll et al. 2009, Trenkel & Berger 2013).

13.5.1.4 DEMERSAL FISH COMMUNITIES

Most of the international effort on developing ecosystem indicators have focused on those for the demersal fish community, usually based on commercial landings data or, less commonly, on catch data from fisheries surveys. Consequently, very many indicators have been proposed – a selection is discussed below.

- **Marine Trophic Index**: MTI is the mean trophic level of fisheries landings (Pauly & Watson 2005) and was recently recommended for use with commercial catch data by the United Nations Biodiversity Convention as a widely applicable and cost-effective indicator for monitoring reductions in biodiversity loss in marine ecosystems (CBD 2004). A gradual decline in trophic level of about 0.2 since industrialised fishing began has been observed in many finfish fisheries around the world (Pauly et al. 1998a, Christensen et al. 2003), ascribed to fisheries targeting high trophic level species and moving on to lower trophic level species as these large species are depleted, a change called 'fishing down the food web'. Essington et al. (2006) noted that 'fishing through the food web', where higher trophic level fish landings are maintained but catch of lower trophic level species increases over time, may occur more often. MTI calculated from total commercial catch will vary with changes in the mix of species targeted by different fisheries over time, the relative importance of different fisheries sectors (e.g., finfish versus invertebrate fisheries), how much of the catch is reported, the quality of identification of species, and for other reasons not necessarily associated with effects of fishing (Caddy et al. 1998, Pauly et al. 1998b, Tuck et al. 2009, Branch et al. 2010). As such, MTI based on scientific surveys is likely to be a better indicator of change in fish communities (Branch et al. 2010).
- **Species-based indicators**: Many indices of diversity have been applied to fish communities (e.g., Peet 1974, Warwick & Clarke 1995, Bianchi et al. 2000, Greenstreet & Rogers 2006). These diversity indices are joint constructs of how many species are present (richness), and how similar their abundances are (evenness). Some indices give additional emphasis to the most important species in a community (dominance). Measures vary in the relative weight given to each of these factors, and on the metric used for similarity between species (e.g., by including a measure of taxonomic distinctiveness or not; Warwick & Clarke 1995). Fishing rarely causes large-scale extirpation so that measures of total species richness are likely to be less sensitive to change in trophic or ecosystem-level properties than measures of evenness. Different measures of evenness respond variously to fishing; they can increase, reduce or be unaffected by fishing depending on the initial characteristics of the ecosystem. A community initially dominated by k-selected[10] species would be expected to become more even and show increasing diversity metrics due to fishing; fishing would be expected to allow the faster growing (initially minor species) to increase at the expense of the slower growing (initially dominant) species. In contrast, diversity and evenness metrics may be expected to decrease after fishing if the ecosystem were originally dominated by r-selected[11] species.
- **Functional group based indicators**: Changes to the relative abundance of different functional groups in an ecosystem can indicate trophic or ecosystem-level changes (Fulton et al. 2005, Methratta & Link 2007, Shannon et al. 2009). Functional groups can be based on various descriptors of ecological niche, such as position in the water column (e.g., pelagic, demersal, benthic), trophic guild/feeding type (e.g., piscivore, pelagic invertebrate feeder, benthic feeder, scavenger), taxonomy (e.g., elasmobranch, gadoid, macrourid), or a combination of multiple ecological and life-history traits (Methratta & Link 2007), which can be combined to suggest high or low resilience (Tuck et al. 2009). A simple and commonly used index is the proportion of piscivorous fish to all fish caught. As piscivorous fish

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[10] Those that produce relatively low numbers of offspring, typically growing more slowly and maturing later.

[11] Those that produce high numbers of offspring, typically growing faster and maturing sooner.

tend to be disproportionately impacted by fishing (Caddy & Garibaldi 2000), their relative abundance in fish assemblages is a measure of ecosystem state and may reveal a trophic or system-level impact of fishing.

- **Size based indicators**: Marine trophic processes tend to be strongly structured by size (Badalamenti et al. 2002, Jennings et al. 2002). Fishing may lead to substantial modifications in the size structure of exploited populations because (a) high-value, generally larger species are targeted by fisheries, (b) fishing gears are size selective, often designed to catch larger fish and let smaller ones escape, (c) the cumulative effect of fishing (over the life of a cohort) leads to fewer older (larger) fish, and (d) long-lived species tend to be affected more as they have lower potential rates of increase. Several size-based metrics have been used to detect trophic and ecosystem-level changes (e.g., Murawski & Idoine 1992, Pope et al. 1987, Pope & Knights 1982, Rice & Gislason 1996). Size-based indicators can be applied at a species or community level. Applied to a given species, possible size-based indicators include: (a) mean length at age; (b) condition (weight at length; e.g., Winters & Wheeler 1994); (c) proportion of large fish; and (d) mean length at maturity in the population. Size-based methods at the community level include: (a) mean length in the community; (b) proportion of large individuals in the community; (c) the biomass size-spectrum; and (d) the diversity size spectrum (Rice & Gislason 1996).
- **Spatial distributions**: Fishing and climate/oceanographic variability/change can alter the geographic distribution of fish species (Perry et al. 2010) and this can indicate an ecosystem-level change. The percentage area of a research survey in which most (typically 90%) of the population occurs has been used as an ecosystem indicator (e.g., Fisher & Frank 2004, Tuck et al. 2009).
- **Diet-based indicators**: The change of diet (or trophic position) of a species of fish may reveal that trophic or ecosystem-level changes have occurred (e.g., Smith & Lucey 2014), but trophic position may change less than the underlying ecosystem structure (Badalamenti et al. 2002). 'Niche width' measured in terms of the range of carbon and nitrogen isotope ratios occupied by a species has also been suggested as indicative of trophic changes in a marine ecosystem especially in relation to upper trophic level predators (Layman et al. 2007), but the utility of this has been questioned (Hoeinghaus & Zeug 2008).

13.5.1.5 TOP PREDATORS

Top predators (upper trophic level consumers) can be used in two ways as indicators of the state of marine ecosystems. First, an OECD core indicator is the overall ecological threat status of species in the ecosystem, often with an emphasis placed on top predators (OECD 2003). Second, particular ecological aspects of selected predator species can be used to indicate changes in ecosystems. For example, top predators are widely used in monitoring the ecosystem effects of fishing krill in the Southern Ocean (Reid et al. 2005, Constable 2006), with information on the breeding of penguins, albatross, petrels, and seals collected, summarised and considered in management annually (CEMP 2004, Agnew 1997). Monitoring top predators as 'bellweathers' of ecosystem health is also increasingly used elsewhere (Boyd et al. 2006, Ainley 2002) as they are recognised as potentially useful downstream integrators of change in the marine ecosystem, exploit marine resources at similar spatial and temporal scales to humans, and receive high public interest. However, given that predators respond in complex ways to many factors simultaneously, ascertaining the appropriate management response to change of a predator-based indicator is difficult (Boyd et al. 2006).

13.5.2 NEW ZEALAND

There has been much work in New Zealand on developing indicators of the marine environment. MPI have carried out a number of projects looking at indicators and time series, including of oceanographic/climate variables (Hurst et al. 2008, Dunn et al. 2007, Pinkerton et al. 2014a), demersal fish communities based on data from scientific trawls (Tuck et al. 2009), and a suite of indicators relevant to deepwater fisheries (Tuck et al. 2014). Other work in New Zealand on marine ecosystem indicators include reports under NIWA Core funding (Pinkerton 2010) and in relation to national environmental reporting (Gilbert et al. 2000, Pinkerton 2007, Pinkerton 2014a).

13.5.2.1 MARINE PRIMARY PRODUCTION

Ocean colour satellite data have been used for more than a decade in New Zealand to investigate spatial and seasonal patterns in phytoplankton abundance and NPP (Murphy et al. 2001, Pinkerton 2007). There is a limited number of data available in New Zealand waters to develop locally tuned estimates of NPP from satellite data, and the concentration of chlorophyll-a is preferred for the purposes of monitoring change in primary production over time (Pinkerton et al. 2014a). Since 2002, mean concentrations of chlorophyll-a in the EEZ have decreased by an average of about 1% per year (Pinkerton, unpublished data). This is likely to be related, at least in part, to oceanographic cycles such as the Interdecadal Pacific Oscillation index[12] and the Southern Oscillation Index,[13] as well as potentially to long-term climate change.

13.5.2.2 LOWER FOOD WEB (MICROBIAL SYSTEM)

Changes to primary production also do not necessarily translate to less food available for higher trophic levels. Virtually all wild-caught seafood in New Zealand are carnivorous, with a mean trophic index of about 4.1 (MacDiarmid et al. 2013) The trophic efficiency by which energy passes between trophic levels is often considered to be about 10% (Pauly & Christensen 1995), meaning that only about one-tenth of the energy consumed by marine organisms is used to build new body mass. This means that each tonne of wild-caught seafood in New Zealand has been supported by over a thousand tonnes of primary production that has been moved through at least two intermediate levels in the marine food web before being consumed by the target species. A change to the lower and middle parts of the New Zealand food web hence have the potential to affect food availability for, and potentially yield of, commercially important fish stocks. At present, there are no data available to monitor for changes in the functioning of the lower trophic levels of New Zealand's marine ecosystems.

13.5.2.3 MIDDLE TROPHIC LEVELS

Middle trophic level organisms in the New Zealand ocean are diverse (more than 21 species of myctophids occur on the Chatham Rise for example; Pinkerton, unpublished data). Although they form the basis of the diet of many commercially-important New Zealand fish species (Dunn et al. 2009a), the basic abundance, distribution and ecology of key middle-trophic level groups like myctophids and hyperbenthic arthropods (prawns and shrimps) are generally poorly known. Two time series of data for middle trophic level organisms in the New Zealand ocean may be useful to investigate trophic and ecosystem-level effects: (a) New Zealand acquired a Continuous Plankton Recorder (CPR) in 2008 and this has been deployed on a transit extending from Oamaru (approximately 45ºS) to the Ross Sea annually since summer 2008–09; approximately 1200 km of this transect are in the subantarctic New Zealand EEZ (Robinson et al. 2013); (b) recent work has shown that multifrequency acoustic backscatter data taken from research vessels during the annual surveys of fish on the Chatham Rise can be used to derive indices of abundance of mesopelagic fish and invertebrates (McClatchie & Dunford 2003, O'Driscoll et al. 2009, Oeffner et al. 2014). Similar acoustic methods could provide time series of middle trophic level species in the Hauraki Gulf and subantarctic plateau in the near future.

13.5.2.4 DEMERSAL FISH COMMUNITIES

There are three series of scientific trawls in New Zealand waters that are particularly valuable for understanding ecosystem dynamics and for monitoring for trophic and ecosystem-level effects at the level of the demersal fish community (Tuck et al. 2009): (a) a scientific trawl survey has been carried out on the Chatham Rise region approximately annually since 1992; (b) a similar survey has been carried out over the subantarctic plateau over the same period but less frequently (Bagley & O'Driscoll 2012, Tuck et al. 2009); (c) a total of 15 trawl surveys have also been carried out in the Hauraki Gulf region between 1980 and 2000. Each of these trawl surveys used a consistent methodology based on scientific bottom trawl gear. Tuck et

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[12] The Interdecadal Pacific Oscillation (also called the Pacific Decadal Oscillation) is a 15–30-year cycle that affects parts of the Pacific Basin, causing variability in climate and oceanography, and has substantial and long-lasting effects on regional ecosystems (Kennedy et al. 2002).

[13] The Southern Oscillation Index is related to the strength of the trade winds in the Southern Hemisphere tropical Pacific (Mullan 1995) and SOI values for May–September are often used as an indicator of El Niño-La Niña Southern Oscillation (ENSO).

al. (2009) used these scientific surveys to investigate change in a series of indicators based on the demersal fish community.

Data from Chatham Rise trawl surveys between 1992 and 2007 showed evidence of increasing evenness (reducing diversity) but no evidence that species were being lost from the food web (Tuck et al. 2009). Some size characteristics of fish in research trawls on the Chatham Rise had changed, with fewer fish longer than 30 cm or heavier than 750 g being taken by trawl gear, although the median length of the catch did not change. Preliminary analysis of the mean trophic level index (MTI) in the demersal fish community of the Chatham Rise (Pinkerton 2010) indicated that this also decreased over the same period, and decreased more in the trawl survey data than in the commercial catch data. The proportion of piscivorous fish and of true demersal (rather than bentho-pelagic) species also declined over this period (Tuck et al. 2009). Somewhat counterintuitively, threatened[14] species and species defined by Tuck et al. (2009) as 'low-resilience', such as dogfish and rays, have increased relative to other species on the Chatham Rise. This was confirmed by independent analyses of Chatham Rise trawl survey data (O'Driscoll et al. 2011) and may be due to a combination of a lack of incentive to catch these species by the fishing fleet and an increase in offal and discards that benefit demersal scavengers. There were changes in the spatial distribution of fish species, with 16 out of 47 species showing changes in the proportion of the study area over which 90% of their abundance by weight was caught. Of these, half showed declining range and half showed increasing range. Tuck et al. (2009) showed that on the Chatham Rise, the species showing contractions of range were generally the more abundant species whereas the species expanding in spatial range were generally the less abundant species. MPI project ZBD2004/02 (Dunn et al. 2009a; Horn & Dunn 2010) examined whether there was evidence of change in the diet of hoki, hake or ling on the Chatham Rise between 1990 and 2009. It appears likely that the importance of fish (primarily myctophids) as a prey item for hoki has increased slightly but steadily between 1990 and 2009, while the importance of euphausiids has declined. In contrast, there were no obvious between-year differences or trends in hake diet from 1990 to 2009 (Horn & Dunn 2010). There were some marked between-year differences in ling diet in this period but no trends detected.

Discards and offal from fisheries is sometimes an important part of the diets of deepwater fish. For example, scavenged fishes accounted for up to a quarter of the diet of smooth skate (*Raja innominata*) in the Chatham Rise region (Dunn et al. 2009a, Forman & Dunn 2012). Anderson & Smith (2005) estimated that 11 000–14 000 t per year of non-commercial species and 600–2100 t per year of hoki are discarded by the New Zealand hoki fishery, leading to the potential for a significant modification of the diet of scavenging species (Forman & Dunn 2012). Interpreting changes in diet from discards in a way that can inform fisheries management is not straightforward. For the Chatham Rise, the changes covered a period of declining hoki spawning biomass (McKenzie 2013) and occurred at the same times as evidence of climate variation, namely a shift the prevalence of Kidson weather types (Kidson 2000) between 1992 and 2007 (Hurst et al. 2012). Disentangling these environmental and fishery drivers of changes to indicators of the demersal fish communities has not yet been attempted in New Zealand although the hypothesis that trophic or environmental factors were responsible for recent changes in hoki recruitment was investigated and was found not to be supported empirically (Francis et al. 2006, Bradford-Grieve et al. 2011).

13.5.2.5 TOP PREDATORS

Information on indicators of change in upper trophic levels in New Zealand are considered in Theme 1 of this report.

13.6 DISCUSSION

Marine ecosystems are complex, show non-linear dynamics (including potential tipping-points) and are subject to a wide range of impacts, including fishing, climate variability and change, coastal eutrophication and habitat change. Any activities that change the composition of species in the ecosystem (both in terms of size, functional group, ecosystem role, and diversity) will affect other groups in the ecosystem through trophic and other connections. A large range of trophic and ecosystem-level effects in marine systems have been documented internationally and these have generally been associated with negative impacts on

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[14] Species deemed more vulnerable according to the IUCN Red List (IUCN 2009); see Tuck et al. (2009).

fisheries (Garcia & Grainger 2005, Valdes et al. 2009, Worm et al. 2009). Understanding the scale and causes of these changes remains scientifically challenging (Rice 2001, Brander 2010, Jennings & Brander 2010, ter Hofstede et al. 2010). There remains substantial debate about the true extent and magnitude of these changes (Hilborn 2007, Murawski et al. 2007) and debate about how to allocate responsibility for these changes among different pressures, including fishing (Benoît & Swain 2008, Holt & Punt 2009, Kotta et al. 2009, Noakes & Beamish 2009, Rijnsdorp et al. 2009, Rice & Garcia 2011, Schiel 2013). Although ecosystem-level changes have rarely been ascribed solely to fisheries drivers, it appears that fishing is likely to make ecosystems less resilient to variability and change in climate/oceanographic forcing (Winder & Schindler 2004, Kirby et al. 2008, 2009). Reduced ecosystem resilience is an ecosystem-level effect that may predominantly occur through trophic mechanisms. Reduced ecosystem resilience may affect the long-term sustainability of harvesting (Hughes et al. 2005), increase ecosystem variability (Salomon et al. 2010), make fisheries less predictable and harder to manage in a variable and changing climate (Badjeck et al. 2010, Brander 2010, McIlgorm et al. 2010), reduce the ability of ecosystems to recover from overfishing (Neubauer et al. 2013), and increase the likelihood or consequence of regime shifts or invasive species (Folke et al. 2004, Salomon et al. 2010).

To date, it has generally not proved possible to realistically (as opposed to theoretically) identify at what point fishing or other pressure may cause serious disruptions in resource productivity or ecosystem function through trophic or ecosystem-level effects. For multi-species fisheries that are managed at a stock level close to B_{MSY} in a way that does not progressively degrade benthic habitat, it is not known whether it is necessary to take trophic and ecosystem-level effects into account more explicitly to ensure long-term sustainability of fisheries (ICES 2005). Some studies (e.g., Jackson et al. 2001, Jennings et al. 2002, Branch 2009), model analyses (Walters et al. 2005, Legovic et al. 2010, Gecek & Legovic 2012, Legovic & Gecek 2012, Ghosh & Kar 2013), and expert groups (Scientific Committee on Oceanographic Research/Intergovernmental Oceanographic Commission working group on indicators; Cury & Christensen 2005) have concluded that harvesting many species in an ecosystem at B_{MSY}[15] can lead to increased chance of fisheries collapse in the medium to long term – an effect called 'ecosystem erosion' or 'ecosystem overfishing' (Murawski 2000, Coll et al. 2008).

ICES (2005) concluded that, for fisheries managed at or close to B_{MSY}, the priority was to avoid fishing practices that drastically changed benthic structure, trophic interactions, food web structures or nutrient cycling (ICES 2005). This is consistent with the widespread consensus that fisheries should be managed within an ecosystem context and by adopting a precautionary approach that includes acknowledging the potentially synergistic effects of fishing and climate change (CBD 2009, Perry et al. 2010, Rice & Garcia 2011). However, there is little consensus on what this actually means in practice (FAO 2008, Ecosystem Principles Advisory Panel 1999, Browman & Stergiou 2004, 2005, Garcia & Cochrane 2005, Murawski 2011). Work by NOAA fisheries (Marasco et al. 2007) towards a pragmatic approach to ecosystem-based fishery management recommended:

- incorporating a broader array of societal goals and uses for ecosystem products and services within a multiple use multiple stressors framework;
- recognising the significance of ocean-climate conditions;
- emphasising food web interactions (recognise that harvest of target species has profound impacts on ecosystem structure and function through trophic interactions);
- employing spatial representation (manage stocks consistent with spatial/habitat variation in productivity);
- increasing and expanding focus on characterising and maintaining viable fish habitats;
- expanding scope of research and monitoring (increased focus on understanding biological interactions/processes, and measuring total fishery removals of target and non-target species);
- acknowledging and responding to higher levels of uncertainty (realistically incorporate uncertainty due to trophic and food web effects into management policy);
- reviewing and improving ecosystem modelling/research.

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[15] The biomass that allows the maximum sustainable yield to be taken.

The role of no-take reserves or marine protected areas (MPAs) in guarding against trophic and ecosystem-level effects remains controversial. A full review of the value of MPAs in this regard is beyond the scope of the present chapter. Suffice to say that some scientists believe strongly that MPAs can be effective at providing an 'ecological safety net' for trophic and ecosystem-level effects (Ballantine 2014, Edgar et al. 2014) whereas other scientists believe MPAs are too few and too small to have any value in this regard (Kaiser 2005, Mora et al. 2006). No-take marine reserves may have the most to contribute to our understanding of trophic and ecosystem effects by providing a 'reference ecosystem' in which populations experience low fishing pressure but a full range of other stressors (such as environmental variability/change, sedimentation, and pollution). Ecosystem changes in the reserve can then be contrasted with adjacent ecosystems exposed to the full range of fishing and other impacts (Micheli et al. 2005).

New Zealand is currently doing better than most countries with regard to many of the recommendations of Marasco et al. (2007). Pitcher et al. (2009) evaluated the performance of 33 countries for ecosystem-based management (EBM) of fisheries in three fields (principles, criteria and implementation). No country rated overall as 'good', only four countries, including New Zealand were 'adequate'. Specific recommendations from Marasco et al. (2007) are relevant to recent research initiatives in New Zealand. The newly announced Sustainable Seas research programme[16] aims to engage more closely with society to ensure that its goals and concerns are heard and addressed. Similarly, the MBIE Marine Futures project led by Dr Simon Thrush has used a multiple use framework to consider how ecosystem resilience can be promoted in the two focus areas of the Hauraki Gulf and Chatham Rise. Hurst et al. (2012) and Dunn et al. (2009b) considered the impact of ocean-climate interactions on New Zealand fisheries. The Ocean Survey 20/20 voyages had an explicit focus on mapping the distribution of seafloor habitats important to fish stocks and associated species (Hewitt et al. 2011). Ecosystem modelling of key New Zealand regions has been an ongoing focus of NIWA core-funded research since 2005, and includes co-funded ecosystem modelling work with MPI (e.g., ZBD2005/05). Data collection towards building up a comprehensive predator-prey database began with the ZBD2004/01 project (Dunn et al. 2009) and continues on the Chatham Rise under NIWA core-funding, with a particular focus on middle trophic level organisms that are abundant. MfE aim to include multi-trophic indicators of marine ecological state in the National Environmental Reporting (Pinkerton et al. 2014, Pinkerton 2014b), DOC are aiming to develop marine ecological integrity indicators (Freeman, pers. comm.), and MPI are actively developing indicators of change in fish communities (Tuck et al. 2009, 2014).

Notwithstanding this progress, most New Zealand stocks are managed on a single-stock basis at close to B_{MSY} (Ministry of Fisheries 2008) irrespective of their role in the ecosystem. The balance of evidence suggests that fishing close to B_{MSY} and in particular using bottom trawling (which impacts on benthic ecosystem function; Thrush & Dayton 2002) is likely to reduce ecosystem resilience and increase ecosystem variability by trophic and ecosystem-level effects (Brock & Carpenter 2006, Carpenter & Brock 2006, van Nes & Scheffer 2007, Guttal & Jayaprakash 2008) and could increase recruitment variability. Fishing is also likely to strengthen bottom-up control of marine ecosystems and make ecosystems more sensitive to the effects of climate change (Kirby et al. 2009, Perry et al. 2010). Greater sensitivity of marine ecosystems to climate variability implies a higher potential for regime shift which may or may not be reversible or desirable (Hsieh et al. 2006). Stronger environmental (bottom-up) forcing of ecosystems suggests a greater likelihood of unexpected changes to fisheries due to extreme environmental events and that these changes may be more severe (Perry et al. 2010, Kirby & Beaugrand 2009).

Time series measurements are crucial to understanding ecosystem function and monitoring for trophic and ecosystem-level effects of fishing. There would seem to be high value in maintaining regular and frequent (annual) surveys of the demersal fish communities of key New Zealand regions (such as the Chatham Rise, Hauraki Gulf and subantarctic plateau). Information on the catches of all species by the fishing fleet is required to monitor for changes in trophically or ecologically important non-QMS species. A key knowledge gap is information to map and monitor abundances, trophic connections and community structure of middle trophic level species, especially

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[16] Beehive. Sustainable Seas National Science Challenge launched. 4 September 2014. Retrieved from http://www.beehive.govt.nz/release/sustainable-seas-national-science-challenge-launched.

mesozooplankton, mesopelagics and hyperbenthics in key fishing areas, such as the Chatham Rise, Hauraki Gulf and subantarctic plateau. Knowledge of the abundance and trophic ecology of small demersal fishes in these regions is notably lacking.

13.7 CONCLUSIONS

1. A range of trophic and ecosystem-level effects in marine systems have been documented internationally, and these have generally been associated with negative impacts on fisheries.
2. Trophic and ecosystem-level effects are not usually brought about by fishing alone, but fishing (especially overfishing but also at or close to B_{MSY}) in multispecies fisheries can make ecosystems less resilient and more sensitive to the effects of environmental variability and change.
3. New Zealand's marine ecosystems are particularly diverse and this provides special challenges in monitoring, understanding and managing fisheries operating in them.
4. There is currently no evidence of a large-scale trophic or ecosystem-level effect impacting New Zealand's deepwater fisheries, but the cause of some changes in New Zealand's marine ecosystem EEZ are not known (e.g., changes to hoki recruitment (Francis et al. 2006, Bradford-Grieve & Livingston 2011); trends in some demersal-fish indicators on the Chatham Rise and other areas (Tuck et al. 2009).
5. It is likely that the reduction in the abundance of sea urchin predators on some rocky reef systems in north-eastern New Zealand due to fishing has contributed to an ecosystem-level effect in these areas, but this effect is unlikely to be widespread in New Zealand coastal areas (Schiel 2013).
6. Multi-species fishing at close to B_{MSY} using predominantly bottom-trawling is likely to make New Zealand's marine ecosystems less resilient (compared to fishing more conservatively compared to B_{MSY} and not using predominantly bottom-trawling) to other anthropogenic disturbance and to environmental variability, including climate change, through trophic and ecosystem-level effects.
7. There are potential, but unknown, trophic and ecosystem-level consequences for fisheries management in New Zealand if populations of marine mammals, such as fur seals, rebuild to levels that some people have suggested existed before humans arrived in New Zealand (see Theme 1 of this report).
8. Time series monitoring of fish communities and middle trophic level species (mesozooplankton, mesopelagics, hyperbenthics) are crucial for understanding and monitoring for trophic and ecosystem-level effects, and the best current sources of these data are trawl surveys to the Chatham Rise, and subantarctic plateau.

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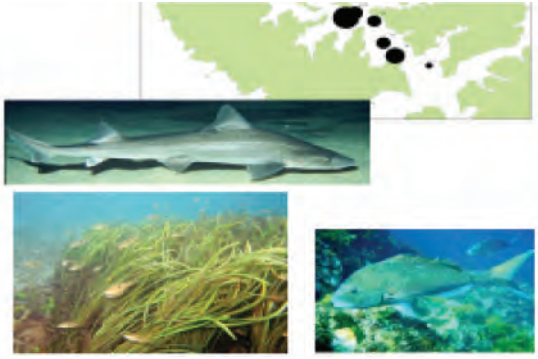
Chapter 14: Habitats of particular significance for fisheries management - Technical Summary

Loss of habitat and its connectivity



Land-based effects, benthic impacts, climate change

1. THE ISSUE IN BRIEF

- Protection of habitats of particular significance for fisheries management (HPSFM) is sanctioned in the Fisheries Act 1996
- HPFSMs have not yet been defined or applied, but some candidate habitats, such as spawning grounds and nursery areas, have been proposed
- Loss of habitat and of the connectivity between habitats have been highlighted as two of the most significant issues facing the health of marine ecosystems in New Zealand
- Like land-based impacts (see Chapter 15), HPSFM also requires an ecosystem-based approach to management

2. HABITAT SIGNIFICANCE

- HPSFMs have been proposed in areas where fish spawn and complete early life stages, but defining HPSFMs remains challenging because habitat significance is species- and life stage-specific and the elements of the habitat that are particularly important can be unclear
- Harbours and coastal areas are identified as spawning sites for several coastal fish species, but offshore areas and features such as seamounts can be relevant for deep-sea and pelagic species spawning and feeding (e.g., the Chatham rise). Overall, most areas to 1500 m of depth are significant to one species or another, but not yet quantified
- Habitat complexity plays a strong role in defining significance for fish species, and habitats built by living organisms (e.g., bryozoan beds and seagrass meadows) are particularly complex. Several existing protection measures already target some of these features
- Habitat complexity and naturalness are also relevant in freshwater, where migratory species like long-finned eels thrive in intact habitats and are negatively affected by loss of connectivity (see box 3)

3. HABITAT CONNECTIVITY LOSS

- Some animals move between different HPSFM throughout their lives
- Lack of connectivity between different HPSFM can be a potential bottleneck, can magnify the effects of habitat loss, and undermine habitat protection measures
- This is most evident in species that migrate over long distances through their life
- An example is freshwater eels that need to return to the sea to reproduce and complete their life cycle and may have their migration pathways blocked by dams

Habitat fragmentation (e.g., dams) can negatively impact species that migrate over long distances

14 HABITATS OF PARTICULAR SIGNIFICANCE FOR FISHERIES MANAGEMENT (HPSFM)

Status of chapter	This chapter has not been updated for AEBAR 2021.
Scope of chapter	This chapter highlights subject areas that might contribute to the management of HPSFM and hence provides a guide for future research.
Area	All of the New Zealand EEZ and Territorial Sea (inclusive of the freshwater and estuarine areas).
Locality hotspots	None formally defined, but already identified likely candidates include areas of biogenic habitat, e.g., Separation Point and Wairoa Hard, and areas identified with large catches and/or vulnerable populations of juveniles, e.g., Hoki Management Areas, packhorse crayfish legislated closures and toheroa beaches.
Key issues	Identifying likely HPSFM and potential threats to them.
Emerging issues	Connectivity and intra-population behaviour variability, multiple use.
MPI research (current)	HAB2007/01 *Biogenic habitats as areas of particular significance for fisheries management*; TOH2007/03 *Toheroa abundance*; ZBD2008/01 *Research on Biogenic Habitat-Forming Biota and their functional role in maintaining Biodiversity in the Inshore Region (5-150M Depths)* – this is also part-funded by Oceans Survey 2020, NIWA and MBIE; ENV2009/07 *Habitats of particular significance for fisheries management: Kaipara Harbour*; ENV2010/03 *Habitats of particular significance for inshore finfish fisheries management*; GMU2009/01 *Spatial Mixing of GMU1 using Otolith Microchemistry*.
NZ government research (current)	Ministry of Business, Innovation and Employment (MBIE) funded programmes (Coastal Conservation Management: C01X0907 *Protecting the functions of marine coastal habitats that support fish assemblages at local, regional and national scales*; C01X1229 *Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific region*; C01X0906 *Impacts of resource use on vulnerable deep-sea communities*. NIWA Core funding in the 'Managing marine stressors' area under the 'Coasts and Oceans' centre, specifically the programme 'Managing marine resources' and the project 'Measuring mapping and conserving (C01X0505)'.
Related chapters/issues	Land-based impacts on fisheries and supporting biodiversity, bycatch composition, marine environmental monitoring.

14.1 CONTEXT

The Fisheries Act 1996, in Section 9 (Environmental principles) states that:

'All persons exercising or performing functions, duties, or powers under this Act, in relation to the utilisation of fisheries resources or ensuring sustainability, shall take into account the following environmental principles:

a. Associated or dependent species should be maintained above a level that ensures their long-term viability:
b. Biological diversity of the aquatic environment should be maintained:
c. Habitat of particular significance for fisheries management should be protected.'

Work is currently ongoing on a guidance document for implementing habitats of particular significance for fisheries management (HPSFM).

This chapter will focus on examples of habitats shown to be important for fisheries and concepts likely to be important to HPSFM. Examples of potential HPSFM include: sources of larvae; larval settlement sites; habitat for juveniles; habitat that supports important prey species; migration corridors; and spawning, pupping or egg-laying grounds. Some of these habitats may be important for only part of the life cycle of an organism, or for part of a year.

The relative importance of habitats, compared with other limiting factors, is largely unknown for most stocks. For example, some stocks may be primarily habitat limited, whereas others may be limited by oceanographic variability, food supply, predation rates (especially during juvenile phases), or a mixture of these and other factors. In the case of stocks that are habitat limited, a management goal might be to preserve or improve some aspect of the habitat for the stock.

Hundreds of legislated spatial fisheries restrictions already apply within New Zealand's Territorial Sea and Exclusive Economic Zone (www.nabis.govt.nz), but until further policy work and research is conducted we cannot be sure of what contribution they make to protecting HPSFM. Examples of these are listed below:

- Separation Point in Tasman Bay, and the Wairoa Hard in Hawke Bay, were created to protect biogenic habitat that was believed to be important as juvenile habitat for a variety of fish species (Grange et al. 2003).
- An area near North Cape is currently closed to packhorse lobster fishing to mitigate sub-legal handling disturbance in this area. This closure was established because of the small size of lobsters caught there and a tagging study that showed movement away from this area into nearby fished areas (Booth 1979).
- The largest legislated closures are the Benthic Protection Areas (BPAs) that protect about 1.2 million km^2 (about 31% of the EEZ) outside the Territorial Sea from contact of trawl and dredge gear with the bottom (Helson et al. 2010).
- Commercial fishers must not use New Zealand fishing vessels or foreign-owned New Zealand fishing vessels over 46 m in overall length for trawling in the Territorial Sea.

In addition to legislated closures, a number of non-regulatory management measures exist. For example:

- Spatial closures:
  - Trawlers greater than 28 m in length are excluded from targeting hoki in four Hoki Management Areas – Cook Strait, Canterbury Banks, Mernoo Bank, and Puysegur Bank (Deep Water Group 2008). These areas were chosen because of the larger number of juveniles caught, relative to adults in these areas.
  - Trawling and pair trawling are both closed around Kapiti Island.
- Seasonal closures:
  - A closure to trawling exists from 1 November until 30 April each year in Tasman Bay.
  - A closure to commercial potting exists for all of CRA 3 for the whole of the month of December each year.

The highly migratory fish plan addresses HPSFM in environment outcome 8.1 'Identify and where appropriate protect habitats of particular significance to highly migratory species, especially within New Zealand waters'. In the deepwater fish plan the Ministry proposes in Management Objective 2.3 'to develop policy guidelines to determine what constitutes HPSFM then apply these policy guidelines to fisheries where necessary'. Inshore fisheries management plans (freshwater, shellfish and finfish) all contain references to identifying and managing HPSFM. These plans recognise that not all impacts stem from fisheries activities, therefore managing them may include trying to influence others to better manage their impacts on HPSFM. Work is underway on a guidance document for HPSFM that will assist in implementing these outcomes and objectives.

14.2 GLOBAL UNDERSTANDING

This section focuses upon those habitats protected overseas for their value to fisheries and discusses important concepts that may help gauge the importance of any particular habitat to fisheries management. This information may guide future research into HPSFM in New Zealand and any subsequent management action.

14.2.1 HABITATS PROTECTED ELSEWHERE FOR FISHERIES MANAGEMENT

Certain habitats have been identified as important for marine species including: shallow sea grass meadows, wetlands, seaweed beds, rivers, estuaries, rhodolith beds, rocky reefs, crevices, boulders, bryozoans, submarine canyons, seamounts, coral reefs, shell beds and shallow bays or inlets (Kamenos et al. 2004, Caddy 2008, Clark 1999, Morato et al. 2010a). Discrete habitats (or parts of these) may have extremely important ecological functions, and/or be especially vulnerable to degradation. For example, seabeds with high roughness are important for many fisheries and can be easily damaged by interaction with fishing gear (Caddy 2008). Examples of these include:

1. The *Oculina* coral banks off Florida were protected in 1994 as an experimental reserve

in response to their perceived importance for reef fish populations (Rosenberg et al. 2000). Later studies confirmed that this area is the only spawning aggregation site for gag (*Mycteroperca microlepis*) and scamp (*M. phenax*) (both groper species), and other economically important reef fish in that region (Koenig et al. 2000). The size of the area within which bottom-tending gears were restricted was subsequently increased based on these findings (Rosenberg et al. 2000).

2. Lophelia cold-water coral reefs are now protected in at least Norway (Fosså et al. 2002), Sweden (Lundälv & Jonsson 2003) and the United Kingdom (European Commission 2003) due to their importance as habitat for many species of fish (Costello et al. 2005).
3. The Western Pacific Regional Fishery Management Council identified all escarpments between 40 m and 280 m as Habitat Areas of Particular Concern (HAPC) for species in the bottom-fish assemblage. The water column to a depth of 1000 m above all shallow seamounts and banks was categorised as HAPC for pelagic species. Certain north-west Hawaiian Island banks shallower than 30 m were categorised as HAPC for crustaceans, and certain Hawaiian Island banks shallower than 30 m were classified as Essential Fish Habitat (EFH) for precious corals. Fishing is closely regulated in the precious-coral EFH, and harvest is only allowed with highly selective gear types that limit impacts, such as manned and unmanned submersibles (Western Pacific Fishery Management Council 1998)

Examples of habitats protected for their freshwater fishery values also exist. For example, the US Atlantic States Interstate fishery management plan (Atlantic States Marine Fisheries Commission 2000) notes the Sargasso Sea is important for spawning, and that seaweed harvesting provides a threat of unknown magnitude to eel spawning. Habitat alteration and destruction are also listed as probably impacting on continental shelves and estuaries/rivers, respectively, but the extent to which these are important is unknown.

It is also possible that HPSFM may be defined by the functional importance of an area to the fishery. For example, large spawning aggregations can happen in midwater for set periods of time (Schumacher & Kendall 1991, Livingston 1990) these could also potentially qualify as HPSFM.

14.2.2 CONCEPTS POTENTIALLY IMPORTANT FOR HPSFM

Many nations are now moving towards formalised habitat classifications for their coastal and ocean waters, which may include fish dynamics in the classifcation, and could potentially help to define HPSFM. Such systems help provide formal definitions for management purposes, and to 'rank' habitats in terms of their relative values and vulnerability to threats. Examples include the Essential Fish Habitat (EFH) framework being advanced in North America (Benaka 1999, Diaz et al. 2004, Valavanis et al. 2008), and in terms of habitat, the developing NOAA Coastal and Marine Ecological Classification Standard for North America (CMECS) (Madden et al. 2005, Keefer et al. 2008), and the European Marine Life Information Network (MarLIN) framework, which has developed habitat classification and sensitivity definitions and rankings (Hiscock & Tyler-Walters 2006).

Habitat connectivity (the movement of species between habitats) operates across a range of spatial scales, and is a rapidly developing area in the understanding of fisheries stocks. These movements link together different habitats into 'habitat chains', which may also include 'habitat bottlenecks', where one or more spatially restricted habitats may act to constrain overall fish production (Werner et al. 1984). Human-driven degradation or loss of such bottleneck habitats may strongly reduce the overall productivity of populations, and hence ultimately reduce long-term sustainable fisheries yields. The most widely studied of these links is between juvenile nursery habitats and often spatially distant adult population areas. Most studies published have been focused on species that use estuaries as juveniles (e.g., blue grouper *Achoerodus viridis* (a large wrasse) (Gillanders & Kingsford 1986) and snapper *Pagrus auratus* (Hamer et al. 2005) in Australia; and gag (*Mycteroperca microlepis*) in the United States (Ross & Moser 1995)), which make unidirectional ontogenetic habitat shifts from estuaries and bays out to the open coast as they grow from juveniles to adults. The extent of wetland habitats in the Gulf of Mexico has also been linked to the yield of fishery species dependent on coastal bays and estuaries. Reduced fishery stock production (of shrimp and the fish menhaden) followed wetland losses and, conversely, stock gains followed increases in the area of wetlands (Turner & Boesch 1987). Juvenile production was limited by the amount of available habitat but, equally, reproduction, larval settlement, juvenile or adult survivorship, or other demographic factors could also be limited by habitat loss or degradation, and these could have knock-on effects to stock characteristics such as productivity and its variability. Other examples include movements that may be bidirectional and regular in nature e.g., seasonal migrations of adult fish to and from spawning and/or feeding grounds, e.g., grey mullet *Mugil cephalus* off Taiwan (Chang et al. 2004).

How habitats are spatially configured to each other is also important to fish usage and associated fisheries production. For example, Nagelkerken et al. (2001) showed that the presence of mangroves in tropical systems significantly increases species richness and abundance of fish assemblages in adjacent seagrass beds. Jelbart et al. (2007) sampled Australian temperate seagrass beds close to (within 200 m) and distant from (more than 500 m from) mangroves. They found seagrass beds closer to mangroves had greater fish densities and diversities than more distant beds, especially of juveniles. Conversely, the densities of fish species in seagrass at low tide that were also found in mangroves at high tide were negatively correlated with the distance of the seagrass bed from the mangroves. This shows the important daily habitat connectivity that exists through tidal movements between mangrove and seagrass habitats. Similar dynamics may occur in more subtidal coastal systems at larger spatial and temporal scales. For example, Dorenbosch et al. (2005) showed that adult densities of coral reef fish, whose juvenile phases were found in mangrove and seagrass nursery habitats, were much reduced or absent on coral reefs located far distant from such nursery habitats, relative to those in closer proximity.

A less studied, but increasingly recognised theme is the existence of intra-population variability in movement and other behavioural traits. Different behavioural phenotypes within a given population have been shown to be very common in land birds, insects, mammals, and other groups. An example of this is a phenomenon known as ‘partial migration’, where part of the overall population migrates each year, often over very large distances, while another component does not move and remains resident. By definition, this partial migration also results in differential use of habitats, often over large spatial scales. Recent work on white perch (*Morone americana*) in the United States shows that this population is made up of two behavioural components: a resident natal freshwater contingent, and a dispersive brackish-water contingent (Kerr et al. 2010). The divergence appears to be a response to early life history experiences that influence individuals’ growth (Kerr 2008). The proportion of the overall population that becomes dispersive for a given year class ranges from 0% in drought years to 96% in high-flow years. Modelling of how differences in growth rates and recruitment strengths of each component contributed to the overall population found that the resident component contributed to long-term population persistence (stability), whereas the dispersive component contributed to population productivity and resilience (defined as rebuilding capacity) (Kerr et al. 2010). Another species, winter flounder *Pseudopleuronectes americanus*, has also shown intra-population variability in spawning migrations; one group stays coastally resident while a second smaller group migrates into estuaries to spawn (De Celles & Cadrin 2010). The authors went on to suggest that coastal waters in the Gulf of Maine should merit consideration in the assignment of Essential Fish Habitat for this species.

Kerr & Secor (2009) and Kerr et al. (2010) argue that such phenotypic dynamics are probably very common in marine fish populations but have not yet been effectively researched and quantified. The existence of such dynamics would have important implications for fisheries management, including the possibility of spatial depletions of more resident forms and variability in the use of potential HPSFM between years. For instance, recent work on snapper in the Hauraki Gulf has shown that fish on reef habitats are more resident (i.e., have less propensity to migrate) than those of soft sediment habitats, and can experience higher fishing removals (Parsons et al. 2011).

The most effective means of protecting a HPSFM in terms of the benefit to the fishery may differ depending on the life-history characteristics of the fish. A variety of modelling, theoretical, and observational approaches have led to the conclusion that spatial protection performs best at enhancing species whose adults are relatively sedentary but whose larvae are broadcast widely (Chiappone & Sealey 2000, Murawski et al. 2000, Roberts 2000, Warner et al. 2000). The sedentary habit of adults allows the stock to accrue the maximum benefit from the protection, whereas the broadcasting of larvae helps ‘seed’ segments of the population outside the protection. However, the role of spatial protection in directly protecting juveniles after they have settled to seafloor habitats (via habitat protection/recovery, and/or reduced juvenile bycatch), or their interaction with non-fisheries impacts has not yet been explicitly considered.

14.3 STATE OF KNOWLEDGE IN NEW ZEALAND

14.3.1 POTENTIAL HPSFM IN NEW ZEALAND

Important areas for spawning, pupping, and egg-laying are potential HPSFM. These areas (insofar as these are known) have been identified and described using science literature and fisheries databases and summarised within two atlases, one coastal (less than 200 m) and one deepwater (more than 200 m). Coastally, these HPSFM areas were identified for 35 important fish species by Hurst et al. (2000b). This report concluded that virtually all coastal areas were important for these functions for one species or another. The report also noted that some coastal species use deeper areas for these functions, either as juveniles, or to spawn (e.g., red cod, giant stargazer) and some coastal areas are important for juveniles of deeper spawning species (e.g., hake and ling). Some species groupings were apparent from this analysis. Elephant fish, rig, and school shark all preferred to pup or lay eggs in shallow water, and very young juveniles of these species were found in shallow coastal areas. Juvenile barracouta, jack mackerel (*Trachurus novaezelandiae*), kahawai, rig, and snapper were all relatively abundant (at least occasionally) in the inner Hauraki Gulf. Important areas for spawning, pupping, and egg-laying were identified for 32 important deepwater fish species (200 to 1500 m depth), 4 pelagic fish species, 45 invertebrate groups, and 5 seaweeds (O'Driscoll et al. 2003). This study concluded that all areas to 1500 m deep were important for either spawning or for juveniles of one or more species studied. The relative significance of areas was hard to gauge because of the variability in the data, however the Chatham Rise was identified as a 'hotspot'.

Areas of high juvenile abundances of certain species may be useful indicators of HPSFM for some species. A third atlas (Hurst et al. 2000b) details species distributions (mainly commercial) of adult and immature stages from trawl, midwater trawl and tuna longline where adequate size information was collected. No conclusions are made in this document, and generalisations across species are inherently difficult, therefore like the previous two atlases, this document is probably best examined for potential HPSFM in a species specific way.

Certain locations within New Zealand already seem likely to qualify as HPSFM under any likely definition. The Kaipara Harbour has been identified as particularly important for the SNA 8 stock. Analysis of otolith chemistry showed that, for the 2003 year class, a very high proportion of new snapper recruits to the SNA 8 stock were sourced as juveniles from the Kaipara Harbour (Morrison et al. 2008). This result is likely to be broadly applicable into the future as the Kaipara provides most of the biogenic habitat available for juvenile snapper on this coast. The Kaipara and Raglan harbours also showed large catches of juvenile rig and the Waitemata, Tamaki and Porirua harbours moderate catches (Francis et al. 2012). Recent extensive fish habitat sampling within the Kaipara harbour in 2010 as part of the MBIE Coastal Conservation Management programme showed juvenile snapper to be strongly associated with subtidal seagrass, horse mussels, sponges, and an introduced bryozoan. Negative impacts on such habitats have the potential to have far-field effects in terms of subsequent fisheries yields from coastal locations well distant from the Kaipara Harbour. Beaches that still retain substantive toheroa populations, e.g., Dargaville and Oreti beaches, may also potentially qualify as HPSFM (Beentjes 2010).

Consistent with the international literature, biogenic (living, habitat forming) habitats have been found to be particularly important juvenile habitat for some coastal fish species in New Zealand. For example: bryozoan mounds in Tasman Bay are known nursery grounds for snapper, tarakihi and John dory (Vooren 1975); northern subtidal seagrass meadows fulfil the same role for a range of fish including snapper, trevally, parore, garfish and spotties (Francis et al. 2005, Morrison et al. 2008, Schwarz et al. 2006, Vooren 1975); northern horse mussel beds for snapper and trevally (Morrison et al. 2009); and mangrove forests for grey mullet, short-finned eels, and parore (Morrisey et al. 2010). Many other types of biogenic habitats exist, and some of their locations are known (e.g., see Davidson et al. 2010 for biogenic habitats in the Marlborough Sounds), but their precise role as HPSFM remains to be quantified. Examples include open coast bryozoan fields, rhodoliths, polychaete (worm) species ranging in collective form from low swathes to large high mounds, sea pens and sea whips, sponges, hydroids, gorgonians, and many forms of algae, ranging from low benthic forms such as *Caulerpa* spp. (sea rimu) through to giant kelp (*Macrocystis pyrifera*) forests in cooler southern waters. Similarly, seamounts are well known to host reef-like formations of deep-sea stony corals (e.g., Tracey et al. 2011), as well as being major spawning or feeding areas for commercial deepwater species such as orange roughy and oreos (e.g., Clark 1999, O'Driscoll &

Clark 2005). However, the role of these benthic communities on seamounts in supporting fishstocks is uncertain, as spawning aggregations continue to form even if the coral habitat is removed by trawling (Clark & Dunn 2012). Hence the oceanography or physical characteristics of the seamount and water column may be the key drivers of spawning or early life-history stage development, rather than the biogenic habitat.

Freshwater eels are reliant upon rivers as well as coastal and oceanic environments. GIS modelling estimates that for longfin eels, about 30% of longfin habitat in the North Island and 34% in the South Island is either in a reserve or in rarely/non-fished areas, with about 49% of the national longfin stock estimate of about 12 000 t being contained in these waterways (Graynoth et al. 2008). More regional examination of the situation for eels also exists, e.g., for the Waikato Catchment (Allen 2010). Shortfin eels prefer slower-flowing coastal habitats such as lagoons, estuaries, and lower reaches of rims (Beentjes et al. 2005). In-stream cover (such as logs and debris) has been identified as important habitat, particularly in terms of influencing the survival of large juvenile eels (Graynoth et al. 2008). Short-fin eel juveniles and adults have also been found to be relatively common in estuarine mangrove forests, and their abundance positively correlated with structural complexity (seedlings, saplings, and tree densities) (Morrisey et al. 2010). In addition oceanic spawning locations are clearly important for eels, the location of these are unknown, although it has been suggested that these may be north-east of Samoa and east of Tonga for shortfins and longfins respectively (Jellyman 1994).

Many of the potential HPSFM are threatened by either fisheries or land-based effects, the reader should look to the land-based effects chapter in this document and the eel section of the Stock Assessment Plenary report for further details.

14.3.2 HABITAT CLASSIFICATION AND PREDICTIVE MODELLING

Habitat classification schemes focused upon biodiversity protection have been developed in New Zealand at both national and regional scales, these may help identify larger habitats which HPSFM may be selected from, but are unlikely to be useful in isolation for determining HPSFM. The Marine Environment Classification (MEC), the demersal fish MEC and the benthic optimised MEC (BOMEC) are national-scale classification schemes that have been developed with the goal of aiding biodiversity protection (Leathwick et al. 2004, 2006, 2012). A classification scheme also exists for New Zealand’s rivers and streams based on their biodiversity values to support the Department of Conservations Waters of National Importance (WONI) project (Leathwick & Julian 2008). Regional classification schemes also exist such as ones mapping the Marine habitats of Northland, or Canterbury in order to assist in Marine Protected Area planning (Benn 2009, Kerr 2010).

Another tool that may help in terms of identifying HPSFM is the predictions of richness, occurrence and abundance of small fish in New Zealand estuaries (Francis et al. 2011). This paper contains richness predictions for 380 estuaries and occurrence predictions for 16 species. This could help minimise the need to undertake expensive field surveys to inform resource management, although environmental sampling may still be needed to drive some models.

14.3.3 CURRENT RESEARCH

Prior to 2007 research within New Zealand was not explicitly focused on identifying HPSFM. However, in line with international trends, this situation has changed in recent times, with recognition of some of the wider aspects of fisheries management.

A number of Ministry and other research projects were commissioned concerning HPSFM in the 2010–11 year. Project ENV200907, ‘Habitat of particular significance to fisheries management: Kaipara Harbour’, is underway and has the overall objective of identifying and mapping areas and habitats of particular significance in the Kaipara Harbour which support coastal fisheries; and identifying and assessing threats to these habitats. Included in this work is the reconstruction of environmental histories through interviews of long time local residents who have experience of the harbour, and associated collation and integration of historical data sources (e.g., catch records, photographs, diaries, maps, and fishing logs). Another output of this work will be recommendations on the best habitats and methods of monitoring to detect change to HPSFM within Kaipara Harbour.

Biogenic habitats on the continental shelf from about 5 to 150 m depths are currently being characterised and mapped through the biodiversity project ZBD2008/01, this will also provide new information on fisheries species utilisation of these habitats. Interviews with 50 retired fishers have provided valuable information on biogenic habitat around New Zealand. A national survey to examine the present occurrences and extents of these biogenic habitats was completed in 2011 in collaboration with Oceans Survey 2020, NIWA and Ministry of Business, Innovation and Employment (MBIE) funding.

A number of other national-scale projects are also underway. A desktop review is collating information on the importance of biogenic habitats to fisheries across the entire Territorial Sea and Exclusive Economic Zone (project HAB2007/01). A project has been approved to review the literature and recommend the relative urgency of research on habitats of particular significance for inshore finfish species (project ENV2010/03).

The Ministry of Business, Innovation and Employment (MBIE) funded project Coastal Conservation Management started in 2009 and runs for six years. This programme aims to integrate and add to existing fish-habitat association work to develop a national-scale marine fish-habitat classification and predictive model framework. This project will also attempt to develop threat assessments at local, regional and national scales. MPI is maximising the synergies between its planned research and this project. As part of this synergy, work on the connectivity and stock structure of grey mullet (*Mugil cephalus*) is underway in collaboration with MPI project GMU2009/01. Otolith chemistry is being assessed for its utility in partitioning the GMU 1 stock into more biologically meaningful management units, and in quantifying the suspected existence of source and sink dynamics between the various estuaries that hold juvenile grey mullet nursery habitats.

In 2012 MBIE also funded the three-year project delivered by NIWA entitled 'Predicting the occurrence of vulnerable marine ecosystems for planning spatial management in the South Pacific region'. The development of predictive models of species occurrence under this project may also aid in identifying HPSFM. Identification of biogenic habitat has been part of the MBIE project 'Vulnerable deep-sea communities' since 2009 (and its predecessor seamount programme), which includes surveys of a range of habitats that may be important for various life-history stages of commercial fish species: seamounts, canyons, continental slope, hydrothermal vents and seeps.

14.4 INDICATORS AND TRENDS

As no HPSFM are defined this section cannot be completed.

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Chapter 15: Land-based effects on the coastal environment - Technical Summary

Degradation of coastal habitats

Sediment, contaminants, connectivity

1. THE ISSUE IN BRIEF

- The interconnections between the terrestrial and the marine environment are widely recognised
- The effects of land-based activities can be detrimental to marine coastal habitats and biological communities, and affect recreational activities and primary production (e.g. marine aquaculture, fisheries)
- Marine biodiversity that underpins fish and shellfish fisheries is impacted by land-based activities
- The influence of these activities is usually greatest near estuaries and the coastline

2. LAND-BASED EFFECTS ON COASTAL ECOSYSTEMS

- Coastal structures (e.g. harbours, outfalls) and inland structures (e.g. hydropower dams) can reduce habitat quality and connectivity for both resident and migratory aquatic life
- Contaminants in freshwater runoff are generally diluted once they reach the sea, but can have local effects
- Land-based activities (e.g. dairy farming) can lead to nutrient enrichment (i.e. eutrophication), which in turn may lead to harmful algal blooms and anoxic events in coastal waters
- Human activities (e.g. driving on beaches) can have adverse effects on shellfish, but their magnitude is unknown
- Land-based pathogens affecting marine organisms are an emerging issue for Māui dolphins (see Chapter 6)

- Different land uses can affect the erosion of soils and their subsequent transport and sedimentation in coastal areas, which is considered one of the highest threats to NZ coastal ecosystems
- The effects of sedimentation include smothering of coastal habitats, such as nursery areas for fish and shellfish
- Coastal infrastructures can change the hydrodynamics near the coastline, potentially increasing sedimentation and eutrophication

Coastal development can alter marine currents and circulation at the local scale

3. MITIGATION MEASURES

- Mitigation measures include habitat restoration of coastal, fluvial and estuary areas, improved water treatment (especially to reduce coliform bacteria), creation of fish passes for migratory species, land-use management, and the inclusion of habitat elements (e.g. holes) in coastal engineering structures
- Mitigation and monitoring of land impacts is a key responsibility of local and regional authorities through the New Zealand Coastal Policy Statement, which is managed by DOC
- Fisheries New Zealand works closely with DOC on these issues

15 LAND-BASED EFFECTS ON THE COASTAL ENVIRONMENT

Status of chapter	This chapter has not been updated for AEBAR 2021.
Scope of chapter	This chapter outlines the main known threats from land-based activities to fisheries, aquaculture and supporting biodiversity. It also describes the present status and trends in land-based impacts.
Area	All of the New Zealand freshwater, EEZ and Territorial Sea.
Focal localities	Freshwater habitats and areas closest to the coast are likely to be most impacted; this will be exacerbated in areas with low water movement. Anthropogenically increased sediment run-off is particularly high from the Waiapu and Waipaoa river catchments on the east coast of the North Island. Areas of intense urbanisation or agricultural use of catchments are also likely to be impacted by bacteria, viruses/diseases, heavy metals or nutrients, or some combination of these.
Key issues	Habitat modification, sedimentation, aquaculture, shellfish, terrestrial land-use change (particularly for urbanisation, forestry or agriculture) water quality and quantity, contamination, recreational activities, consequences of increased pollutants to seafood production, freshwater management and demand.
Emerging issues	Impacts on habitats of particular significance to fisheries management (HPSFM), linkages through rainfall patterns to climate change, shellfish bed closures, habitat remediation, domestic animal diseases in protected marine species, proposed aquaculture expansion, water abstraction impacts.
MPI research (latest)	ZBD2008/01 *Research on Biogenic Habitat-Forming Biota and their functional role in maintaining Biodiversity in the Inshore Region (5–150 m depths)* – this is also part-funded by Oceans Survey 2020, NIWA and MBIE.
NZ government research (current)	Ministry of Business, Innovation and Employment (MBIE) funded programmes: UOCX0902 *After the outfall: recovery from eutrophication in degraded New Zealand estuaries*; CO1X1005 *Management of Cumulative Effects of Stressors in Aquatic Ecosystems*. NIWA core-funded research on this topic occurs in two areas. Firstly, the 'Managing marine ecosystems' programme, specifically the projects 'Measuring mapping and conserving', 'Ecosystem-based management of coasts and estuaries', 'Coastal management' (C01X0907) and 'Marine Futures' (C01X0227) (Note that the latter two finish 30 September 2014). Secondly, in the 'Fisheries' Centre, the EAFM programme deals with ecosystem-based management approaches in conjunction with the 'Coasts and Oceans' centre. Some funding within these areas will be aligned to the Sustainable Seas Science Challenge in the near future in which the focus is on ecosystem based management of the marine environment.
Related chapters/issues	Habitats of particular significance for fisheries management (HPSFM), marine environmental monitoring.

15.1 CONTEXT

Land-based activities that may have impacts on seafood production are primarily regulated under the Resource Management Act 1991 (and subsequent amendments). Fisheries are controlled under the Fisheries Act 1996, this includes marine and freshwater resources management of aquatic life (under Part 2 of the Fisheries Act). Both the Fisheries Act 1996 and the MPI Strategy 'Our Strategy'[1] state that New Zealand's the primary sector's utilization of natural marine and freshwater fish resources needs to be sustainable.

The government's 'Fresh Start for Freshwater Programme'[2] (led by MfE and MPI) aims to create a water management

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[1] Ministry for Primary Industries. Our Strategy. Retrieved from http://www.mpi.govt.nz/about-mpi/our-strategy.

[2] Ministry for the Environment. Fresh Start for Fresh Water. Retrieved from http://www.mfe.govt.nz/more/cabinet-papers-and-related-material-search/cabinet-papers/freshwater/fresh-start-fresh-water.

system that allows us to make more transparent and better targeted and informed decisions on fresh water. Businesses and water users will have more certainty so that they can plan and invest. All New Zealanders will have a greater say on the water quality they want for their lakes and rivers. The Coastal Policy Statement (2010) also has relevance to matters of fisheries interest, e.g., Policy 20(1) (paraphrased) controls the use of vehicles on beaches where (b) harm to shellfish beds may result. MPI also works with other agencies, principally DOC, MfE and regional councils and through the Natural Resource Cluster to influence these processes to ensure consideration of land-based impacts upon seafood production. The New Zealand aquaculture industry has an objective of developing into a billion dollar industry by 2025.[3] Government supports well-planned and sustainable aquaculture through its *Aquaculture Strategy and Five-year Plan*. One of the desired outcomes of actions by the New Zealand government is to enable more space to be made available for aquaculture. This outcome is likely to heighten the potential for conflict between aquaculture proponents and those creating negative land-based effects.

An MPI-funded survey of scientific experts (MacDiarmid et al. 2012) addressed the vulnerability to a number of threats of marine habitat types within the New Zealand's Territorial Sea and Exclusive Economic Zone (EEZ). Each vulnerability score was based on an assessment of five factors including the spatial scale, frequency and functional impact of the threat in the given habitat as well as the susceptibility of the habitat to the threat and the recovery time of the habitat following disturbance from that threat. The study found that the number of threats and their severity were generally considered to decrease with depth, particularly below 50 m. Reef, sand, and mud habitats in harbours and estuaries and along sheltered and exposed coasts were considered to be the most highly threatened habitats. The study also reported that over half of the 26 top threats fully, or in part, stemmed from human activities external to the marine environment itself. The top six threats in order were:

1. ocean acidification,
2. rising sea temperatures resulting from global climate change,

3rd equal. bottom trawling fishing,
3rd equal. increased sediment loadings from river inputs,
5th equal. change in currents from climate change,
5th equal. increased storminess from climate change.

The reader is guided to MacDiarmid et al. (2012) for more detail including tables of threats-by-habitat and habitats-by-threat. Climate change and ocean acidification, although they can be considered land-based effects, are covered under the chapters in this document called 'New Zealand's Climate and Oceanic Setting' and 'Biodiversity'.

Land-based effects on seafood production and biodiversity in this context are defined as resulting either from the inputs of contaminants from terrestrial sources or through engineering structures (e.g., breakwaters, causeways, bridges), that change the nature and characteristics of coastal habitats and modify hydrodynamics. The major route for entry of land-based contaminants into the marine environment is associated with freshwater flows (rivers, streams, direct runoff and ground water), although contaminants may enter the marine environment via direct inputs (e.g., landslides) or atmospheric transport processes.

Different land-use effects act over different scales; for example localised effects act on small streams and adjacent estuarine habitats, large scale effects extend to coastal embayments and shelf ecosystems. Associated risks will vary according to location and depend on the relevant ecosystem services (e.g., high value commercial fishery stocks) and their perceived sensitivities. The risk from stormwater pollutants will be more important near urban areas and the effects of nutrient enrichment will be more important near intensively farmed rural areas.

The risk from land-based impacts for seafood production is that they will limit the productivity of a stock or stocks. For example, the bryozoan beds around Separation Point in Golden Bay, were protected from fishing in 1980, partly because of their perceived role as nursery grounds for a variety of coastal fish species (Grange et al. 2003). Recent work has suggested that the main threat to these bryozoans is now sedimentation from the Motueka River, which may inhibit recovery of any damaged bryozoans (Grange et al. 2003, Morrison et al. 2009). Any declines in this bryozoan

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[3] Aquaculture New Zealand. Strategy. Retrieved from http://aquaculture.org.nz/about-us/strategy.

bed and associated ecological communities could also affect the productivity of adjacent fishery stocks.

Fisheries New Zealand mainly manages in the marine environment, therefore this topic area will be dealt with first. The main freshwater fisheries management Fisheries New Zealand is involved in is the freshwater eel fishery; this will be dealt in later sections, as relevant.

15.2 GLOBAL UNDERSTANDING

15.2.1 LAND-BASED INFLUENCES

It has been acknowledged for some time now that land-based activities can have important effects on seafood production. The main threats to the quality and use of the world's oceans are (GESAMP 2001):

- alteration and destruction of habitats and ecosystems;
- effects of sewage on human health;
- widespread and increased eutrophication;
- decline of fishstocks and other renewable resources; and
- changes in sediment flows due to hydrological changes.

Coastal development is projected to impact 91% of all inhabited coasts by 2050 and will contribute to more than 80% of all marine pollution (Nellemann et al. 2008). The importance of different land-based influences differ regionally but the South Pacific Regional Environmental Programme (SPREP, which includes New Zealand) defines waste management and pollution control as one of its four strategic priorities for 2011–15 (SPREP 2010).

Influences, including land-based influences, seldom work in isolation; for example the development of farming and fishing over the last hundred years has meant that increased sediment and nutrient runoff has to some degree occurred simultaneously with increased fishing pressure. However, the impact of these influences has often been studied in isolation. In a review on coastal eutrophication, Cloern (2001) stated that *'Our view of the problem [eutrophication] is narrow because it continues to focus on one signal of change in the coastal zone, as though nutrient enrichment operates as an independent stressor; it does not reflect a broad ecosystem-scale view that considers nutrient enrichment in the context of all the other stressors that cause change in coastal ecosystems'*. These influences (in isolation or combination) can also cause indirect effects, such as decreasing species diversity that then lessens resistance to invasion by non-indigenous species or species with different life-history strategies (Balata et al. 2007, Kneitel & Perrault 2006, Piola & Johnston 2008). Studies that research a realistic mix of influences are rare, but valuable.

The most important land-based effect in New Zealand is arguably increased sediment deposition around our coasts (Morrison et al. 2009, MacDiarmid et al. 2012). This deposition has been accelerated due to increased erosion from land-use, which causes gully and channel erosion and landslides (Glade 2003). Inputs of sediments to our coastal zone, although naturally high in places due to our high rainfall and rates of tectonic uplift (Carter 1975), have been accelerated by human activities (Goff 1997). New Zealand represents only about 0.3% of the land area that drains into the oceans (Griffiths & Glasby 1985, Milliman & Syvitski 1992), yet sediment inputs make up about 1% of the estimated global detrital input to the oceans (Carter et al. 1996). Sediment deposition can be an important influence, particularly in areas of high rainfall, tectonic uplift, and forest clearances, or areas where these activities coincide. Sediments are known to erode from the land at an increased rate in response to human use, for example, estimates from a largely deforested tropical highland suggest erosion rates 10–100 times faster than pre-clearance rates (Hewawasam et al. 2003). Increased sediment either deposited on the seafloor or suspended in the water column can negatively impact invertebrates in a number of ways including: burial, scour, inhibiting settlement, decreasing filter-feeding efficiency and decreasing light penetration, generally leading to less diverse communities, with a decrease in suspension feeders (Thrush et al. 2004). These impacts can affect the structure, composition and dynamics of benthic communities (Airoldi 2003, Thrush et al. 2004). Effects of this increased sediment movement and deposition on finfish are mostly known from freshwater fish and can range from behavioural (such as decreased feeding rates) to sublethal (e.g., gill tissue disruption) and lethal as well as having effects on habitat important to fishes (Morrison et al. 2009). These effects differ by species and life-stage and are dependant upon factors that include the duration, frequency and magnitude of exposure, temperature, and other environmental variables (Servizi & Martens 1992).

Increased nutrient addition to the aquatic environment can initially increase production, but with increasing nutrients there is an increasing likelihood of harmful algal blooms and cascades of effects damaging to most communities above the level of the plankton (Kennish 2002, Heisler et al. 2008). This excess of nutrients is termed eutrophication. Eutrophication can stimulate phytoplankton growth, which can decrease the light availability and subsequently lead to losses in benthic production from seagrass, microalgae or macroalgae and their associated animal communities. Algal blooms then die and their decay depletes oxygen and blankets the seafloor. The lack of oxygen in the bed and water column can lead to losses of finfish and benthic communities. These effects are likely to be location specific and are influenced by a number of factors including: water transparency, distribution of vascular plants and biomass of macroalgae, sediment biogeochemistry and nutrient cycling, nutrient ratios and their regulation of phytoplankton community composition, frequency of toxic/harmful algal blooms, habitat quality for metazoans, reproduction/growth/survival of pelagic and benthic invertebrates, and subtle changes such as shifts in the seasonality of ecosystems (Cloern 2001). The effects of eutrophication abound in the literature, for example, the formation of dead (or anoxic) zones is exacerbated by eutrophication, although oceanographic conditions also play a key role (Diaz & Rosenberg 2008). Dead zones have now been reported from more than 400 systems, affecting a total area of more than 245 000 km^2 (Diaz & Rosenberg 2008). This includes anoxic events from New Zealand in coastal north-eastern New Zealand and Stewart Island (Taylor et al. 1985, Morrissey 2000).

Other pollutants such as heavy metals and organic chemicals can have severe effects, but are more localised in extent than sediment or nutrient pollution (Castro and Huber 2003, Kennish 2002). Fortunately the concentration of these pollutants in most New Zealand aquatic environments is relatively low, with a few known exceptions. Examples of this include naturally elevated levels of arsenic in Northland,[4] cadmium levels in Foveaux Strait oysters (Frew et al. 1996) and levels of nickel and chromium within the Motueka river plume in Tasman Bay (Forrest et al. 2007). The high cadmium levels have caused market access issues for Foveaux Strait oysters. Some anthropogenically generated pollutants such as copper, lead, zinc and PCBs are high in localised hotspots within urban watersheds. In the Auckland region these hotspots tend to be in muddy estuarine sites and tidal creeks that receive runoff from older urban catchments (Auckland Regional Council 2010). There is a lack of knowledge on the impacts of these pollutants upon fisheries.

Climate change is likely to interact with the effect of land-based impacts as the main delivery of land-based influences is through rainfall and subsequent freshwater flows. Global climate change projections include changes in the amount and regional distribution of rainfall over New Zealand (IPCC 2007). More regional predictions include increasing frequency of heavy rainfall events over New Zealand (Whetton et al. 1996). This is likely to exacerbate the impact of some land-based influences as delivery peaks at times of high rainfall, e.g., sediment delivery (Morrison et al. 2009).

Physical alterations of the coast are generally, but not exclusively (e.g., wetland reclamation for agriculture), concentrated around urban areas and can have a number of consequences on the marine environment (Bulleri & Chapman 2010). Changes in diversity, habitat fragmentation or loss and increased invasion susceptibility have all been identified as consequences of physical alteration. The effects of physical alterations upon fisheries remain largely unquantified; however the habitat loss or alteration portion of physical alterations will be dealt with under the habitats of particular significance for fisheries management (HPSFM) section.

An area of emerging interest internationally is infectious diseases from land-based animals affecting marine populations. Perhaps the most well-known example of this is the canine distemper outbreak in Caspian seals that caused a mass mortality in the Caspian Sea in 2000 (Kennedy et al. 2000).

15.2.2 HABITAT RESTORATION

Habitat restoration or rehabilitation has been the subject of much recent research. Habitat restoration or rehabilitation rarely, if ever, replaces what was lost and is most applicable in estuarine or enclosed coastal areas as opposed to exposed coastal or open ocean habitats (Elliott et al. 2007). Connectivity of populations is a key consideration when evaluating the effectiveness of any marine restoration or rehabilitation (Lipcius et al. 2008). In the marine area, seagrass replanting methodologies are being developed to

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[4] NIWA. Ocean Survey 2020. Retrieved from https://www.niwa.co.nz/coasts-and-oceans/research-projects/oceans-2020.

ensure the best survival success (Bell et al. 2008) and artificial reefs can improve fisheries catches, although whether artificial reefs boost population numbers or merely attract fish is unclear (Seaman 2007). In addition, the incorporation of habitat elements in engineering structures, e.g., artificial rockpools in seawalls, shows promise in terms of ameliorating the impacts of physical alterations (Bulleri 2006). Spatial approaches to managing land-use impacts, such as marine reserves, will be covered under the section about HPSFM.

Freshwater rehabilitation has been reviewed by Roni et al. (2008). Habitat reconnection, floodplain rehabilitation and instream habitat improvement are all suggested for improving habitat and local fish abundances. Riparian rehabilitation, sediment reduction, dam removal, and restoration of natural flood regimes have shown promise for restoring natural processes that create and maintain habitats, but there is a lack of long-term studies to gauge their success. Wild eel fisheries in America and Europe have declined over time (Allen et al. 2006, Atlantic States Marine Fisheries Commission 2000, Haro et al. 2000). Declines in wild eel fisheries have been linked to a number of factors including: barriers to migration; hydro turbine mortality; and habitat loss or alteration. Information to quantitatively assess these linkages is however often lacking (Haro et al. 2000).

15.3 STATE OF KNOWLEDGE IN NEW ZEALAND

Land-based effects will be most pronounced closest to the land, therefore freshwater, estuarine, coastal, middle depths and deepwater fisheries, will be affected in decreasing order. The scale of land-use effects will, however, differ depending upon the particular influence. The most localised are likely to be direct physical impacts; for example, the replacement of natural shorelines with seawalls; although even direct physical impacts can have larger-scale impacts, such as affecting sediment transport and hence beach erosion, or contributing to cumulative effects upon ecosystem responses. Point-source discharges are likely to have a variable scale of influence, and this influence is likely to increase where a number of point-sources discharge, particularly when this occurs into an embayed, low-current environment. An example of this is Waitemata Harbour in Auckland where there are multiple stormwater discharges (Hayward et al. 2006). The influences on the largest scale can be from diffuse-source discharges such as nutrients or sediment (Kennish 2002). For example, the influence of diffuse-source materials from the Motueka river catchment in Golden Bay on subtidal sediments and assemblages and shellfish quality can extend up to tens of kilometres offshore (Tuckey et al. 2006; Forrest et al. 2007), with even a moderate storm event extending a plume greater than 6 km offshore (Cornelisen et al. 2011). Terrestrial influences on New Zealand’s marine environment can, at times, be detected by satellites from differences in ocean colour and turbidity extending many kilometres offshore from river mouths (Gibbs et al. 2006).

All coastal areas are unlikely to suffer from land-based impacts in the same way. The quantities of pollutants or structures differ spatially. Stormwater pollutants, seawalls and jetties are more likely to be concentrated around urban areas. Nutrient inputs are likely to be concentrated either around sewage outlets or associated with areas of intensive agriculture or horticulture. Sediment production has been mapped around the country and is greatest around the west coast of the South Island and the east coast of the North Island (Griffiths & Glasby 1985, Hicks & Shankar 2003, Hicks et al. 2011). Notably the catchments where improved land management may result in the biggest changes to sediment delivery to coastal environments are likely to be the Waiapu and Waipaoa river catchments on the East coast of the North Island. In addition to this, the sensitivity of receiving environments is also likely to differ; this will be covered in subsequent sections.

An MPI-funded project (IPA2007/07) reviewed the impacts of land-based influences on coastal biodiversity and fisheries (Morrison et al. 2009). This review used a number of lines of evidence to conclude that in this context, sedimentation is probably New Zealand’s most important pollutant. The negative impacts of sediment include decreasing efficiency of filter-feeding shellfish (such as cockles, pipi, and scallops), reduced settlement success and survival of larval and juvenile phases (e.g., paua, kina), and reductions in the foraging abilities of finfish (e.g., juvenile snapper). Indirect effects include the modification or loss of important nursery habitats, particularly biogenic habitats (green-lipped and horse mussel beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelps/seaweeds, and a range of other structurally complex species). Inshore filter-feeding bivalves and biogenic habitats were identified as the most likely to be adversely affected by sedimentation. Eutrophication was also identified as a potential threat from experience overseas. This review identified knowledge gaps and made suggestions for more relevant research on these influences:

- identification of fisheries species/habitat associations for different life stages, including consideration of how changing habitat landscapes may change fisheries production;
- better knowledge of connectivity between habitats and ecosystems at large spatial scales;
- the role of river plumes;
- the effects of land-based influences both directly on fished species, and indirectly through impacts on nursery habitats;
- a better spatially based understanding, mapping and synthesis of the integrated impacts of land-based and marine-based influences on coastal marine ecosystems.

The locations where addressing land-based impacts is likely to result in a lowering in risk to seafood production or increased seafood production, excluding those already mentioned, are undefined.

A national-scale threat analysis has been completed for biogenic habitats, given their likely importance for fisheries management as nursery areas (Morrison et al. 2014b). The sparse data available (often anecdotal accounts), shows that strong declines in biogenic habitats have occurred, which appear largely attributable to land-based effects (e.g., sedimentation and elevated nutrient levels), and fishing impacts. Examples include the extensive loss of seagrass meadows (e.g., large areas in Whangarei, Waitemata, Manukau, Tauranga and Avon-Heathcote estuaries), green-lipped mussel beds (about 500 km^2 in the Hauraki Gulf), bryozoan beds (about 80 km^2 in Torrent Bay, about 800 km^2 in Foveaux Strait), and deepwater coral thickets on seamounts. Cumulatively, the magnitude and extent of biogenic habitat losses are likely to have been very substantial, but are unknown, and probably will never be able to be calculated. Other biogenic habitat species for which evidence points to historical losses include horse mussels, kelp forests, oyster beds, and sponges, both in assemblages where they tend to dominate, and as part of mixed biogenic habitat assemblages. A better understanding of the threats to these biogenic habitats is recommended.

The Kaipara Harbour has been identified as a system that supports important fisheries functions both for the harbour proper, and for the wider west coast North Island ecosystem (Morrison et al. 2014a). This report detailed fish-habitat associations in the harbour and concluded that increased sedimentation, and to a lesser extent the possibility of eutrophication, was probably the greatest threat to these fisheries.

The threat of sedimentation has prompted much concern and action by land managers and local communities (Morrison et al. 2014a). For example, in the Kaipara Harbour the southern subtidal seagrass meadows area is especially important as a juvenile nursery for snapper and trevally and based on its high value as a juvenile fish nursery habitat, the Auckland Council has listed this area as an Ecologically Significant Area (ESA) in its draft unitary plan. There are significant collaborative CRI/Northland Regional Council/Auckland Council sediment erosion and transport research programmes currently under way in Kaipara Harbour catchment and the harbour itself. There are also local initiatives around tree planting and the improvement of riparian and other forms of land management. The fish/fisheries habitat work described here engages and collaborates with the IKHMG and Kaipara Research Advisory Group (KRAG), and this type of collaboration/interaction between fisheries habitat research, other scientific research programmes, and management agencies is one promising way for these issues to be addressed.

Another study investigated correlations between environmental variables and flounder abundance for the Manukau and Mahurangi harbours (McKenzie et al. 2013). Consistent correlations were obtained for a variety of environmental variables for juvenile sand and yellowbelly flounder (YBF) in the Manukau, but not in Mahurangi Harbour. The influence of environmental variables on adult YBF catch in the Manukau Harbour was even more evident. These correlations suggested that decreasing oxygen and increasing ammonia and turbidity may have negatively affected yellowbelly flounder recruitment success. When these results were considered alongside the declining trends in flatfish abundance in the FLA 1 fishery, estuarine water quality may be a significant factor affecting the sustainability of the flatfish fishery.

Marine restoration studies published in New Zealand have focused on the New Zealand cockle *Austrovenus stutchburyi*. The first of these studies identified a tagging methodology to aid relocation of transplanted individuals (Stewart & Creese 2002). Subsequent studies stressed the use of adults in restoration and the importance of site selection, either from theoretical or modelling viewpoints (Lundquist et al. 2009, Marsden & Adkins 2009). Detailed restoration methodology has been investigated in

Whangarei Harbour and recommends replanting adults at densities between 222 and 832 m-2 (Cummings et al. 2007).

Multiple influences in areas relevant to seafood production in New Zealand have been addressed by three studies. A field experiment near Auckland showed greater effects on infaunal colonisation of intertidal estuarine sediments when three heavy metals (copper, lead and zinc) were in combination compared to each in isolation (Fukunaga et al. 2010). A survey approach looking at the interaction of sediment grain size, organic content and heavy metal contamination upon densities of 46 macrofaunal taxa across the Auckland region also showed a predominance of multiplicative effects (Thrush et al. 2008). However influences can work in unexpected directions; as in a study on large suspension feeding bivalves off estuary mouths where the anticipated negative impacts from sediment were not observed and these species benefitted from food resources generated from the estuaries (Savage et al. 2012).

Toheroa populations are currently closed to all but customary harvesting but have failed to recover to former population levels even though periodic (and sometimes substantial) pulses in young recruits have been detected in both Northland and Southland (Beentjes 2010, Morrison & Parkinson 2008). Current thinking suggests that a mix of influences are probably responsible for these declines including overharvesting, land-use changes leading to changes in freshwater seeps on the beaches, and vehicle traffic (Morrison et al. 2009, Williams et al. 2013). A number of discrete pieces of research have been completed in this area. A review of the wider impact of vehicles on beaches and sandy dunes has been completed, and suggested that more research was needed on the impacts of vehicle traffic on the intertidal (Stephenson 1999). A four-day study over a fishing contest on Ninety Mile Beach showed the potential of traffic to produce immediate mortalities of juvenile toheroa, but the temporal importance of this could not be gauged (Hooker & Redfearn 1998). Mortalities of toheroa from the Burt Munro Classic motorcycle race on Oreti beach have been quantified and recommendations made for how to minimise these, but again the importance of vehicle traffic for toheroa survival over longer time periods was unclear (Moller et al. 2009). Notably, similar negative impacts from driving were observed on juvenile tuatua (*Paphies donacina*) on a Pegasus Bay beach (Marsden & Taylor 2010). The impact of a range of influences upon toheroa at Ninety Mile Beach has been investigated by Williams et al. (2013). The main factors identified that potentially affect toheroa abundance were food availability, climate and weather, sand smothering/sediment instability, toxic algal blooms, predation, harvesting, vehicle impacts, and land-use change. To investigate the causal mechanisms operating, a combination of monitoring, experimental, and modelling studies may be necessary.

Rhodolith beds have been surveyed in the Bay of Islands and high diversity was reported even in areas of abundant fine sediments (Nelson et al. 2012). It is unclear if the increasing sedimentation occurring in the Te Rawhiti Reach is negatively impacting rhodoliths and whether this atypical rhodolith bed (i.e., with abundant fine sediments) is at risk if current sedimentation and mobilisation rates continue.

The protozoan *Toxoplasma gondii* has been identified as the cause of death for 7 of 28 Hector's and Māui dolphins examined since 2007 (W. Roe, Massey University, unpubl. data, 31 July 2012). Land-based runoff containing cat faeces is believed to be the means by which *Toxoplasma gondii* enters the marine environment (Hill & Dubey 2002). A Hector's dolphin has also tested positive for *Brucella abortus* (or a similar organism) a pathogen of terrestrial mammals that can cause late pregnancy abortion, and has been seen in a range of cetacean species elsewhere. This resulted in the Department of Conservation's suggested research priorities in the 'Review of the Maui's dolphin Threat Management Plan: Consultation paper', including objectives to determine the presence, pathways and possible mitigation of the threat from *Toxoplasmosis gondii* (Department of Conservation and Ministry for Primary Industries 2012). The recently established Māui dolphin Research Advisory Group[5] confirmed risk factors to Māui dolphin from *Toxoplasma gondii* as a priority area for future research.

The effects of large-scale habitat loss and modification on eels in New Zealand are clearly significant, but difficult to quantify (Beentjes et al. 2005). Significant non-fisheries mortality of New Zealand freshwater longfin and shortfin eels are caused by mechanical clearance of drainage channels, and damage by hydro-electric turbines and flood control pumping. Eels prefer habitat that offers cover and

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[5] Department of Conservation. Māui Dolphin. Retrieved from http://www.doc.govt.nz/nature/native-animals/marine-mammals/dolphins/maui-dolphin.

in modified drains aquatic weed provides both daytime cover and nighttime foraging areas. Loss of weed and natural debris can thus result in significant displacement of eels to other areas. In addition, wetlands drainage has resulted in greatly reduced available habitat for eels, particularly shortfins, which prefer slower-flowing coastal habitats such as lagoons, estuaries, and lower reaches of rims. Water abstraction is one of a number of information requirements identified in Beentjes et al. (2005) to better define the effects on eel populations.

A number of Integrated Catchment Management (ICM) projects are underway in New Zealand. These take a holistic view to land management incorporating aquatic effects; this approach could help restore water quality of both fresh and coastal waters. An overview of these projects is given in a Ministry for the Environment Report on integrated catchment management (Environmental Communications Limited 2010). Many of these projects employ restoration techniques such as riparian planting, but few assessments of the effectiveness of riparian planting exist. One assessment of the effect of nine riparian zone planting schemes in the North Island on water quality, physical and ecological indicators concluded that riparian planting could improve stream quality; in particular, rapid improvements were seen in terms of visual clarity and channel stability (Parkyn et al. 2003). Nutrient and faecal contamination results were more variable. Improvement in macroinvertebrate communities did not occur in most streams and the three factors needed for these were canopy closure (which decreased stream temperature), long lengths of riparian planting and protection of headwater tributaries. A modelling study also demonstrated the long time lag needed to grow large trees, which then provide wood debris to structure channels, which achieves the best stream rehabilitation results (Davies-Colley et al. 2009). Although some of these studies extend into the marine realm (at least in terms of monitoring) it is difficult to gauge the impact of these activities upon fisheries or aquaculture, particularly on wider scales because ICM studies have been localised at small scales.

15.3.1 CURRENT RESEARCH

An MPI biodiversity project also has components that address land-based effects; the threats to biogenic habitats are addressed in project ZBD2008/01 (for more detail see the Biodiversity chapter).

A Ministry of Business, Innovation and Employment (MBIE)-funded project [6] of particular relevance is 'Nitrogen reduction and benthic recovery' (UOCX0902, University of Canterbury). This research aims to determine the trajectories and thresholds of coastal ecosystem recovery following removal of excessive nutrient loading (called 'eutrophication') and earthquake impacts. This will be achieved by monitoring the effects of diverting all of Christchurch's treated wastewater discharge from the eutrophied Avon-Heathcote (Ihutai) Estuary and the subsequent earthquake induced disturbances to this diversion.

15.4 INDICATORS AND TRENDS

A national view of the impacts of land-based influences upon seafood production does not exist; this could be facilitated by better coordination and planning of the many disparate marine monitoring programmes operating around the country. Monitoring of marine water quality and associated communities is carried out through a variety of organisations, including universities, regional councils and aquaculture or shell fisheries operations. Regional council monitoring of water quality and associated biological communities is often reported through websites such as the Auckland Regional Council environmental monitoring data, or summary reports such as the Hauraki Gulf state of the Environment 2011 report (Auckland Regional Council 2011). Water quality and associated marine communities may also be monitored for a regional council as part of a consent application or as a stipulation for a particular marine development. However the data from aquaculture and shellfisheries water quality monitoring are not generally available.

Improved coordination and planning of marine monitoring has been achieved in some countries, e.g., the United Kingdom. [7] The Marine Environmental Monitoring Programme (ZBD2010-42), is a step towards this goal, more information is available on this project in the Biodiversity chapter of this document. This project identifies remote sensing of sea surface particulate matter in nearshore waters as a possible indicator of changes in sediment inputs in the future, but this requires algorithm validation for New

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[6] Ministry of Business, Innovation and Employment. Who got funded? Retrieved from http://www.msi.govt.nz/update-me/who-got-funded.

[7] CEFAS. Marine monitoring. Retrieved from http://wavenet.cefas.co.uk/Smartbuoy.

Zealand waters. Possible national-scale proxies for coastal faecal contamination may exist after collating information from sanitation area monitoring for shellfish harvesting and/or coastal bathing beaches.

High faecal coliform counts (primarily from mammal or bird faeces) can impact upon the value gained from shellfish fisheries and aquaculture. Area closures to commercial harvesting usually depend on an area's rainfall/runoff relationship and areas closer to significant farming areas or urban concentrations are likely to be closed more frequently, due to high faecal coliform counts, than areas where the catchment is unfarmed or not heavily populated. For example, Inner Pelorus sound is likely to be closed more frequently than outer Pelorus Sound (Marlborough Sounds). For coastal areas of the Marlborough Sounds, the Coromandel Peninsula and Northland closures can range from a few days to over 50% of the time in a given year (Brian Roughan, New Zealand Food Safety Authority, pers. comm.). Certain fisheries may be limited by the amount of time where water quality is sufficient to allow harvesting, e.g., the cockle fishery in COC 1A (Snake bank in Whangarei harbour) was closed for 101, 96, 167, 86, 117 and 118 days for the 2006–07, 2007–08, 2008–09, 2009–10, 2010–11 and 2011–12 fishing years, respectively, due to high faecal coliform counts from sewage spills or runoff.[8] Models also now exist that allow real-time prediction of *E. coli* pulses associated with storm events (e.g., Wilkinson et al. 2011), which may help harvesters to better cope with water quality issues.

The Ministry for the Environment (MfE) also reports on freshwater quality. River water quality indicators that have been assessed have direct relevance to the eel, and other freshwater fisheries, and this water will flow through estuaries and enter the marine environment. The National River Water Quality Network (NRWQN) has national coverage, and has been running for over 20 years and has recently reported upon the following eight variables: temperature, dissolved oxygen, visual clarity, dissolved reactive and total phosphorous, and ammoniacal, oxidised and total nitrogen (Ballantine & Davies-Colley 2009). Dissolved oxygen showed few meaningful trends and the ammoniacal nitrogen data suffered from a processing artefact. An upward, although not significant trend in temperature and an improvement of water clarity were seen at the national scale. However, a negative correlation was seen between water clarity and percent of catchment in pasture, which suggests that any expansion of pasture lands may have impacts on clarity. Strong increasing trends over time were seen in oxidised nitrogen, total nitrogen, total phosphorous and dissolved reactive phosphorous. These latter trends all signify deteriorating water quality and are mainly attributable to increased diffuse-source pollution from the expansion and intensification of pastoral agriculture.

Total nitrogen and phosphorous loads to the coast in New Zealand have been modelled and were estimated at 167 300 and 63 100 t yr-1, respectively (Elliot et al. 2005).[9] The main sources of nitrogen and phosphorous were from pastoralism (70%) and erosion (53%), respectively. Dairying contributes 37% of the nitrogen load from only 6.8% of the land. The total amount of land used for dairy farms increased by 47% (1.4 to 2.0 million ha) from 1986 to 2002.[10] These statistics provide strong circumstantial evidence that the expansion in dairying is primarily responsible for the observed declines in water quality from agricultural sources.

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[8] Statistics supplied by New Zealand Food Safety Authority in Whangarei. Notably the fishery has not been operating since November 2012.

[9] This is a known underestimate because streams with catchments less than 10 km^2 were excluded from this calculation.

[10] Statistics NZ (2006) Fertiliser Use and the Environment. Retrieved from http://www3.stats.govt.nz/environment/Fertiliser_use_and_the_environment_Aug06.pdf.

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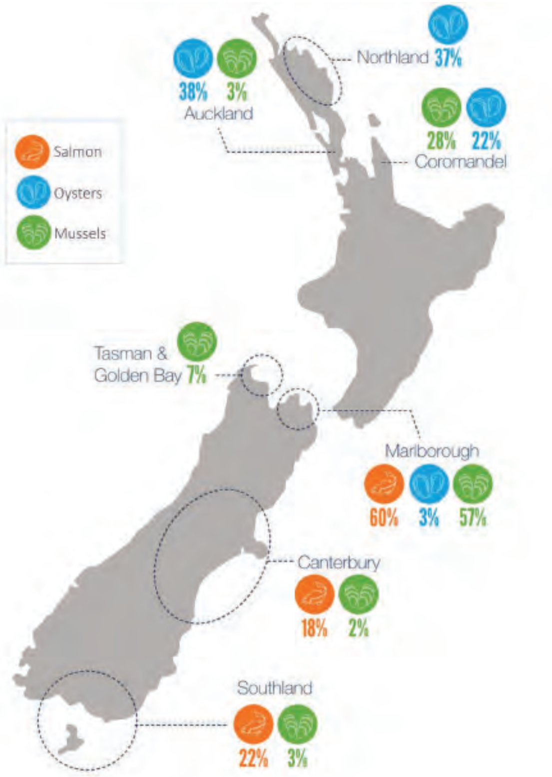
Chapter 16: Ecological effects of marine aquaculture - Technical Summary

Aquaculture is a growing sector in NZ



Ecological effects are considered as part of any development

1. THE ISSUE IN BRIEF

- Aquaculture is the fastest growing food producing industry, supplying nearly half of the world's seafood
- NZ aquaculture industry is valued at over $600m and the NZ Government's strategy supports the potential for aquaculture to be a $3bn. industry by 2035.
- Marine aquaculture in NZ is dominated by green-lipped mussels, chinook (king) salmon, and Pacific oysters
- Aquaculture activities can have ecological effects on marine ecosystems. Some are positive, others less so. Research is ongoing to ensure aquaculture remains sustainable in New Zealand

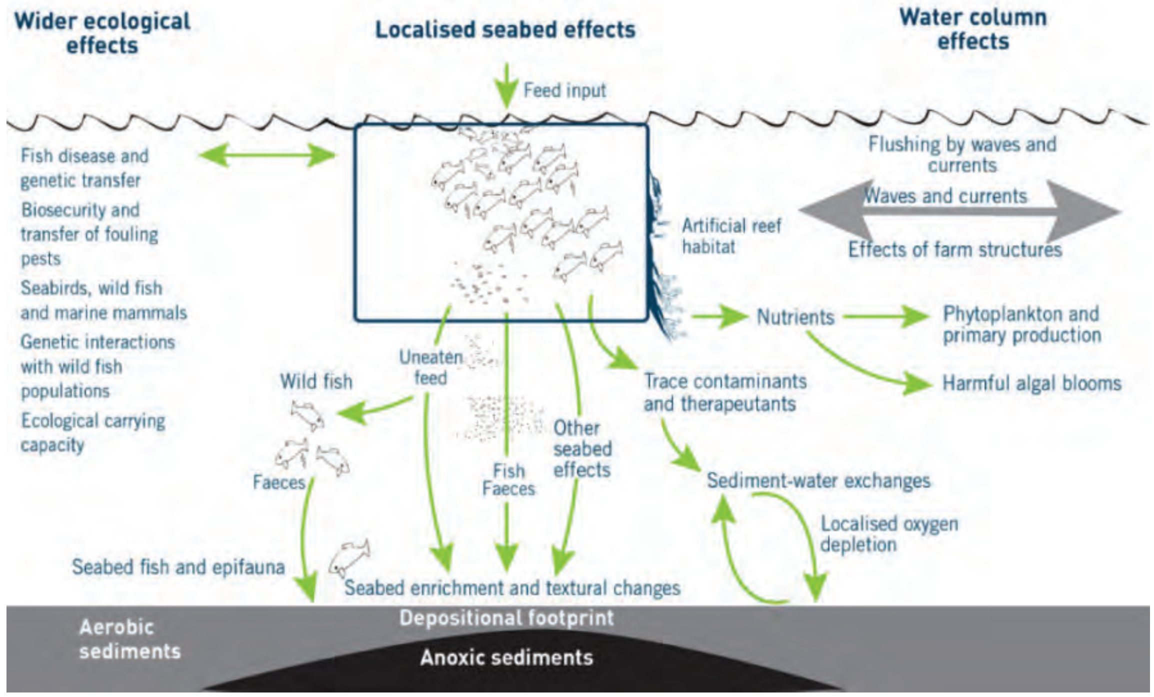
2. ECOLOGICAL EFFECTS

Different types of aquaculture (see examples A and B below) can affect marine ecosystems at different spatial scales

Potential local seabed effects include:

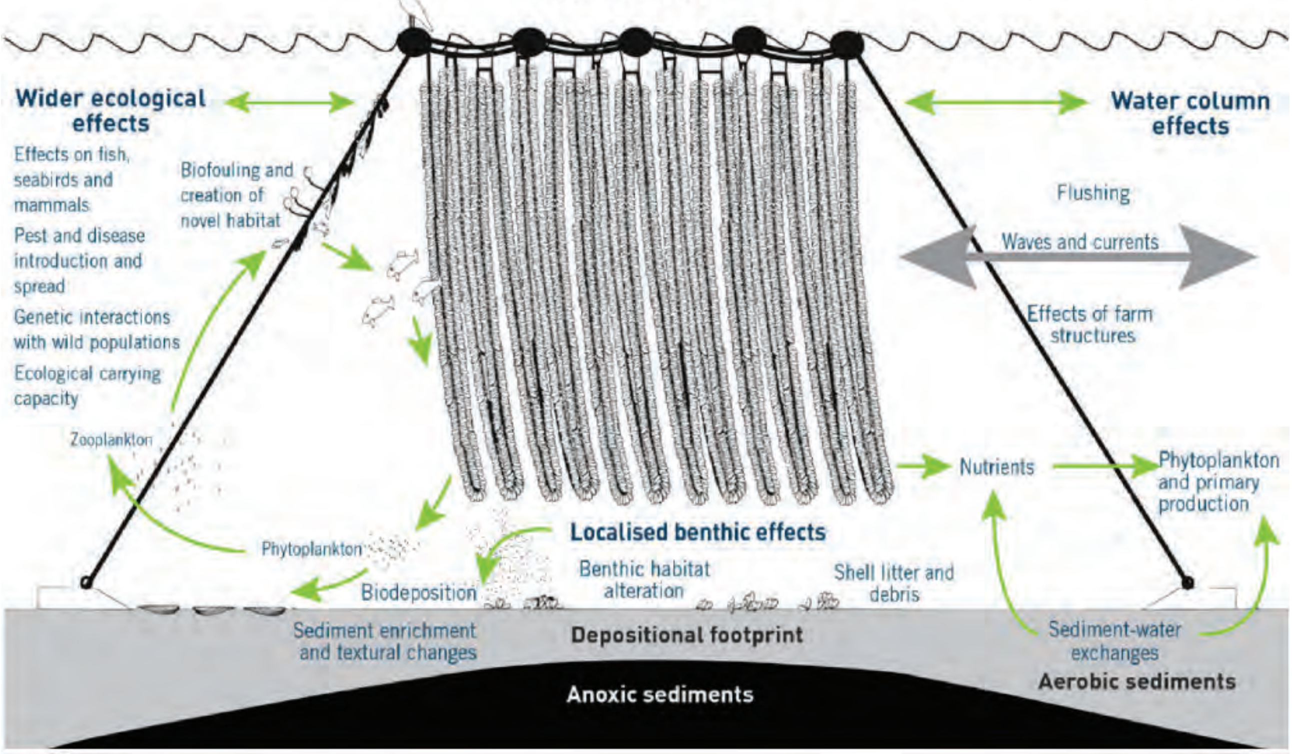
- Organic enrichment of the seabed by biodeposition beneath the farm (A+B)
- Smothering of benthic organisms and changes to the physical composition of sediments through deposition of shell litter and debris (A+B)

Example A: Finfish Farm



Schematic representation of the potential effects of marine finfish aquaculture (edited from Forrest et al. 2007)

Box 2. *Continued*


Example B: Mussel Farm

Schematic representation of the potential effects of marine shellfish aquaculture (edited from Keeley et al. 2009)

Potential local water quality effects include:

- Phytoplankton depletion and changes in planktonic community composition by cultured filter feeders (B)
- Nutrient enrichment effects from fish pellets and excretory products (A)
- Depletion of dissolved oxygen by respiration of farmed organisms (A+B)

Potential wider ecosystem effects include:

- Facilitation of the introduction, establishment, and spread of pests, parasites, and diseases (A+B)
- Disturbance, displacement, or entanglement of marine mammals and birds (A+B)
- Attraction of predators/scavengers and attraction of wild fish to aquaculture structures (A+B)
- Creation of habitats in the water column and on the seabed supporting increased biodiversity (B)
- Reduction of nutrient loads in the water column originating from other human activities (B)

3. MITIGATION OF INSHORE AQUACULTURE EFFECTS

- Aquaculture planning and consenting processes in New Zealand are managed by Regional Councils under the Resource Management Act 1991, and the potential effects of farms are considered in the consenting process
- Careful siting of farms is required to ensure adequate flushing, avoiding critical breeding and foraging areas and sensitive benthic habitats
- Consent conditions are set by Regional Councils to minimise potential ecosystem effects. In addition several voluntary measures have been developed
- Best management practices for monitoring and managing effects of salmon farming on benthic habitats and water quality have been developed for the Marlborough Sounds
- Biosecurity risk management (e.g., of diseases or parasites) is guided by industry codes of practice and an Aquaculture Biosecurity Handbook

4. KNOWLEDGE GAPS

- Cumulative effects and the carrying capacity of coastal regions where aquaculture activities take place
- Ecological effects of potential new farmed species
- Effects of open ocean aquaculture on migrating/protected species and on the offshore marine environment

16 ECOLOGICAL EFFECTS OF MARINE AQUACULTURE

Status of chapter	This chapter has been updated for AEBAR 2021.
Scope of chapter	The known ecological effects of marine aquaculture operations in New Zealand.
Area	The New Zealand EEZ and territorial sea. Presently aquaculture operations are located coastally, however, options for open ocean aquaculture are currently being explored.
Focal localities	Northland, Coromandel, Auckland, Bay of Plenty, Marlborough Sounds, Tasman Bay and Golden Bay, Canterbury, Southland.
Key issues	Uncertainty in predictions, cumulative effects, benthic effects of aquaculture in open ocean environments.
Emerging issues	Marine spatial planning, integration of monitoring datasets, effects of open ocean aquaculture, diversification of farmed species, pathogens affecting salmon, and parasites of flat oysters and pāua. The role of seaweeds in New Zealand aquaculture development.
Fisheries New Zealand research (current)	SEA2018-13 and AQU2021-01 *Validation of an eDNA multi-trophic metabarcoding enrichment stage monitoring tool.* Aquaculture research budget *Method to report environmental performance of aquaculture*.
New Zealand research (current)	Sustainable Farming Fund projects: The use of seaweed aquaculture for bioremediation (2 projects). MBIE Endeavour fund project: Greenwave regenerative ocean farming of seaweed. Phase II of the Sustainable Seas Challenge Projects: *Ecological responses to cumulative effects* and *Tools for managing cumulative effects*; *The Moana Project. Re-imagining aquaculture: inventing low-impact, offshore mobile technology that transforms finfish production.* Science for Technological Innovation National Science Challenge project: *Molecular biosensors to detect and monitor toxins from harmful algal blooms*. Strategic Science Investment Fund project: *Monitoring tools and farm management systems that quantify and minimise both the environmental effects and compliance costs of aquaculture*.
Related issues	Land-based effects, marine biodiversity, habitats of particular significance for fisheries management, climate change.

16.1 CONTEXT

Aquaculture is the world’s fastest growing major food production sector and in 2018 supplied 49.4% of the supply of aquatic food globally (excluding seaweeds) (FAO 2020). The aquaculture sector in New Zealand provided over 3000 jobs in 2018 and generated over $600 million in revenue (Aquaculture New Zealand 2019). In 2018 the Oceania region (which includes New Zealand and Australia) produced only 0.25% of the world’s aquaculture production (approximately 205 000 t); globally 82 million t were produced (FAO 2020).

The annual value of New Zealand aquaculture exports from 1998 to 2019 has been dominated by green-lipped mussels ($336 million in 2019), salmon ($115 million in 2019), and pacific oysters ($21 million in 2019) (Aquaculture New Zealand 2020). New Zealand has 15 000 km of coastline, making it one of the longest in the world. There are currently around 20 000 hectares of water space allocated for marine farming. Almost all the current commercial production comes from near-shore areas which account for 40% of the total allocation. The main current aquaculture locations in New Zealand are shown in Figure 16.1.

There are opportunities to grow the sector in New Zealand by improving the value of existing farming space, and by extending into land-based and open ocean aquaculture. Based on annual sales, records of growth, and development opportunities, Fisheries New Zealand predicts that New Zealand’s aquaculture industry could grow to $3 billion in annual sales by 2035.

The potential benefits and opportunities offered by aquaculture are recognised by the New Zealand Government, which developed an Aquaculture Strategy in 2019 (Fisheries New Zealand 2019). One of the four key outcomes provided in the strategy is that aquaculture will lead in sustainable environmental practices. The National Environmental Standard (NES) for marine aquaculture (MA), came into effect on 1 December 2020. The NES-MA provides a nationally consistent framework for regional councils to manage existing aquaculture activities. The NES-MA:

- increases regulatory consistency and certainty,
- ensures environmental effects are appropriately managed, and
- increases industry confidence to promote investment.

Sustainable development of aquaculture in New Zealand needs to be supported by good quality information on ecological effects to enable appropriate decision making. The aquaculture unit of MPI funded a collaborative project between NIWA and the Cawthron Institute to review the ecological effects of aquaculture (PRM2010-36). The MPI (2013) overview report and related literature reviews summarise the potential ecological effects of different types of aquaculture, discuss the magnitude and significance of those effects, consider management and mitigation options, and describe key knowledge gaps. This chapter largely summarises the findings of that larger document (MPI 2013), which should be referred to for further details, references, or clarification. In addition, MPI has supported and will continue to support work to improve environmental performance through the development of best practice standards. Recent developments in best practice guidelines to monitor and manage environmental effects of aquaculture are reported in the relevant sections below.

Increased industry interest in 'open ocean' salmon farming has resulted in four applications for open ocean sites in Marlborough and Southland. The state of knowledge of the potential effects of open ocean salmon farming in a New Zealand context has recently been reviewed (Bennett et al. 2020, Keeley 2020). The initial consent hearing from the New Zealand King Salmon Blue Endeavour open ocean salmon farm application has elicited a significant body of work and professional opinion on the potential environmental effects of offshore aquaculture. The outcome of this application is not yet known.

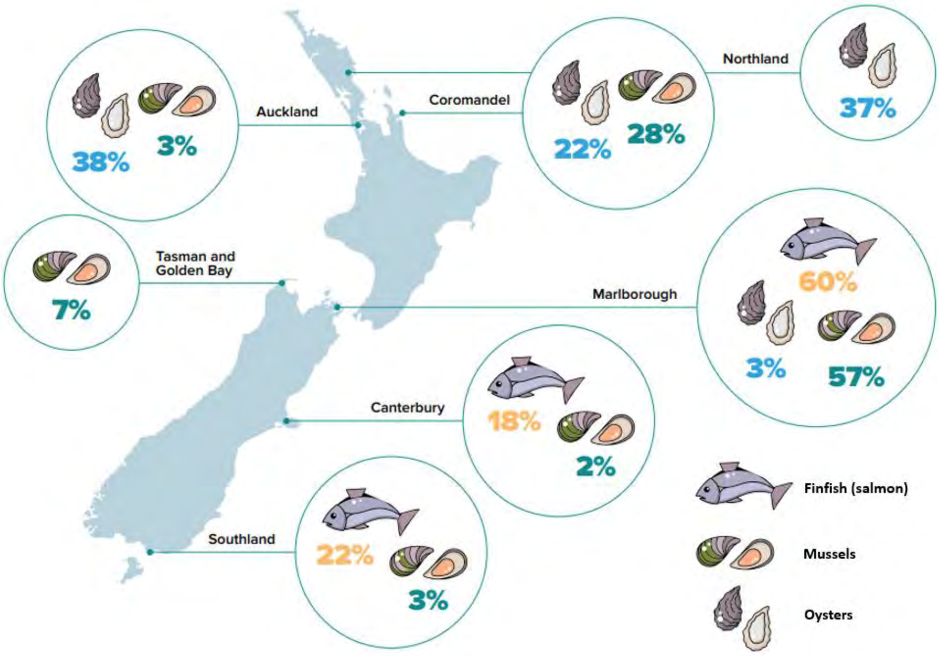


Figure 16.1: Geographic locations of main marine farming areas in New Zealand (Fisheries New Zealand 2019).

In 2012 an expert panel approach was used to trial a method for prioritising the ecological effects from aquaculture (see summary in Figures 16.2 and 16.3) (Stoklosa et al. 2012). This process brought together 17 knowledgeable participants from across a range of interested parties (central and local government, aquaculture industry, and scientists), to attempt to gain consensus on the relative importance of a range of ecological effects from aquaculture.

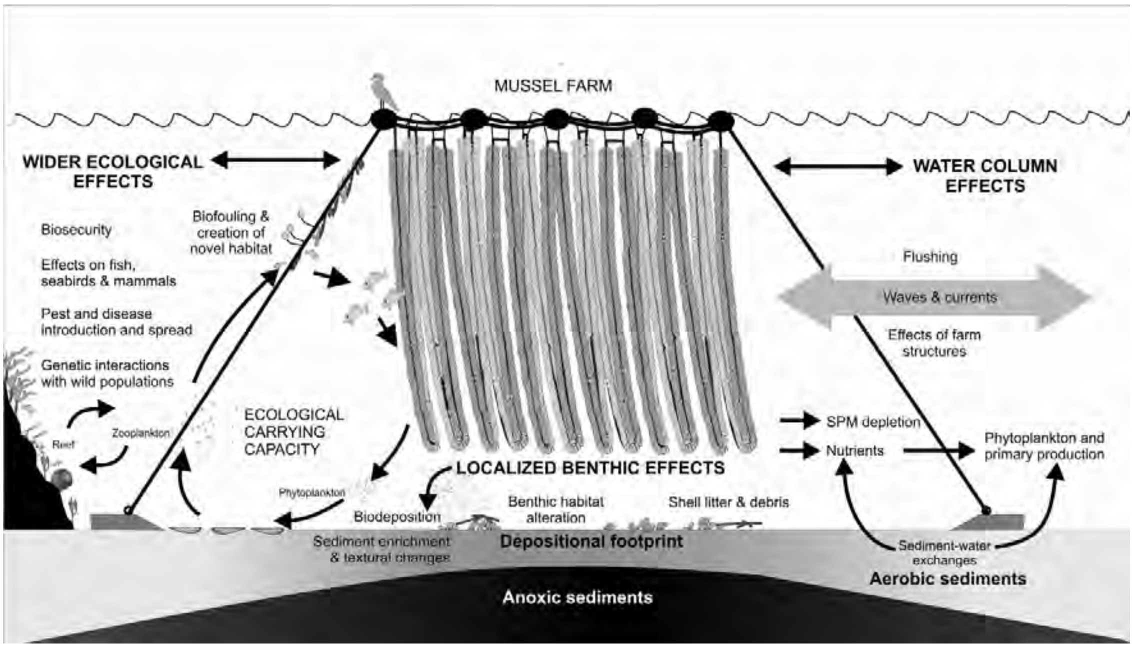


Figure 16.2: Schematic of actual and potential ecological effects from mussel farming (Keeley et al. 2019).

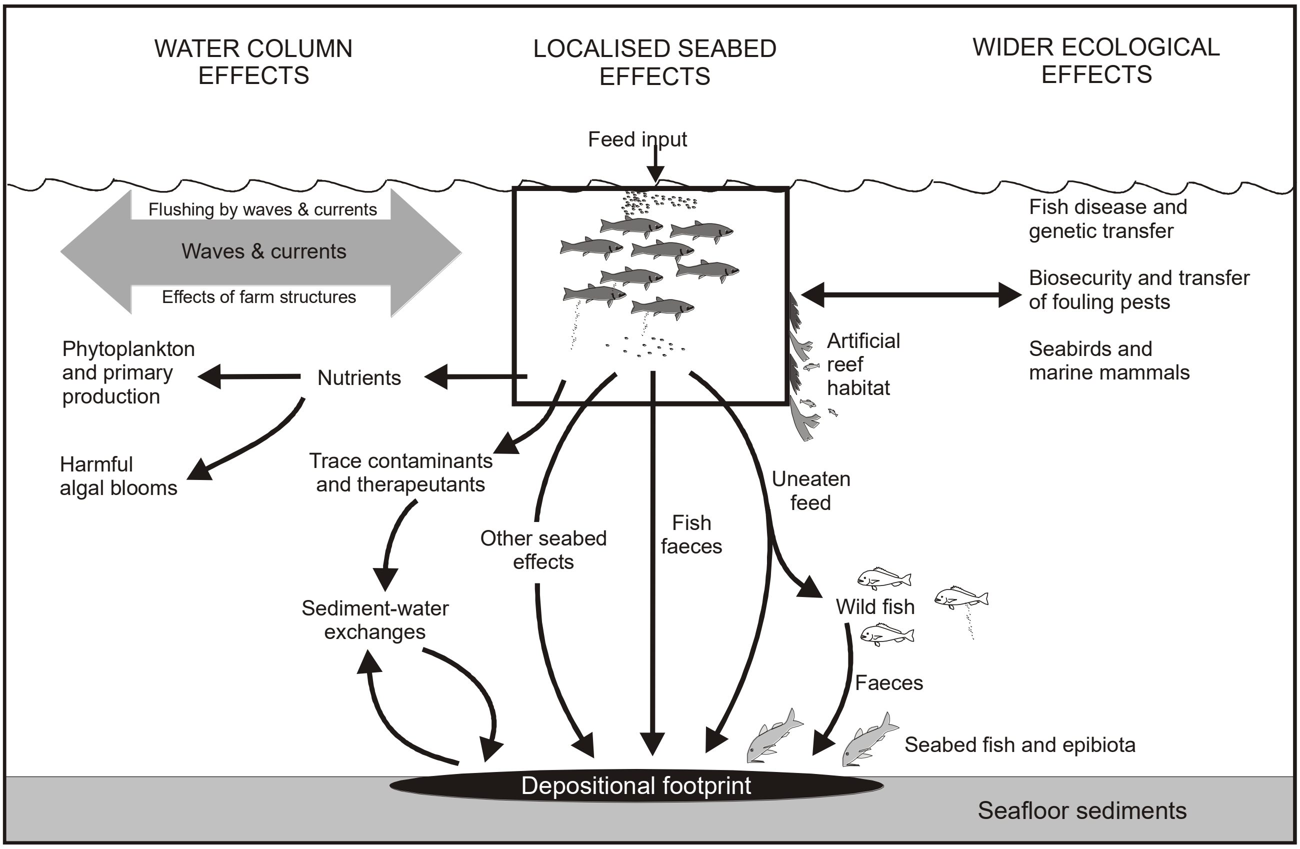


Figure 16.3: Schematic of actual and potential ecological effects from feed-added farming (from Forrest et al. 2007c).

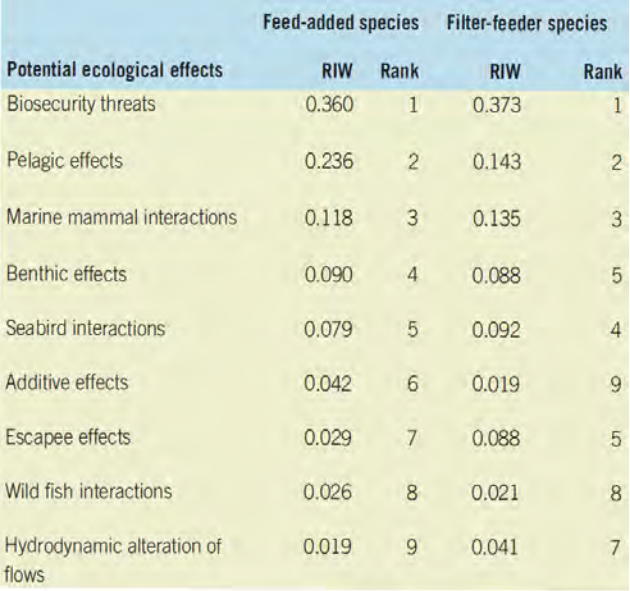
The results of this process are only indicative, and are focused on inshore aquaculture, but for both feed-added and filter-feeding species the same three issues were identified as most important; these were (in decreasing order of importance): biosecurity threats, pelagic effects, and marine mammal interactions (Table 16.1). Notably the score for the threat from biosecurity was more than 50% greater than the next highest score and the threat of pelagic effects was rated as markedly higher for feed-added species than it was for filter-feeders. Other potential ecological threats considered were of lesser importance and are listed below the top three (in no particular order), along with an explanation of what was considered under each term. Interactions between threats and large-scale effects were not covered within this prioritisation exercise.

1. Biosecurity threats — how aquaculture may influence risks associated with pests and diseases.
2. Pelagic effects — aquaculture effects on the water column (excluding those explicitly dealt with by other chapters in the MPI 2013 literature review), at approximately the scale of the farm.
3. Marine mammal interactions — aquaculture effects on marine mammals.

- Benthic effects — aquaculture effects on the seafloor.
- Seabird interactions — aquaculture effects on birds.
- Effects from additives — effects of chemicals used in aquaculture upon the environment.
- Escapee effects — effects of escaped farmed species upon the environment.
- Wild fish interactions — aquaculture effects on non-farmed fish populations.
- Hydrodynamic alteration of flows — aquaculture effects on the water movement at scales greater than the farm scale.

These topic areas will be discussed further under each of their headings below (in the order above). In addition, note that stressors do not act in isolation, and any aquaculture impacts will occur within the context of (and potentially interacting with) other anthropogenic stressors and ongoing natural processes. The interacting and cumulative effects of aquaculture will be discussed in section 16.1.10 of this chapter. While the 2012 study focused on inshore aquaculture, the priority areas are also considered to be aligned with issues relating to open ocean developments.

Table 16.1: Trial prioritisation of potential classes of aquaculture effects from Stoklosa et al. (2012). Results of pair-wise comparisons using the Analytical Hierarchy Process (Saaty 1987) from the phase two workshop of the Aquaculture Ecological Guidance Project. RIW = relative importance weight. Order is decreasing in importance for the feed-added species.[1]



	Feed-added species		Filter-feeder species	
Potential ecological effects	**RIW**	**Rank**	**RIW**	**Rank**
Biosecurity threats	0.360	1	0.373	1
Pelagic effects	0.236	2	0.143	2
Marine mammal interactions	0.118	3	0.135	3
Benthic effects	0.090	4	0.088	5
Seabird interactions	0.079	5	0.092	4
Additive effects	0.042	6	0.019	9
Escapee effects	0.029	7	0.088	5
Wild fish interactions	0.026	8	0.021	8
Hydrodynamic alteration of flows	0.019	9	0.041	7

16.1.1 BIOSECURITY THREATS

Biosecurity threats posed by aquaculture activities have been reviewed by Forrest et al. (2011) for finfish and Keeley et al. (2009) for other species, and then compiled and summarised by MPI (2013). In 2016 MPI in collaboration with Aquaculture New Zealand published a technical document which provides information on the types of biosecurity risks and risk organisms and draws on national and international best practice to suggest objectives and management options for an integrated approach to on-farm biosecurity management (Georgiades et al. 2016). This section draws heavily from these sources, and the reader is referred to them for more detail.

16.1.1.1 INTRODUCTION

Aquaculture facilities (including hatcheries and open water operations) have an inherent risk of pest and disease introduction, exacerbation, or spread. Biosecurity is a set of preventive measures designed to exclude, eradicate, or

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[1] Notably there was a chapter by MPI (2013) on the potential effects from genetic manipulation and polyploidy. However, genetic manipulation is controlled by the Environmental Protection Authority (EPA) and is not authorised for use in aquaculture. Polyploidy was also considered by the risk assessment workshop participants to be relatively rare in aquaculture and therefore this topic area was not considered by the prioritisation.

effectively manage the risk posed by pests and pathogens[2]. Biosecurity risk organisms include animals, plants, and micro-organisms capable of causing diseases (e.g., the ostreid herpes virus in pacific oysters) or otherwise adversely affecting New Zealand's natural, traditional, socio-cultural, or economic values (e.g., the sea squirt *Styela clava* and the Mediterranean fanworm *Sabella spallanzini*). In an aquaculture context, biosecurity also encompasses indigenous species already present in the environment that become enhanced as a result of culture operations (Forrest et al. 2011).

The primary source of entry for biosecurity risk organisms into New Zealand is through international shipping (Cranfield et al. 1998, Kospartov et al. 2010, Bell et al. 2011). Aquaculture pathways for on-farm introduction of pathogens and pests are provided by the open design of near-shore aquaculture sites and the input of seawater or freshwater into land-based systems (Peeler 2005, Johansen et al. 2011, Fitridge et al. 2012), broodstock and stock introductions, stock movements, shared infrastructure, equipment and vessels, staff, contractors or visitors, wildlife, and feed (Georgiades et al. 2016).

Aquaculture production systems may increase biosecurity risk, through acting as reservoirs or exacerbators (Okamura & Feist 2011, Peeler & Taylor 2011). Reservoirs host risk-organisms that can then spread by either natural or human-mediated mechanisms. Exacerbators create incubators/stepping stones for otherwise benign or low impact pests, pathogens, or parasites (both native and exotic species). Densely farmed populations may exacerbate the likelihood of disease outbreaks as stressed or weaker individuals that are more susceptible to infection may transmit the pathogens or parasites to healthy individuals (Handlinger et al. 2006, Robertsen 2011). Propagule pressure from pests and diseases may be exacerbated by densely farmed populations, if not managed effectively on site. In addition, farm infrastructure provides potential habitat for the settlement of biofouling organisms.

Considerable effort is placed on preventing incursions of pests, parasites, and diseases into the New Zealand environment. This is because the introduction, proliferation, and spread of risk species in New Zealand can have effects on marine and freshwater environments that are often difficult to manage, resulting in permanent and irreversible impacts (Forrest et al. 2011). The few successful efforts to eradicate aquatic invasive species (AIS) have several common elements (Locke et al. 2009b), which are unlikely to occur in combination:

- early detection and correct identification of the invader,
- pre-existing authority to take action,
- the ability to sequester the AIS to prevent dispersal, (or else the AIS had very limited dispersal capabilities),
- political and public support for management and eradication,
- acceptance of some collateral environmental damage,
- plans in place to minimise re-infestation of pests and diseases through surveillance and subsequent response actions, and
- follow-up monitoring to verify the completeness of the eradication.

The potential to develop treatment responses to marine AIS incursions was reviewed by Cahill et al. (2021). The review provides a snapshot of the current state-of-play for marine pathway treatments for AIS. Some treatment options are well understood from viewpoints of efficacy, safety, biosecurity, quality control, and compliance. However further work is required to deliver implementation ready AIS treatment protocols that can be reliably applied at scale.

Environmental factors including depth, wave climate, temperature regime, and currents influence dispersal of waste, disease agents, and pests and therefore play a significant role in determining the potential biosecurity risk for a given site.

For example, individual farms within any one Aquaculture Management Area (AMA) in Nelson Bays could function as a source of infection to other AMAs in Golden Bay (Zeldis et al. 2011b) via the transfer of viral or bacterial pathogens. Dispersion potential (within farms, between farms, or between blocks of farms), which is largely controlled by hydrodynamics, will also be influenced by temperature, because temperature can regulate metabolic growth and the proliferation of bacteria/viruses etc. that are shed as free-living single-celled organisms (Zeldis et al. 2011b).

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[2] Defined here as an agent of disease, e.g., a bacterium or virus.

Temperature and salinity can also affect the biosecurity risks associated with individual species by controlling their range. For example, in the proliferation of invasive pacific oysters, the southern distribution is limited to Nelson-Marlborough, because water temperatures further south are too low for successful reproduction (Quale 1969, Askew 1972, Dinamani 1974). Salinity can vary with season, climatic variation (Scavia et al. 2002), and the catchment rainfall, with catchments that are dry in summer producing less runoff, elevating coastal salinities, which then affect the distribution of fouling species (S. Handley, NIWA, unpublished data). Farm stocks that may be susceptible to biosecurity risks are usually at greatest risk in summer. Summer is when temperatures, and hence metabolic rates of farmed animals, are highest, dissolved oxygen levels in the water are lowest (hence the risk of oxygen deprivation is highest), and the proliferation of biofouling populations is also greatest (S. Handley, NIWA, unpublished data).

Aquaculture space allocation in New Zealand has predominantly been driven by constraint mapping, allocating space in areas that do not conflict with other users and stakeholders (e.g., Handley & Jeffs 2003). This strategy increases potential biosecurity risks by encouraging development of aquaculture at environmentally less favourable sites. The use of ecosystem-based approaches to aquaculture development that incorporate tools like GIS can incorporate biosecurity risks (if known) to optimise site selection even in data poor environments (Aguilar-Manjarrez et al. 2010, Soto et al. 2008, Silva et al. 2011). In addition, models, which provide simulations of pest and pathogen spread, can be used to inform site selection. Biosecurity New Zealand have recently commissioned MetOcean Solutions Ltd to develop an application to examine coastal connectivity which could contribute to spatial planning.

16.1.1.2 SIGNIFICANCE OF EFFECTS

It is generally recognised that adverse ecological effects arising from pests, parasites, and pathogenic species associated with aquaculture can result in a range of threats including (Molnar et al. 2008):

a. disruptions to entire ecosystem processes with wider abiotic influences,
b. disruptions to wider ecosystem function, and/or keystone species or species/assemblages of high conservation value (e.g., threatened species),
c. disruptions to single species with little or no wider ecosystem impact, and
d. little or no disruption.

The infection of marine farms by pest organisms can lead to the development of significant infestations on farm structures, which may then:

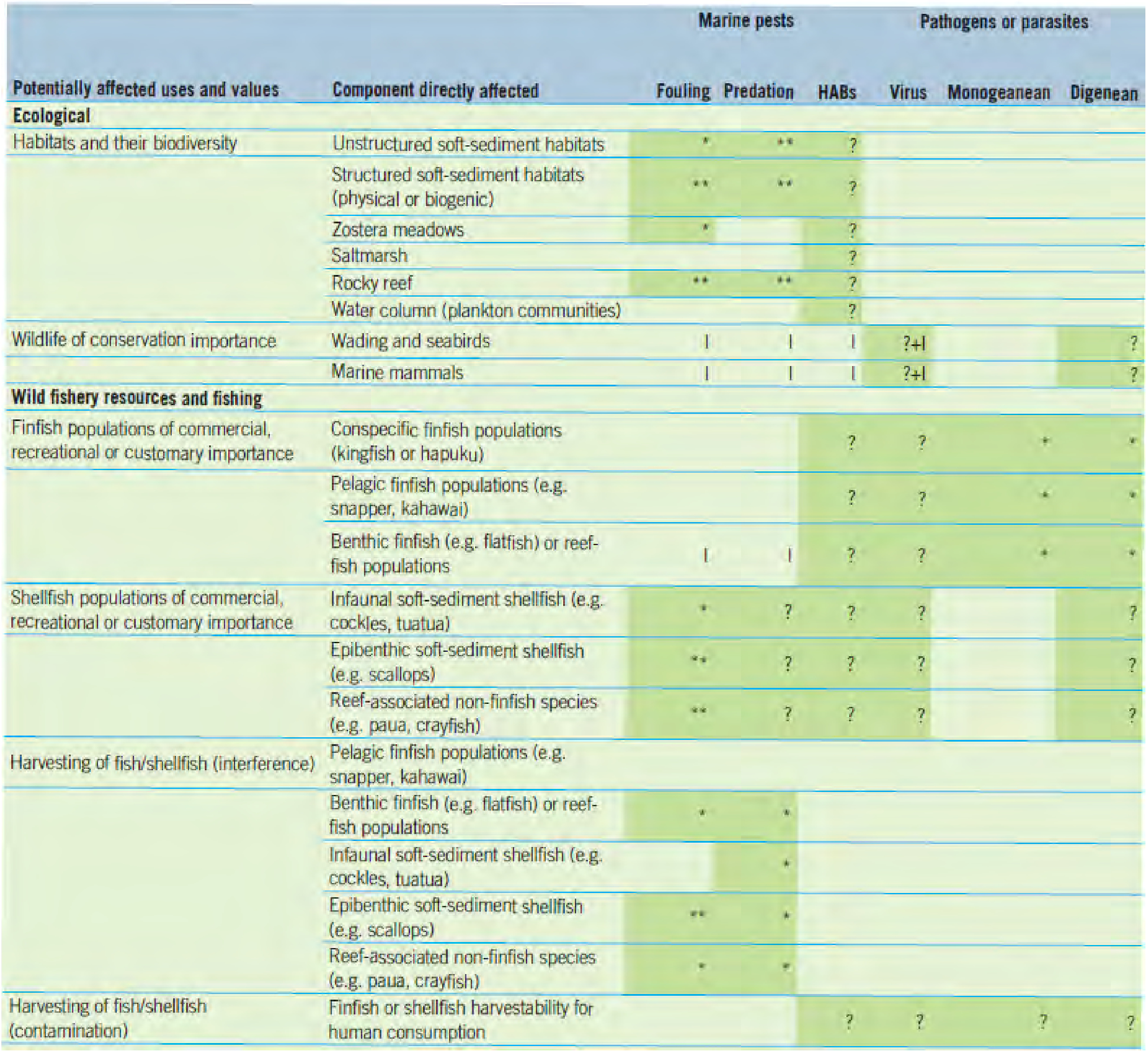
1. act as a reservoir for subsequent spread to natural ecosystems and to the stock being cultured,
2. increase drag on cages and anchoring systems in high current areas, which in turn increases the chance of escapee effects if stocks are infected with pathogens or parasites (Forrest et al. 2011),
3. restrict water exchange, and thereby cause a build-up of waste, decrease in oxygen levels, and reduction in food availability (Fitridge et al. 2012)
4. harbour pathogens whose spread may be facilitated by aquaculture activities (Georgiades et al. 2016), and
5. compete for food with cultured species and thereby reduce stock growth (Georgiades et al. 2016).

Examples of significant effects from pest fouling organisms on aquaculture activities in New Zealand include documented impacts from infestation of marine farms with *Undaria* and the colonial tunicate *Didemnum vexillum* (e.g., Forrest & Taylor 2002 and L. Fletcher, Cawthron, unpublished data). As well as attached fouling organisms, aquaculture structures may also act as recruitment substrata for mobile pelagic or benthic species (e.g., jellyfish, ctenophores, sea star *Asterias amurensis*, sea cucumbers, or the crab *Carcinus maenas*; Forrest et al. 2009, 2011).

An example of the ecological effects stemming from a pathogen is the outbreak of pilchard herpes virus that was thought to have stemmed from pilchards imported for tuna aquaculture feed in South Australia. This event caused widespread mortality of pilchards, which led to starvation and the recruitment failure of little penguins, which prey on pilchards (Dann et al. 2000). The pathogens ostreid herpes virus microvariant 1 (Keeling et al. 2014) and *Bonamia ostreae* (Lane 2016) have been detected in New Zealand oysters, and disease outbreaks and management decisions have had significant impacts on New Zealand's pacific oyster (*Magallana gigas*) and flat oyster (*Ostrea chilensis*) industries, respectively (Castinel et al. 2015, Sim-Smith et al. 2016, Ross et al. 2017). The costs of the biosecurity responses to these pathogens have been far greater than that for macro-fouling organisms.

Any attempt to assess the significance of potential effects of invasive pests, pathogens, or parasites in terms of their magnitude will be limited by the lack of robust information on the affected environments, inherent difficulties in making reliable predictions regarding the invasiveness of different species, and hence inferences regarding their direct or indirect effects (Forrest et al. 2011). The potential effects of pests and pathogens are given in Table 16.2 for finfish aquaculture in the Waikato region.

Table 16.2: Matrix illustrating the often unknown effects of pests, pathogens, and parasites associated with finfish aquaculture in the Waikato region. Examples are given of direct interactions (shaded cells) between potential biosecurity hazards and values in the Waikato region, and indirect effects (I). Direct interactions designated as: may be an important incremental risk above that already occurring (**), and probably a minor incremental risk (*). ? = direct interaction possible but significance unknown. From Forrest et al. (2011).



		Marine pests			Pathogens or parasites		
Potentially affected uses and values	**Component directly affected**	**Fouling**	**Predation**	**HABs**	**Virus**	**Monogeanean**	**Digenean**
Ecological							
Habitats and their biodiversity	Unstructured soft-sediment habitats	*	**	?			
	Structured soft-sediment habitats (physical or biogenic)	**	**	?			
	Zostera meadows	*		?			
	Saltmarsh			?			
	Rocky reef	**	**	?			
	Water column (plankton communities)			?			
Wildlife of conservation importance	Wading and seabirds	I	I	I	?+I		?
	Marine mammals	I	I	I	?+I		?
Wild fishery resources and fishing							
Finfish populations of commercial, recreational or customary importance	Conspecific finfish populations (kingfish or hapuku)			?	?	*	*
	Pelagic finfish populations (e.g. snapper, kahawai)			?	?	*	*
	Benthic finfish (e.g. flatfish) or reef-fish populations	I	I	?	?	*	*
Shellfish populations of commercial, recreational or customary importance	Infaunal soft-sediment shellfish (e.g. cockles, tuatua)	*	?	?	?		?
	Epibenthic soft-sediment shellfish (e.g. scallops)	**	?	?	?		?
	Reef-associated non-finfish species (e.g. paua, crayfish)	**	?	?	?		?
Harvesting of fish/shellfish (interference)	Pelagic finfish populations (e.g. snapper, kahawai)						
	Benthic finfish (e.g. flatfish) or reef-fish populations	*	*				
	Infaunal soft-sediment shellfish (e.g. cockles, tuatua)		*				
	Epibenthic soft-sediment shellfish (e.g. scallops)	**	*				
	Reef-associated non-finfish species (e.g. paua, crayfish)	*	*				
Harvesting of fish/shellfish (contamination)	Finfish or shellfish harvestability for human consumption			?	?	?	?

16.1.1.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

One of the key outcomes of the Aquaculture Strategy (Fisheries New Zealand 2019) is to ensure the industry is resilient by protecting aquaculture from biological harm and supporting it in adapting to climate change. Strengthening biosecurity management by shifting from reactive responses to planned and active management has been identified as an objective to help achieve this outcome.

Biosecurity control of aquaculture activities currently occurs through resource consent conditions, codes of practice, and import health standards. The resource consenting process under the Resource Management Act (RMA) considers biosecurity via factors such as farm spacing, zoning,[3] staged development, and epidemiological units.

Biosecurity New Zealand in collaboration with Aquaculture New Zealand published a technical reference document containing detailed information about preventative on-farm biosecurity (Georgiades et al. 2016) and an aquaculture biosecurity handbook (MPI 2016), which provides guidance and practical and cost-effective options for biosecurity. The purpose of the reference document and handbook is to assist the industry in strengthening their on-farm biosecurity practices by providing technical information on the types of biosecurity risks, risk pathways, and risk organisms and suggesting biosecurity objectives and management options based on national and international best practice.

Biosecurity New Zealand has updated a marine pest identification guide which is available for marine farmers to use on the farm. Marine farmers must report any organism classified as 'Unwanted' or 'Notifiable' to the MPI pest and diseases hotline.

Import health standards are controlled by Biosecurity New Zealand and include requirements that must be met in the exporting country, during transit, and on arrival. For example, existing standards cover:

- import of juvenile yellowtail kingfish (*Seriola lalandi*) from Australia, and
- import of fish food and fish bait from all countries.

Possible prevention approaches that could be considered are summarised here as pathway management or on-farm management (Forrest et al. 2011).

Pathway management should focus on controls and surveillance on pathways from:

i. international source regions or pathways that are novel,
ii. pathways from domestic source regions known to be infected by recognised high-risk pests and pathogens, and
iii. pathways along which the frequency of transfers is considerably greater than that occurring as a result of other human activities.

Broadly there are two approaches to management of pathway risk (Forrest & Blakemore 2002), either a) avoid transfers on high-risk pathways, or b) treat pathways to minimise risk. Surveillance strategies for pathways can focus on entry surveillance, routine surveillance, or targeted surveillance of high-risk areas. Entry surveillance includes activities such as routine screening at airports, ports, and mail centres. MPI also commissions routine surveillance in ports and harbours around New Zealand. Targeted surveillance may be undertaken when activities such as harvest, grading, or transfer of stock from hatcheries or between sites is undertaken.

Good on-farm biosecurity management should include farm cleaning and disinfection, and surveillance (MPI 2013). Farm cleaning guidelines should deal with factors such as frequency of movement, source, and destination (i.e., locations) of movement of gear, stock, vessels, and waste and mortality disposal. Routine surveillance, undertaken on and around marine farms, is often the first point of detection of pests, pathogens, and diseases. Biosecurity measures adopted by farmers should be practical and fit for purpose, with biosecurity procedures for each farm based on their own site-specific conditions. Ideally each individual biosecurity procedure should be implemented with the understanding that it will work within a wider biosecurity plan (Georgiades et al. 2016).

New Zealand experience suggests that even when pest organisms become well established, the benefits gained from even limited management success have the potential to greatly outweigh the consequences of uncontrolled biofouling (Forrest et al. 2007a). To be effective, however, management requires buy-in from all marine aquaculture stakeholders whose activities can spread pest and pathogen organisms. Aquaculture companies can assist by:

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[3] The World Organisation for Animal Health's (OIE) online aquatic animal health code (http://www.oie.int/en/international-standard-setting/aquatic-code/access-online) suggests establishing zones and using compartmentalisation (through geographical separation) to manage biosecurity and epidemiological risks.

a. developing Biosecurity Management plans, protocols, and associated Standard Operating Procedures (SOPs) for all freshwater/seawater sites and operations,
b. identifying existing and future pests and pathogens that threaten the aquaculture industry,
c. implementing surveillance of farm structures and associated vessels and infrastructure, to check for biofouling,
d. implementing regular robust cleaning and disinfection protocols of farm vessels, gear, and equipment used by farm personnel and visitors,
e. regular screening of broodstock and adult stock health,
f. managing stocking densities,
g. fallowing of sites following harvest/end of a production cycle,
h. developing coordinated and site-specific response plans for high-risk pests and pathogens before they become established,
i. developing effective contingency plans to manage effects of adverse events on the integrity of farm structures and farm operations,
j. effective training of staff in biosecurity practices and identification of diseases and signs of stress and behavioural changes in stock,
k. regular audits of biosecurity plans and incorporation of changes into ongoing training for personnel,
l. reporting unusual or higher-than-normal mortalities to Biosecurity New Zealand's Pest and Disease Hotline (0800 80 99 66),
m. managing existing pests on aquaculture structures to control biofouling and limit its impacts on stock and structures (management needs to be undertaken in a biosecure way).

For vectors of spread such as service vessels and farm equipment, preventative management options include:

i. maintenance of effective antifouling coatings for service and harvesting vessels (including dive boats),
ii. hull inspections, and hull cleaning and disinfection as necessary,
iii. cleaning and disinfection of dive equipment, and
iv. early eradication of pests from farm structures through daily checks and removal of organisms before they become well established.

However, once incursions have occurred, the use of reactive eradication treatments is necessary. There has been extensive international effort to develop reactive treatments for marine AIS, but this effort has largely not translated into implementation-ready biosecurity treatment protocols (Cahill et al. 2021). Eradication techniques that have been used in an attempt to control biofouling and pests include:

- direct treatments such as acetic acid, lime and hot water. (Carver et al. 2003, Coutts & Forrest 2005, Locke et al. 2009a, Morrisey et al. 2009),
- indirect treatments such as changing culture systems or practices (Handley & Jeffs 2003, Handley 2002, Handley & Bergquist 1997), and
- biological control agents (NRC 2010, Hidu et al. 1981, Enright et al. 1983, 1993, Cigarria et al. 1998).

Perhaps the best method for controlling the spread of disease is through the use of management practices that call for the inspection and batch testing of animals to ensure that infected animals are not moved into areas that do not currently have endemic infections (WWF 2010). Disease testing in New Zealand is currently conducted on an ad hoc basis and it is recommended that a national disease testing and surveillance system that facilitates routine disease testing of stock be established (Georgiades et al. 2016). In New Zealand, in the absence of enforced stock transfer protocols, management of stock, gear, and vessel transfers between geographic zones by voluntary codes of practice developed by industry could be used to minimise risks, e.g., the New Zealand Mussel Industry Council Ltd. code of practice for transfer of mussel seed (NZMIC 2001).

The different prospective farmed groups: feed-added (referred to as finfish), filter-feeders (referred to as shellfish), and lower trophic level species (seaweeds, e.g., *Undaria*, and sea cucumbers) and their potential impacts and management measures were covered in the literature review (MPI 2013) and are summarised in Table 16.3.

Table 16.3: Matrix of biosecurity management options and their relevance to key aquaculture groups (updated from MPI 2013). [Continued on next page]

Management Measure	**Description**	**Finfish**	**Shellfish**	**Seaweeds**	**Sea Cucumbers**
Import					
Import Health Standards	Import of Seedstock	y	n	n	n
Border Surveillance	Prevent import of macroscopic pests	y	y	y	y
Regulations on fouling on vessels / bilge water release.	Prevent import of macroscopic pests/fouling organisms/harmful algae	y	y	y	y
Planning and Development					
Site Selection	Meet biological requirements of stock	y	y	y	y
Zoning	Pathogen exposure from other farms, processing plants, rivers, sewage	y	y	y	y
Vessel Berthing	Segregate vessels based on local/regional movement	y	y	y	y
Targeted Surveillance	Routine monitoring of predetermined range of species	y	y	y	y
Farm Practices					
Fouling					
Management of nets and equipment	Regularly remove fouling from equipment	y	y	n	?
Antifouling	Treat equipment to prevent fouling	y	?	n	?
Transfer of equipment between sites/regions	Prevent transfer of potentially contaminated equipment between sites.	y	y	y	y
Husbandry					
Appropriate stock husbandry	Minimise stress to reduce the risk of disease becoming established	y	y	y	y
Management of feed so as not to attract birds/fish	Limit the opportunity to transfer between sites/wild stocks	y	n	n	n
Routine monitoring linked to husbandry activity	Manage stock within environmental limits.	y	y	y	y

Table 16.3 [continued from previous page]: Matrix of biosecurity management options and their relevance to key aquaculture groups (updated from MPI 2013).

Remove mortalities	Limit potential for reservoir of disease	y	n	n	n
	Reduce attraction of predators.	y	n	n	n
Use of processed feeds	Heat-treat feeds to kill pests/pathogens	y	n	n	y
Surveillance	Observe and record causes of mortality, unusual fouling, etc.	y	y	y	y
Stock Transfer					
Hatchery testing for disease	Prevent diseases stock being sent to sites	y	y	y	y
Single year class sites	Prevent disease transmission between year classes	y	n	y	y
Harvest					
Isolate waste streams from growing areas	Prevent reintroduction of pest/pathogens to harvest sites	y	y	y	y
Fallow sites	Reduce opportunities for reintroduction of pets/pathogens from intermediate hosts.	y	y	y	y
Education					
Codes of practice	Educate and alert staff to biosecurity requirements	y	y	y	y
Public information	Alert public to biosecurity risks	y	y	y	y
Eradication					
Culling	Cull diseased stock to remove pathogen/pest	y	y	y	y
Fallowing	Remove stock from an area to allow host mediated pathogen to die out	y	y	y	y
Manual removal of macroscopic organisms	Eradicate individual pest organisms early in the invasion process.	y	y	y	y
Treatment technologies	Treat the whole farm or bay to remove pests.	y	y	y	y
Pharmaceutical treatment	Treat the individual affected stocks to remove pathogens/pests.	y	n	n	n

16.1.2 PELAGIC EFFECTS

There is a large volume of international literature on the effects of shellfish and salmon farming on the pelagic environment and much of this material is referenced in three local reviews: finfish (Forrest et al. 2007a), shellfish (Keeley et al. 2009), and oysters (Forrest et al. 2007b) and summarised by MPI (2013); the reader is referred to these for more detail. Guidelines for monitoring pelagic effects have recently been developed by Fisheries New Zealand (Giles et al. 2021).

16.1.2.1 INTRODUCTION

This section deals with near-field (approximately at the scale of the farm) pelagic effects (those seen in the water column). This should be read in conjunction with the benthic effects (where wastes from the pelagic zone settle) and the cumulative effects sections (where far-field pelagic effects are seen).

The pelagic zone is the zone where:

- Filter-feeders extract phytoplankton, micro-zooplankton, and organic particulates from the water column, which can reduce food available to other consumers (Zeldis et al. 2004).
- Dissolved oxygen (DO) is extracted by respiration of farmed organisms and this can potentially lead to DO depletion when cages are heavily stocked or where they are located in shallow sites with weak flushing (La Rosa et al. 2002). Excessive DO depletion in the water column could potentially stress or kill the fish and other animals. Sediment DO depletion may result in the release of toxic by-products (e.g., hydrogen sulphide) into the water, which can also have adverse effects on fish and other organisms (Forrest et al. 2007a).
- Fish pellets and the excretory products and waste products of cultured and fouling organisms are received.

Wastes excreted can either be as a particulate 'cloud' that disperses rapidly (as for finfish) or be bound in long strands composed of digested and undigested plankton (as for filter-feeders) (Reid 2007). The difference in shellfish and finfish faeces can result in different biochemical impacts on the pelagic zone (Reid 2007). Dissolved farm waste has the potential to increase ambient DIN (Dissolved Inorganic Nitrogen), and the potential effects of this are usually experienced away from the farm (and these are discussed in the cumulative effects section 16.3.10).

16.1.2.2 SIGNIFICANCE OF EFFECTS

The significance of these key primary impacts depends on the assimilation capacity (or carrying capacity) of the environment. Local hydrodynamics, water depth, and ambient oxygen levels are the most critical criteria for determining the pelagic impacts of aquaculture (Zeldis 2008a, Zeldis et al. 2010, 2011a). In shallow areas with slow currents, effects will be more pronounced compared with a deep site with strong flow and good flushing. In the New Zealand situation where most shellfish farms are located in well-flushed areas, nutrient enrichment beyond the farm boundaries is presently difficult to detect (Zeldis 2008a). In addition, there are siting design and management factors that will greatly influence potential impacts.

- Dispersive environments (such as open ocean) will dilute inputs and generally reduce the concentration of nutrient inputs.
- Density of farms in a unit volume of water; more farms will generally have more effect.
- Stocking density: higher stocking densities will generally have more effect, and this may differ seasonally.
- Feed conversion ratio (FCR for feed-added species): FCR is a measure of the efficiency of growth relative to feed used, and the global range is 1.1 to 1.7 on average (Reid 2007). The lower the FCR the less waste will be produced.
- Cage designs and orientation to prevailing current direction. This will impact on drag on passing water masses, flushing of cages, and settlement of biofouling organisms.

For salmon farms in the Marlborough Sounds, localised and direct water quality effects are elevated nitrogen and reduced dissolved oxygen (Elvines et al. 2019b). These effects are largely periodic and are associated with periodic feed/excretion/respiration patterns (Tomasso 1994). Notably the immediate impact signal of a salmon farm decays rapidly with distance, primarily due to mixing with more distant waters, but also due to uptake by primary producers (Elvines et al. 2019b). Indeed, the signal is rarely detectable at distances greater than a few hundred metres from the pens (e.g., Knight & Beamsley 2012, Bennett et al. 2018).

Seaweed aquaculture is receiving increasing interest in New Zealand. Seaweeds tend to extract nutrients and CO_2 from seawater and are generally considered beneficial for the pelagic environment (Stenton-Dozey et al. 2020).

Sea cucumbers are benthic deposit feeders and are normally cultured at low densities. It is expected that they would have little in the way of ecological effects on the pelagic environment (Zamora et al. 2018); see MPI (2013) for further discussion.

16.1.2.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Pelagic effects can be partially controlled through careful site selection. More dispersive sites provide multiple advantages in terms of environmental effects on the water column. Open ocean sites tend to have stronger currents and provide good conditions for turbulent mixing, thereby increasing the rate of dilution and dispersal of farm-derived wastes (Welch et al. 2019)

Farm design, orientation, and stocking rates should be appropriate to the site. Good farm management (e.g., compliance with The New Zealand Finfish Aquaculture Environmental Code of Practice (NZFSA 2007)) should include reducing biofouling on nets by regular cleaning and removal of biofouling waste. Monitoring, adaptive management, and the use of Integrated Multi Trophic Aquaculture (IMTA) are also potential mitigation measures (see section 16.3.10 for further discussion). Notably pelagic effects are reversible after removal of the farm.

Models are an important component in determining pelagic effects at a site. Several regional councils have commissioned biophysical modelling of the coastal environment to inform aquaculture planning and consent processes (e.g., Hadfield et al. 2014, Knight et al. 2014, and Broekhuizen et al. 2015). Modelling is an iterative process, and it is envisioned that hydrodynamic and aquaculture effects models will continue to be improved, such that over time more complex processes (e.g., cumulative effects) will be encompassed (Knight et al. 2014).

Best management practice for salmon farms in the Marlborough Sounds have recently been developed for water quality standards and monitoring protocols (Elvines et al. 2019b). The focus of the best management practice is on the broader scale nutrient enrichment effects within a region, rather than on localised effects around the farms themselves (see section 16.3.10).

16.1.3 MARINE MAMMALS

The reader is referred to Clement et al. (2021), Würsig (2021), and MPI (2013) for more detail.

16.1.3.1 INTRODUCTION

Fisheries New Zealand have recently published guidelines on best practices and technologies available to minimise and mitigate the interactions between finfish open ocean aquaculture and marine mammals (Clement et al. 2021). These guidelines draw on several overseas studies (Würsig & Gailey 2002, Kemper et al. 2003, Wright 2008) and a recent New Zealand focused review (Würsig 2021) that have characterised the possible interactions between marine mammals and aquaculture, which include the following.

- Attraction or avoidance, because marine farm structures represent a new physical, visual, or acoustic obstruction that marine mammals may choose to ignore, investigate, or avoid. Attraction might alter natural foraging and passage patterns and potentially lead to interactions, and also to incidents.

- Interactions with farm structures, defined as events when a marine mammal makes physical contact with the farm structure, which may lead to
- incidents that cause injury (e.g., rope cut, abrasion), entrapment, or entanglement (live or fatal) of a marine mammal.

The guidelines focus on mitigation of interactions through site selection, design, and operation of farm infrastructure. Because open ocean aquaculture is at an early development stage in New Zealand, carefully designed monitoring programmes will be required to assess the effectiveness of mitigation measures.

The physical location of the farm within important habitats or migration routes of New Zealand marine mammal species is the main factor that leads to potentially adverse interactions or avoidance issues. Once a farm is within the habitat or migration route of a species, the types of gear and equipment employed, as well as operational procedures around regular farm activities, influence the probability and scale of the impacts given above.

Mitigation options include:

- discourage activities that could attract marine mammals to the farm,
- minimise artificial lighting, where and when possible, to reduce attraction of prey fish and predators,
- avoid un-tensioned and/or loose ropes, lines, or nets on farms at all times,
- consider the use of predator-resistant materials in the construction of farms, and
- minimise predator exclusion nets, or design nets to minimise the likelihood of entrapment.

16.1.3.2 SIGNIFICANCE OF EFFECTS

Incidences of marine mammal entanglement with aquaculture operations are reasonably infrequent in New Zealand. This is due in part to the relatively small scale of this industry and operational procedures that minimise entanglement risk (Forrest et al. 2007c). Studies in New Zealand have so far only addressed interactions between mussel farms with Hector's (Slooten et al. 2001) and dusky dolphins (Markowitz et al. 2004, Vaughn & Würsig 2006, Duprey 2007, Pearson et al. 2007). Collectively these works suggest that, although some marine mammal species are not completely displaced from regions as a whole, they do not appear to be utilising habitats occupied by shellfish farms in the same manner as before the farms were established. Interactions between dolphins, fur seals, and salmon farms also occur. New Zealand King Salmon report all marine mammal interactions on their website.

The interactions between marine mammals and aquaculture have been considered in relation to any larger scale and offshore developments in New Zealand waters (Würsig 2021). The development of multiple farms or several types of overlapping aquaculture may lead to exclusion of some species from particular bays or regions depending on the species and its sensitivity to such activities. For depleted populations (e.g., southern right whales or Hector's and Māui dolphins), the issues of low population size and a fairly isolated population structure make these species more vulnerable to such impacts than other species.

16.1.3.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Farm locations need to be carefully selected to minimise the likelihood of overlap with marine mammal migration routes and/or known habitats. In Admiralty Bay, where overlap with dusky dolphins was a concern, and distribution patterns were not well known, three years of presence monitoring was required before commencement of aquaculture development (Mulcahy & Peart 2012).

The risks associated with physical interactions can be further minimised by adopting operational and maintenance guidelines and standards for farm structures as well as any noise-generating equipment (British Columbia Shellfish Growers Association 2001, SAD 2011). Some examples include enclosing predator nets at the bottom, keeping nets taut, using mesh sizes of less than 6 cm (Kemper et al. 2003), keeping nets well maintained (e.g., repairing holes), and reducing feed waste.

In Admiralty Bay, surface lines were removed from the water over winter to minimise interactions when dolphins are more active foragers (Mulcahy & Peart 2012).

Unfortunately, detailed information on abundance, distribution, and critical habitats is available for only a handful of New Zealand's marine mammal populations. Monitoring records of the presence (and absence) of marine mammal species in the vicinity or general region of a farm site (see for example Clement & Elvines 2019), and any detailed observations of their time spent under or around the farm structure should be compiled where possible. Future research needs to focus also on those species most likely to come in contact with aquaculture. In addition, ongoing research into the types of design, maintenance features, and operational procedures that minimise entanglement risk should be supported. For example, cage technology in South Australia has developed and improved to the point where predators are excluded by the cage structures rather than predator nets (Taylor et al. 2010).

16.1.4 BENTHIC EFFECTS

This area is covered for inshore aquaculture by Forrest et al. (2007c) and summarised by MPI (2013); and for offshore aquaculture by Bennett et al. 2020 and Keeley 2020. The reader is referred to these documents for more detail. Guidelines for monitoring pelagic effects have recently been developed by Fisheries New Zealand (Giles et al. 2021).

16.1.4.1 INTRODUCTION

The benthic effects of aquaculture can be classified as:

- Organic enrichment and smothering, which can lead to (Forrest et al. 2007c):
    - localised biodeposition leading to enrichment of the seabed and associated microbial processes, and chemical and biological changes (including to infauna and epifauna, e.g., Christensen et al. 2003, Keeley et al. 2009),
    - widespread biodeposition, in intensive filter-feeder cultivation, can potentially lead to a reduction in natural deposition rates,
    - widespread biodeposition leading to mild enrichment in naturally depositional areas which has the potential for effects on reefs, inshore habitats, and sensitive taxa,
    - smothering of benthic organisms and changes in sediment physical composition, and
    - sediment contamination (copper and zinc, covered in the additives section 16.3.6).
- Biofouling and drop-off of debris, which can lead to:
    - smothering and changes to physical composition of sediments (Keeley et al. 2009),
    - creation of habitat structure (Davidson & Brown 1999), and
    - aggregations of predators and scavengers (Inglis & Gust 2003).
- Seabed shading by structures, which can change localised productivity under the farm (Huxham et al. 2006).

The magnitude and spatial extent of seabed effects from finfish farms are a function of a number of inter-related factors, which can be broadly considered as farm attributes and physical environment attributes.

Farm attributes that can affect the mass load of organic material deposited to the seabed include the following:

- fish stocking density and settling velocities of fish faeces (Magill et al. 2006),
- the type of feed and feeding systems, the feeding efficiency of the fish stock, and the settling velocities of waste feed pellets, and
- the type of cage structure can also influence depositional effects through differences in fish holding capacity, which affects feed loadings and may affect feeding efficiencies. Furthermore, cage design and position may affect the site’s hydrodynamics; any reductions in flow will reduce waste dispersal and flushing, potentially resulting in depositional effects that are more localised but also more pronounced.

The capacity of the environment to disperse and assimilate farm wastes is a function of the attributes of the site (primarily water depth, sediment type, and current speed), although assimilative capacity may also vary seasonally in relation to factors such as water temperature. Consequently, sites located in deep water (over 30 m) and exposed to strong water currents (over 15 cm s^{-1} on average) will have more widely dispersed depositional footprints with less intense enrichment than shallow, less well-flushed sites (e.g., Molina Dominguez et al. 2001, Pearson & Black 2001, Aguado-Gimenez & Garcia-Garcia 2004).

16.1.4.2 SIGNIFICANCE OF EFFECTS

The benthic effects from feed-added and filter-feeder aquaculture result from farm derived organic waste settling on the seafloor. The higher volume of waste and uneaten food involved in feed-added farming and its more particulate nature generally means that effects from feed-added aquaculture are greater than those seen from filter-feeder aquaculture, and these can be seen further away (within 1 km for feed-added species as opposed to within 100 m for filter-feeders; Forrest et al. 2007c). In extreme cases the settlement of organic waste can overwhelm the assimilative capacity of the seabed fauna and can lead to anoxia and outgassing of hydrogen sulphide and methane. At low-flow sites very little resuspension occurs and effects are largely constrained to the local environment (Forrest et al. 2007c). At high-flow sites, however, biodeposits can be resuspended, exported, and eventually deposited in a very diffuse form in neighbouring low-flow areas (e.g., in enclosed bays or inlets). If depositional inputs are sufficiently elevated, then there is potential for effects in the form of increased far-field deposition. This may result in very mild, but potentially spatially extensive, organic enrichment.

The ecological effects of farming seaweeds are likely to be significantly less than for farming species that require artificial feed and excrete solid waste. Seaweeds are largely extractive and benthic impacts are likely to be restricted to drop-off of fouling organisms and shading. Overseas, seaweed farming has been recorded as having positive effects on benthic fauna (Visch et al. 2020).

The benthic effects of sea cucumber culture will depend on the culture technology used. Sea cucumbers caged or ranched below other aquaculture species may be expected to consume deposited organic matter from the seabed. Apart from the organic matter reduction, sea cucumbers reduce the accumulation of phytopigments, enhance bacterial activity, the mineralisation processes, and also increase nutrient dissolution into interstitial water and the water column (Stenton-Dozey et al. 2020, Keeley et al. 2009).

Fish farm and mussel farm studies in New Zealand and overseas indicate timescales for recovery of benthic habitats once farms are removed. These recovery times ranged from a few months in well-flushed areas where effects are minor, to a few years in poorly flushed areas where moderate or strong enrichment has occurred (see MPI 2013).

There is growing international recognition that, rather than be viewed solely as causing adverse environmental impacts, shellfish aquaculture may actually contribute toward the resilience of coastal ecosystems (Lindahl et al. 2005, Coen et al. 2011, Arreguín-Sánchez 2013, Saurel et al. 2014, Rose et al. 2015, Kluger et al. 2016, 2017, Bricker et al. 2018). A review of the potential ecological and ecosystem services provided by mussel farms in the Marlborough Sounds has recently been completed (Stenton-Dozey & Broekhuizen 2019). Species diversity found on mussel droppers was akin to that associated with present day benthic algae meadows, rhodolith beds, bryozoan thicket, and calcareous tube worms, whereas diversity found on mussel-culture derived reefs was akin to soft bottom non-calcareous tubeworm mounds that attract a predominance of scavengers, detritivores, and predators. The review suggested that mussel farms may to some degree compensate for the loss of biodiversity supported by biogenic reefs by providing habitats that increase the abundance of organisms that once that would have been plentiful in the Marlborough Sounds (Stenton-Dozey & Broekhuizen 2019).

16.1.4.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management measures for mitigating benthic impacts for aquaculture are similar to those for mitigating pelagic impacts (section 16.1.2.3). Site selection is important for the same reasons, to maximise the dispersive properties of the site, but should also try to avoid potentially sensitive or valuable benthic habitats (conservation areas, reefs, etc.) (see MacDiarmid et al. 2013 for examples of sensitive habitats). Depositional modelling should be used to predict benthic effects from a range of farming scenarios to inform decisions regarding optimum (sustainable) site-specific feed capacities.

Modelling techniques are continuing to evolve and increase in their ability to accurately predict the depositional footprint of finfish aquaculture (Elvines et al. 2021). Further work is required to accurately assess near-bottom flows in complex habitats that impact on particle resuspension (Smeaton & Vennell 2021).

In New Zealand we do not yet have any experience with monitoring benthic effects of finfish or mussel farming in open ocean settings, and there are relatively few examples to draw on from overseas. The characteristics of open ocean environments create new logistical challenges for monitoring. For example, farming in these new open ocean environments will likely encounter new habitats and species for which little is known about their tolerance for organic waste (Keeley 2020). The logistics of accurately monitoring benthic impacts in open ocean environments may be complicated by working at depth and in strong tidal currents.

The application of Environmental Quality Standards (EQS) that encompass a range of effects measures can be useful for assessing impacts from aquaculture. For example, best management practice guidelines (BMP) have been developed for monitoring benthic impacts from salmon farms in the Marlborough Sounds (Keeley et al. 2015a, Keeley et al. 2019). These guidelines provide consistent and clear requirements for the management and monitoring of existing farms, based around an agreed set of environmental quality standards with accompanying rationale. Details are provided about how and when to conduct the surveys, along with consequences in the event of non-compliance. New standards will be required for open ocean aquaculture.

Staged development and Modelling-Ongrowing-Monitoring (MOM) approaches are also potentially beneficial for establishing and managing environmental effects (MPI 2013).

16.1.5 SEABIRD INTERACTIONS

The reader is referred to MPI (2013) and Gaskin et al. (2021) (and references therein) for more detail.

16.1.5.1 INTRODUCTION

In New Zealand, the negative effects (to seabirds) of both feed-added aquaculture and filter-feeder aquaculture have centred on entanglement (resulting in birds drowning) and habitat exclusion and displacement from feeding grounds. The location of the farm within the range of seabirds and the conservation status (which is a measure of the risk of extinction) of these seabird species are the main factors that may lead to issues of sustainability and conservation concern. Of particular concern are the location of farms in relation to breeding and feeding sites and the operational procedures of regular farm activities (which, for example, can affect the likelihood of entanglement).

Fisheries New Zealand have recently published guidelines on best practices and technologies available to minimise and mitigate the interactions between finfish open ocean aquaculture and seabirds (Gaskin et al. 2021).

Within these guidelines, the main effects that need to be addressed include the following.

- Attraction or avoidance, because marine farm structures represent a new physical, visual, or acoustic obstruction that seabirds may choose to ignore, investigate, or avoid. Attraction might alter natural foraging and passage patterns and potentially lead to interactions, and also to incidents.

- Interactions with farm structures, defined as events when a seabird makes physical contact with the farm structure, which may lead to

- Incidents that cause injury (e.g., rope cut, abrasion), collision, entrapment, or entanglement (live or fatal) of a seabird.

Other potential negative effects may include disturbance of breeding colonies and bird feeding, blockage of the digestive tract following ingestion of foreign objects, and the spread of pathogens or pest species.

In contrast, a potential beneficial effect includes the provision of roost sites closer to foraging areas (Lalas 2001), saving energy and enabling more efficient foraging; this is most likely to benefit shags, gulls, and terns (MPI 2013). The attraction and aggregation of small fish around marine farm structures (Grange 2002) may provide enhanced feeding opportunities for piscivorous seabirds but also may increase attraction to the farm with the associated risk of negative interactions.

16.1.5.2 SIGNIFICANCE OF EFFECTS

Siting of a farm close to a seabird breeding colony is very likely to have an immediate adverse effect that will continue as long as the duration of the farm. There are few reports of negative seabird interactions with shellfish farms (Butler 2003, Lloyd 2003); however, there are reports of seabird deaths as a result of entanglement in salmon farm nets. New Zealand King Salmon publish details of all seabird incidents (https://www.kingsalmon.co.nz/our-environment/). The potential effects of habitat exclusion by feed-added farms in New Zealand are considered to be insignificant given the small area occupied in relation to the large total area of suitable habitat available for foraging seabirds (MPI 2013).

16.1.5.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

At present, potential risks are identified on a farm-by-farm basis. The most obvious is the choice of site for a farm to avoid disturbance to sensitive breeding colonies of seabirds. Good operating practices (for feed-added farms) such as enclosing predator nets above and below cages, controlling litter, minimising the use of lights at night, keeping nets taut, and using mesh sizes less than 6 cm, all minimise the chances of negative seabird interactions. Given the current relatively small size of the aquaculture industry in New Zealand, the overlap of farming activities with the feeding areas of seabirds is unlikely to present significant issues. However, if larger farms are developed at open ocean sites the potential range and frequency of interactions will need to be reassessed (Gaskin et al. 2021, MPI 2013).

There are significant knowledge gaps concerning almost all seabird species in New Zealand. Detailed information on the time-specific distribution, abundance, and critical habitats is lacking. Also missing is information on key prey species of seabirds, particularly those that may be affected by aquaculture. In addition, there should be ongoing monitoring (where an issue is identified) and research into the operation, design, and maintenance of farm structures that minimise disturbance and entanglement risks. Little is known about the exclusion distance needed for different species of foraging and feeding seabirds; for example, proposed exclusion distances for king shags in the Marlborough Sounds range from 100 m to 1000 m (Davidson et al. 1995, Taylor 2000), but Lalas (2001) noted that king shags resting ashore or on emergent objects only flew off when approached to within 30 m.

Resource consent conditions may require that management plans be developed for specific species. Where knowledge gaps exist, these management plans may provide provisions which address these gaps. For example, the King Shag Management Plan, established by New Zealand King Salmon in accordance with their consent conditions, requires that aerial surveys be conducted to monitor king shag population abundance. New Zealand King Salmon must determine and take subsequent action if any population decline is a result of farm activity. King shags, classified as 'Nationally Endangered' by DOC (Robertson et al. 2021), have a small population size and restricted range and concerns have been raised regarding the effect of aquaculture activities on populations in the Marlborough Sounds (Bell 2019b). Three surveys, conducted in 2015, 2018, and 2019, have revealed fluctuations in king shag population numbers with a reduction in birds between 2015 and 2018 and a subsequent increase by 2019 (still 6% below that seen in 2015) (Bell et al. 2019).

It is difficult to interpret these fluctuations in numbers as several factors, such as annual variations in breeding success or roost sites being missed during surveys, may influence the results. Nests were counted in a boat survey in 2019 which also provided opportunity to count chicks and observe behaviours (Bell 2019a). Further research using GPS devices into the foraging behaviour of king shags is currently underway (Bell 2020, Bell 2019b); this may provide further insight into the potential effects of aquaculture activities on this species. Of eleven king shags tagged, four foraged close to mussel farms and all eleven roosted on mussel farms at times, and one of the birds was found to forage almost exclusively within mussel farms (Bell 2020).

16.1.6 EFFECTS FROM ADDITIVES

Assessments of the use and impact of chemicals used in aquaculture in New Zealand are focused on salmon farming (Forrest et al. 2007c, 2011, Wilson et al. 2009, Burridge et al. 2010, Clement et al. 2010, MPI 2013); the reader is referred to these documents for more detail.

16.1.6.1 INTRODUCTION

Additives such as antibiotics, antibacterials, and other therapeutants may be introduced intentionally at aquatic farms (MPI 2013). These therapeutants could potentially affect non-target organisms (phytoplankton, zooplankton, and sediment bacteria) and lead to an increase in resistant bacteria and/or parasites (GESAMP 1997, Forrest et al. 2007c, 2011).

Metals from fish feed (zinc) and antifouling agents used on structures (copper) may be unintentionally introduced to the marine environment at aquaculture sites (MPI 2013). The main concern with these metals is their toxicity to animals (Forrest et al. 2007c, 2010, Clement et al. 2010).

16.1.6.2 SIGNIFICANCE OF EFFECTS

Currently, no therapeutants are known to be used in the farming of bivalves and lower trophic level species and no antibiotics or pesticides have been used on salmon farms in New Zealand (Seafood Watch 2020). However, culture of native species and/or the emergence of disease may lead to the need to use therapeutants in the future.

Assessments at salmon farming sites in the Marlborough Sounds revealed locally elevated copper and zinc levels (with maxima exceeding ANZECC (2000) sediment quality guideline values; Hopkins et al. 2006). Copper antifouling paint has not been used at salmon farms in New Zealand since 2015 (Elvines et al. 2019a). Potential adverse effects from high zinc exposures range from interference with growth at low concentrations to behavioural abnormalities at high concentrations (Eisler 1993, Burridge et al. 2010); but elevated metal concentrations do not necessarily indicate adverse ecological effects because they may not be bioavailable (Sneddon & Tremblay 2011, Forrest et al. 2007c).

16.1.6.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

The use of therapeutants is closely controlled for all species grown for human consumption. Food safety standards regulate the acceptable concentrations of metals, chemicals, and additives in food products. New Zealand salmon farmers must also comply with the New Zealand Salmon Farmers Association's Finfish Aquaculture Environmental Code of Practice, with harvesting and processing in accordance with New Zealand food safety standards.

'Best management practice' should minimise the potential future use of therapeutants, and hence help mitigate potential effects. The most important means to reduce and manage the overall antibiotic usage would be to support development of targeted disease management strategies (biosecurity protocols) and alternative therapies, in particular vaccines.

The potential for environmental issues from therapeutant use in the future will need to be assessed on a farm-by-farm basis. The persistence of therapeutants in the environment, the induction of resistance of targeted organisms, and the effects on non-target organisms are the main knowledge gaps.

Although the use of copper antifouling has largely been phased out of New Zealand salmon farms, zinc remains as a feed additive. Careful management of feed wastage is already prioritised due to the cost of feed and need to manage environmental impacts. Studies on the bioavailability and forms of the metals will give better understanding of their toxicity; a focus is needed on sub-lethal effects on individual species and the broader effects on benthic communities.

16.1.7 ESCAPEE EFFECTS

The subject of escapee effects from aquaculture is well covered for finfish in the reviews by Forrest et al. (2007c) for New Zealand and Jensen et al. (2010) for Norway, and for shellfish by Keeley et al. (2009) and summarised by MPI (2013). The reader is referred to these sources for more detail.

16.1.7.1 INTRODUCTION

It is useful to recognise that the human-mediated transfer of numerous marine organisms to New Zealand and around the coastline is an issue with a long history that continues today. Historically, this reflects deliberate transplants of marine organisms (including salmon), and more recently the inadvertent transfer of a range of native and non-indigenous marine species (including fish), especially via vessel movements (e.g., Hayward 1997, Cranfield et al. 1998). The alteration to marine ecosystems and transfer of fish diseases via these unmanaged mechanisms is well recognised (Ruiz et al. 2000, Hilliard 2004), and hence any incremental risk from finfish culture should be considered within this broader context.

The effects of escapees from aquaculture vary considerably in relation to the following factors (Forrest et al. 2007c):

- the numbers involved in the escape episode,
- the location of the farm in relation to wild populations and its size, distribution, and health,
- whether the species is native (e.g., hāpuku, kingfish) or introduced (e.g., salmon),
- whether the brood stock is hatchery bred or wild sourced,
- the fish harvest size in relation to reproductive maturity and the ability of gametes to survive and develop in the wild, and
- the ability of escapees to survive and reproduce in the wild, as determined by their ability to feed successfully and interbreed with wild stocks.

The main effects of escapees (Forrest et al. 2007c) for feed-added species are:

- competition for resources with wild fish and related ecosystem effects from escapee fish (e.g., through predation),
- alteration of the genetic structure of wild fish populations by escapee fish and potential loss of genetic integrity in the wild populations, and
- transmission of pathogens from farmed stocks to wild fish populations.

The main factors controlling the number of fish escaping, and their subsequent effects, are the integrity of the nets used to contain the fish and the amount of difference between the wild fish and farmed fish in terms of their genetics and their pests and diseases.

16.1.7.2 SIGNIFICANCE OF EFFECTS

The likelihood of escapee effects in New Zealand is low, based on the current small size of the industry, limited overlap of wild and farmed populations (i.e., salmon; Deans et al. 2004) and the broad home range (i.e., kingfish and hāpuku), and likelihood of high genetic diversity in these native species (Paul 2002, Forrest et al. 2007c). If escapee effects are seen on wild populations they are, however, likely to be irreversible and could potentially be at a national scale.

16.1.7.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management strategies to minimise escapees are usually based upon maintaining net integrity. In Norway reporting of escapes, and estimation of numbers escaped, is mandatory and therefore provides a baseline to improve upon (Jensen et al. 2010). In New Zealand escapee events are not reported to any central authority. At this time no knowledge is available on the potential effect that escaped farmed kingfish or hāpuku could have upon the wild populations. The potential for genetic interactions between hatchery/selectively bred farmed shellfish and wild shellfish populations has not yet been investigated in New Zealand.

16.1.8 EFFECTS ON WILD FISH

The reader is referred to MPI (2013) and Taylor and Dempster (2019) (and references therein) for more detail.

16.1.8.1 INTRODUCTION

The installation of marine farming structures introduces new three-dimensional structures to the marine environment. These structures provide habitat for colonisation by fouling organisms and associated biota (Glasby 1999, Connell 2000, DeAlteris et al. 2004). These newly colonised structures and the habitat they create tend to attract wild fish species seeking foraging habitat, detrital food sources, and/or refuge from predators (e.g., DeAlteris et al. 2004). Finfish farms also provide additional feed sources to wild fish through waste feed.

The presence of farms can lead to degradation or loss of habitat beneath or within close proximity to new farm structures that may have consequences in terms of spatial overlap with critical spawning grounds and/or migration routes. Submerged artificial lighting at night is frequently used on finfish farms to control maturation and increase productivity (e.g., Porter et al. 1999). The lighting can enhance the attraction of wild fish to farm structures (Cornelisen & Quarterman 2010).

The main effects associated with the creation of artificial habitats, and attraction of wild fish species to aquaculture structures, include:

- enhanced predation on wild fish by higher trophic level predators (e.g., seals and seabirds) and predation by cultured fish on wild fish trapped within cage structures,
- consumption of waste feed by wild fish (Felsing et al. 2004, Dempster et al. 2005),
- changes in recreational fishing patterns and pressure (Taylor and Dempster 2019), which could increase fishing pressure on species attracted to structures, and
- larval fish depletion by filter-feeders (as observed by Davenport et al. 2000 and Lehane & Davenport 2002) and/or potential trophic interactions (e.g., alteration of plankton composition and food availability).

16.1.8.2 SIGNIFICANCE OF EFFECTS

In general, the effects of aquaculture on wild fish populations are likely to be small in comparison with the effects on other aspects of the marine ecosystem, such as effects on the seabed. The effects of farming hāpuku or kingfish on wild fish are expected to be generally similar to those from farming king salmon already in New Zealand. Modelling of larval egg depletion (Broekhuizen et al. 2002) and other work suggest that although shellfish aquaculture could have an impact on recruitment to fisheries, the scale of this effect will largely be governed by the extent of the culture, the behaviour and characteristics of larvae, and the flow dynamics of the regions in question (MPI 2013).

The effects of farming filter-feeders are likely to be less than those of farming feed-added species (due to the lack of waste food acting as an attractant). The extent of impacts from the farming of seaweeds and sea cucumbers is likely to have a lesser impact than feed-added or filter-feeding aquaculture. Although seaweed cultivation may offer an attractive habitat for fish aggregation (Radulovich et al. 2015) it does not provide additional food or significantly modify the benthic habitat (MPI 2013).

16.1.8.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Management options identified by MPI (2013) to minimise effects on wild fish include proper site selection, which requires assessment of potential impacts of farm developments on wild fish stocks. Assessments should identify proximity and impact to critical, sensitive, or protected habitats and species, with particular reference to potential impacts on spawning grounds or juvenile habitats. Careful management of feed quality and feeding practices should minimise waste feed inputs to the surrounding environment and minimise effects on wild fish populations. The effects of finfish farms on wild fish populations in New Zealand are not well documented and knowledge gaps exist, particularly with regard to the effects of finfish farms on fish movements and various reproductive stages (e.g., larval settlement).

16.1.9 HYDRODYNAMIC EFFECTS

The reader is referred to MPI (2013), and references therein, for more detail.

16.1.9.1 INTRODUCTION

Hydrodynamic conditions are an important determinant of the suitability of a site for aquaculture, as well as the spatial size and magnitude of the environmental effects. Here, hydrodynamics refers to the physical attributes of the water including:

- currents,
- stratification, and
- waves.

Current speed is a key factor determining the exchange of water through the cage, areas over which deposition occurs, where the dissolved material is transported and how it is dispersed, and the resuspension of material. Stratification refers to the layering of water caused by differences in temperature and salinity. Stratification can play a strong role in oxygen depletion by restricting vertical transport of oxygen from the surface to deeper waters. Waves can break up stratification, play a key role in determining which species can inhabit an area, and resuspend material.

16.1.9.2 SIGNIFICANCE OF EFFECTS

Aquaculture operations can have a number of effects on hydrodynamics. The drag from cages can affect currents, causing wakes, turbulence, and flow diversion (Helsley & Kim 2005, Venayagamoorthy et al. 2011). Low-velocity areas have a higher probability of issues of deposition, oxygen depletion, and ammonium build-up. There are likely to be interactions between stratification and fish cages in the form of selective blocking, restricted underflow, generation of internal waves, and vertical mixing (Plew et al. 2006). Fish swimming may also play a role in enhancing mixing and causing upwelling within cages (Chacon-Torres et al. 1988). Wave energy is attenuated by marine farm structures, and this will result in a shadow of reduced wave activity behind the farmed areas (Chan & Lee 2001, Lader et al. 2007).

Some physical effects may affect other physical processes directly, for example attenuation of wave energy affecting surf or coastal sediment transport; it is generally more important to consider how physical effects influence ecological processes. For example, the physical effect of reduced current speeds caused by drag from aquaculture structures (Helsley & Kim 2005, Venayagamoorthy et al. 2011) may result in an increase in the flushing time of a bay (Plew 2011). This in turn may lead to increased nutrient concentrations. Reductions in wave energy near the coast may change the mix of species inhabiting an area.

16.1.9.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

The physical hydrodynamic effects will interact strongly with pelagic and benthic processes. Selection of suitable indicators for physical changes should ideally be based on their relative importance in determining the habitat for ecological communities in an area. However, it is this link between the physical and ecological changes that is often the least understood area of hydrodynamic impacts.

16.1.10 CUMULATIVE IMPACTS

The following section draws heavily on previous reviews of the environmental effects of finfish (Forrest et al. 2007c) and non-finfish aquaculture (Keeley et al. 2009). Complementary information on the local scale ecosystem effects of aquaculture in relation to the water column is provided in section 16.1.2: Pelagic effects. The reader is referred to MPI (2013), and references therein, for more detail.

16.1.10.1 INTRODUCTION

The previous sections (16.1.1–16.1.9) have focused on issue-specific ecological effects of aquaculture developments on the marine environment. Our understanding of these effects is largely based on farm-scale assessments and monitoring; the potential for wider-ecosystem effects (e.g., far-field benthic enrichment, effects on fish populations, migrating mammals, etc.) is acknowledged but is far less well understood. As aquaculture develops and the number of farms in coastal waters increases, wider-ecosystem issues become more important to consider due to the cumulative environmental effects that could arise from multiple farms combined with additional anthropogenic stressors affecting, and possibly interacting with, natural marine processes (see Figure 16.4 for an example of multiple stressors interacting with natural processes).

Within the context of aquaculture development in the marine environment, cumulative effects are defined here as:

> *Ecological effects in the marine environment that result from the incremental, accumulating, and interacting effects of an aquaculture development when added to other stressors from anthropogenic activities affecting the marine environment (past, present, and future activities) and foreseeable changes in ocean conditions (i.e., in response to climate change).*

A number of examples of potential cumulative impacts of aquaculture exist, three of these will be given here to illustrate the definition above:

- Drop-off of mussels, shells, and biofouling organisms to the seabed beneath mussel farms can lead to the creation of reef-like habitat and alter the composition and abundance of benthic organisms beneath farms (see section 16.1.4). Where this occurs in high densities such as the ribbon-like developments in the Marlborough Sounds, this could lead to additive (cumulative) effects on the wider ecosystem due to alteration of a larger proportion of the benthos.
- The presence of farm structures, where aquaculture involves numerous farms situated along the coast, could also have cumulative effects on near-shore currents and waves, which in turn could affect important processes (e.g., larval transport, nutrient exchange) along the shoreline (see section 16.1.9).
- As aquaculture development intensifies, there is likely to be an increase in man-made structures and boat traffic, increasing the risk of invasion and establishment of pests. Cumulative degradation of the marine environment from multiple stressors compromises habitat quality and could enhance biosecurity risks by increasing productivity and proliferation of pest species such as invasive macroalage (e.g., *Undaria*) and invertebrates (e.g., the bivalve *Theora lubrica* and tunicate *Styela clava*) that thrive on the benthos under conditions of high organic enrichment (section 16.1.1 provides comprehensive information on methods to minimise biosecurity risk that are applicable to wider, regional scales).

Because of limited resources and uncertainty in understanding all the potentially complex interactions between aquaculture, other stressors, and the environment, it is necessary to focus on those aspects of aquaculture most likely to contribute to cumulative environmental change. Hence, increasing emphasis has been placed on assessing the contribution of aquaculture to cumulative changes in nutrient conditions and primary production and, in turn, the knock-on effects on the wider ecosystem (see Hargrave et al. 2005, Volkman et al. 2009, and chapters therein). All forms of aquaculture addressed in this report contribute to these nutrient effects, whether through nutrient emissions to the water column and seabed, or the net extraction of plankton (filter-feeding bivalves) and nutrients (nutrient uptake by macroalgae) from the water column. The following sections focus on the potential far-field nutrient implications of aquaculture.

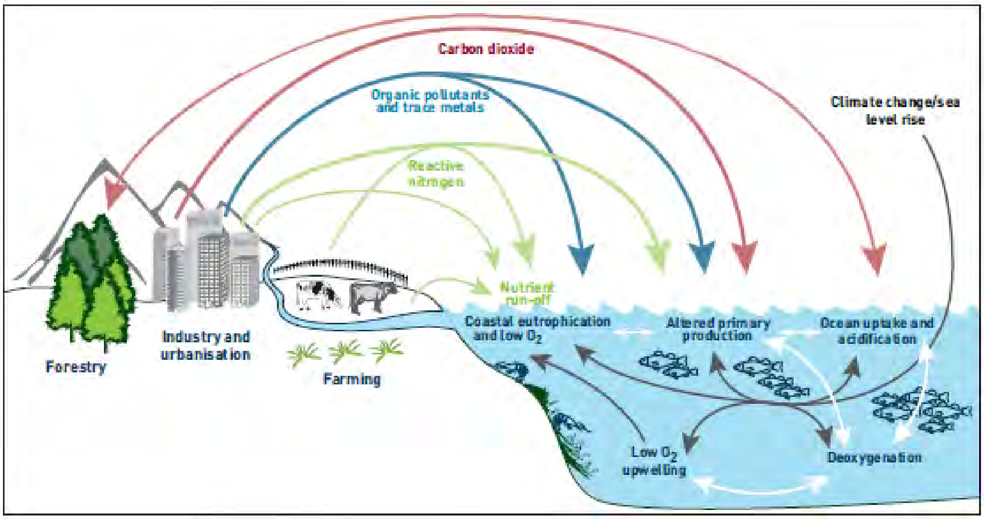

Figure 16.4: Conceptual diagram of anthropogenic influence in marine ecosystems.

16.1.10.2 SIGNIFICANCE OF EFFECTS

A particular concern with the potential expansion of fish farms is the potential risk of eutrophication (SEPA 2000, Hargrave et al. 2005, Díaz et al. 2012). Eutrophication is the process where excessive nutrient inputs to a water body result in accelerated primary production (phytoplankton and macroalgae growth) and flow-on effects to the wider environment such as reduced water clarity, physical smothering of biota, or extreme reductions in dissolved oxygen because of microbial decay (Degobbis 1989, Cloern 2001, Paerl 2006). On a global scale, runoff from land-based agriculture has been identified as the primary driver of intense eutrophication of coastal environments; however, feed-added forms of aquaculture have been singled out as an important emerging contributor to nutrient enrichment (Díaz et al. 2012).

Nutrients of varying particulate and dissolved organic and inorganic forms are added to the environment as a result of feed-added aquaculture. Particulate organic nitrogen (PON) and phosphorus (POP) are primarily deposited onto the seabed as fish faeces but also as waste feed pellets and particles. Farmed fish also excrete dissolved inorganic nutrients such as ammonium. Smaller particles of feed in the water column (through the addition of feed and/or via resuspension) can be consumed by other organisms such as zooplankton and shellfish, which, through subsequent excretion, in turn contribute to the dissolved nutrient pool. The dissolved inorganic nutrients from feed-added aquaculture combined with other sources of nutrient inputs can fuel the growth of phytoplankton (Wu et al. 1994) and at high concentrations can cause harmful phytoplankton blooms (Sorokin et al. 1996). In New Zealand's temperate waters, nitrogen may be the nutrient that limits phytoplankton growth under certain conditions, e.g., when concentrations are generally low, and light is plentiful (MacKenzie 2004, Howarth & Marino 2006). However, nutrients from finfish farms are only one source of nutrients in the marine environment, and, like other sources, their inputs vary over time, e.g., salmon farms in the Marlborough Sounds increase feed levels by about 50% during summer months, which is also the period of greatest light availability for primary production. Internationally there have been experiences of blooms of species that produce biotoxins, some of which can be directly toxic to fish and others which can accumulate in shellfish and affect consumers. As far as is known to date, salmon farming in New Zealand has not given rise to any harmful phytoplankton blooms and such effects are unlikely in the near future unless considerable new development occurs (Forrest et al. 2007c).

The risk of exceeding the assimilative capacity and accelerating eutrophication will be dictated by the physical characteristics of a region, such as retention time, water depth, and ambient nutrient concentrations, combined with the intensity and types of existing and planned aquaculture and upstream land-based developments. There is compelling evidence that bivalve aquaculture can affect nutrient cycling and the quantity and quality of food (plankton) across a range of spatial scales from local to system wide (Prins et al. 1998, Cerco & Noel 2007, Coen et al. 2007). In turn, the quantity and quality of food available to other consumers could be affected (Prins et al. 1998, Dupuy et al. 2000, Pietros & Rice 2003, Leguerrier et al. 2004), with consequences for local populations of higher trophic level organisms such as fish.

In some regions where numerous farms with high-density cultures occur, there is the potential risk of exceeding the region's capacity to sustain high shellfish production and the wider ecosystem itself. An example is Pelorus Sound, where questions around the concept of carrying capacity arose following observed decreases of about 25% in green-lipped mussel yields between 1999 and 2002 (Zeldis et al. 2008). These reductions were attributed to climatic forcing conditions and inter-annual variability in phytoplankton biomass over multi-year time scales (Zeldis et al. 2008). This suggests that this region is close to sustainable production limits during years of naturally low primary production.

16.1.10.3 MANAGEMENT OPTIONS AND KNOWLEDGE GAPS

Internationally there is a limited understanding of the cumulative effects of multiple stressors on marine ecosystems in the long term. A critical requirement for understanding these effects is to have good information about existing environmental conditions, and continued monitoring to provide long time series datasets from which to validate models and to quantify and forecast changes occurring in the wider environment. Monitoring programmes, the use of models and existing data, and suggested management approaches for future aquaculture development, which contribute to cumulative effects management in New Zealand, are discussed below.

16.1.10.3.1 MONITORING PROGRAMMES

Management of cumulative effects in the marine environment can be conducted using a two-tiered approach that not only considers the contribution of effects from individual developments, but also an overall regional assessment of wider environmental change in response to the many stressors impacting on the marine environment (e.g., Dubé 2003). Critical to regional assessments of cumulative effects in the marine environment is accessibility and coordination of datasets, including those derived from consent monitoring at individual farms, and long-term State of the Environment monitoring programmes by regional councils. Standardisation of monitoring requirements for aquaculture is an important step to ensure the usefulness of consent monitoring datasets within broader-scale assessments. The requirements for assessment and management of cumulative effects fall beyond the scope of a single consent applicant or industry and are best dealt with through regional councils (e.g., Dubé 2003, Hargrave et al. 2005, Zeldis 2008a, 2008b) or central government departments (Morrisey et al. 2009, Zeldis et al. 2011a, 2011b).

Much work has been completed in New Zealand to guide the implementation of regional scale monitoring programmes. These include the following.

- The use of the Limits of Acceptable Change adaptive framework to develop a management framework for the 3000 ha Wilson Bay Aquaculture Management Area, in the eastern Firth of Thames. This involved agreement by stakeholders, both to levels of acceptable change in indicators and to management responses if monitoring showed that these changes have been exceeded (Zeldis et al. 2006).
- The production of a series of reports for the Waikato Regional Council to guide the implementation of a regional programme to monitor water quality and benthic habitats. Three reports were produced: the first provides the rationale and key elements of a regional monitoring framework that integrates monitoring associated with consented activities and wider State of the Environment monitoring (Forrest & Cornelisen 2015); the second describes the ecological effects of aquaculture in the Waikato coastal management area and identifies priority issues (Forrest et al. 2015); and the third recommends methodologies and standards for monitoring the seabed, water column, and the wider environment in relation to the potential effects of aquaculture (Keeley et al. 2015b)
- The development of water quality standards and a framework to monitor and manage potential water column nutrient enrichment from salmon farms in the Marlborough Sounds. These best management practice guidelines were developed by a multi-stakeholder working group (Elvines et al. 2019b). The guidelines were informed by international examples of best practice and customised to the biophysical conditions of the Marlborough Sounds; however, much of the monitoring and management framework may be broadly applicable to other finfish farms or feed-added aquaculture, and/or existing salmon farms in low-flow locations. The aim of the monitoring protocol is to provide early detection, or warning signs, of a deterioration in water quality from

nutrient enrichment at a regional scale. A tiered monitoring approach is recommended. This consists of routine regional scale monitoring (Tier 1), which encompasses the State of the Environment monitoring by regional councils, against the water quality standards. Tier 2 monitoring is triggered when the water quality standards are exceeded, to determine whether salmon farm inputs are likely to be the primary cause. Tier 3 monitoring can be initiated on a farm-by-farm basis if more intensive monitoring is required. Tiers 2 and 3 monitoring protocols are not explicitly defined in the guidelines, because these would be performed and designed on a farm-by-farm basis (Elvines et al. 2019b).

- The development of benthic environmental quality standards and a framework to monitor and manage potential benthic habitat impacts from salmon farms in the Marlborough Sounds. These best management practice guidelines were developed by a multi-stakeholder working group (Keeley et al. 2015a, Keeley et al. 2019). The recommended monitoring protocol seeks to provide consistent and clear requirements for the benthic monitoring and management of existing farms, based around an agreed set of environmental quality standards with accompanying rationale. The monitoring protocol focuses on farm-scale monitoring of effects in the immediate vicinity of the farm and near-field and far-field effects. An option is provided to include cumulative effects monitoring sites stationed in areas that are potentially predisposed to organic accumulation or are otherwise of concern, e.g., a nearby depression or an area close to habitats of ecological significance.

A project, commissioned by MPI, to provide recommendations for a national marine environmental monitoring programme for New Zealand (Hewitt et al. 2014) identified several variables that could, with small extensions in data collection, together with the development of analytical and reporting techniques, be reported at a national level. These variables cover various aspects of physics, chemistry, and biology, and research has demonstrated both strong links to other components of the ecosystem and strong responses to anthropogenic stressors (Hewitt et al. 2014).

16.1.10.3.2 MODELLING AND DATA USE

Modelling has an important role to play in understanding, prediction, and management of cumulative effects. New Zealand has access to extensive modelling capability; yet in most analyses the uncertainty in model accuracy remains high due to insufficient field data for their calibration and validation. For example, underlying hydrodynamic models require sufficient time series data on currents and water column stratification, and more advanced biogeochemical models require validated estimates of inputs (e.g., surface water, groundwater, marine parameters) and losses (denitrification, burial rates) of nutrients specific to New Zealand's coastal waters.

Spatial modelling tools offer a way to estimate the extent to which the cumulative effects of aquaculture may be approaching ecological carrying capacity on 'bay-wide' and 'regional' scales. However, knowledge gaps are still evident in these models; particularly in the biological aspects (e.g., feeding behaviour and growth of the shellfish), which are still areas of active research. Some generalisations have been proposed for carrying capacity, but these are not always in agreement. Using 'sustainability performance indicators', Gibbs (2007) suggests that the retention (flushing) time for a water body should not exceed 5% of the clearance time of farmed mussels to minimise cumulative effects on the wider ecosystem. The proposed bivalve aquaculture standards suggest that if the clearance time for the farmed bivalves divided by the retention time of the water body is less than 1, and the area occupied by the farms is less than 10% of the total area of the water body, then ecological impacts are likely to be acceptable (Bivalve Aquaculture Dialogue 2010).

ECOPATH modelling (Christensen et al. 2000) was applied to assess the potential of Tasman Bay for mussel aquaculture development. This indicated that significant ecosystem energy flow changes occurred at mussel biomass levels less than 20% of a mussel-dominated ecosystem, thus implying that ecological carrying capacity limits may be much lower than production carrying capacity limits (Jiang & Gibbs 2005). Typically, modelling is therefore used to determine the ecological carrying capacity of each system.

For analyses of cumulative effects related to eutrophication, there is currently a very limited scientific understanding of the transport, fate, and ecological consequences of nutrient loading from different sources and, in turn, how they cumulatively affect marine ecosystems (Olsen et al. 2008). A review commissioned by MPI highlighted a range of approaches used internationally to understand and manage the effects of nitrogen loading from aquaculture on the marine environment (Hartstein & Oldman 2015). The review found commonalities between approaches used in several countries in that they all (1) set some environmental trigger/s at which point a review of production, management, or some other intervention occurs, (2) provide an understanding of the dynamics of the system (through modelling and/or monitoring), and (3) quantify the impacts of the existing and/or proposed aquaculture production in the context of the natural variability of the system.

The trigger level approach adopted overseas (and being developed in New Zealand) is recommended as the standard for assessing how a system is responding to the establishment or expansion of aquaculture areas and should become an integral part of adaptive management plans. The process of establishing such triggers necessitates development of a thorough understanding of the marine system being considered through a combination of modelling and monitoring. With such tools in place, the relative roles of natural variability and the impacts of potential aquaculture development scenarios can be assessed.

A precautionary approach necessitates establishment of conservative thresholds or limits to minimise risks and the extent of cumulative effects. Nutrient mass-balance models can provide guidance on nutrient loading rates in a region under various scenarios and on gauging proximity to conservative critical nutrient loading rates (Olsen et al. 2008). The mass-balance approach has facilitated the development of system-wide nutrient budgets and estimates of carrying capacity for feed-added aquaculture in Golden Bay and Tasman Bay (Zeldis 2008b, Zeldis et al. 2011a, 2011b) and the Firth of Thames (Zeldis 2008a, Zeldis et al. 2010).

Projects currently underway which may help to address some of the knowledge gaps include the following.

- Phase II of the Sustainable Seas Challenge. Phase II includes two projects under the theme 'Degradation and recovery'. One of the projects will investigate ecological responses to cumulative effects and the other will explore tools to manage cumulative effects. Multiple stakeholders are helping to co-develop these projects, including Fisheries New Zealand. Phase II of the Challenge commenced in 2019 and results are expected in 2024.
- The Moana project by MetOcean Solutions. This project is currently underway and supported by Fisheries New Zealand. The project aims to deliver an open-access archive of oceanography data for New Zealand; accurate historical wave, wind, and current data at high spatial and temporal resolutions; an accurate description of current flows around New Zealand; and data access tools and products. These long-term data sets will help to inform the assessment of potential effects of aquaculture and feed into ecological and depositional models.

16.1.10.3.3 MANAGING FUTURE DEVELOPMENTS

Managing cumulative effects to achieve sustainability ultimately requires regional approaches to manage developments and activities in a holistic, ecosystem-based management (EBM) framework which utilises spatial planning (Crain et al. 2008). In the absence of over-arching EBM programmes and a robust scientific base for adaptive management in response to cumulative effects, a precautionary approach is warranted in future developments of feed-added aquaculture. Development should be conducted in a staged manner based on conservative limits of expansion taking into account the level of uncertainty associated with potential ecological effects. Important tools and components of a precautionary approach include the following.

1. The use of models and existing data to gauge limits to development[4] within the context of a region's assimilation capacity (i.e., ecological carrying capacity).
2. Establishment of wider-ecosystem, long-term monitoring programmes that include establishment of baseline conditions of a region and adoption of limits of acceptable change.
3. Mitigation of effects through continual improvement of on-farm practices, potentially

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[4] Areas may not be suitable for any development of aquaculture.

including improved feed technologies and the use of Integrated Multitrophic Aquaculture (IMTA, Figure 16.5). IMTA combines farming of different species to potentially ameliorate environmental effects.

4. Targeted monitoring and research to validate and improve accuracy of predictive models (see, for example, Elvines et al. 2021) and to understand the role of feed-added aquaculture in driving cumulative effects.

An overseas example of the precautionary approach is the M-O-M system (Modelling–Ongrowing fish farms–Monitoring), which has been used in Norway to provide information for adaptive management of salmon farming (Ervik et al. 1997, Hansen et al. 2001).

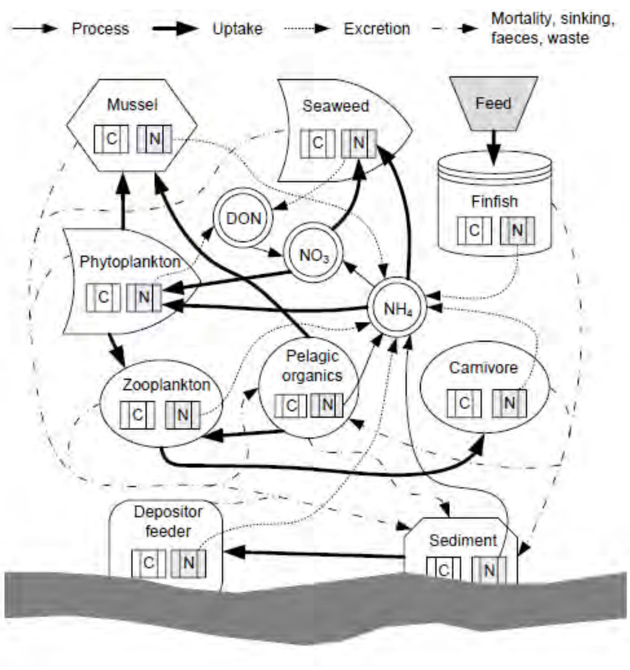

Figure 16.5: Conceptual diagram of IMTA model in terms of carbon (C) and nitrogen (N) biomass (from J Ren. NIWA pers. comm.).

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Chapter 17: Antarctic Science - Technical Summary

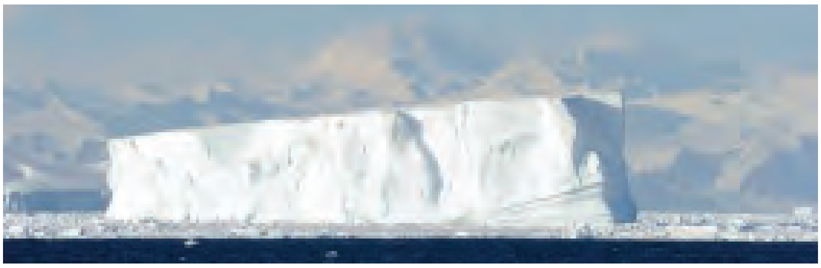
1. THE ISSUE IN BRIEF

- The Convention for the Conservation of Antarctic Marine Living Resources (CCAMLR) was established in 1982. It was the first international agreement to explicitly focus on conservation and require that fisheries management considers the effects of fishing on dependent and associated species.
- Under the Antarctic Treaty system, Antarctica is designated as a natural reserve, devoted to peace and science, and New Zealand is committed to preserving and protecting Antarctica and the Southern Ocean for present and future generations. New Zealand's interest in CCAMLR is the health and conservation of the Southern Ocean, with a main focus on the Ross Sea ecosystem where there is a bottom longline fishery for toothfish involving vessels from multiple countries, including New Zealand.
- Now the Ross Sea toothfish fishery is managed to meet very precautionary decision rules, among the most precautionary management of any fisheries globally. New Zealand leads the stock assessment research for this fishery. The current status of the stock is healthy and well above target level.
- The world's largest Marine Protected Area (MPA) was established in the Ross Sea in 2017. New Zealand, along with other countries, contributes to the research and monitoring of the MPA
- New Zealand also undertakes regular compliance monitoring of activities within the MPA, and also in the Ross Sea fishery to confirm all fishing vessels are complying with the rules in force

2. IMPACTS OF TOOTHFISH FISHING ON THE ROSS SEA ENVIRONMENT

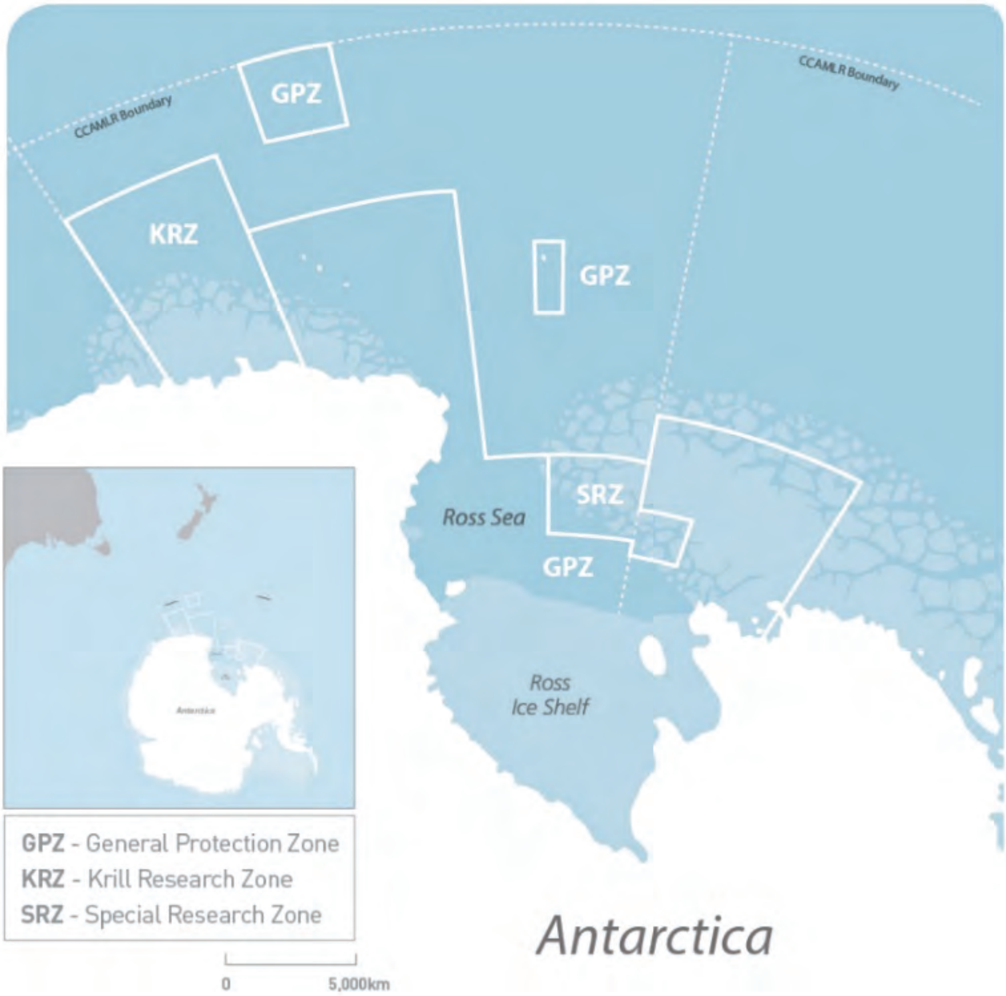
- Fish bycatch: The main bycatch species in the Ross Sea region are macrourids or grenadiers, icefish, skates, eel cods, and deepsea (morid) cods. There are several approaches in place to mitigate fish bycatch, including catch limits. A skate tagging research programme is also underway to assess the abundance of skates and to understand the survivability of returned live skates
- Seabird mortality: Following the introduction of some of the strictest management requirements anywhere in the world, seabird bycatch rates in Southern Ocean fisheries are now negligible. In the Ross Sea toothfish fishery, only 2 seabird mortalities have occurred since 2007
- Mammal mortality: There have been no reported mortalities of marine mammals in the toothfish longline fisheries in the Ross Sea or Amundsen Sea regions
- Benthic communities: The impacts of bottom longlining on benthic communities are not well understood and are recognised as in need of attention. Several benthic focussed research programmes are underway to verify and quantify the impacts of fishing gear on the benthos, including the use of deep-sea cameras to photograph and map the seabed where fishing takes place
- Effects on top predators: Predators can potentially be affected by changes in the abundance of their prey due to fishery removals. Weddell seals, killer whales, and sperm whales are known to feed on toothfish. Adélie penguins feed on silverfish, which are prey for toothfish. Limited information is known on these effects, but given the health of the toothfish stock these effects are considered to be negligible
- General environment: The broader impacts of fishing on the marine environment such as marine pollution, lost fishing gear, and biosecurity, are strictly managed by rules that align with global best practices

3. ROSS SEA REGION MARINE PROTECTED AREA



- CCAMLR members unanimously supported the establishment of the Ross Sea region Marine Protected Area, which came into force in 2017. New Zealand and USA were the co-proponents that designed it and developed its objectives

- The MPA has multiple objectives including providing a reference area to better understand the ecosystem effects of climate change and fishing, and to preserve a representative portion of the Ross Sea environment (including benthic and pelagic marine environments), and to protect core foraging areas for land-based predators
- A Scientific Research and Monitoring Plan has been developed. New Zealand along with other nations, including Italy, South Korea, and the USA are actively conducting research to feed into the first scientific review of the MPA in 2022



The MPA consists of:
- A 'no take' General Protection Zone (a fully protected area where no commercial fishing is permitted) split into three separate areas
- A Krill Research Zone which allows for controlled research fishing for krill, in accordance with the objectives of the MPA
- A Special Research Zone which allows for limited research fishing for krill and toothfish

17 ANTARCTIC SCIENCE

Status of chapter	This chapter was updated for AEBAR 2021.
Scope of chapter	This chapter outlines the following for the Ross Sea sector of the Southern Ocean (Ross Sea and Amundsen Sea): the ecosystem structure, the toothfish fishery, the nature of ecosystem-fishing interactions, the management approach, trends in key indicators of fishing effects, and major sources of uncertainty and research priorities.
Area	Ross Sea region; Amundsen Sea region (CCAMLR areas 88.1 and 88.2).
Focal localities	Areas with significant fisheries interactions include the Ross Sea shelf and slope, Amundsen Sea seamounts and shelf, and areas of the Pacific-Antarctic Ridge.
Key issues	Fisheries in the region are managed by CCAMLR according to the principles of conservation given in Article 2 of the Convention for the Conservation of Antarctic Marine Living Resources. Effects of fishing are considered in the following categories: (1) bycatch species, (2) prey of target species, (3) predators of target species, (4) ecosystem at the system-level, and (5) benthic habitat.
Emerging issues	Role of Marine Protected Areas in the Ross Sea region.
Fisheries New Zealand research (current)	ANT2019-01.
New Zealand government research (current)	MBIE: C01X1226 *Ross Sea Climate and Ecosystem*. MBIE: END18301 *Ross Sea Region Research and Monitoring Programme*. NIWA Core Funding, Coasts & Oceans Programme 4: ‘Structure and function of marine ecosystems’ Antarctica New Zealand: Antarctic SIF
Related chapters/issues	Benthic (seabed) impacts; Trophic and ecosystem-level effects; Biodiversity. Fisheries Plenary Report May 2021 Chapter on Toothfish (Fisheries New Zealand 2021).

17.1 CONTEXT

17.1.1 THIS CHAPTER

This chapter discusses the ecosystem effects of fishing for toothfish (principally Antarctic toothfish) in the Ross Sea region (150° E to 150° W) and the Amundsen Sea region (150° W to 105° W) (Figure 17.1). There is currently no krill fishing in the Ross Sea and Amundsen Sea regions.

The focus is on the ecosystem effects of fishing rather than the management of the toothfish stock itself. The stock assessment for Antarctic toothfish in the Ross Sea region is updated every two years (most recently in 2021, Grüss et al. 2021c) and a summary is available as part of the May Plenary Report (Fisheries New Zealand 2021). Research towards a stock assessment for Antarctic toothfish in the Amundsen Sea region was last completed in 2016 (Mormede et al. 2016) and an assessment is planned for 2022.

Section 17.1 presents a brief history of Southern Ocean fisheries, the present management framework for toothfish, and overviews of the life history, fishery, and management of toothfish fisheries in the Ross Sea and Amundsen Sea regions, including the Ross Sea region Marine Protected Area (RSrMPA).

Section 17.2 gives a characterisation of the Ross Sea ecosystem; no characterisation is yet available for the Amundsen Sea ecosystem.

Section 17.3 presents information on the major ecosystem effects of fishing in five categories:

- Effects of fishing on bycatch species,
- Effects of fishing on prey species,
- Effects of fishing on predator species,
- Trophic and system-level effects,
- Effects of fishing on habitats.

Section 17.4 summarises information on indicators and trends for the ecosystem effects of fishing in the Ross Sea and Amundsen Sea regions in the same five categories.

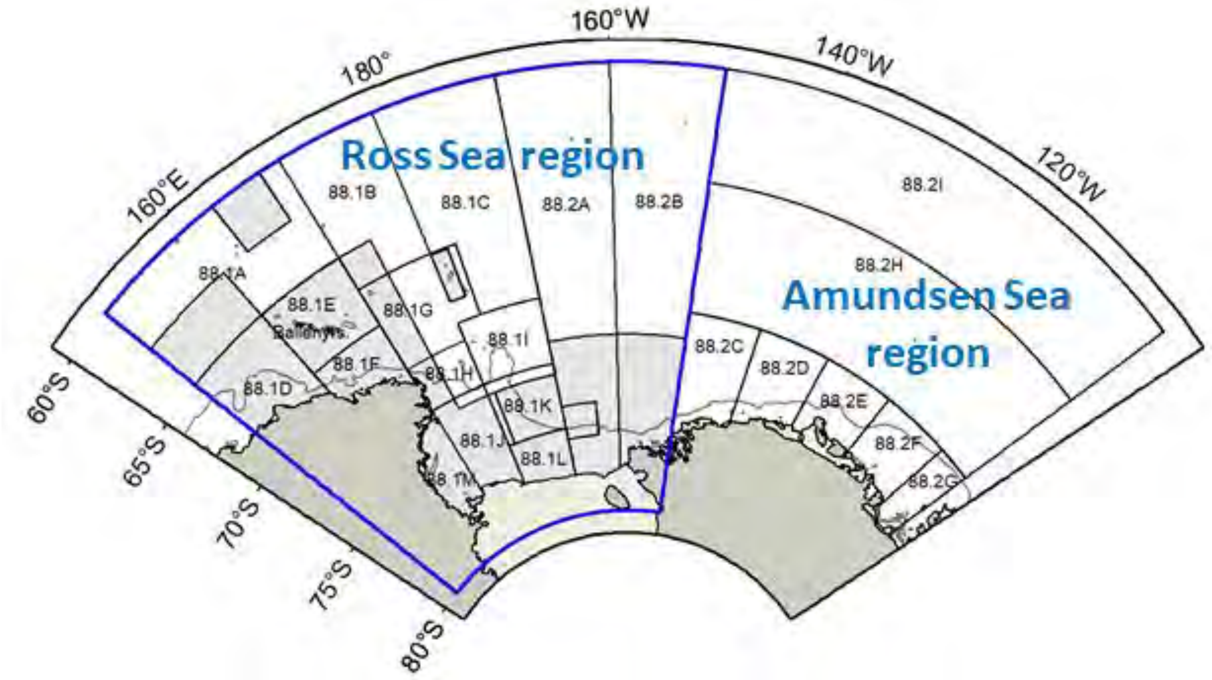


Figure 17.1: The Ross Sea and Amundsen Sea regions, which span CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources) Subareas 88.1 and 88.2. CCAMLR small scale research units (SSRUs) and the depth contour at 1000 m are shown. The Ross Sea region is made up of 88.1 and 88.2A, B. The Amundsen Sea region includes SSRUs 88.2C–I. The Ross Sea *slope* region (with depths of approximately 1000–3000 m) is mainly contained in SSRUs 88.1H, 88.1I and 88.1K. Areas that are shallower than about 1000 m are called *shelf* regions (comprising 88.1J, 88.1L, 88.1M for the Ross Sea shelf). Shaded regions indicate the Ross Sea region MPA boundaries and include the Special Research Zone, Krill Research Zone, and General Protected Zones (i), (ii), and (iii).

17.1.2 SOUTHERN OCEAN FISHERIES

A brief history of fisheries in the Southern Ocean[1] is given by CCAMLR (Commission for the Conservation of Antarctic Marine Living Resources)[2] and this section summarises that text. Seal harvesting in the Southern Ocean began in 1790. By 1825, some populations of fur seal were hunted close to extinction, and sealers began hunting elephant seals and some species of penguins for their oil. Sealing continued on a small scale into the 20th century, but there has been no commercial sealing in Antarctica since the 1950s.

Whaling in the Southern Ocean area began in 1904 and multiple species of whales found in the region were extensively exploited (see section 17.2.3 for more details about whaling in the Ross Sea region). A moratorium on commercial whaling was introduced in 1987. Whale sanctuaries were established in the Indian Ocean in 1979 and Southern Ocean in 1994. Management of whales is today the responsibility of the International Whaling Commission (IWC). There are indications that some species of whale are recovering, but the low abundance of some of the largest species has made total numbers difficult to estimate.

Large-scale fishing for finfish in the Southern Ocean began in the late 1960s. Overall trends in fishery catches have varied widely, reflecting intense fishing during the 1960s and 1970s prior to the establishment of CCAMLR. Such fishing led to the overexploitation of some finfish species in the mid-1970s and 1980s. This overfishing, along with interest in large-scale exploitation of Antarctic krill, raised concerns about the sustainability of Southern Oceans fisheries.

In the 1980s and 1990s, fishing in the Southern Ocean focused on krill (*Euphausia superba*), Patagonian toothfish (*Dissostichus eleginoides*), mackerel icefish (*Champsocephalus gunnari*) and, to a limited extent, squid

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[1] The Southern Ocean extends from the coast of the Antarctic continent northwards to the Antarctic Polar Front and represents approximately 15% of the world's ocean area.

[2] CCAMLR, History: The Southern Ocean. Retrieved from https://www.ccamlr.org/en/organisation/history.

and crab. Since the 1990s there has been growing interest in fisheries targeting Antarctic toothfish (*Dissostichus mawsoni*) adjacent to the Antarctic continent.

At the Eighth Antarctic Treaty Consultative Meeting in 1975, the Parties adopted the recommendation that noted the need to promote protection, scientific study, and rational use of Antarctic marine living resources. This led to a Conference on the Conservation of Antarctic Marine Living Resources (CAMLR), which resulted in the CAMLR Convention.

New Zealand was a founding member of the CAMLR Convention which entered into force in 1982. The area of jurisdiction of the CAMLR Convention is approximately south of the circumpolar Antarctic Polar Front (Antarctic Convergence) in the Southern Ocean (Figure 17.2). The position of the Antarctic Polar Front varies seasonally and geographically, but is generally located near 50° S in the Atlantic and Indian sectors of the Southern Ocean and 60° S in the Pacific sector.

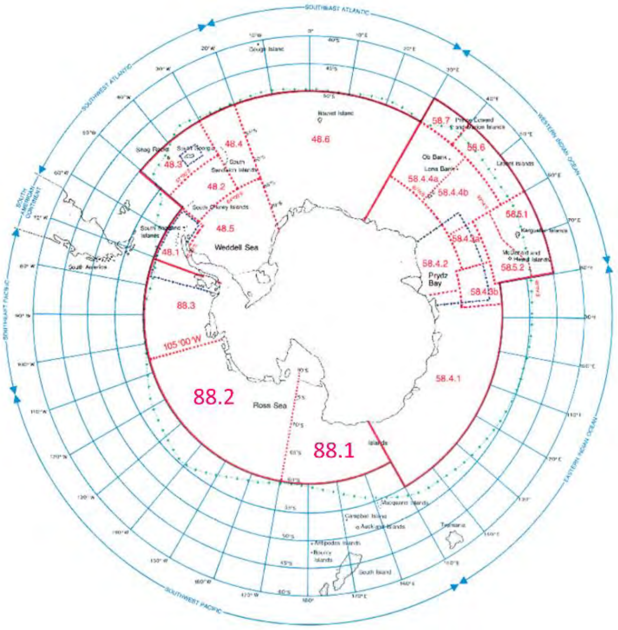

Figure 17.2: Boundary (solid pink line) of area managed according to the Convention for the Conservation of Antarctic Marine Living Resources. Dashed pink lines delineate CCAMLR statistical subareas.

17.1.3 CCAMLR'S MANAGEMENT

The aim of the CAMLR Convention is to conserve the marine life of the Southern Ocean while allowing rational use of marine resources, including commercial fishing (CAMLR Convention 1980). The CAMLR Convention was the first international fishing agreement to explicitly require that management considers the effects of fishing on dependent and associated species as well as on the target species. The principles of conservation are given in Article II of the CAMLR Convention and allow fishing in the CCAMLR Area subject to:

(a) maintenance of the size of harvested population at levels which ensure stable recruitment;
(b) maintenance of ecological relationships between harvested, dependent, and related populations;
(c) prevention or minimisation of the risk of changes in the marine ecosystem that are not reversible in 20–30 years.

The regulatory framework for CCAMLR-managed fisheries recognises five types of fisheries: (1) new fishery; (2) exploratory fishery; (3) established fishery; (4) lapsed fishery; and (5) closed fishery.

Both the Ross Sea region and Amundsen Sea region toothfish fisheries are managed as 'exploratory fisheries' by CCAMLR. Exploratory fisheries are not allowed to expand faster than the acquisition of information necessary for managing the fishery within CCAMLR's management objectives. In addition, notification and permission are required each year prior to fishing (Conservation Measure CM 21-02). Finally, a fishery remains an exploratory fishery until sufficient information is available on appropriate catch and effort levels and the potential impacts on dependent and related species.

Decisions in CCAMLR are made by consensus among member states. At present members of the Commission include 25 States[3] and the European Union (acting as a single member). A further 10 countries[4] have acceded to the Convention. CCAMLR's Secretariat facilitates the implementation of the CAMLR Convention. Measures to manage Southern Ocean fisheries are implemented by means of a series of Conservation Measures (CMs), which

[3] Argentina, Australia, Belgium, Brazil, Chile, China, European Union, France, Germany, India, Italy, Japan, Republic of Korea, Namibia, Netherlands, New Zealand, Norway, Poland, Russian Federation, South Africa, Spain, Sweden, Ukraine, United Kingdom, United States of America, Uruguay.

[4] Bulgaria, Canada, Cook Islands, Finland, Greece, Mauritius, Islamic Republic of Pakistan, Republic of Panama, Peru, Vanuatu.

are published annually[5] following decisions by the CCAMLR members at the annual Commission meetings in October. Scientific information, analyses, and discussion to inform management are brought together annually by the CCAMLR Scientific Committee, which in turn is informed by several working groups, including: (1) Working Group on Ecosystem Monitoring and Management (WG-EMM); (2) Working Group on Fish Stock Assessment (WG-FSA); (3) Working Group on Statistics, Assessments and Modelling (WG-SAM); (4) Working Group on Incidental Mortality Associated with Fishing (WG-IMAF); (5) Subgroup on Acoustics, Survey and Analysis Methods (SG-ASAM).

The Scientific Committee on Antarctic Research (SCAR) has also advised CCAMLR in respect of key scientific areas for research. Work to develop greater collaboration between CCAMLR and IWC has been underway since 2013, especially with regard to managing the trophic impact of fishing for krill on baleen whales and other krill predators. Of particular relevance is the IWC-Southern Ocean Research Partnership (SORP).[6]

17.1.4 ANTARCTIC AND PATAGONIAN TOOTHFISH

Antarctic toothfish (*Dissostichus mawsoni*, Norman 1937) is endemic to the Southern Ocean, with a circumpolar distribution. The species is found in higher latitudes south of the Antarctic Convergence (Gon & Heemstra 1990). Patagonian toothfish (*Dissostichus eleginoides*, Smitt 1898), often marketed as '*Chilean sea bass*', shares many similarities with Antarctic toothfish but has a more northern distribution being rarely found in latitudes south of the Antarctic Convergence at about 65° S (Figure 17.3). A species profile, covering aspects of the biology, fisheries, and stock assessment of both toothfish species was completed by Hanchet (2010) and Hanchet et al. (2015a).

For Antarctic toothfish in the Ross Sea region, spawning dynamics and early life history is described by Hanchet et al. (2008), Hanchet et al. (2015a), Ghigliotti et al. (2018), Parker et al. (2019), and Behrens et al. (in press).

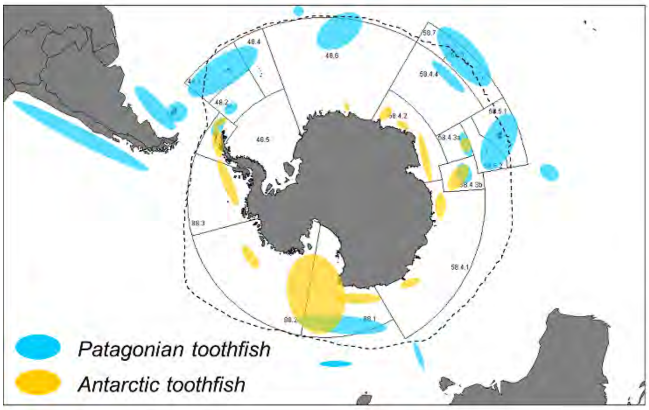

Figure 17.3: Distributions of Antarctic and Patagonian toothfish in the Southern Ocean. Approximate location of Antarctic Convergence shown by the dotted line.[7]

The toothfish stocks in the Amundsen Sea region are managed separately by CCAMLR from those in the Ross Sea region. Parker et al. (2014) reviewed the information from genetic studies, otolith microchemistry, stable isotopes, tagging, size and age structure, growth dynamics, and egg and larval dispersal simulations. The study concluded that it is likely that juveniles (less than 80 cm total length) from the two stocks mix in the shelf region, but that there is very limited mixing of adults between the Ross Sea and Amundsen Sea regions. Dispersal modelling indicated eggs (and then juveniles) can be transported from spawning areas in the Ross Sea into the Amundsen Sea (Behrens et al. in press). The Amundsen Sea stock probably includes juveniles and adults along the continental margin of the Amundsen and Bellingshausen seas, and a spawning region in the seamount complex of SSRU 88.2H (Amundsen Sea). Further information is needed to improve knowledge of the toothfish stock structure in the Amundsen Sea region (Delegations of New Zealand, Norway and the United Kingdom 2014).

For Antarctic toothfish in the Ross Sea and Amundsen Sea regions, spawning is thought to take place to the north of the Antarctic continental slope, during winter (Hanchet et al. 2008). The first winter longline survey of Antarctic toothfish conducted during June and July 2016 in the northern Ross Sea region confirmed toothfish spawning in this region (Stevens et al. 2016). A second survey in the same area in 2019, conducted in September and October, found eggs at the eyed-embryo stage in surface waters and

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[5] CCAMLR Publications. Retrieved from https://www.ccamlr.org/en/publications/publications.

[6] International Whaling Commission. The Southern Ocean Research Partnership (IWC-SORP). Retrieved from https://iwc.int/sorp.

[7] NIWA. Antarctic Toothfish Fishery in the Ross Sea. Retrieved from https://www.niwa.co.nz/fisheries/research-projects/the-ross-sea-trophic-model/toothfish-fishery.

confirmed spawning had been completed by mid-August (Parker & DiBlasi 2020).

More information on the life history and stock structure can be found in the Fisheries Assessment Plenary (Fisheries New Zealand 2021).

The stock structure of Patagonian toothfish in the Ross Sea and Amundsen Sea regions is less well known. Patagonian toothfish in the Ross Sea region are believed to come from a stock which is widely distributed beyond Macquarie and Campbell Plateau and into the high seas.

17.1.5 ROSS SEA REGION TOOTHFISH FISHERY

A characterisation of the fishery in the Ross Sea region is given by Grüss et al. (2021a). Fishing for toothfish began in the Ross Sea region in 1997. The Ross Sea region is the major fishing area for Antarctic toothfish in the Southern Ocean (Hanchet et al. 2008). Most of the catch in the Ross Sea region (over 99%) is Antarctic toothfish (an average of 2860 t y^{-1} since 2005). Catches of Patagonian toothfish, taken mainly from the north-west of the Ross Sea region, have averaged only about 0.1% of the total reported toothfish catch over the period 2010–2021.

The toothfish fishery in the Ross Sea region saw a steady expansion of effort (number of sets) from 1998 to 2001, and an almost three-fold increase in 2004, which led to the increases in catches shown in Figure 17.4. Since 2005, effort has been more stable. All fishing for toothfish in the Ross Sea and Amundsen Sea regions uses baited longlines. In earlier years, most vessels fished with the autoline system, but these have been joined by vessels fishing with Spanish lines and, more recently, trotlines.

The average Illegal, Unreported, and Unregulated (IUU) catch of toothfish in Subarea 88.1 was estimated to be 92 t in 2002, 240 t in 2004, 28 t in 2005 and 272t in 2008. Following the recognition of methodological issues regarding the estimation of IUU catch levels since 2011, evidence of IUU presence or activity has continued to be recorded, but no corresponding estimates of the IUU catch for *Dissostichus* spp. have been provided. One IUU-listed fishing vessel was observed in Subarea 88.1 during 2006 and 2010 and unmarked fishing gear, potentially from an IUU vessel, was reported in 2016 (CCAMLR 2021a).

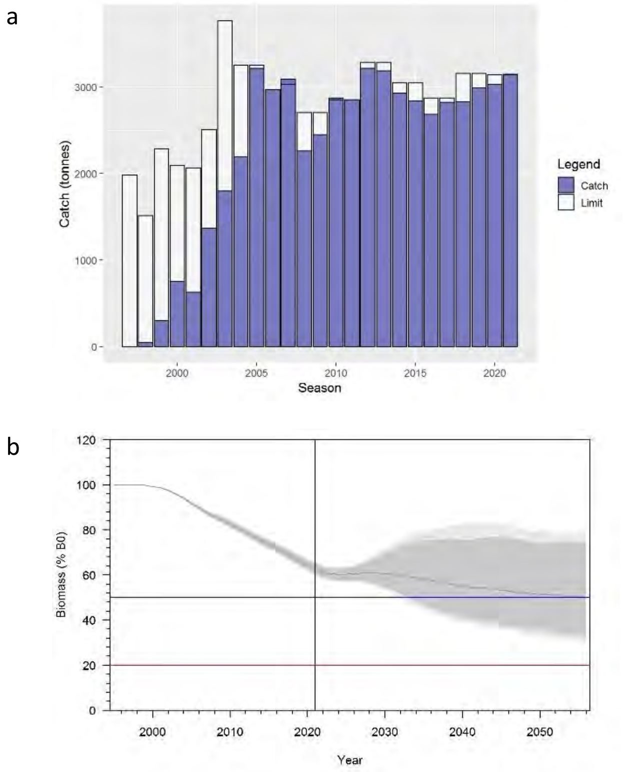

Figure 17.4: [a] Catch and catch limit, [b] assessed spawning stock biomass (with 5th–95th percentiles in grey) for Antarctic toothfish in the Ross Sea region (CCAMLR subareas 88.1 and 88.2A, B). Antarctic fishing years are labelled as the later year of the season (e.g., the 1997–98 fishing season is labelled '1998'). [CCAMLR 2021a; Grüss et al. 2021a, Grüss et al. 2021c].

Annual research surveys of sub-adult (70–110 cm) toothfish have been carried out in the southern Ross Sea since 2011 to provide an estimate of any changes in recruitment (e.g., Devine et al. 2021).

Spatial information on fishing in the 88.1/88.2 is often described using CCAMLR Small-Scale Research Units (SSRUs; Figure 17.1). Although most SSRUs have been fished over time, the proportion of effort in each SSRU has varied considerably each year and with different ice conditions. Two of the three slope SSRUs (88.1H and 88.1I) have been the most consistently fished SSRUs (Figure 17.5). In years with ice conditions favourable to fishing the fishery also extends into 88.1K.

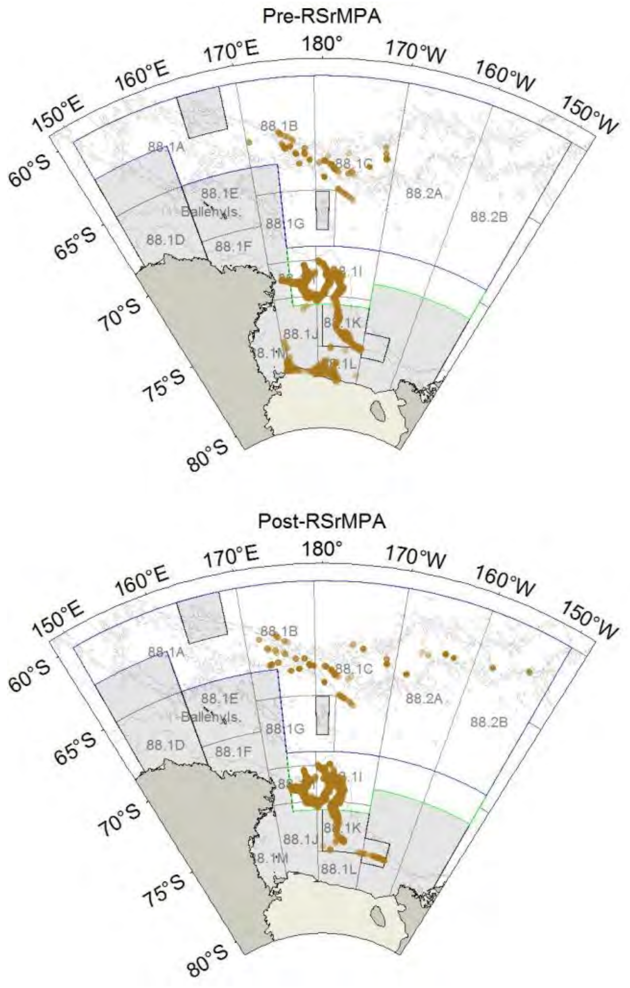

Figure 17.5: Spatial distribution of total toothfish catch (t) in the Ross Sea region before implementation of the Ross Sea region Marine Protected Area (2014–2017) and after (2018–2021).

The length of the fishing season in the Ross Sea fishery has changed over time. In the first few years the fishery was mainly carried out from January to March, and between 2001 and 2003 extended into April and May. Since 2006, fishing starts on 1st December (ice permitting) and is usually finished by early February (Grüss et al. 2021a).

The Ross Sea region toothfish fishery was first certified by the Marine Stewardship Council (MSC) in November 2010 and was recertified in 2015 and 2021. The MSC Fisheries Standard is designed to assess whether a fishery is well managed and sustainable. There are three core principles that every fishery must meet:

1) sustainable fish stock,
2) minimising environmental impact,
3) effective management.

17.1.6 AMUNDSEN SEA REGION FISHERY

The Amundsen Sea toothfish fishery is designated as an exploratory fishery by CCAMLR and a characterisation of the fishery in this region is given by Grüss et al (2021b). The toothfish fishery in the Amundsen Sea region has been operating since 2003, with an annual catch of 106–753 t since 2006 (Grüss et al. 2021b).

The main fishery in this area operated in the northern SSRU until 2013 and then decreased effort following reductions in the catch limit (Grüss et al. 2021b). The total catch distribution for 1997–2021 is shown in Figure 17.6. Within the northern waters, fishing has concentrated mainly on one seamount (the furthest north); effort and catch spreads to other seamounts depending on ice access, number of vessels present, and remaining catch limit at the time. Parker (2014) showed high local exploitation rates and indications of localised depletion on some individual seamounts in the north. Currently, data quality is impacted by the low spatial overlap between locations of released tagged fish and fishing effort in the subsequent year and reductions in fishing effort in the area, exacerbated by lower catch limits (Grüss et al. 2021b).

Only seven Patagonian toothfish have been caught in the Amundsen Sea region since 2004.

More data are required before a robust stock assessment for the Amundsen Sea region can be developed (Mormede et al. 2016, Parker et al. 2021).

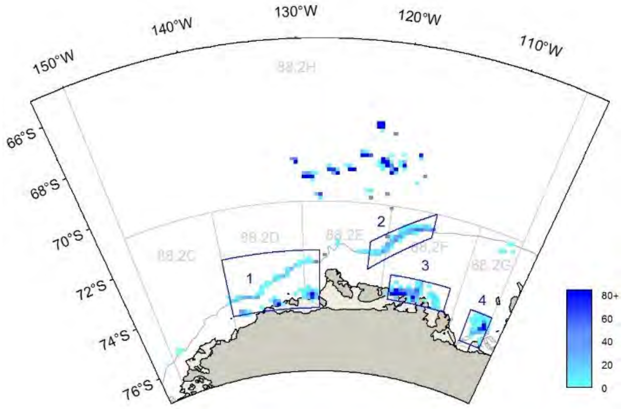

Figure 17.6: Spatial distribution of total toothfish catch (t) in the Amundsen Sea region from 1997 to 2021.

17.2 ROSS SEA ECOSYSTEM

17.2.1 OVERVIEW

Although annual primary productivity is still low on a global scale, seasonally, the shelf waters of the Ross Sea are amongst the most biologically productive areas of the Southern Ocean (Arrigo & van Dijken 2004). Irradiance, iron, and macronutrients (nitrate, silicate) variously limit algal growth through the year (Smith et al. 2003). The high latitude position of the Ross Sea means that primary production is highly seasonal, driven by the annual light/dark cycle and the freezing and thawing of the sea surface. Insolation sets the dominant limit on primary production from autumn to spring; in summer, macronutrients are not depleted and iron appears to limit primary production (e.g., Sedwick et al. 2000, Arrigo et al. 2003). Organisms have various strategies for survival through the winter, including storage of lipids or other high-energy products, winter quiescence, vertical migration, adoption of a wide range of feeding styles, and adaptation of breeding cycles, including migrating in and out of the region (some whales, seals, birds) (Battaglia et al. 1997).

Sea ice plays a key structural role in influencing the ecology of the Ross Sea (Thomas & Dieckmann 2002, Arrigo & Thomas 2004). The mean monthly sea ice cover in the Ross Sea varies from 5% ice-free in winter to 70% ice-free in January (Arrigo & van Dijken 2004), with ice reaching a maximum thickness around November of about 2 m. The Ross Sea polynya[8] is the major structural oceanographic feature of the Ross Sea (Jacobs & Comiso 1989). The dynamics of phytoplankton in the open water of the Ross Sea polynya are very different to those in the marginal ice zone around the polynya. Although ice extent in the Ross Sea region is increasing (Comiso 2003), sea ice in the Ross Sea itself has been decreasing and getting thinner as the Ross Sea polynya has become larger and more persistent (Parkinson 2002).

The upper surface of the ice provides a habitat for a number of seabirds and mammals (Ackley et al. 2003). At the same time, the ice itself, especially the underpart, which is in contact with the water, constitutes a unique habitat for microalgae and bacteria. This provides a food source for associated microfauna and meiofauna and the cryopelagic fauna of the surface water layer immediately below the ice (Garrison 1991, Brierley & Thomas 2002, Arrigo & Thomas 2004). Present estimates suggest that the contribution of epontic[9] algae to total primary production in the Ross Sea is a few percent (Arrigo et al. 1997, Pinkerton et al. 2010a).

The flow of energy from primary production in the water column and sea ice in the Ross Sea consumers is channelled mainly through the copepods. However, the trophic connection between primary producers and copepods is usually not direct. Heterotrophic flagellates and larger heterotrophic microplankton (including dinoflagellates, tintinnids, other ciliates, and eggs and developmental stages of metazoans) graze primary production and often form a large part of the diet of many copepods (Umani et al. 1998, Caron et al. 2000).

Two species of krill are found in the Ross Sea: *Euphausia crystallorophias* and *E. superba*. *E. crystallorophias* is only found over the shelf and *E. superba* is found primarily along the continental slope. Although they form an important link between the water column, sea ice, and larger predators, they are believed to be less productive and have slower turnover rates than the large epipelagic copepods (*Calanoides acutus*, *Calanus propinquus*, *Rhincalanus gigas*, and *Metridia gerlachei*) (Voronina 1998, Tarling et al. 2004). Neither species of krill seems to be as abundant in the Ross Sea as *E. superba* is in the Scotia Sea, where a commercial krill fishery operates and specialist krill predators dominate the ecosystem.

In addition to krill, Antarctic silverfish (*Pleuragramma antarctica*) are a major link between mesozooplankton (mainly copepods) and the larger predators. *Pleuragramma antarctica* are found in the diet of all large animals (seabirds, seals, toothed and baleen whales, toothfish, many other species of fish, and squid) (DeWitt 1970, Laws 1984, Eastman 1985, Vacchi et al. 2017). Throughout their life history the distribution of Antarctic silverfish is thought to include the whole Ross Sea shelf and slope (Hubold 1985), and their juveniles dominate the Ross Sea ichthyoplankton.

More than 100 species and 18 families of fishes have been recorded from the Ross Sea shelf and slope (Chernova & Eastman 2001, Eastman & Hubold 1999, Stewart & Roberts 2001, Bradford-Grieve & Fenwick 2001). Little is known of the abundance of many of these fish species.

The fish fauna of the Ross Sea region can be divided into: (1) a coastal (shelf) fauna, (2) a continental slope fauna, and

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[8] Polynya is a stretch of open water surrounded by ice.

[9] Epontic refers to organisms closely associated with sea ice.

(3) a northern, deeper, oceanic fauna. The shelf fish fauna is dominated (over 90% of biomass) by the four notothenioid families (Nototheniidae, Artedidraconidae, Bathydraconidae, and Channichthyidae), which are endemic to high-latitude Antarctic waters (La Mesa et al. 2004). The benthic shelf fish fauna is species-rich, but the number of species decreases with depth, particularly past the shelf break. Many species have a circum-Antarctic distribution. The Ross Sea slope fish fauna is dominated (in terms of biomass) by the macrourids *Macrourus whitsoni* and *M. caml*, skates (especially *Bathyraja eatonii*), icefish (*Chionobathyscus dewitti*), and eel cods (*Muraenolepis* sp.). To the north of the Ross Sea shelf, the fish fauna is dominated by the small pelagic lanternfishes (myctophidae), especially *Electrona antarctica*, *E. carlsbergii*, *Gymnoscopelus braueri*, and *G. nicholsi*; Antarctic silverfish are not found north of the Ross Sea slope.

Cephalopods (squid and octopods) are likely to be important components of the Ross Sea ecosystem because they appear in the diets of many predators (Rodhouse 2013), but their abundance and trophic roles are poorly known (Okutani 1995, Thompson et al. 2012).

Avian abundance in the Ross Sea region is dominated by penguins. About 38% of the world population of Adélie penguins (*Pygoscelis adeliae*) reside in the Ross Sea, breeding at 35 rookeries (Figure 17.7) with a total of about 1 million breeding pairs (Young 1981, Kooyman & Mullins 1990, Lyver et al. 2014). There are more than 40 000 pairs of emperor penguins (*Aptenodytes forsteri*) breeding between Cape Roget and Cape Crozier, and at Cape Colbeck (Young 1981, Harper et al. 1984, Kooyman & Mullins 1990, Wienecke 2011). There are a significant number of non-breeders and juvenile birds in addition to these breeders.

Seals are the most common marine mammals in the Ross Sea region, with more than 200 000 crabeater seals (*Lobodon carcinophaga*) alone (Ainley 1985). Weddell seals (*Leptonychotes weddellii*) are likely to be the second-most common seal in the Ross Sea, with estimates for the larger Ross Sea region of 32 000 individuals (Stirling 1969, Ainley 1985, Stewart et al. 2003), or about 45% of the entire Pacific sector population. There is debate over the degree to which Weddell seals are migratory. Some individuals may remain in residence year-round in the fast ice at latitudes as high as 78° S in McMurdo Sound. Others, particularly newly weaned and sub-adult animals, might disperse north and east from the continent in the Ross Sea and may spend the winter in the pack ice north of the Ross Sea (Goetz 2015). Smaller numbers of Ross seal (*Ommatophoca rossii*) and leopard seal (*Hydrurga leptonyx*) breed in the region, but abundances are not well known (Ainley 1985, Pinkerton et al. 2010a). Southern elephant seals (*Mirounga leonina*) are also present in the Ross Sea region but the nearest breeding colony is on Macquarie Island.

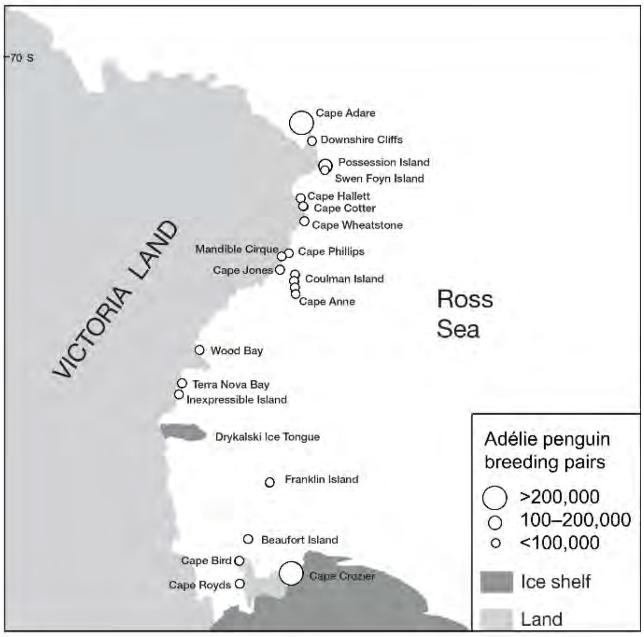

Figure 17.7: The number of breeding pairs of Adélie penguins in the Ross Sea from aerial census methods between 2001 and 2013 (Lyver, unpublished data).

The movements of minke and other baleen whales are poorly understood. In the summer, baleen whales present include minke whale (*Balaenoptera bonaerensis*), fin whale (*Balaenoptera physalus*), humpback whale (*Megaptera novaeangliae*), sei whale (*Balaenoptera borealis*), and blue whale (*Balaenoptera musculus*). They tend to congregate in a feeding zone associated with the pack ice north of the Ross Sea slope where krill are abundant. Over the Ross Sea shelf, humpback and sei whales are largely absent (Ainley 1985, Pinkerton et al. 2010b), although minke whales are relatively common in summer. Antarctic minke whales penetrate deep into coastal sea ice and, together killer whales and beaked whales, have the southernmost distribution of all cetaceans in the Ross Sea region.

Toothed whales present in the Ross Sea region include sperm whale (*Physeter macrocephalus*), killer whale (*Orcinus orca*), southern bottlenose whale (*Hyperoodon planifrons*), and Arnoux's beaked whale (*Berardius arnuxii*). Information on the seasonal abundance of toothed whales in the Ross Sea is rather limited, coming primarily from infrequent surveys of their distribution and numbers (e.g., Ainley 1985). There are at least three different types of killer whale in the Ross Sea region (Pitman et al. 2001, Pitman & Ensor 2003, Pitman 2003, Eisert et al. 2015). Both Type B (penguin and seal-eating) and Type C (fish-eating) killer whales occur in the McMurdo Sound region (extreme south-west of the Ross Sea), but Type C are by far the more common form. Both Type B and Type C killer whales appear to undergo seasonal northward migration and have been sighted in New Zealand waters (Visser 1999, Eisert et al. 2015).

The Ross Sea benthic fauna has high diversity in some taxa, but lacks crabs and lobsters and has low diversity of some major groups such as gastropods, bivalves, polychaetes, and amphipods. There is a dominance of sessile animals, and benthic communities may be multi-storeyed (i.e., occurring in different layers in some areas). Gigantism is found amongst sponges, pycnogonids, amphipods, isopods, and polychaetes.

A review of the biodiversity of the Ross Sea was provided by Bradford-Grieve & Fenwick (2001). However, in contrast, relatively little is known about the biodiversity, structure, or dynamics of the ecosystem of the Amundsen Sea region.

17.2.2 TROPHIC MODELLING

Species in an ecosystem are connected in many ways, but one of the main types of connection is trophic, i.e., the feeding of one organism on another within the food web (McCann et al. 1998, Pace et al. 1999, Frank et al. 2005). Research on the structure of the food web of the Ross Sea has culminated in complex qualitative descriptions (e.g., Smith et al. 2007, Smith et al. 2012) and a quantitative mass-balance model (Pinkerton et al. 2010a, 2016).

The Ross Sea trophic model describes food web structure in a typical year during the period 1990–2000 when fishing has not reduced the toothfish population (Mormede et al. 2015). Biomass and flows were modelled in terms of organic carbon density (gC m^{-2}) as a proxy for energy flow (Figure 17.8). The Ross Sea trophic model covers an area of 637 000 km^2, which includes the Ross Sea shelf and slope and includes 41 trophic groups. The modelling framework for the trophic model is a mass-balance similar to that of Ecopath (Christensen & Walters 2004, Christensen et al. 2008), but non-trophic transfers (including the release of material from sea ice to the water column and vertical detrital flux) were included. Detailed information on the estimation of the parameters is available online from the NIWA website.[10] Revisions and updates to the model are detailed by Pinkerton et al. (2016).

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[10] NIWA. Ross Sea Ecosystem and Trophic Model. Retrieved from https://www.niwa.co.nz/fisheries/research-projects/the-ross-sea-trophic-model/ross-sea-ecosystem-and-trophic-model.

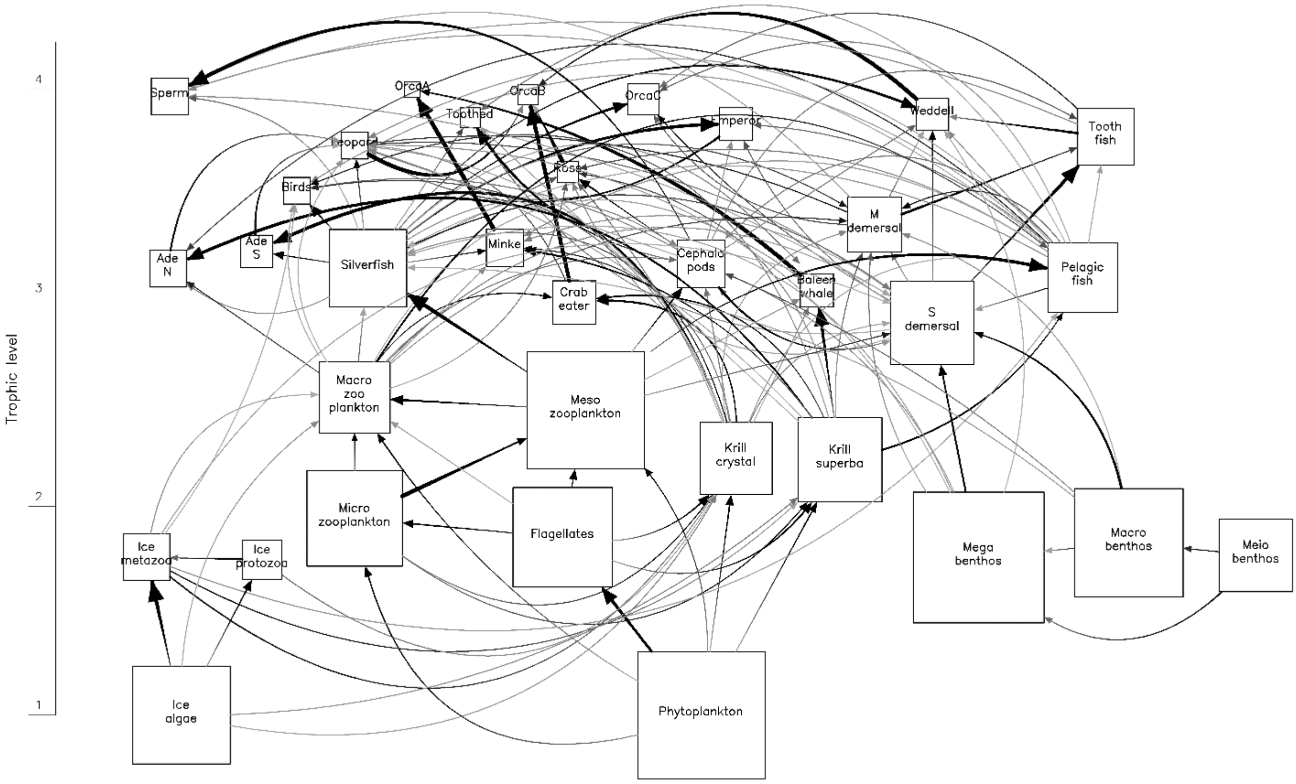

Figure 17.8: Ross Sea trophic model flow diagram, with arrows showing the direction of organic carbon flow. Bacterial and detrital groups omitted for clarity. Bigger boxes indicate more biomass. Boxes are positioned vertically according to trophic level. Thicker/darker lines show higher flows in or out of the group in proportion to total flows in or out of the group. Sperm = sperm whale; Orca-A = Type A killer whale; Orca-B = Type B killer whale; Orca-C = Type C killer whale; Toothed = other toothed whales; Minke = minke whale; Crabeater = crabeater seal; Weddell = Weddell seal; Leopard = leopard seal; Ross= Ross seal; Ade N = Adélie penguins from northern Ross Sea breeding colonies (north of and including Wood Bay); Ade S = Adélie penguins from southern Ross Sea breeding colonies; Birds = flying birds; M demersal = medium-sized (40–100 cm total length) demersal fish ; S demersal = small demersal fish (<40 cm total length). [Pinkerton et al. 2010a, 2016].

17.2.3 HISTORICAL HUMAN EFFECTS ON THE ROSS SEA ECOSYSTEM

The Ross Sea has been identified as the one of the ocean regions least affected by human activity (Halpern et al. 2008). Major industrial sealing did not affect the Ross Sea, although an estimated 2000 Weddell seals were killed in southern McMurdo Sound to supply dog food to early polar expeditions and permanent stations (Scott Base and McMurdo Station) (Stirling 1971, Ainley 2009).

Blue, fin, and sei whales were taken from the continental slope of the Ross Sea in the 1920s–70s but little whaling was carried out over the Ross Sea shelf itself (Ainley 2009). The removal of an estimated 9330 blue whales from the Ross Sea region (Ainley 2009) may have represented most of the local population of this species. Subsequent industrial whaling for minke whales during the 1970s–80s was largely confined to waters north, east, and west of the Ross Sea (Ainley 2009), and the minke whale population seems to have recovered after whaling ceased in the 1980s (Branch 2006). Catches of southern right whales (*Eubalaena australis*) and sperm whales in the Ross Sea region were also low and confined to waters north of the Ross Sea slope (Whitehead 2000, Ainley 2009). In the early 1980s whalers from the former Soviet Union killed more than 900 killer whales belonging to multiple types in one season (Pitman 2003), which represents a significant perturbation to a population estimated at about 3000 animals (Ainley 2009).

Before the advent of the toothfish fishery in 1997 there was no commercial fishing for finfish in the Ross Sea region.

17.3 ECOSYSTEM EFFECTS OF FISHING IN THE ANTARCTIC

17.3.1 INTRODUCTION

CCAMLR's approach to management recognises that species in an ecosystem are linked (Constable et al. 2000, Kock 2000). Target species are often important components of the ecosystem. Changing their abundance may substantially impact related and dependent species, and affect whole-system dynamics and resilience (Murawski 2000, ICES 2005). To develop management in the Ross Sea and Amundsen Sea regions consistent with CCAMLR's principles of conservation has required the management scope to extend beyond single-stock reference points (Hanchet et al. 2014).

Research and management of the ecosystem effects of fishing are more advanced in the Ross Sea region than in the Amundsen Sea region. The toothfish fishery in the Ross Sea has been operating for longer than in the Amundsen Sea, and the development of a stable stock assessment model in the former (since 2004) has allowed more focus on ecosystem effects of fishing there.

17.3.2 EFFECTS ON BYCATCH SPECIES

Seabird mortality

There are two potential impacts of the fishery on seabirds in the Ross Sea region: (1) direct mortality of flying birds from interaction with fishing gear; (2) indirect impacts on seabirds due to trophic effects (e.g., changes in availability of prey for seabirds – see Section 17.3.5). Extensive measures to mitigate the direct effects of fishing on seabirds in the Ross Sea have been in place since the initiation of the fishery (Reid et al. 2010; CMs 24-02, 25-02). These include the use of streamer lines, the use of weights or weighted lines to enable faster line sink rates, and no discharge of offal south of 60° S. Since the beginning of the fishery in 1997, only two seabirds have been reported as caught by fishing vessels.

Mammal mortality

There has also been no reported bycatch of marine mammals on longlines in the toothfish fisheries of the Ross Sea or Amundsen Sea regions.

Fish bycatch

A detailed characterisation of the bycatch in the toothfish fishery in the Ross Sea region was carried out by Stevenson et al. (2012). Moore & Parker (2021) provided an update on recent catches and data availability for the Ross Sea toothfish fishery. An updated characterisation of the main bycatch species in the fishery is planned for 2022. Fishery bycatch in the Amundsen Sea region has not yet been characterised in detail. The main bycatch species in the Ross Sea region are macrourids or grenadiers (*Macrourus whitsoni* and *M. caml*), icefish (mainly *Chionobathyscus dewitti*), skates (mainly *Amblyraja georgiana*), eel cods (*Muraenolepis* spp.) and deepsea (morid) cods (mainly *Antimora rostrata*). A small bycatch of rock cods and ice cods is also taken.

Spatial distributions of fish bycatch are shown in Figure 17.9.

The highest catches for macrourids, skates, eel cods, and icefish occur on the Ross Sea continental slope, near Iselin Bank. Since the implementation of the Ross Sea region Marine Protected Area, catches have become more concentrated in this area (Figure 17.9) (Moore & Parker 2021). Deepsea (morid) cods have a more northern range and relatively higher catches occur over seamounts in the Pacific-Antarctic Ridge. Rock cods and ice cods tend to occur at shallower depths, especially over the Ross Sea shelf and around the Balleny Islands.

Except for skates and rays, the main bycatch species in the toothfish fishery are also the main prey items for toothfish (Fenaughty et al. 2003, Stevens et al. 2014). One of the reasons is the paucity of other large teleost or squid prey in the Ross Sea region (Bradford-Grieve & Fenwick 2001, Smith et al. 2012). For macrourids and icefish, it is likely that the predation release effect (see Section 17.3.3; Soulé et al. 1988, Prugh et al. 2009) may be stronger than the direct effect of fishing mortality on these species.

Macrourids

The main bycatch species in the Ross Sea are macrourids, which form around 5% of the total catch by weight and about 30% of the total catch by number per year, with recent catches at around 120 t y^{-1} (Moore & Parker 2021). Macrourid bycatch in the Ross Sea region was considered to be almost exclusively *M. whitsoni* (Regan 1913) until samples collected on the IPY-CAML voyage in 2008 led to the identification of a new species, *M. caml* (Smith et al.

2011, McMillan et al. 2012). The relative proportion of *M. caml* to *M. whitsoni* in the catch has not been assessed, but observers have been gathering information on this since 2013. Preliminary results suggest *M. caml* and *M. whitsoni* are sympatric by depth (both appearing to be abundant between depths of 900 and 1900 m) (Pinkerton et al. 2013).

There are several approaches in place to mitigate macrourid bycatch, including Subarea and amalgamated SSRU limits on the amount of bycatch (CM 41-09, 41-10), introduced in 2002. Macrourid catch limits were initially based on analogy to fisheries in other areas of the Southern Ocean, but more recently limits have been based on the results of the IPY-CAML trawl survey of the Ross Sea slope. Bycatch limits for macrourids were exceeded in a number of SSRUs during the early period of the fishery but since 2007 the total macrourid catch has always been less than half of the macrourid catch limit. To help prevent localised depletion of macrourids, 'move-on' rules were introduced in the 2001/02 season (CM 33-03). These rules require a vessel to move to another location at least 5 n. miles distant if the bycatch of any one species is equal to or greater than 1 t in any one set. An additional measure in CM 33-03 makes vessels responsible for managing their individual macrourid bycatch by requiring a vessel to cease fishing in an SSRU for the remainder of the season if its macrourid catch exceeds 16% of its catch of *Dissostichus* spp.

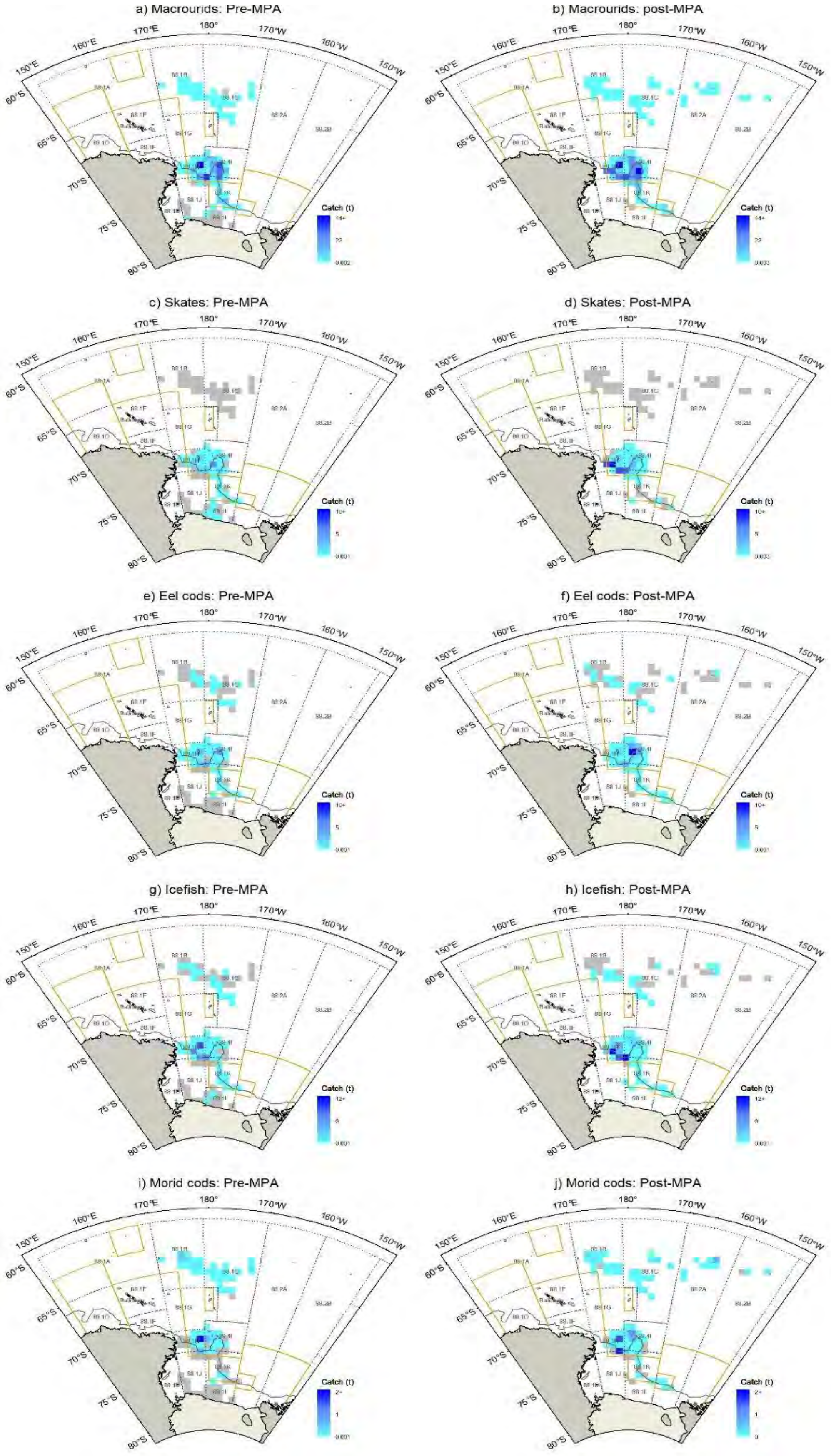

Figure 17.9: Densities of retained catches of macrourids (a, b), skates (c, d), eel cods (e, f), icefish (g, h), and morid cods (i, j) by weight before (2014–2017; left column) and after (2018–2021; right column) implementation of the Ross Sea region Marine Protected Area. From Moore & Parker (2021).

Managing the effects of fishing on macrourids has been applied to all species of macrourid combined. Although the two species of macrourid seem to occur in the same places and at the same depths, their longevities and ages at maturity differ; females of *M. whitsoni* only become sexually mature at 79% of L_{inf} (maximum length) as opposed to about 50%L_{inf} as is common for most macrourids including *M. caml* (Pinkerton et al. 2013). This may make *M. whitsoni* more vulnerable than *M. caml* to the effects of fishing (Reynolds et al. 2005). Although there remains the potential for fishing to affect the two species of macrourid differently in the Ross Sea region, perhaps necessitating species-specific management, information is not yet available to develop this.

Skates and rays (rajids)

Skates and rays (rajids) are the second highest group of bycatch species. Since the start of the fishery, skates have comprised less than 1% of the catch brought onboard (by weight) because most are cut off alive at the surface (Moore & Parker 2021). The main skate caught is the Antarctic starry skate (*Amblyraja georgiana*).

Rajids are required to be brought onboard or alongside the hauler to be checked for tags from historical tagging and for their condition to be assessed. All rajids which are caught alive and with 'a high probability for survival' are released alive at the surface; any 'dead or injured skates' are retained onboard (CM 33-03). The retained catch of rajids is very low (1 t y^{-1}) and has never exceeded the bycatch limit for rajids (Large et al. 2015).

In the Ross Sea region, the highest catch rates of the Antarctic starry skate are in 850–1350 m, whereas Eaton's skates (*Bathyraja eatonii*) are generally caught in 750–850 m. Catch rates are much lower than those of starry skates (Mormede & Dunn 2010). There have been some measurements of skate survival rates in longline fisheries in the Southern Ocean (Endicott & Agnew 2004), but little data on survival of skates caught shallower than 1200 m. Skate survivorship experiments in South Georgia (Subarea 48.3) show that some skates (n=95 fish) survive the capture event, at least for 12 hrs following capture, and that survival rates are higher at shallower depths (Endicott & Agnew 2004). There is also a move-on rule in place to help prevent localised depletion of rajids (CM 33-03). Potential methods for monitoring skates in the Ross Sea region were reviewed by O'Driscoll et al. (2005), who concluded that a tag-recapture experiment was likely to be most successful for monitoring skates.

A preliminary stock assessment based on skate tag-recapture data and ancillary fishery data was completed by Dunn et al. (2007). They identified several problems with the data currently being collected and made the following recommendations: improve species identification, improve detection of tagged skates, increase number of skates measured and sexed, validate the estimates of age and growth, revise skate tagging protocols, and undertake additional survivorship experiments. Following the CCAMLR 'Year of the Skate' in 2008–09, an updated characterisation of skate catches was carried out by Mormede & Dunn (2010). They noted that, up to and including the 2010 season, a total of 14 000 skates had been tagged and released and a total of 179 skates had been recaptured. The return rates for tagged skates in CCAMLR fisheries is typically lower than from tagging programmes elsewhere in the world and the reasons for this are unclear (McCully et al. 2013).

A 2-year programme was implemented in the 2019/20 and 2020/21 fishing seasons in the Ross Sea region to tag and release skates for population size estimation and to validate the thorn ageing method for Antarctic starry skate. During this programme a total of 8 506 skates were tagged and released in the Ross Sea region, with a further 484 individual skates voluntarily tagged in the Amundsen Sea region. Preliminary findings suggest skates do not move great distances, although results are limited by both the small number of recaptures (n=44 from the programme) and limited time at liberty of tagged individuals (Moore et al. 2021).

The medium-term research plan (Delegations of New Zealand, Norway and the United Kingdom 2014; Section 17.3.9) has identified that further analysis is needed to understand the effect of fishing on rajids in the Ross Sea region.

Icefish

Icefish are caught in low numbers with bottom longline and trawl gears throughout the Southern Ocean (CCAMLR 2014). In the Ross Sea region, the bycatch of icefish in the toothfish fishery is typically less than 10 t y^{-1}, with a peak of about 25 t reported in 2020 (Moore & Parker 2021). Since 2014, a total of 8 species codes have been used for icefish caught in the Ross Sea fishery (Moore & Parker 2021). Although a large amount of catch is reported as ‘unspecified icefish’, the most common species in the catch is *C. dewitti* (Sutton et al. 2008, Moore & Parker 2021).

In the Ross Sea region, *C. dewitti* becomes sexually mature aged about four years and the oldest fish aged was 12 years old from a sample size of 296 fish (Sutton et al. 2008). Icefish are a major prey of toothfish, comprising 20–25% by weight of the prey of sub-adult and adult toothfish on the Ross Sea slope. Icefish are less common over the northern seamounts in the Ross Sea region where they comprise less than 5% by weight of the diet of toothfish.

Eel cods

Muraenolepididae (eel cods) occur over the continental shelf and slope of cold temperate and Antarctic southern hemisphere (Nelson 2006). They are caught in low numbers with bottom longline and trawl gears throughout the Ross Sea region (Parker et al. 2012). On northern Iselin Bank where the median catch rate in the Ross Sea region is highest, catches are less than 0.01 kg per hook. Overall catches in Subarea 88.1 have been stable throughout the fishery at typically less than 10 t y^{-1}, with a peak catch of about 20 t in 2007 when 19 sets reported greater than 100 kg of *Muraenolepis* spp. (Parker et al. 2012, Moore & Parker 2021).

Morphological identification of eel cod species continues to be difficult and previous identifications of *Muraenolepis microps* from the Ross Sea region are now considered incorrect (Parker et al. 2012). Thirteen *Muraenolepis* specimens captured in the Ross Sea region during the 2008 IPY/CAML voyage were identified, based on morphology, as *M. evseenkoi* (identification by Te Papa Tongarewa Museum of New Zealand).

Genetic methods appear to be more effective than morphology at identifying eel cod species and are increasingly used (Fitzcharles 2014, Fitzcharles et al. 2021). Genetic identification of more than a hundred specimens indicates that eel cods on the Ross Sea slope are exclusively *M. evseenkoi* (Fitzcharles 2014). Eel cods caught over the Pacific-Antarctic fracture zone in the north of the Ross Sea region were identified genetically as predominantly *M. evseenkoi* with a single specimen of *M. microcephalus* (Fitzcharles 2014).

The biological studies published on species in the genus *Muraenolepis* suggest a relatively fast growing, semelparous[11] life history with a maximum age of 11 years (Parker et al. 2012). In the Ross Sea, eel cods selected by longline gear are almost exclusively female, and a localised area of high catch rates occurs on Iselin Bank on Ross Sea slope. Eel cods comprise a total of about 11% by weight of prey of sub-adult toothfish and about 14% by weight of prey of adult toothfish on the Ross Sea slope (Stevens et al. 2014).

Fishing is likely to affect eel cods in the Ross Sea region by a combination of predation release (fewer toothfish consuming eels cods) and fishing mortality (increased overall mortality), which act in opposition. The overall effects of fishing on this bycatch species depend on factors such as the distribution pattern and total biomass of eel cods, as well as their productivity. Further directed sampling to determine species composition, life-history attributes, reproductive strategy, sex-specific distribution, and any trends in biomass is needed from the Ross Sea area and throughout the CCAMLR Convention Area (Parker et al. 2012).

Deepsea (morid) cods

Catches of deepsea (morid) cods are dominated by *Antimora rostrata*. This species has a wide spatial distribution, north to the New Zealand EEZ where it is called ‘violet cod’ or ‘blue antimora’. The stock structure of this species is unknown. The species forms less than 2% of the diet of toothfish on the Ross Sea slope, but about 20% by weight of diet over the northern (seamount) region of the Ross Sea (Stevens et al. 2014).

Rock cods and ice cods

Rock cods and ice cods (Nototheniidae) comprised less than 0.01% of the total retained catch between 1998 and 2020 (Moore & Parker 2021).

The highest catch rates for rock cods occur in a narrow depth band of 400–600 m (Stevenson et al. 2012).

Four different codes have been used to record rock cod and ice cod catches in the Ross Sea region and it is likely that different species dominate this group in different SSRUs. In SSRUs 88.1E and 88.1G, the highest mean catch rates are likely to mainly comprise the striped rock cod

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[11] ‘Semelparous’ means the adults breed once in their life then die.

(*Lepidonotothen kempi*); this species was the most abundant species caught in research trawls and observed on videos during the BioRoss and IPY-CAML biodiversity surveys (Clark et al. 2010). Catches on the Ross Sea shelf are likely to mainly comprise the deepwater notothen (*Trematomus loenbergii*); this species was the most commonly caught species in the sub-adult toothfish survey over the southern Ross Sea shelf (Hanchet et al. 2012).

17.3.3 EFFECTS ON PREY SPECIES

Fishing can reduce predation on prey species by removing parts of the predator population (i.e., the target species for fishing operations). This can lead to mesopredator (or predation) release (Soulé et al. 1988, Prugh et al. 2009).

Empirical meta-analysis suggests that predation release tends to be weaker in pelagic marine and terrestrial systems than in benthic marine and freshwater systems (Shurin et al. 2002). Predation release tends to be stronger where the predator is large and mobile, has high metabolic rate, where prey species are long-lived, functional predator diversity is low, and predator intraguild predation is weak or absent (Borer et al. 2005, Heithaus et al. 2008).

Many of these factors are present in the Ross Sea. On the Ross Sea continental slope, where the majority of the regional Antarctic toothfish population feeds (Hanchet et al. 2008), toothfish are likely to be by far the major predators of macrourids, icefish, and eel-cods (Pinkerton et al. 2010a, Stevens et al. 2014, Pinkerton & Bradford-Grieve 2014). There are no other piscine predators of the size of Antarctic toothfish over the Ross Sea shelf and slope (Smith et al. 2012). Some prey species of toothfish have relatively high longevities and low productivity rates. Macrourids tend to be long-lived (Bergstad 1995, Kelly et al. 1997) and, in the Ross Sea region, otolith ageing found maximum recorded ages of 27 years for *M. whitsoni* (n=227) and 62 years for *M. caml* (n=319) (Pinkerton et al. 2013). In contrast, *C. dewitti* and *M. evseenkoi* are faster-growing and shorter-lived species, with maximum recorded ages of around 11 years (Sutton et al. 2008, Parker et al. 2012). One mitigating factor against strong top-down changes to prey species is the relatively low consumption rate of toothfish, which is likely to be only one to two times its body mass per year because of its large size and the cold water (Pinkerton et al. 2010a).

Models of specific cases of predation release in marine systems are few (Prugh et al. 2009) partly because reliable information on marine predators is often scarce (Heupel et al. 2014). A number of approaches have been used to investigate ecological interactions in marine systems including full-ecosystem models (Plagányi 2007, Rose et al. 2010) and mixed-trophic impact analysis (Ulanowicz & Puccia 1990).

Mixed trophic impact analysis was applied to the Ross Sea trophic model (Pinkerton et al. 2010a, Pinkerton & Bradford-Grieve 2014) and suggested a strong trophic connection between toothfish and medium-sized demersal fish (mainly macrourids and icefish). In the Ross Sea trophic model, toothfish consumed 64% of the annual production of medium-sized demersal fish. This led to the strongest, top-down impact in the whole multiple-step analysis of Pinkerton & Bradford-Grieve (2014) who concluded that at least some piscine prey of toothfish will experience a relatively strong predation-release effect as the abundance of toothfish is reduced by fishing.

Such 'whole system' approaches tend not to consider interactions over small spatial scales or affecting only parts of populations, and their ability to reliably represent the dynamics of whole ecosystems remains limited (Beckage et al. 2011, Planque 2015). Modelling predation release within a key subset of the whole marine system may be more robust and hence more useful for fisheries management (Plagányi 2007, Plagányi et al. 2014).

To explore the potential effects of the toothfish fishery on these medium-sized demersal fish, a minimum realistic model (MRM) of Antarctic toothfish, macrourids, and icefish was developed (Mormede et al. 2014d). This was spatially explicit and dynamic, and based on a model of predator-prey interactions for the Ross Sea Region. The MRM included age-based population dynamics of toothfish, macrourids, and icefish, and included natural mortality, predation mortality, and fishing mortality on all three species. The MRM suggested that the predation release caused by the fishery effect on toothfish abundance was greater than the direct fishing mortality on both prey species and that icefish were expected to show a larger increase in biomass through time than macrourids (Mormede et al. 2014d). This may affect the proportions of macrourids and icefish in the diet of toothfish over time (Mormede et al. 2014d).

17.3.4 EFFECTS ON PREDATOR SPECIES

Three species are known to prey on toothfish in the Ross Sea region: Weddell seals, Type C killer whales, and sperm whales. Other species discussed below may also consume toothfish. In assessing the potential consequences of fishing to its predators of toothfish, two factors are important:

1. To what extent is the predator population *ecologically dependent* on toothfish as a prey item? This includes aspects such as the proportion of toothfish in the predator's diet and whether alternative prey items are available (and at what additional ecological cost to the predator). Also relevant is whether toothfish is especially important as prey at a particular time of year, in a particular area, or to a particular part of the predator population.
2. To what extent will the fishery reduce the availability of toothfish to the predators at ecologically relevant scales – i.e., taking into account temporal, spatial, and population factors?

Weddell seal

There remains uncertainty over the degree to which Weddell seals are ecologically dependent on toothfish as prey (Pinkerton et al. 2008, Eisert et al. 2013). Nutritional analysis of Ross Sea prey suggests that toothfish may represent a unique high-energy food resource for Weddell seals that may not be replaceable by other prey, in particular during periods of high energy demand such as late-stage lactation and the post-breeding recovery of body weight and condition for adult females (Eisert et al. 2013).

Changes to toothfish availability near Weddell seal breeding colonies in the period between pupping and weaning could affect survival of Weddell seal pups and lactating mothers, and fertility rates in the following season, and hence have a compounding impact on Weddell seal populations in these areas (e.g., Pinkerton et al. 2008, Eisert et al. 2013).

Eisert et al. (2013) recommended that the assumed dominance of Antarctic silverfish in Weddell seal diets should be re-examined given the known biases of methods used to derive diet estimates; although large (over 30 g) silverfish occurring at high densities are likely to be a valuable nutritional resource to Weddell seals, smaller size classes of silverfish are unlikely to be adequate to meet the estimated energy requirements of adult Weddell seals.

Killer whale

Killer whales are considered to constitute a single species throughout the world (Rice 1998) but there are at least four different forms (or 'ecotypes') of killer whale in the Antarctic (Pitman & Ensor 2003). The Ross Sea (or 'Type C') killer whale ecotype is believed to feed almost entirely on fish. There is strong circumstantial evidence that toothfish are an important prey item for Type C killer whales in the Ross Sea region (Torres et al. 2013, Eisert et al. 2013, 2014).

The evidence includes:

(1) Killer whale population ecology includes high consumption rates, low abundances, low production rates, often specialised diets, and unknown potential for foraging innovation.

(2) Type C killer whales near McMurdo Sound have been commonly observed carrying toothfish in their mouths (Eisert et al. 2013, 2014).

(3) Comparison of the relative nutrient density of toothfish with silverfish and other prey shows that toothfish represent a high-energy food resource of much higher quality than other potential prey in the Ross Sea region (Eisert et al. 2014). While equivalent energy-dense non-fish prey is available in the Ross Sea (e.g., penguins or seals), observations in northern hemisphere killer whale populations suggest that switching from fish to endotherm (warm blooded) prey is unlikely (Barrett-Lennard et al. 1996, Barrett-Lennard 2011).

(4) An important recent finding is the high incidence of suckling calves observed in Type C killer whale groups in McMurdo Sound (Eisert et al. 2014). Caring for young (less than six months old) calves greatly increases the energy requirement of lactating females, not only for milk production, but also because mothers assist their calves through drafting, which increases their own locomotory costs. Revised estimates of energy requirements indicate that lactating female killer whales of the fish-eating ecotype require toothfish to meet their elevated demand.

(5) Densities of other alternative potential prey (Antarctic silverfish, cryopelagic fish) seem too low to justify killer whales coming to the Ross Sea for feeding and the development of a fish-eating ecotype (Eisert et al. 2014).

However, other information on the potential feeding by killer whales on toothfish was inconclusive:

(1) It is not known to what extent toothfish forage pelagically or how deep type C killer whales can dive. Type C killer whales in the Ross Sea can routinely dive to 200–400 m, with a maximum of over 700 m (Torres et al. 2013, Eisert et al. 2015). This is deep enough to reach demersal prey over much of the Ross Sea shelf, but foraging times would be short at these depths.

(2) Stable isotope analysis of tissue from killer whales and toothfish was inconclusive, due to the unknown rate of skin turnover and the stable isotope gradient across the killer whale known range (Krahn et al. 2008, Graham & Bury 2019).

The balance of evidence suggests that toothfish are likely to form a significant part of the diet of Type C killer whales in McMurdo Sound in summer, but it is not possible to say whether toothfish are an important prey item to Type C killer whales in other locations on the Ross Sea shelf (e.g., Terra Nova Bay, Bay of Whales, Sulzberger Bay) or at the scale of the whole Ross Sea shelf and slope (Torres et al. 2013, Eisert et al. 2014).

Evidence derived from limited satellite tagging and photo-identification shows that Type C killer whales undergo long-distance travel from the southern Ross Sea to New Zealand waters and into subtropical regions (Figure 17.10; Eisert et al. 2015).

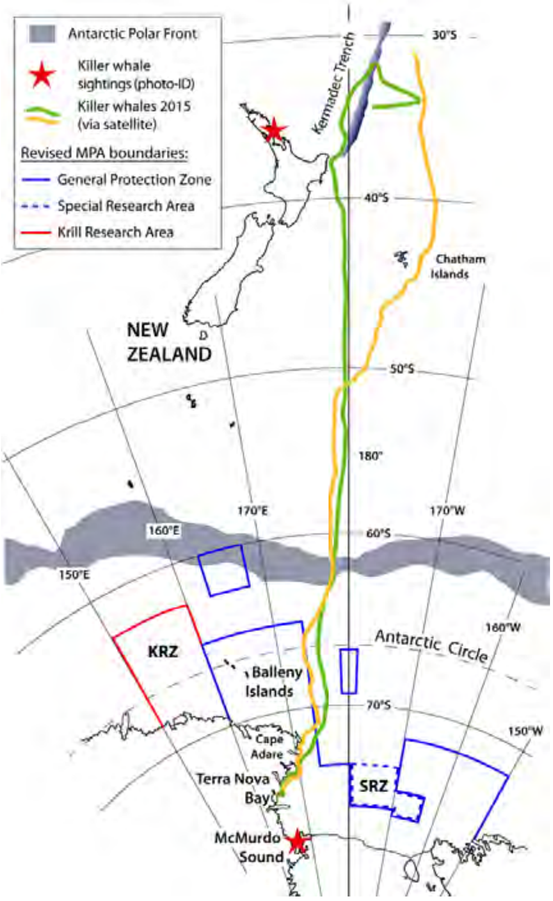

Figure 17.10: Satellite tracking of Type C killer whales showing ecological connectivity for this species between the Ross Sea and New Zealand (Eisert et al. 2015).

Analysis of photo-ID data indicates that Type C killer whales from the Terra Nova Bay area of the Ross Sea show a high degree of seasonal site fidelity. Individual whales returned over different years to areas of ecological significance, including New Zealand waters north and east of East Cape, the Kermadec Trench region, and the Ross Sea (Eisert et al. 2015).

Sperm whale

Sperm whales are migratory and are distributed from the tropics to the pack ice edges in both hemispheres. The subtropical convergence at about 40° S marks the southern limit of females and young males; only the larger males penetrate further south (Lockyer & Brown 1981, Knox 2007). Sperm whales are the largest toothed whale and sexually dimorphic (males reach 16 m and 45 t, females 11 m and 15 t; Whitehead 2018).

Present and historical occurrence of sperm whales along the Ross Sea continental slope remains unclear. Kasamatsu & Joyce (1995) reported a southernmost sighting of a sperm whale at 74° S on the Ross Sea slope and summarised data collected in sighting surveys between 1976–77 and 1987–88 during mid-December to mid-February. The IWC data from the 1990s showed sperm whale sightings in the area defined by 70–78° S, 150–180° E (along the Ross Sea shelf edge). However, sperm whales have rarely been sighted on the Ross Sea slope from fishing vessels (Fenaughty, pers. comm.). We are not aware of any systematic surveys of sperm whales in the Ross Sea or Amundsen Sea regions in the last 30 years.

Sperm whales are deep divers. They typically stay submerged for about 45 min (up to two hours) and commonly dive to about 600 m, but may be able to reach depths of >2000 m (Clarke 1976, Watkins et al. 1993, Perrin et al. 2002, Whitehead 2018). Sperm whales in the Southern Ocean and Pacific subantarctic waters are reported as feeding primarily on squid and secondarily on fish (Clarke 1980, Knox 2007, Evans & Hindell 2004). Knox (2007) gives the ratio of squid to fish in their diet as 9:1.

Yukhov (1971, 1972) and Abe & Iwami (1989) described Antarctic toothfish as prey items but proportions are not known. Yukhov (1971, 1972) examined large numbers of stomachs from 12–18 m long (i.e., subadult and adult) sperm whales from the Pacific Ocean sector of the Antarctic from 1965 to 1969 and found that the main prey were cephalopods but that Antarctic toothfish (97–160 cm total length) were also frequently found in the sperm whale stomachs. Although some records were associated with seamounts and ridges, many occurred over deep water (more than 4000 m) suggesting that the sperm whales were feeding pelagically (Yukhov 1972).

Other potential predators of toothfish

Other possible predators of toothfish in the Ross Sea region include southern elephant seals, Arnoux's beaked whales, and colossal squid.

Southern elephant seals (*Mirounga leonina*) enter the Ross Sea only in the summer from breeding and feeding grounds further to the north. They are likely to be mainly feeding on small pelagic fish, squid, and crustaceans (Walters et al. 2014). However, their deep diving capability (about 1500 m) and occurrence around the Ross Sea slope, as well as photographic evidence from the Antarctic Peninsula region (Eisert and Smellie, pers. comm.) suggests that elephant seals may consume Antarctic toothfish.

Very little is known about the predation of Arnoux's beaked whales (*Berardius arnuxii*) on toothfish but this is unlikely to be significant. These whales are known to occur in the Ross Sea to 77° S (Eisert, pers. comm.), are capable of diving to depths where toothfish occur on the Ross Sea slope, and are predominantly small fish and squid eaters (Walker et al. 2002, Ohizumi et al. 2003).

Beak-shaped bite marks on toothfish caught on longlines suggest some depredation on toothfish by colossal squid (*Mesonychoteuthis hamiltoni*). However, stable isotope analysis of tissue of this species of squid suggests that it is unlikely to feed on toothfish.

Effects of fishing on availability of toothfish as prey

There are four ways in which the fishery could alter the availability of toothfish for predators in the Ross Sea region:

(1) *Smaller stock size*. Fishing leads to fewer fish available as prey.

(2) *Local depletion by fishing within a season*. Fishing may locally reduce toothfish abundance (catch rates typically decline when an area is fished). If fishing occurs in an area where predators forage, the availability of toothfish to predators may be reduced for some time. In 2008, CCAMLR set a zero allowable catch for SSRU 88.1M (along the Victoria Land coast), which had the effect of moving fishing effort away from the known foraging grounds of Weddell seals and Type C killer whales in the south-west Ross Sea.

(3) *Reduced recruitment*. The number of sub-adult toothfish available in the southwest Ross Sea could decline if there was reduced toothfish recruitment. The stock assessment suggests that toothfish spawning biomass in 2021 was about 62.7% B_0 (Grüss et al. 2021c). At this level, recruitment of toothfish is not estimated to be reduced. Based on the stock-recruit relationship with steepness assumed at 0.75 in the stock assessment (Mormede et al. 2014a), recruitment is predicted to be reduced to about 92% of unfished recruitment when the spawning stock biomass reaches 50% of its unfished status.

(4) *Density-dependent* or *stock-contraction effects*. As has been seen in some other species elsewhere (Swain & Sinclair 1993, Hutchings 1996, Atkinson et al. 1997, Fisher & Frank 2004), fishing may change movement patterns and distribution of toothfish throughout the Ross Sea region. Changes in the distribution of toothfish that affect

abundance at the edges of the toothfish range may be important to their predators.

17.3.5 TROPHIC AND SYSTEM-LEVEL EFFECTS

Changes in the abundance of one species may impact other species that are neither its predators nor its prey. These are called 'second order' trophic effect, or ecosystem-level effects and can include trophic cascades and regime shifts. Well-documented, oceanographic-induced regime shifts in marine ecosystems have historically had substantial, long-lasting, and typically (but not always) negative effects on fisheries. A review of trophic and ecosystem level effects of fishing is given in Chapter 13: Trophic and ecosystem-level effects.

Trophic effects arising from fishing are more likely to be important if the target species has a key role or is of high trophic importance in the ecosystem (Fletcher et al. 2002, Fletcher 2005). An estimate of trophic importance, using mixed trophic impact analysis (Ulanowicz & Puccia 1990) was applied to the Ross Sea (Pinkerton & Bradford-Grieve 2014) based on the Ross Sea trophic model (Section 17.2.2). This concluded that Antarctic toothfish has moderate trophic importance in the Ross Sea food web as a whole. The analysis did not support the hypothesis that changes to toothfish abundances due to fishing will cascade through the Ross Sea regional ecosystem by simple trophic effects. Pinkerton & Bradford-Grieve (2014) did not rule out cascading effects on the Ross Sea ecosystem due to changes in the abundance of toothfish, but noted that for such changes to occur, a mechanism other than simple trophic interactions would need to be involved. Instead, Pinkerton & Bradford-Grieve (2014) found that trophic importance was highest in the middle-trophic level organisms of the Ross Sea food web. Antarctic silverfish, krill, small demersal and pelagic fishes, cephalopods, and mesozooplankton were identified as having key roles in maintaining ecosystem resilience.

17.3.6 EFFECTS ON HABITATS

Vulnerable Marine Ecosystems (VMEs) constitute areas that may be vulnerable to impacts from fishing activities. Taxa considered to comprise VMEs vary geographically. Essentially, VMEs are ecosystems with organisms that create biogenic structures, are fragile relative to the fishing gears in question, are rare or endemic, or have life-history traits that imply slow recovery from disturbance (Rogers et al. 2008, FAO 2008).

In 2007 CCAMLR adopted Conservation Measure (CM) 22-06 requiring Member countries to assess and manage adverse effects of bottom fishing on VMEs in the Convention Area. The *New Zealand Antarctic Bottom Fishing Impact Assessment Workshop* in 2007 identified 14 groups of taxa indicative of habitats or communities where VME organisms occur (Parker et al. 2008). A CCAMLR guide to VME taxa was produced in 2009.[12]

All fishing for toothfish in the Ross Sea and Amundsen Sea regions is by longline, which are laid on or close to the seabed and held down by weights and grapples (Fenaughty 2008). Structure-forming benthic invertebrates can be damaged by the longlines, especially during their hauling (recovery from depth) when the longlines may move laterally. Benthic invertebrates that have been brought to the surface attached to lines in the Ross Sea region include anemones (Actiniaria), stony corals (Scleractinia), gorgonians (Gorgonacea), sponges (Porifera), and ascidians (Ascidiacea) (Parker & Bowden 2009).

The potential for the longlines to significantly affect a particular group of structure-forming benthic habitat in the Ross Sea is related to the spatial scale of the area of contact between fishing gear and the seafloor as a proportion of the total area in which the habitat is present. An impact assessment method developed by Sharp et al. (2009) showed that regardless of the distribution of VME taxa (for which actual spatial distributions are unknown) the cumulative impact on VME organisms of all historical longline fishing effort in the Ross Sea region has been very low. At a very fine scale (i.e., spatial cells measuring 0.05° latitude by 0.167° longitude) fewer than 5% of cells within fishable depths have been fished. Average impacts within fished cells are less than 0.1% total mortality of vulnerable taxa; estimated impact in the single most heavily impacted cell is less than 5% (Sharp 2010). These low impacts reflect both the spatially restricted area within which the fishery operates and the very narrow spatial footprint of individual longlines.

A spatially explicit production model was developed and used to simulate likely population level effects (including

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[12] CCAMLR. VME Taxa Classification Guide 2009. Retrieved from https://www.ccamlr.org/en/system/files/VME-guide.pdf.

recovery) arising from benthic impacts from longline fishing effort in the Ross Sea region (Dunn et al. 2010). Simulations included different productivity assumptions, impact, and spatial scale, with and without management by areal closures. The results of the simulations suggested that management action of areal closures in the Ross Sea region would improve the outcome for VMEs, but that the improvement was very small, given the already low level of impact.

Research has not found significant correlation between the occurrence of VMEs and toothfish abundance within areas fished for toothfish in the Ross Sea region (Parker & Mormede 2009, Parker et al. 2010, Parker & Smith 2010). Dunn et al. (2010) recommended further work on simulating effects of fishing on VMEs, including investigating how changes in the distribution of future fishing may result in alternative impacts or how different assumptions of the underlying distributions of benthic organisms may influence the results.

17.3.7 SYNERGISTIC EFFECTS OF FISHING AND CLIMATE CHANGE

There is increasing understanding of the potential impacts of climate change on fisheries (Valdes et al. 2009, Rice & Garcia 2011, IPCC 2014). Fishing can also act synergistically with climate variation/change and lead to ecosystem-level change (e.g., Winder & Schindler 2004, Brierley & Kingsford 2009, Kirby et al. 2009, Perry et al. 2010; see also Chapter 13).

If change to the level of toothfish recruitment in the Ross Sea or Amundsen Sea regions did occur (for example due to effects of fishing and climate change), would the current monitoring and management framework be able to detect this, and after how long?

Changes in age structure caused by changes in recruitment strength would most likely be detected from the fisheries catch data, but would not be apparent until the relevant cohort was of sufficient age to be fully selected by the fishery. Even then, the signal may be confounded with changing effort patterns. Without specific monitoring of sub-adult toothfish in the Ross Sea any substantial change in recruitment would not likely be detected until some years after it occurs.

This delay in detecting any effect of the fishery on recruitment was one reason for the start of the sub-adult survey for Antarctic toothfish over the southern Ross Sea shelf in 2012 (Hanchet et al. 2012). There have been six surveys to date (Hanchet et al. 2012, Parker et al. 2013b, Mormede et al. 2014c, Hanchet et al. 2015b, Dunn et al. 2016, Large et al. 2017). These surveys use a consistent, stratified design for sampling sub-adult toothfish to better estimate recruitment variability and provide an early-warning of changes in toothfish recruitment. It is likely that this survey, if continued on the same basis as at present, would detect changes to recruitment about five years after it occurred. In contrast, in the absence of a fishery-independent survey, the relevant cohort would not be available to the commercial fishery for approximately 10 years, and it is possible that any recruitment signal in the fishery-dependent data would be confounded by the effects of variable or uncontrolled commercial fishery selectivity. The survey hence reduces risk of changes in recruitment of toothfish in the Ross Sea region being detected too late for management to respond.

17.3.8 REVERSIBILITY OF ECOSYSTEM EFFECTS OF FISHING

Principle (b) of Article II of the CAMLR Convention requires the “maintenance of ecological relationships” in the ecosystem. Principle (c) of Article II of the CAMLR Convention states also that changes due to fishing should be “reversible over two or three decades”.

It has been suggested that trophic interactions can affect the ability of fish populations (and by extension, related or dependent species in the ecosystem) to regain their former characteristics following exploitation (Hutchings 2000, Steele & Schumacher 2000). Predictions as to the reversibility of ecosystem effects of fishing would be limited by three key factors. First, there is presently no information with which to estimate density-dependent effects of changes to toothfish (Abrams 2014). Second, the reversibility of different types of ecosystem effects of fishing will vary, so any theoretical investigations of reversibility will need to be carried out for each effect of fishing separately. Third, it is not known whether trophic and ecosystem-level effects or genetic or behavioural factors may come into play should fishing for toothfish cease. At present, there is very limited scientific ability to predict the dynamics of ecosystems (Planque 2015).

Keith & Hutchings (2012) concluded that “emergent and demographic Allee [density-dependent] effects, coupled with altered interspecific interactions, render questionable the presumption that the recovery of heavily depleted populations can be reliably forecasted by population dynamical behaviour during the decline.” However, in this context, ‘heavily depleted’ means depleted to much lower levels than the CCAMLR target of 50% B_0 so issues of reversibility are likely to be relevant only in the case of *significant overdepletion* of toothfish or arising from *substantial ecosystem effects* in other parts of the system. The focus in CCAMLR and in the Ross Sea and Amundsen Sea regions has hence been on preventing significant overdepletion of target species and on developing indicators for changes in dependent or related species (e.g., CEMP 2004, Delegations of New Zealand, Norway and the United Kingdom 2014). At present, evidence does not suggest that significant overdepletion of target species or substantial ecosystem effects are occurring in the Ross Sea and Amundsen Sea regions.

17.3.9 RESEARCH PRIORITIES

The medium-term (5–10 year) research priorities for the Antarctic toothfish fishery in the Ross Sea and Amundsen Sea regions were updated in 2014 (Delegations of New Zealand, Norway and the United Kingdom 2014). The first two sections of the medium-term research plan (MTRP, Table 17.1) prioritised research to assess, monitor and maintain the reproductive potential of the toothfish population. The third section dealt with issues related to the ecosystem effects of fishing, including reversibility of any effects of fishing.

Table 17.1: Medium-term research plan (MTRP) priorities with regard to the ecosystem effects of the fishery for toothfish in the Ross Sea and Amundsen Sea regions (Delegations of New Zealand, Norway and the United Kingdom 2014). The other two parts of the MTR plan are not shown. These are to (1) reduce uncertainty in toothfish model parameters; and (2) reduce management uncertainty.

Section	Key research priorities	
Maintenance of ecosystem structure and function	(i)	To determine the temporal and spatial extent of the overlap in the distribution of toothfish and its key predators (in particular killer whales and Weddell seals).
	(ii)	To investigate the abundance, foraging ecology, habitat use, functional importance and resilience of key toothfish predators (in particular killer whales and Weddell seals).
	(iii)	To develop methods of monitoring changes in relative abundance of key prey/bycatch species (in particular macrourids and icefish) on the Ross Sea slope and hence assess the potential impact of the toothfish fishery on these species.
	(iv)	To monitor diet of toothfish in key areas, especially on the Ross Sea slope.
	(v)	To simulate the effect of the fishery on populations of toothfish, its predators, and its prey (using Minimum Realistic Models or similar).
	(vi)	To develop quantitative and testable hypotheses as to the ‘second-order’ effects (such as trophic cascades, regime shift) and ensure data collection is adequate to monitor for any risks deemed reasonable.
	(vii)	To assess the impact of the toothfish fishery on Patagonian toothfish.
	(viii)	To estimate survivorship of released skates.
	(ix)	To develop semi-quantitative and spatially explicit risk assessments for macrourids and Antarctic skates (*A. georgiana*), especially in the slope fishery of the Ross Sea.
	(x)	To develop methods to assess whether the potential impacts of the toothfish fishery on the ecosystem are likely to be reversible in two to three decades.

The research priorities for the ecosystem effects of fishing were:

1. Further analysis is needed to understand the effect of fishing on rajids in the Ross Sea region.

2. To improve our understanding of the effect of fish on the prey assemblage of toothfish, especially in the most heavily-fished area of the Ross Sea slope, further information on the two species of macrourid separately is needed. In particular, information is needed on the relative abundances of *M. whitsoni*, *M. caml*, the relative catch of the two species across the Ross Sea and Amundsen Sea regions and the relative amount of the two species consumed by toothfish. Some of this research is underway. For example, the RV *Tangaroa* voyages to the Ross Sea in February 2015 and in January-February in 2019 included depth-stratified demersal trawl surveys of the Iselin Bank (SSRU 88.1I) and the results are being analysed. New Zealand observers have been identifying some of the macrourid bycatch in the Ross Sea region to species level since 2012 (i.e., separating *M. caml* from *M. whitsoni*), and macrourid prey found in the stomachs of toothfish during diet analysis will be identified to species level.

3. The minimum realistic model of interactions between toothfish and key prey species (especially macrourids and icefish) should be further developed. This modelling

enables the potential impacts of the fishery on key prey species to be evaluated to generate hypotheses of future change and to design monitoring tools for ecosystem effects.

4. Ongoing monitoring of toothfish diet is recommended, as is the monitoring of the icefish and macrourid populations (especially in SSRUs 88.1H and 88.1K) through the development of age frequencies (length measurements and ageing) (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014d).

5. Our ability to determine to what extent Weddell seal, Type C killer whale, and sperm whale populations in the Ross Sea and Amundsen Sea regions are ecologically dependent on toothfish requires further information on their diet and improved information on their seasonal and spatial abundances.

17.3.10 MARINE PROTECTED AREAS

The CAMLR Convention provides the overarching basis for marine resource conservation in the Southern Ocean. It includes a role for marine protected areas (MPAs). The CCAMLR position on MPAs is given online.[13] A decision was made at the World Summit on Sustainable Development (WSSD) in 2002 in Johannesburg, South Africa, to achieve a representative network of MPAs by 2012. CCAMLR responded to the WSSD target by aiming to establish a representative network of MPAs in the CAMLR Convention Area by 2012.

Globally, spatial fishing closures have been proposed as one way that fisheries management can manage, avoid, or mitigate the risk of ecosystem effects of fishing.[14] Although there are different types, in general, an MPA is a kind of spatial fisheries management that provides protection for all or part of the natural resources it contains. MPAs do not necessarily exclude fishing, research, or other human activities. MPAs in which no fishing is allowed are often referred to as 'no-take areas'. Other uses may still be permitted.

The Ross Sea region MPA was approved by CCAMLR at its Commission meeting in October 2016. Conservation Measure 91-05 (2016) (CM 91-05)[15] details the specificities of the MPA. The boundaries of the MPA can be found in the CM (Figure 17.11). It has an area of 1.55 million km^2. It came into force in December 2017 and the period of designation is 35 years. The full chronology and scientific basis for the design and designation of the Ross Sea region MPA by CCAMLR is summarised in Delegations of New Zealand and the United States of America (2014).

The MPA limits activities inside its boundaries to meet conservation, habitat protection, ecosystem monitoring, and fisheries management objectives (Table 17.2). The MPA is divided into three zones:

- the General Protection Zone, which corresponds to 72% of the MPA, is a 'no-take' zone, which prohibits commercial fishing;
- the Special Research Zone (SRZ), which permits some commercial fishing as a part of scientific research;
- the Krill Research Zone (KRZ), which permits some harvesting of krill as a part of scientific research.

A management plan has been agreed and provides further details about the features or areas within the MPA associated with the specific objectives, as well as the management measures and administrative arrangements for achieving them (Annex 91-05/B of CM 91-05).

A Scientific Research and Monitoring Plan was developed for the October 2017 CCAMLR Commission meeting (Dunn et al. 2017). Priority elements for the plan can be found in Annex 91-05/C of Conservation Measure 91-05.

The Conservation Measure defining the Ross Sea is due for review at least every 10 years to evaluate whether the specific objectives of the MPA are still relevant or being achieved and to evaluate the delivery of the research and monitoring plan.

There are no proposals to establish MPAs in the Amundsen Sea region.

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[13] CCAMLR. Marine Protected Areas. Retrieved from https://www.ccamlr.org/en/science/marine-protected-areas-mpas.

[14] Scientific Consensus Statement on Marine Reserves and Marine Protected Areas, https://www.nceas.ucsb.edu/Consensus.

[15] CCAMLR. Conservation Measure 91-05 (2016). Retrieved from https://www.ccamlr.org/en/measure-91-05-2016.

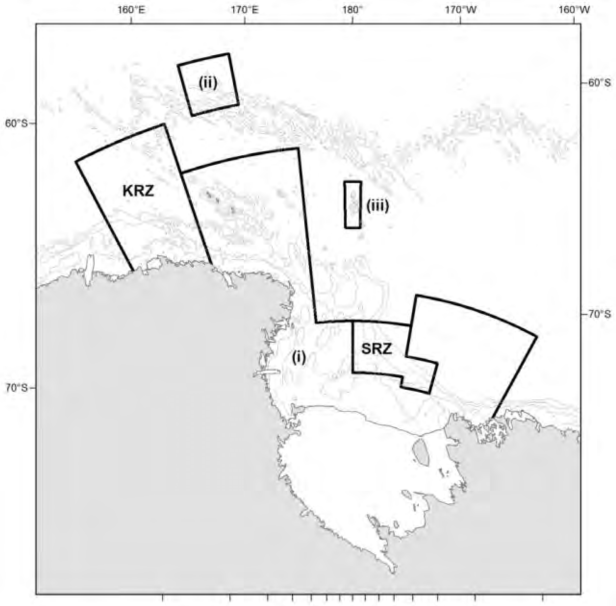

Figure 17.11: Map of the Marine Protected Area in the Ross Sea region. The black lines indicate the boundaries of the General Protection Zone (composed of areas (i), (ii), and (iii)), the Special Research Zone (SRZ), and the Krill Research Zone (KRZ). Depth contours are at 500 m, 1500 m, and 2500 m.

Table 17.2: Objectives of the Marine Protected Area in the Ross Sea region.

1	To conserve ecological structure and function throughout the Ross Sea Region at all levels of biological organisation, by protecting habitats that are important to native mammals, birds, fishes, and invertebrates.
2	To provide a reference area in which fishing is limited, to better gauge the ecosystem effects of climate change and fishing, and to provide other opportunities for better understanding the Antarctic marine ecosystem.
3	To promote research and other scientific activities (including monitoring) focused on marine living resources.
4	To protect a representative portion of benthic and pelagic marine environments.
5	To protect large-scale ecosystem processes responsible for the productivity and functional integrity of the ecosystem.
6	To protect core distributions of trophically dominant pelagic prey species.
7	To protect core foraging areas for land-based predators or those that may experience direct trophic competition from fisheries.
8	To protect coastal locations of particular ecological importance.
9	To protect areas of importance in the lifecycle of Antarctic toothfish.
10	To protect known rare or vulnerable benthic habitats.
11	To promote research and scientific understanding of krill, including in the Krill Research Zone in the north-western Ross Sea region.

17.4 INDICATORS AND TRENDS

17.4.1 EFFECTS ON BYCATCH SPECIES

Rajids (skates and rays) are the bycatch group deemed at most risk from a direct effect of fishing in the Ross Sea region (Delegations of New Zealand, Norway and the United Kingdom 2014). No information or indicators as to the ecological effects of fishing on rajids in the Ross Sea and Amundsen Sea regions are available. Before the 2008 fishing season skates were cut off in the water with the hook attached. Starting in the 2008 fishing season, skates that were not already tagged (i.e., recaptured tagged fish) and which were deemed to be in reasonable condition were required to be cut from longlines (CM 33-03). This led to a fall in the number of rajids landed onboard and an increase in numbers released (Figure 17.12).

Macrourid bycatch in the Ross Sea and Amundsen Sea regions increased to a maximum in 2005 and 2006, respectively, as the fisheries expanded and then decreased (Figure 17.13).

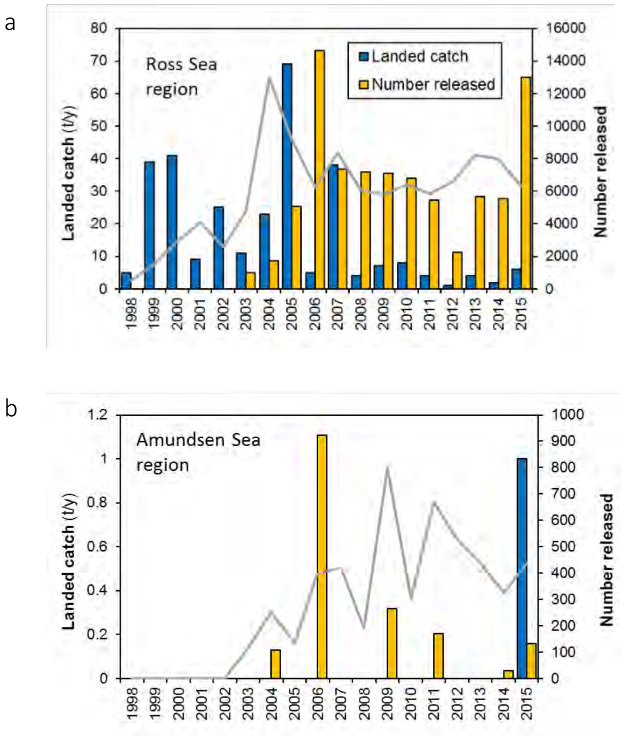

Figure 17.12: Catch of rajids (skates and rays). [a] Ross Sea region; [b] Amundsen Sea region. Weight of landed rajids (blue) and numbers cut off alive (orange). The relative effort (number of sets) is shown as the grey line.

Predation release of macrourids and icefish is expected to be larger than fishing mortality and may lead to increased abundance over time (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014d). Analysis of the rates of bycatch for macrourids (*M. whitsoni* and *M. caml*), icefish (principally *Chionobathyscus dewitti*), eel cods (*Muraenolepis* spp.), and deepsea cods (*Antimora rostrata*) has been carried out (Moore & Parker 2021), using standardisation to control for area and vessel reporting.

It is likely that changes to CCAMLR management rules aimed at reducing bycatch of macrourids together with more targeted fishing practices have led to decreases in the catch of macrourids in the Ross Sea slope region. These changes in fishing locations and practices are also likely to have affected catch rates for bycatch species so that changes in catch rates in Figure 17.14 probably do not reflect changes in population sizes.

Alternative methods to look for changes in the population abundance of macrourids over time are being explored, including acoustics (O'Driscoll et al. 2012, Ladroit et al. 2014). Also, 'catch-curve' analysis to explore changes in the total mortality rate of macrourids is being investigated.

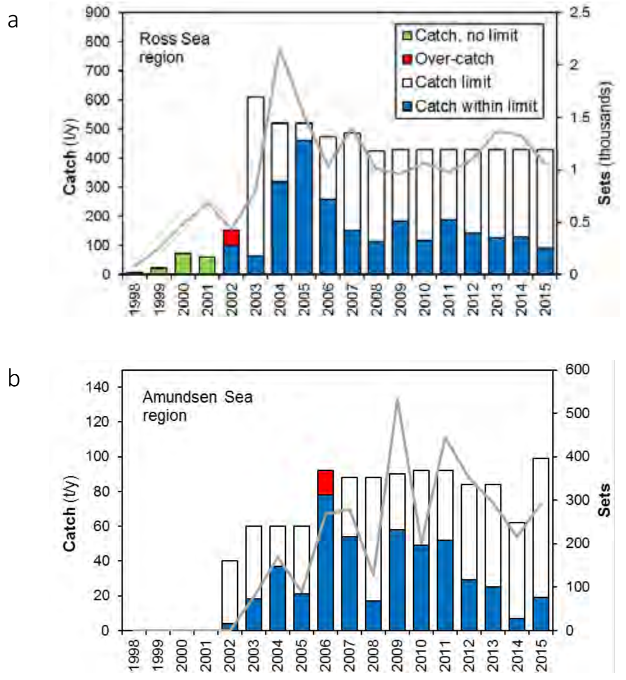

Figure 17.13: Macrourid bycatch (bars) and total fishing effort in terms of number of sets (grey line). [a] Ross Sea region; [b] Amundsen Sea region. White bars shows where the catch limit exceeds the catch, and red bars indicate that catch exceeded the catch limit in that year.

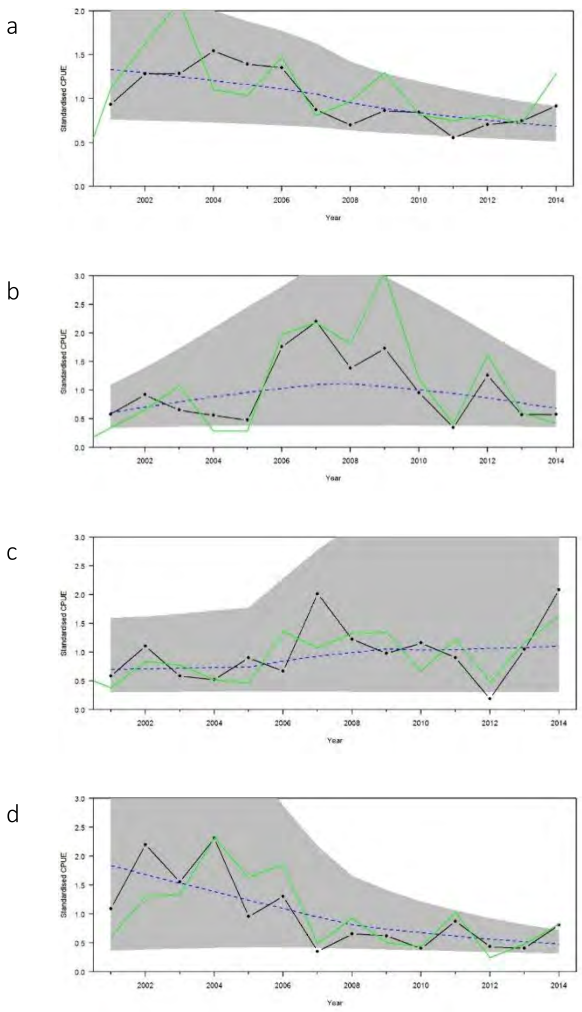

Figure 17.14: Raw (green), standardised (black), trends (blue) and 95% confidence intervals (grey) for catch-per-unit-effort (CPUE) in groups of bycatch species on the Ross Sea continental slope (small-scale research units (SSRUs) 88.1 H and I). [a] Macrourids; [b] icefish; [c] eel cods; [d] deepsea cod.

17.4.2 EFFECTS ON PREY SPECIES

Both mixed trophic impact analysis and the minimum realistic model of trophic interactions between toothfish, macrourids, and icefish in the Ross Sea region suggest that the toothfish fishery is likely to cause predation release in prey species, especially on the Ross Sea slope (Pinkerton & Bradford-Grieve 2014, Mormede et al. 2014d). The differential strength of the predation release on macrourids and icefish would be likely to lead to a change in the diet of toothfish over time in favour of more icefish being consumed (Mormede et al. 2014d).

Stevens et al. (2014) found no significant temporal change in the diet of toothfish between 2003 and 2010 based on examination of stomach contents of toothfish on the Ross

Sea slope. Pinkerton et al. (2014) found a small but significant reduction in the trophic level of toothfish between 2006 and 2014 in a direction consistent with more icefish and fewer macrourids being consumed.

Monitoring for changes in the diet of toothfish, with a focus on the Ross Sea slope, is a research priority (Delegations of New Zealand, Norway and the United Kingdom 2014) and is continuing through periodic collection of toothfish stomachs and analysis of toothfish tissue samples by stable isotope analysis to test for changes in trophic level over time.

17.4.3 EFFECTS ON PREDATOR SPECIES

At present, no indicators are available to monitor changes to the ecological state of known predators of toothfish (Type C killer whales, Weddell seals, and sperm whales) in the Ross Sea or Amundsen Sea regions, and this is a recognised priority for future research (Delegations of New Zealand, Norway and the United Kingdom 2014). The fact that Type C killer whales, and potentially sperm whales, move between the Ross Sea and the EEZ, gives New Zealand a key role in the management of risks to these species.

Information is available on the extent to which fishing is likely to have reduced the availability of toothfish to predators of toothfish. Two factors are important when considering indicators for changes to the availability of toothfish as relevant to toothfish predators.

First, different predators forage over different spatial scales so that spatial patterns in changes to toothfish abundance over time are important. For example, the foraging ranges of lactating Weddell seals are constrained by the seals having to return to fast-ice colonies to feed the pups. Foraging range of Type C killer whales in the Ross Sea is not known, but it appears that the McMurdo Sound is important (Eisert et al. 2015). Sperm whales are unlikely to venture south of the Ross Sea slope.

Second, the size of toothfish consumed by predators is important because different size classes of toothfish will be affected differentially over time by fishing. Weddell seals consume toothfish of total length (TL) 60–110 cm (median TL about 80 cm; Kim et al. 2011, Ainley & Siniff 2009). Although information on the size of toothfish taken by killer whales is scarce, Type C killer whales appear to take larger toothfish than Weddell seals. In the McMurdo Sound region at least, a Type C killer whale was observed with an approximately 150 cm TL toothfish (Eisert et al. 2015). This size of toothfish coincides with the modal size classes (130–159 cm TL) of toothfish caught in McMurdo Sound by scientists (Ainley et al. 2013). For sperm whales, because of their ability to access the entire water column, it is likely that all sizes of toothfish present in the Ross Sea slope region are available as prey.

Simulations of changes to the abundance of toothfish by geographic area were generated by the spatial population model of Mormede et al. (2014b). This model estimates the distribution of age classes of toothfish in the Ross Sea region.

Over the Ross Sea continental shelf (where Weddell seals and Type C killer whales overlap in distribution with toothfish), the spatial population model suggests that the biomass of sexually mature toothfish (greater than about 110–130 cm TL; Parker & Marriott 2012) was about 74% B_0 in 2013 and will decrease to about 57% B_0 in 2048 (Pinkerton et al. 2016). In SSRUs 88.1H and 88.1I on the Ross Sea slope (where sperm whales may occur and feed on toothfish) the spatial model suggests that total toothfish biomass (all lengths) in 2013 was 77% of that before fishing, and that this will decrease to 60% of the pre-exploitation biomass by 2048.

Changes in the length-frequency distribution of toothfish taken over the Ross Sea shelf by the fishery, in the SRZ and the south of 70° S, between 1998 and 2021 are summarised by Grüss et al. (2021a) and shown in Figure 17.15.

For the southern part of the Ross Sea shelf, the sub-adult survey catches a lower proportion of toothfish over 150 cm TL than the commercial fishery over the whole shelf (Figure 17.16; Devine et al. 2021) but again, changes to the proportion of large toothfish in the sub-adult survey over this period are not obvious. Furthermore, the standardised catch rates from a research longline survey of pre-recruit toothfish (70–110 cm TL) in the southern Ross Sea in 2012 were similar to those made by the same vessel fishing in the area earlier in the fishery, between 1999 and 2003 (Hanchet et al. 2012).

In the vicinity of McMurdo Sound, scientific droplining (through ice holes) had suggested large decreases in toothfish abundance since the 1970s (Ainley et al. 2013), but Parker et al. (2015) obtained catch rates of toothfish similar to those prior to the advent of the toothfish fishery (Figure 17.17). Results from Parker et al. (2015) suggest that either large old fish have returned to McMurdo Sound following a temporary environmentally driven absence, or that they remained locally present but were not detected in the areas sampled.

17.4.4 TROPHIC AND SYSTEM-LEVEL EFFECTS

The Ross Sea is home to about a third of the world population of Adélie penguins. Between 2001 and 2013 the number of breeding pairs of Adélie penguins at colonies in the southwestern Ross Sea more than doubled (Figures 17.18a,b) from about 235 000 to more than half a million (Lyver et al. 2014).

It is not known what has caused this increase but it is likely that changing ice patterns (Stammerjohn et al. 2008) play a primary role. Some researchers (Ainley et al. 2013, Lyver et al. 2014) previously suggested that reduced toothfish abundance in association with the Antarctic toothfish fishery had reduced predation on Antarctic silverfish and that the observed magnitude of the population response led to increases in the abundance of this species, which is known to be an important prey for Adélie penguins, especially during chick rearing. Other small fish are also taken by Adélie penguins. However, a predation release model of this effect acting via silverfish was not consistent with the magnitude of any plausible fishery-associated predation release. The mass of silverfish released from predation due to the effects of fishing was estimated to be equivalent to less than 2% of the biomass of silverfish estimated to be consumed annually by Adélie penguins (Pinkerton et al. 2016). Even if toothfish consumed only silverfish, the predicted predation release effect would still not be sufficient to explain the observed increase in the number of Adélie penguins in the southern Ross Sea (Pinkerton et al. 2016, Figure 17.18c).

The reasons for the increase in Adélie penguin numbers in the Ross Sea region are still not known. The fact that similar colony growth rates were seen for several Adélie penguin colonies in the south-west Ross Sea suggests that large-scale factors were responsible (Whitehead et al. 2015). The paucity of census data for the northern Ross Sea metapopulation makes it difficult to discern trends there (Lyver et al. 2014).

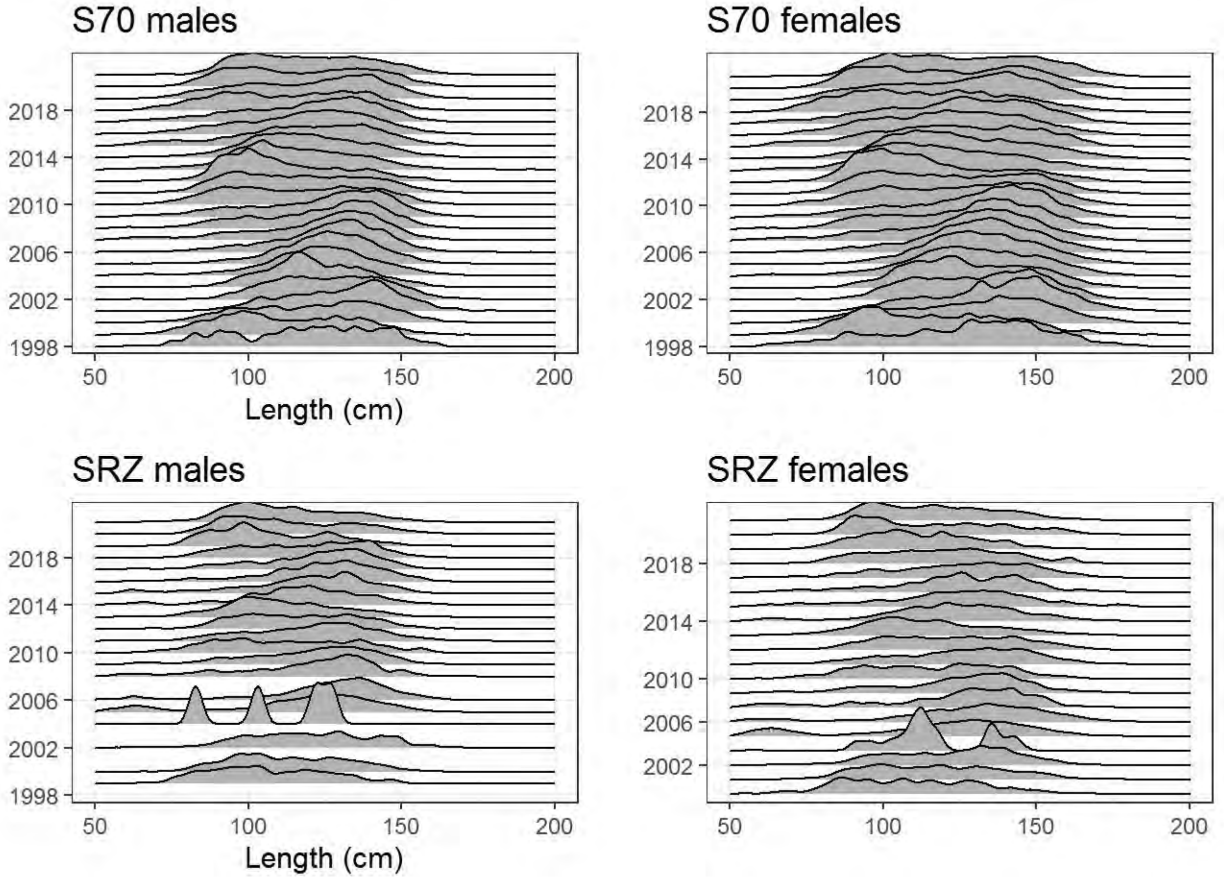

Figure 17.15: Estimated proportion of fish at length by sex for all vessels in the shelf region (south of 70° S and the SRZ) of the Ross Sea, for the years 1998–2021 (Grüss et al. 2021a).

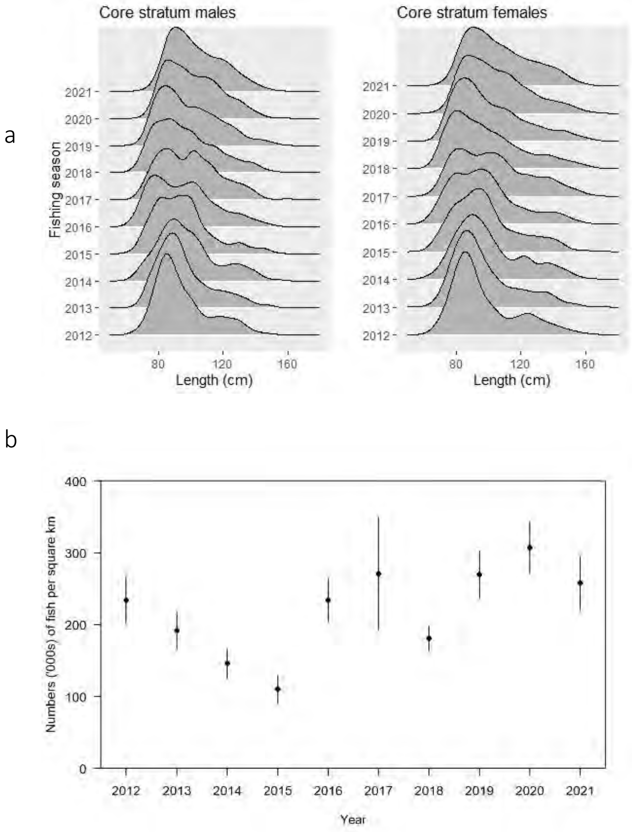

Figure 17.16: [a] Toothfish length frequency distributions in the Ross Sea shelf core strata (A–C) for the 2012–21 sub-adult surveys (Devine et al. 2021). [b] Estimated abundance index (numbers) for Antarctic toothfish in the core strata of the Ross Sea shelf survey 2012–2021. Error bars indicate the 95% confidence intervals (Devine et al. 2021).

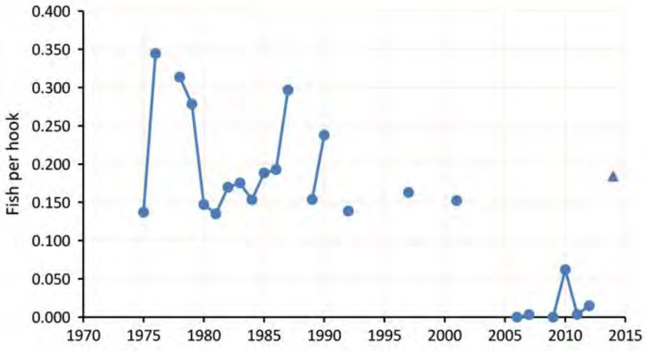

Figure 17.17: Catch rates (fish per hook) for toothfish sampled in McMurdo Sound, Antarctica, 1975–2014. Circles indicate pre-2013 data recalculated from Ainley et al. (2013) and triangle indicates the 2014 value from Parker et al. (2015). [Parker et al. (2015), figure 3].

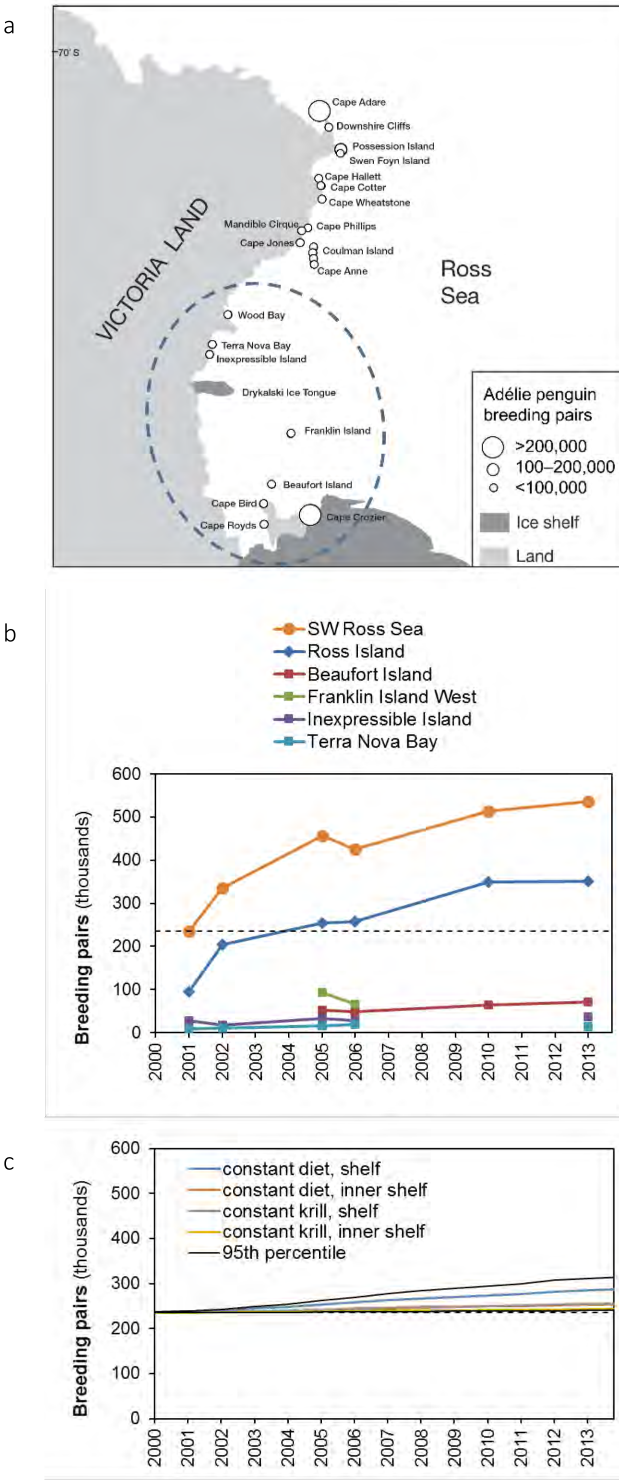

Figure 17.18: [a] Location and sizes of breeding colonies of Adélie penguins in the Ross Sea region. Those forming the 'southwest (SW) metapopulation' are enclosed in the dashed ellipse and forage over the Ross Sea shelf between chick hatching and fledging. [b] Changes to the total number of Adélie penguins breeding in the SW colonies (orange line) driven largely by increases in numbers breeding on Ross Island (blue line, capes Crozier, Bird, and Royds). [c] Greatest modelled effect of number of Adélie penguins that could be supported from additional silverfish released from predation by the toothfish fishery. [Pinkerton et al. 2016].

17.4.5 EFFECTS ON HABITATS

There are no indicators available to assess effects on the benthic habitat of fishing for toothfish in the Ross Sea or Amundsen Sea regions.

The status of selected habitat-forming benthic invertebrates likely to be physically impacted by fishing (vulnerable marine ecosystems, VMEs) was simulated (Dunn et al. 2010) under various scenarios of future fishing and assuming no correlation between distributions of VMEs and fishing. Predicted changes to the status of selected VMEs were small at the scale of the Ross Sea region, even with no specific management of VME impacts (Figure 17.19).

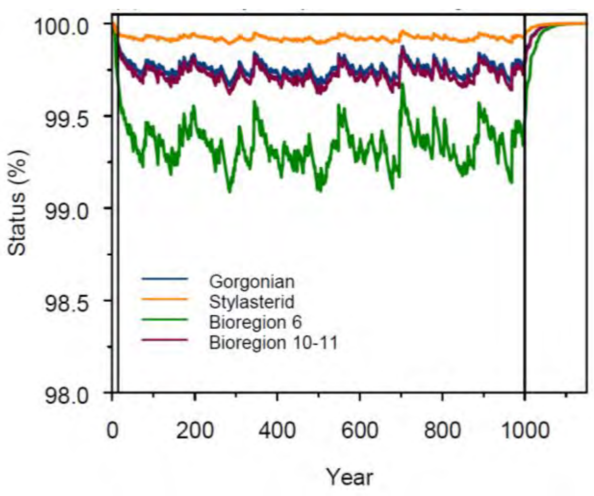

Figure 17.19: Simulated changes in the status of selected vulnerable marine ecosystems (VMEs) over time. Here, a status of 100% indicates that the habitat has the same extent and biomass as before fishing began and 0% indicates habitat removal at the scale of the Ross Sea region. Results are based on the medium-scale benthic habitat model of Dunn et al. (2010). The runs are for VMEs characterised as Gorgonian, Stylasterid, or for all VMEs in three indicative areas of the Ross Sea that have been identified as having different benthic biological conditions (benthic bioregions; Sharp et al. 2010). The benthic model assumes historical fishing pattern intensity up to 1000 years.

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THEME 5: MARINE BIODIVERSITY

Chapter 18: Biodiversity - Technical Summary

Declining marine biodiversity

Habitat loss, debris, climate change

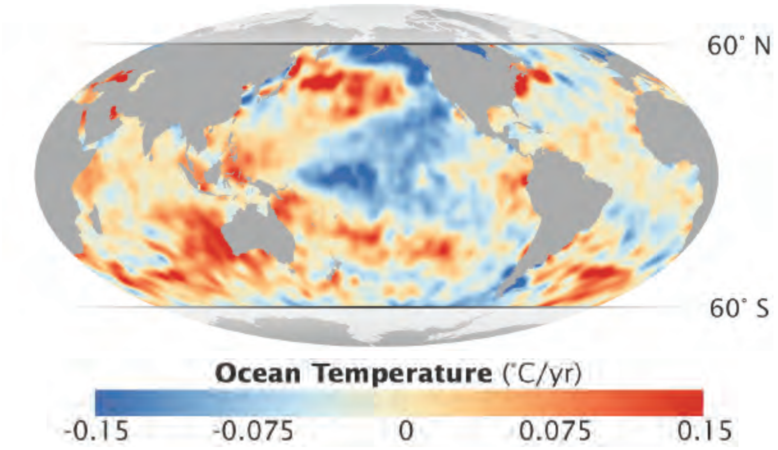
1. THE ISSUE IN BRIEF

- Ecosystems, biodiversity, and the productivity of the marine environment are under threat worldwide from the cumulative effects of human pressures on the ocean
- The contribution of biodiversity to a healthy functioning marine ecosystem is not easily quantified and the rate of decline is confounded by poorly documented biodiversity
- Since the launch of NZ's Biodiversity Strategy 2000, Fisheries New Zealand has run a Marine Biodiversity Research Programme with 67 projects to date, addressing biodiversity knowledge gaps
- Cumulative effects and the effects of climate change on biodiversity have yet to be quantified

2. NEW ZEALAND'S MARINE BIODIVERSITY

- NZ marine biodiversity is rich with ~ 18,000 known species and a further 13,000 species yet to be described
- NZ marine biodiversity forms 8% of global marine biodiversity, and approximately 40% of species identified so far are endemic
- The number of species discovered continues to climb annually (Lundquist et al. 2014). Our state of knowledge about marine biodiversity is assessed as being sixth across 18 nations (Costello et al. 2010)
- Identification guides are available for a wide range of biota

3. THREATS TO BIODIVERSITY AND RELATED WORKSTREAMS

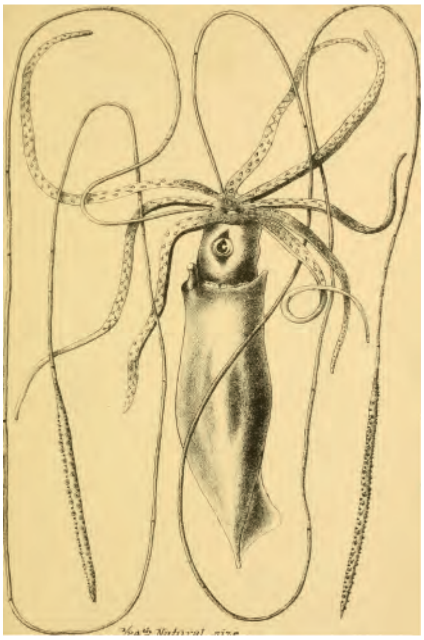


2003–2012 sea surface temperature trend from NASA

- Direct threats to biodiversity and marine ecosystems from human activities in NZ include biosecurity risks, commercial fishing, marine debris pollution (including plastics), noise pollution, nutrient and sediment run-off, recreational fishing, and seabed mining
- Climate change and ocean acidification pose an external threat to biodiversity (see also Chapter 12). Some useful indicators of change have been identified, but tracking ecosystem-wide health is more elusive
- There are six biodiversity workstreams at Fisheries New Zealand

1. Development of ecological measures for more integrated fisheries management
2. Tools and methods for improved assessment of biodiversity health
3. Monitoring and assessing the effects of environmental change on fisheries and biological productivity
4. Sustainable development of the blue-green economy
5. Evaluate and safeguard natural capital for future generations
6. Progressing integrated management approaches to meet international obligations

4. BIODIVERSITY RESEARCH PROGRESS AND UPTAKE



- Strategic uptake of biodiversity research

The Marine Biodiversity Research Programme enables Fisheries New Zealand and MPI to meet longer term commitments to New Zealand's international and national obligations regarding the distribution and identification of species, the environmental footprint of fishing, spatial management, and cumulative effects of environmental change

- Several biodiversity projects have resulted in management and policy uptake

Examples include land-sea interactions; spatial marine protection; national environmental monitoring reporting and environmental risk assessment; innovative modelling approaches; Vulnerable Marine Ecosystem move-on rules in the South Pacific Regional Fisheries Management Organisation; Marine Environmental Classification; eco-certification; substrate-disturbance modelling and sedimentation; bioindicators

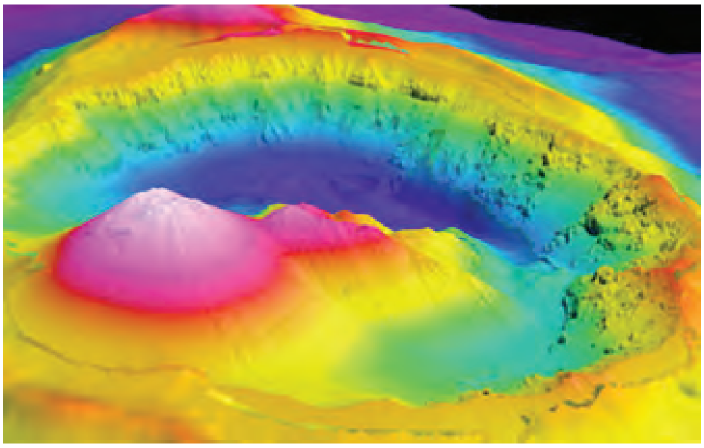
***Architeuthis longimanus* - Transactions and proceedings of the New Zealand Institute (1887)**

5. SEABED MAPPING

- Methods to quantify biota across large tracts of seabed have gradually improved through the development of new technology

- However, seabed mapping to identify habitats and biodiversity hotspots is a long-term goal that is far from complete in NZ waters

- More integrated fisheries management approaches require greater understanding of biodiversity and habitat distribution



6. ONGOING RESEARCH

- Public awareness of biodiversity and environmental effects of resource use has increased. Current research directions in this programme are being driven by Fisheries New Zealand and MPI goals to improve environmental sustainability, and cross-government goals to better manage biodiversity outcomes through the refresh of the NZ Biodiversity Strategy
- Climate change is measurably affecting NZ waters. Current research focuses on identifying climate change risks to the marine environment and the seafood sector, biodiversity, deepwater corals, biogenic habitats, fish, shellfish, plankton, primary productivity, and regime shifts
- Improving monitoring methods to measure change and assess future scenarios has become important. Methods of incorporating climate change indicators into stock assessment, and methods to assess species movements, are underway
- The requirement to integrate land-based activities and downstream effects with the marine environment, and the need to better include Treaty Partners in research planning and decision making, have also been recognised

18 BIODIVERSITY

Status of chapter	This chapter has been updated for AEBAR 2021.
Scope of chapter	This chapter outlines the current status and trends in Fisheries New Zealand marine biodiversity research, links between biodiversity and ecosystem function, and innovative methods and indicators for evaluating marine biodiversity and ecosystem health in New Zealand. An overview of research within the Fisheries New Zealand Marine Biodiversity Programme for 2000–20 is described within the context of international and national policy obligations, whole-of-government research initiatives and aligned international and national research programmes.
Area	New Zealand Territorial Sea, Exclusive Economic Zone, and Extended Continental Shelf; South-west Pacific Region associated with South Pacific Regional Fisheries Management Organisation (SPRFMO); Southern Ocean and Ross Sea region
Focal issues	• The chapter outlines new workstreams that respond to New Zealand's new biodiversity strategy 'Te Mana o te Taiao' and Fisheries New Zealand strategic priorities in the environment. The new workstreams are summarised below and aim to develop robust methods that estimate biodiversity patterns and trends, to develop and use as suitable indicators for national and international reporting, and to meet domestic and international obligations within constraints of limited data. • Map and document the identity, abundance, and distribution patterns of New Zealand's marine biodiversity in this extremely large area of responsibility (about 5.8 million km^2) which is far from complete. • Improve understanding of links between biodiversity and ecosystem function in near-shore and offshore marine ecosystems, and effects of cumulative stressors on ecosystem health. • Develop new approaches and indicators to evaluate efficacy of current spatial measures and management actions to protect marine biodiversity. • Increase recognition of connections between land-based stressors (e.g., sediment and nutrients) and the health of near-shore biodiversity and ecosystems (see Chapter 15). • Evaluate the risks of climate change to marine biodiversity and marine ecosystems. Identifying areas of high biodiversity remains a challenge, particularly for environmental impact evaluation or assessing response to climate change scenarios.
Emerging issues	• Use of environmental data to inform stock assessment of fish stocks and a move towards Ecosystem Based Fisheries Management as signalled in Fisheries Management Plans. • Assessment of marine debris and pollution, particularly in the coastal zone. • Urgent need for ecosystem approaches to marine resource management, to allow development of the blue economy within environmental constraints and facilitate integration of socioeconomic factors into fisheries management. • Trophic dynamics and fishing. Trophic modelling studies from parts of New Zealand suggest tight coupling among components of the food web. • Exploration of new technologies for measuring marine biodiversity (e.g., eDNA). • Working more closely with DOC and MfE on marine biodiversity initiatives to meet the directions signalled under the new Oceans and Fisheries portfolio and the Future of Fisheries report released by the Office of the Prime Minister Chief Science Advisor, and the Marine Protected Areas Science Advisory Group (MSAG).

Fisheries New Zealand Research (current)	Current research includes: ZBD2020-11 *Review of land-based effects on coastal fisheries and kaimoana and their habitats;* ZBD2020-09 *Cumulative effects of stressors on scallops and scallop habitats in the Marlborough Sounds;* ZBD2020-08 *Role of low and mid trophic fish in the Hauraki Gulf;* ZBD2020-07 *Recovery of Seamount Communities;* ZBD2020-06 *Recovery of biogenic habitats;* ZBD2019-11 *Development of Electronic Automated Reporting System (EARS) to improve seabird bycatch monitoring* (see Seabirds Chapter); ZBD2019-01 *Quantifying benthic habitats Part 2*; ZBD2019-04 *Plastics and marine debris across the ocean floor in New Zealand waters*; ZBD2018-01 *5 year continuous plankton survey (Phase 3)*; ZBD2018-02 *Climate change, fish distribution meta-analysis*; ZBD2018-03 *Climate variability, trends, and fish population parameters*; ZBD2018-05 *Ecosystem function and regime shifts in the Subantarctic*; ZBD2016-11 *Quantifying benthic biodiversity across natural gradients*; ZBD2016-07 *Multiple Stressors on Coastal Ecosystems in situ;* ZBD2016-04 *Organic Carbon Recycling in Deepwater*; ZBD2014-09 *Climate change risks and opportunities;* ZBD2014-03 *Sublethal effects of environmental change on fish populations.*
External research	The Biodiversity Programme has synergies with the Aquatic Environment Working Group (AEWG), Fisheries New Zealand 'Fisheries Change Programme', the Natural Resource Sector, Aotearoa Circle, biodiversity research (DOC), the Marine Protected Areas Science Advisory Group (MSAG) established by the Ministry for the Environment in 2016 has commissioned relevant biodiversity work that links to this Biodiversity Programme, environmental reporting (MfE), the new Fisheries and Oceans ministerial portfolio and Te Mana o te Taiao - Aotearoa New Zealand Biodiversity Strategy 2020. Awareness and understanding of climate change has been a key topic in recent years. Research programmes and database initiatives at research institutes and universities, e.g., NIWA Strategic Science Investment Funding – Oceans, Coasts and Estuaries, and Fisheries, World Register of Marine Species (WoRMS), OBIS, New Zealand Organisms Register, MBIE National Science Challenge 'Sustainable Seas', MBIE-funded projects CARIM (Coastal Acidification: Rates, Impact and Management) and the recent Moana Project are all important in ensuring that synergies across government are optimised.
Related chapters/issues	Cumulative effects, land-based effects, protected areas, benthic impacts, ecosystem approaches to fisheries and marine resource management, ocean context.

18.1 INTRODUCTION

This chapter summarises the development and progress of the twenty-year Fisheries New Zealand Biodiversity Research Programme 2000–2020 and reviews the work commissioned by Fisheries New Zealand in the context of national and global concerns about the maintenance of biodiversity initially under the New Zealand Biodiversity Strategy (NZBS, Anon 2000) and now the new Biodiversity Strategy 'Te Mana O Te Taiao' (Aotearoa New Zealand Biodiversity Strategy 2020).

The recognition of increasing societal expectation to use fisheries management measures that will achieve biodiversity conservation was signalled in the policy document Fisheries 2030 (Ministry of Fisheries 2009) in its long-term commitment to ecosystem-based fisheries management and to ensuring that *"biodiversity and the function of ecological systems, including trophic linkages, are conserved"*. Although New Zealand's environmental performance with regard to fishing is perceived to be relatively high on an international scale, Fisheries New Zealand is not complacent about the ongoing requirement to monitor and provide evidence that measures to achieve biodiversity conservation needs are being met. This includes the need to better understand and mitigate the effects of fishing in the areas impacted by fishing, and to evaluate the effectiveness of management approaches in non-fished areas such as marine reserves and Benthic Protection Areas (BPAs). Assessing the effects of fishing on the aquatic environment and risks to biodiversity and the aquatic environment are supported through the Ministry for Primary Industries Strategic Intentions 2021-25 https://www.mpi.govt.nz/dmsdocument/48589-Strategic-Intentions-2021-2025 and Fisheries New Zealand's National Fisheries Plans (deepwater, highly migratory species, inshore finfish, inshore shellfish, and freshwater fisheries) as well as the Aquatic Environment Medium Term Research plan (draft).

Fisheries New Zealand is also one of several government agencies with a strong interest and a statutory management mandate in the Ross Sea region of Antarctica through the Antarctic Marine Living Resources Act 1981. Fisheries New Zealand's Antarctic science contributes strongly to New Zealand's whole-of-government involvement in contributions to the Commission for the Convention on Antarctic Marine Living Resources (CCAMLR) and the Antarctic Treaty.

There are a range of societal values beyond commercial, customary, and recreational take from the sea that are recognised as part of 'strengthening our society' in New Zealand. These include aesthetic and cultural values as well as other economic values such as tourism and other forms of marine recreation (Le Heron et al. 2016). To link socioeconomic values of biodiversity to science supporting fisheries management will require a multi-disciplinary approach that is only just beginning in New Zealand (Lundquist et al. 2016).

18.1.1 NEW ZEALAND BIODIVERSITY STRATEGY REFRESH 2020

In June 2000, the 'New Zealand Biodiversity Strategy – Our Chance to Turn the Tide' (NZBS) was launched as part of New Zealand's commitment to the international Convention on Biological Diversity 1993 (Anon 2000). To meet long-term goals of the NZBS (i.e., to halt the decline of biodiversity in New Zealand and protect and enhance the environment), a comprehensive plan with stated objectives and actions was developed to address biodiversity issues in terrestrial, freshwater, and marine systems. This document has been succeeded by a new Aotearoa Biodiversity Strategy 2020, 'Te Mana o te Taiao' (TMOTT) available at https://www.doc.govt.nz/globalassets/documents/conservation/biodiversity/anzbs-2020.pdf.

In the marine environment, biodiversity decline is characterised primarily in relation to megafaunal taxa (marine mammals, seabirds, see Chapters 4–8) and other protected species such as sharks, which may experience changes in threat status and risk of extinction. Biodiversity is also influenced by environmental degradation due to habitats being diminished or removed, and by the disruption of ecosystem structure, function, and the disruption of ecological processes (e.g., biological cycling of water, nutrients, and energy), species invasion, and hybridisations that affect the diversity of marine species and their life history strategies. Measuring the decline of marine biodiversity is complicated by the 'shifting baseline syndrome', a common obstacle to useful biodiversity assessment and monitoring (Soga & Gaston 2018). Furthermore, the size range of organisms sampled is often limited to macroscopic or larger. Changes (declines) in the diversity of smaller-sized organisms below the sampling threshold that may be critical to marine ecosystem health and well-being are therefore likely to be missed (Azam & Malfatti 2007).

The task of implementing TMOTT is led by the Department of Conservation (DOC), with significant input from the Ministry for Environment (MfE), and Fisheries New Zealand.

Te Mana o te Taiao - Aotearoa New Zealand Biodiversity Strategy 2020 comes at a time when perspectives on biodiversity are changing the way in which it is assessed, managed, and valued. Various knowledge systems are being drawn upon to take a more informed and holistic approach to biodiversity (e.g., Māori knowledge and scientific disciplines, e.g., Home - Te Ahu o Rehua). It is more widely recognised that human well-being relies on healthy, functioning ecosystems and this is changing the way biodiversity is valued and cared for (e.g., Short 2015, Díaz et al. 2018). These themes are strongly reflected in Te Mana o te Taiao which states that "*people are a part of nature… we can only thrive when nature thrives*". The strategy places much emphasis on collaboration between Treaty partners and promotes objectives and goals that unify New Zealanders in stewardship of our unique biodiversity and indigenous species. This new strategy will likely be updated in the future to be aligned with the Convention on Biological Diversity 'Post-2020 Global Biodiversity Framework' which is currently in its first draft stage. The objectives of TMOTT are provided in Table 18.1.

Table 18.1: The objectives of TMOTT set out what the strategy aims to achieve by 2050. Objectives of particular relevance to Fisheries New Zealand are in bold (2-5, 9-13).

1. Governance, legislation and funding systems are in place and enable delivery of the strategy outcomes

2. Treaty partners, whānau, hapū, iwi and Māori organisations are rangatira and kaitiaki

3. Biodiversity protection is at the heart of economic activity

4. Improved systems for knowledge, science, data and innovation inform our work

5. Mātauranga Māori is an integral part of biodiversity research and management

6. Aotearoa New Zealand is making a meaningful contribution to biodiversity globally

7. All New Zealanders have the skills, knowledge and capability to be effective

8. Resourcing and support are enabling connected, active guardians of nature

9. Collaboration, co-design and partnership are delivering better outcomes

10: Ecosystems and species are protected, restored, resilient and connected from mountain tops to ocean depths

11. Management ensures that Biological threats and pressures are reduced through management

12. Natural resources are managed sustainably

13. Biodiversity provides nature-based solutions to climate change and is resilient to its effects

18.2 THE FISHERIES NEW ZEALAND BIODIVERSITY RESEARCH PROGRAMME

The Ministry of Fisheries responded to the NZBS in 2000 with the establishment of the Marine Biodiversity Research Programme (MBRP) to address aspects of biodiversity in NZBS Theme 3 that would complement research under the Aquatic Environment Programme. The focus of the MBRP was initially to map and describe marine biodiversity throughout New Zealand and in the Ross Sea, particularly to conduct benthic habitat surveys and describe the taxonomy of new species. DOC and Ministry of Fisheries/MPI research on protected species and marine spatial protection was largely dealt with outside the MBRP.

In more recent years, the MBRP focus has incorporated the effects of climate change on marine biodiversity.

The core purpose of the Marine Biodiversity Research Programme is *"To ensure that biodiversity, marine habitat diversity, and ecosystem services that underpin the sustainability of wild caught fisheries productivity and ecosystem resilience are understood and maintained."*

The Biodiversity Research Programme is guided by a multi-stakeholder biodiversity research advisory group (BRAG), chaired by Fisheries New Zealand. The research commissioned at different phases of the Programme have evolved and the workstreams refreshed in 2017 have been further modified here to reflect the new TMOTT. An overview of the revised Biodiversity Programme Workstreams (2021) is given in Table 18.2

Table 18.2: Revised Biodiversity Programme Workstreams to reflect TMOTT. Note that Workstream 1 is completely new, and that the international Workstream 6 from the previous version has been removed.

New Workstreams to match TMOTT	Purpose	Related Objectives and 2025 Goals from Te Mana o te Taiao - Aotearoa New Zealand Biodiversity Strategy 2020
Workstream 1 Iwi and community-based biodiversity projects (New workstream)	Engage with local communities, whānau, hapū, iwi, and Māori organisations to promote collaborative research efforts that connect biodiversity to local knowledge and decision-making. This workstream includes projects related to indigenous/taonga species and aims to improve understanding through multiple scientific disciplines and knowledge systems including mātauranga Māori and citizen science. Iwi Fisheries Forums will provide a platform to support this workstream.	• Objective 2 o Goal 2.2 o Goal 2.3 • Objective 5 o Goal 5.3 • Objective 9 o Goal 9.1
Workstream 2 Baseline knowledge for EBFM (incorporates former Workstream 1)	Identify key biodiversity and ecosystem components related to fisheries activity that will inform mitigation and spatial planning in New Zealand to meet development goals for Ecosystem-Based Fisheries Management (EBFM). This workstream aims to improve understanding about the key drivers of biodiversity distribution and abundance. The outputs will be used to inform and develop performance measures and environmental standards or guidelines for decision-makers.	• Objective 10 o Goal 10.1.1 o Goal 10.4.1 o Goal 10.5.1 • Objective 13 o Goal 13.1.1 o Goal 13.2.1
Workstream 3 Tools and methods for biodiversity and ecosystem assessment (incorporates former Workstream 2)	Work synergistically across government to produce tools such as ID guides, ecological maps or atlases of seabed habitats, ecosystem and biodiversity indicators and report cards. Genetic tools and emerging technology (e.g., electronic monitoring, cameras on boats) are included in this workstream.	• Objective 4 o Goal 4.1 o Goal 4.5
Workstream 4 Functional threats to biodiversity (incorporates former Workstream 3)	Investigate the effects of multiple stressors (e.g., sedimentation, cumulative effects) and environmental change (e.g., climate change, regime shifts, trophic effects) on ecosystems, their functions, and fisheries. This includes working with local and national government, Crown Research Institutes, and universities to gain a national and regional record of long-term changes through the development of robust indicators, metrics, and monitoring programmes.	• Objective 10 o Goal 10.2.1 • Objective 12 o Goal 12.7.1 • Objective 13 o Goal 13.3.1
Workstream 5 Sustainability for future generations (Former Workstreams 4 and 5)	Ensure the socio-ecological and sustainable use of natural resources. This includes understanding the effectiveness of protected/closed areas, preserving cultural practices as they relate to fisheries and biodiversity, maintaining a social licence to operate, and identifying creative solutions that allow for sustainable development of the marine economy with biodiversity at its core.	• Objective 3 o Goal 3.1 o Goal 3.2 o Goal 3.3 • Objective 10 o Goal 10.4.1 o Goal 10.5.1 o Goal 10.6.1 o Goal 10.8.1 • Objective 12 o Goal 12.1.1 o Goal 12.5.1

18.3 MARINE BIODIVERSITY TRENDS IN NEW ZEALAND

18.3.1 NEW ZEALAND'S MARINE BIODIVERSITY

New Zealand's ocean territory is vast, covering approximately 5.8 million km^2 including the Territorial Sea (TS) and the recent continental shelf extension. It is very large relative to the area of land and includes approximately 15 000–18 000 kilometres of coastline extending from the subtropical north to the cool sub-Antarctic waters in the south. New Zealand also has a rich marine biodiversity that is globally significant with up to 38% of all marine species (46% for Animalia) estimated as endemic (Gordon et al. 2010, Lundquist et al. 2014) and comprises up to 8% of global marine biodiversity. These estimates do not include undiscovered species, which are likely to increase the proportion of endemics.

The most recent summary of knowledge about marine biodiversity in New Zealand is provided by Gordon (2009, 2010, 2012) and Gordon et al. (2010), which estimates a tally of 17 987 living species in the EEZ, including 4320 known undescribed species in collections. Since 2010, further updates using all records available within OBIS, NIWA, and Te Papa Tongarewa collections have assessed the spatial distribution of biodiversity records and suggested metrics for reporting on the status of marine biodiversity (Lundquist et al. 2014). Species diversity for the most intensively studied animal phyla (Cnidaria, Mollusca, Brachiopoda, Bryozoa, Kinorhyncha, Echinodermata, Chordata) is more or less equivalent to that in the ERMS (European Register of Marine Species) region, an area 5.5 times larger than the New Zealand EEZ (Gordon et al. 2010), suggesting that the New Zealand region biodiversity is proportionately richer than the ERMS region. In the Southern Hemisphere though, New Zealand has approximately 50% lower species richness than Australia (in terms of species by area), noting that Australia's EEZ also includes substantial tropical and subtropical regions.

The approximately 18 000 known marine species and associated ecosystems around New Zealand deliver a wide range of environmental goods and services that sustain considerable fishing, aquaculture, and tourism industries, as well as drive major biogeochemical and ecological processes. An estimate of undiscovered marine biodiversity suggests another 13 000 species, with several factors suggesting that this estimate of total marine species is conservative. Such factors include the difficulty of sampling such a large region with depths exceeding 10 000 m, the geomorphological and hydrological complexity created by tectonic and seismic activity, limited water column sampling and limited benthic sampling (especially below 1500 m), and rates of new species descriptions, currently about 50 per year. Inflating estimates of undiscovered marine biodiversity is the potentially very large numbers of parasitic and commensal protists (especially microsporidia) and parasitic animals such as myxozoans and nematodes, as well as free-living nematodes. Expectations of massive oceanic microbial diversity (e.g., Sogin et al. 2006) have yet to eventuate; an expert assessment recently determined that between one-third and two-thirds of marine eukaryotic species may be undescribed and previous estimates of there being more than one million such species appear highly unlikely (Appeltans et al. 2012). A recent review of New Zealand's marine biogenic habitat revealed the lack of knowledge on biodiversity associated with such habitats and indicated the potential to increase marine biodiversity records in the New Zealand region (Anderson et al. 2019).

18.3.2 MECHANISMS FOR INCREASING MARINE BIODIVERSITY RECORDS

Scientific research has provided information about the predicted distribution and abundance of marine biodiversity in some areas of New Zealand's coasts and oceans. Advances in the marine protection of the Ross Sea Region have been made and available information has been used to assess habitat types at greatest risk from disturbance, particularly fishing (Clark & Rowden 2009, Clark & Tittensor 2010, Hewitt et al. 2011a, 2011b, Floerl & Hewitt 2012). Many ecosystems within New Zealand waters remain poorly sampled however, and the efficacy of current spatial protection measures for biodiversity in New Zealand is unknown. Further, the proportion of different marine habitat types that should be or can be protected to maintain a healthy aquatic environment is also unknown (Lundquist et al. 2015).

A number of initiatives have been supported by Fisheries New Zealand and its predecessors to meet the goals of the former NZBS. Commitments included the creation of NABIS (the National Aquatic Biodiversity Information System; http://www.nabis.govt.nz), for making data on marine biodiversity more accessible. DOC surveys and monitors aspects of marine biodiversity, particularly in marine reserves and in relation to protected and threatened marine species. Periodic marine 'BioBlitzes' around New Zealand yield surprising numbers of new species, even in well-studied areas, have the advantage of engaging children, parents, and teachers in discovery, and involve research scientists who then describe the new taxa (e.g., Harper et al. 2009). More of these citizen science projects can be found under the MBIE funded Curious Minds programme (https://www.curiousminds.nz). The museums of Auckland, Canterbury, Otago, and the Museum of New Zealand (Te Papa Tongarewa) also conduct biodiversity sampling expeditions and national collections of specimens have been set up within museums and at NIWA (https://niwa.co.nz/our-services/online-services/nic).

Marine biodiversity research in New Zealand is largely supported through public good funding and is conducted in both universities and CRIs. Both have contributed to New Zealand's high profile for marine biodiversity on the international scientific network through participation in global initiatives such as the Census of Marine Life (CoML) (http://www.coml.org), as well as to local programmes that have improved understanding of the role of biodiversity in the marine ecosystem.

In 2010, New Zealand contributed to the CoML, an unprecedented global collaboration among researchers from more than 80 nations to assess and explain the diversity, distribution, and abundance of life in the oceans. NIWA scientists were part of the team that led CenSeam (http://www.coml.org/global-census-marine-life-seamounts-censeam), the seamount component of CoML, and New Zealand scientists played significant roles in a number of other CoML programmes. The New Zealand International Polar Year-Census of Antarctic Marine Life (IPY-CAML) voyage to the Ross Sea in 2008 was also a major contribution to CoML (O'Driscoll 2009).

The CoML-facilitated activities led to better assessments of global marine biodiversity, resulting in an increase in the total number of known marine species by about 20 000, from 230 000 in 2000 to about 250 000 in 2010. The digital archive (the Ocean Biodiversity Information System OBIS; http://www.iobis.org) has now grown to almost 80 million distribution records, and the Census helped to create the first comprehensive list of the known marine species and facilitated an initiative which resulted in a web-based database of over 80 000 species, the Encyclopaedia of Life (http:/www.eol.org).

A summary of the overall state of knowledge about marine biodiversity by Costello et al. (2010) places New Zealand 6th out of 18 national regions based on the collective knowledge assembled by CoML National and Regional Implementation Committees (NRIC) and comparison of the Spearman rank correlation coefficients between known diversity (total species richness, alien species, and endemics) and available resources, such as numbers of taxonomic guides and experts.

18.3.3 THREATS TO BIODIVERSITY

Understanding of New Zealand's coastal marine environment and its land-sea interactions has progressed since the launch of the NZBS, although knowledge about the state of the marine environment and marine biodiversity at a national scale remains limited (Lundquist et al. 2014). Recent knowledge about New Zealand's and the Ross Sea's marine biodiversity suggests that it may generally be in better shape than that of many other countries (Costello et al. 2010, Gordon et al. 2010). New Zealand was placed 12th out of 18 regions in terms of overall threat levels to biodiversity, overfishing, and alien species invasion (first place being the most threatened). Habitat loss and ocean acidification were identified as the biggest threats to marine biodiversity and marine habitats in New Zealand (Costello et al. 2010, MacDiarmid et al. 2012).

Key marine environment and biodiversity related stressors are outlined in Environment Aotearoa 2015 (Ministry for the Environment & Statistics New Zealand 2015). New Zealand's marine biodiversity is affected by many uses of the marine environment, including fishing, aquaculture, shipping, petroleum and mineral extraction, renewable energy, tourism, and recreation (Royal Society of New Zealand 2012). Impacts from changing land use, including agriculture, urban run-off, and coastal development can also affect marine biodiversity through processes such as sedimentation and eutrophication (Morrison et al. 2009, Larned et al. 2018). The most conspicuous impacts in the marine environment include habitat degradation and reductions in mussels, cockle and scallop beds, seagrass meadows, bryozoan and tubeworm mounds, sponge gardens, kelp forests, and biogenic habitats provided by these organisms (Larned et al. 2018). There are ongoing concerns about the decline of some key species (Ministry for the Environment 2016), localised impacts on habitats and conditions (Thrush & Dayton 2002, Cryer et al. 2002, Clark et al. 2010a, 2010b, Gordon et al. 2010), and emerging threats to the marine environment (MacDiarmid et al. 2012).

The past 750 years of human activity has impacted on marine environments. For example, depletion of fur seals and sea lions occurred from the earliest days of human settlement, not just with European arrival (Smith 2005, 2011). There was also a pulse of sedimentation coinciding with the initial clearance of 40% of New Zealand forests within 200 years of Polynesian settlement (McWethy et al. 2010). Impacts have occurred in remote areas, as well as near population centres, and to sea depths in excess of 1000 m (Carroll et al. 2014, MacDiarmid et al. 2014, 2016a, 2016b, 2018, Maxwell & MacDiarmid 2016, Pinkerton et al. 2015, Fisheries New Zealand 2018).

Although New Zealand has reasonable archaeological, historical, and contemporary data on the decline in abundance of some individual marine species, current trends in the status of New Zealand's marine biodiversity are difficult to determine for several reasons. These include a lack of both pre-disturbance baseline and recent information, and a lack of a nationally coordinated approach to assessing and monitoring marine biodiversity (Lundquist et al. 2014). A multidisciplinary study 'Taking Stock' that examined records from two locations in New Zealand's coastal marine shelf system provided significant insight into marine biodiversity changes over a 1000 year period since early human settlement (MacDiarmid et al. 2016a). Evidence from local ecological knowledge, archaeological digs, museum archives, fisheries records, natural history records, and other sources, shows that top predators, particularly marine mammals and some species of shellfish and fish have taken a major hit in abundance and distribution since human settlement, but that intermediate components of the food chain remain largely intact.

Threat status and susceptibility of biodiversity to capture by fishing activity is discussed in other chapters for marine mammals, seabirds, and other protected species (see Chapters 4–8). Progress has been made on evaluating threats and risks to the marine environment and components within it (e.g., Currey et al. 2012, MacDiarmid et al. 2011, 2012, 2014, Ministry for Primary Industries 2013; Ministry for Primary Industries & Department of Conservation 2013, Larned et al. 2018) and some of these have been followed up with a Spatially Explicit Risk Assessment (Richard et al. 2017, 2020). Marine mammals, seabirds, and reptiles are regularly assessed using the New Zealand Threat Classification System (NZTCS). However, most fish and invertebrates have not been assessed (reviewed by Lundquist et al. 2014). For example, only a small fraction (less than 5%) of New Zealand's marine invertebrate fauna have been evaluated for their threat status and many taxa remain 'data deficient' or unlisted (Freeman et al. 2010, 2013). These are important assessments about the state of knowledge and can help to focus where attention is needed across our marine biota.

MacDiarmid et al. (2012) undertook an expert assessment of the impact of 65 potentially hazardous human activities on 62 identifiable marine habitats in New Zealand's Territorial Sea and 200 nautical mile Exclusive Economic Zone (EEZ). Experts concluded that many of the biggest threats stemmed from human activities outside the marine environment itself. The two biggest threats identified by participants were ocean acidification and ocean warming. Seven other threats deriving from global climate change all ranked in the top 20 threats indicating the importance of global climate change to New Zealand's marine ecosystems.

Climate change can have an adverse impact on the spatial patterns of marine biodiversity and ecosystem function through changes in species distributions, species mix, and habitat availability, particularly at critical stages of species life histories (Lundquist et al. 2011, Poloczanska et al. 2013, Poloczanska et al. 2016). Understanding the dynamics of climate change and predicting the impacts on food webs and fisheries productivity has improved and is a substantial research topic in many parts of the world (e.g., Brown et al. 2010, Blasiak et al. 2017, Phillips & Perez-Ramirez 2017), including New Zealand (e.g., CARIM; http://www.carim.nz). Recently, in New Zealand, Cummings et al. (2021) examined the likely effects of climate change on the New Zealand seafood sector, and this is described further in section 18.4.7.

Understanding the resilience of biodiversity and its rate of change in response to the cumulative effects of multiple stressors across large spatial scales (e.g., ocean acidification, temperature increase, and oxygen depletion) remains an ongoing topic of investigation. A study of global patterns of climate change impacts predicted dramatic species turnovers of over 60% of present biodiversity by 2050. This was based on model projections of changes in the distributional ranges of a sample of 1066 exploited marine fish and invertebrates using a newly developed dynamic bioclimate envelope model (Cheung et al. 2009).

The study showed that climate change and other stressors may lead to numerous local extinctions in the sub-polar regions, the tropics, and semi-enclosed seas, as well as ecological disturbances that potentially disrupt ecosystem services.

The loss of marine biodiversity and loss of functionality associated with climate change and ocean acidification are of increasing concern worldwide (e.g., Guinotte et al. 2006, Ramirez-Llodra et al. 2011), as well as in New Zealand (e.g., Royal Society of New Zealand 2012). In late June 2011, two science-based reports heightened concerns about the critical state of the world's oceans in response to climate change. One focused on the potential impacts of ocean acidification on fisheries and higher trophic level ecology and took a modelling approach to scaling from physiology to ecology (Le Quesne & Pinnegar 2012), and the other assessed the critical state of the world's oceans in relation to climate change and other stressors (Rogers & Laffoley 2011). Implications of global climate change have been further summarised by the Intergovernmental Panel on Climate Change (IPCC 2018).

Global-scale threats associated with the potential effects of ocean acidification on microbial diversity and their roles in biogeochemical processes have yet to be quantified but could have EEZ-wide implications (Bostock et al. 2012). The growing arrival of non-indigenous (sometimes invasive) marine species is also a threat to local biodiversity (e.g., Cheung et al. 2009, Coutts & Dodgshun 2007, Cranfield et al. 2003, Gould & Ahyong 2008, Russell et al. 2008, Williams et al. 2008).

Last year, United Nations officials reported that the world's nations largely failed to achieve the Aichi targets of the previous global agreement on biodiversity, made in 2010 https://news.un.org/en/story/2020/09/1072292.

18.3.4 NATIONAL BIODIVERSITY LEGISLATION, REGULATIONS, AND POLICIES

A broad range of legislation, regulations, and policies are in place to manage and regulate uses of the marine environment, to protect marine biodiversity, to improve management of the coastal and marine environment and to meet worldwide consumer demands for improved sustainability. The development and implementation of fisheries management is one of the goals for Te Mana o te Taiao and for the Ocean Fisheries Workplan under the Oceans and Fisheries Ministerial Portfolio. It is also a recommendation from the 'The future of commercial fishing in Aotearoa New Zealand' prepared by the Prime Minister's Chief Science Advisor https://www.pmcsa.ac.nz/topics/fish/

Managing fisheries in a more holistic way through Ecosystem Based Fisheries Management (EBFM) has been signalled by Fisheries New Zealand through National Fisheries Plans as a priority to make progress on over the next 3 –5 years.

However, there is currently no integrated national oceans policy, and ocean management is fragmented across agencies and statutes (Lundquist et al. 2016; Davies et al. 2018a). Initiatives and research funded through the Sustainable Seas National Science Challenge are developing a framework for cumulative effects management across land and sea resource management sectors, industries, and communities — an internal component of an Ecosystems Based Management (EBM) approach. A further Sustainable Seas project has examined whether current legislation, policy, and institutional practice are sufficient to enact EBM, and what additional approaches could be acted on to further enhance EBM.

Marine spatial planning and marine reserves have allowed for stakeholder driven processes to inform marine management and allocation of space for marine protection (Davies et al. 2018b) and other uses of the marine environment (e.g., aquaculture). The Hauraki Gulf Forum initiated the Sea Change – Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan in 2013, taking a holistic and integrated approach to marine management and restoring the marine productivity in the area (http://www.seachange.org.nz), with a recent announcement from government to support implementation of the plan (November 2018). MPI commissioned a small knowledge review to help inform possible restoration efforts for the Hauraki Gulf (Morrison 2021).

A Marine Protected Areas Science Advisory Group (MSAG) was established by MfE in 2016. The MSAG comprises marine science staff from DOC, Fisheries New Zealand, and MfE. The role of the MSAG is to discuss, commission, undertake, and review science — to build the scientific foundation to enable the implementation of a representative network of marine protected areas and advise on emerging issues relating to marine protected areas and spatial management.

Recent work undertaken by the MSAG includes: the development of a New Zealand Seafloor Community Classification; guidance for the use of decision-support tools for identifying optimal areas for biodiversity conservation; collation of data for mapping key ecological areas in the New Zealand marine environment; principles for MPA network design in New Zealand and a review of the scientific basis; and application of quantitative targets for marine protection.

Reports associated with work are available through the DOC Marine Protected Areas Research Programme webpage: https://www.doc.govt.nz/our-work/mpas-research-programme/

18.3.5 INTERNATIONAL BIODIVERSITY OBLIGATIONS

As mentioned above, it has been widely recognised that the Convention on Biological Diversity's (CBD) 2020 Global Biodiversity Framework (including the Aichi targets) did not progress sufficiently and the threats to biodiversity and the extinction of species have continued throughout the period.

The UN Environment Management Group identifies five main direct drivers of biodiversity loss and degradation: changes in the use of land and sea; over-exploitation of biodiversity; climate change; pollution; and invasive alien species.

Today, with a decade left to achieve the Sustainable Development Goals (SDGs), the 2020 SDGs Report (https://unstats.un.org/sdgs/report/2021/) paints a daunting picture, one worsened by the COVID-19 pandemic addition. Most of the SDGs related to biodiversity and environmental sustainability will not be achieved under current trajectories, posing a risk to the overall achievement of the 2030 Agenda. https://unemg.org/wp-content/uploads/2021/04/EMG-Biodiversity-WEB.pdf

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) was established in 2012 and provides a mechanism to assess the state of the planet's biodiversity, its ecosystems, and the essential services they provide to society (http://www.ipbes.net). This international platform is similar in function to the Intergovernmental Panel on Climate Change (IPCC) in terms of bringing together international expertise and will review information on the provisioning of biodiversity for ecosystem services, stimulate science and innovation on this research topic, and interact with national and international management agencies to integrate IPBES results into policy and management.

The 2019 IPBES Global Assessment Report on Biodiversity and Ecosystem Services (IPBES 2019) states that around 25 per cent of species in assessed animal and plant groups are threatened, suggesting that some one million species already face extinction, and also warns that current biodiversity loss is increasingly putting human security and well-being at risk. This also echoes similar calls from the IPCC in its special report on Global Warming of 1.5 °C (IPCC 2018), the European Environment Agency's (EEA) Sustainability transitions: policy and practice, the CBD's fifth Global Biodiversity Outlook (GBO-5), UNEP's Global Environment Outlook (GEO-6), and FAO's report on The State of the World's Biodiversity for Food and Agriculture (FAO 2019). The full text of the First Global Integrated Marine Assessment, conducted by some of the world's foremost experts on ocean issues for policymakers, was released online in 2016 and stresses the need to act immediately to halt the degradation of the world's oceans and marine biodiversity (https://www.unep.org/resources/report/first-global-integrated-marine-assessment-world-ocean-assessment-i).

To address both the biodiversity crisis and the climate crisis, there is growing scientific research suggesting that half of the planet must be kept in a natural state. Some papers have suggested that the number should be even higher, and some have suggested slightly lower. Despite this, experts agree that a scientifically credible and necessary interim goal is to achieve a minimum of 30% protection by 2030 (https://www.hacfornatureandpeople.org/why-30x30) and one study has assessed where protection would be most effective to achieve this goal (Zhao et al. 2020).

The Convention on Biological Diversity is developing new targets for the decade to 2030 which will replace the Aichi targets set for 2011–2020 and will also lead towards the longer 2050 goal: living in harmony with nature. These targets may inform any future updates to Te Mana o te Taiao, New Zealand's new Biodiversity Strategy which informs the workstreams of Fisheries New Zealand's Biodiversity Research Programme.

Most of New Zealand's commercial fisheries are wild-caught and continuation of their productivity is dependent on the retention of a healthy functioning marine ecosystem. The 'licence to operate' is mandated by the Fisheries Act 1996 and requires strict compliance with sustainable and environmentally responsible use of fish stocks. Compliance is also required with other legislation such as the Marine Mammals Protection Act 1978 and a range of international obligations such as the United Nations Convention on the Law of the Sea (UNCLOS). Under the Quota Management System, considerable monitoring of fishing activity and the environmental footprint of commercial operators is required.

18.3.6 CURRENT CHALLENGES AND AGENDAS

The economy of the sea is a significant part of the overall economy in New Zealand and has potential for growth, particularly in aquaculture, oil and gas, and minerals (Ministry of Business, Innovation and Employment 2017). It is important that the aquatic environment and biodiversity are not adversely affected by new or increasing activities, be they in the seafood sector or other natural resource industries (Fisheries Act 1996, Exclusive Economic Zone and Continental Shelf (Environment Effects) Act 2012).

The large-scale threats to the marine environment and biodiversity include climate change and ocean acidification, increasing exploitation of resources (living or non-living), and the cumulative effect of multiple uses of the marine environment (e.g., renewable energy, commercial fisheries, recreational fisheries, aquaculture, and hydrocarbon and mineral extraction) (Kingsford et al. 2009, Lundquist et al. 2011). These threats are increasingly being recognised in policy and government circles (e.g., Office of the Prime Minister's Science Advisory Committee 2013; Statistics New Zealand 2013; Royal Society of New Zealand 2012; Capson & Guinotte 2014).

Despite this recognition, progress on tackling marine-related climate change effects and investment in long-term monitoring of the marine environment and ready access to data remains slow (although see Table 18.3 for a number of data-collecting initiatives that are contributing to long-term historical datasets, noting that these are primarily of environmental drivers rather than measures of biodiversity itself). Long-term monitoring and environmental reporting has been recognised as a major gap by the government and in September 2015 the Environmental Reporting Act (2015) was passed into law. The new framework for environmental reporting divides the environment into five environmental domains. Under each domain three main types of information are reported on: pressures, states, and impacts. 'Our marine environment 2016' was the first report released under the new legislation (Ministry for the Environment 2016). In 2015, a number of challenges were identified in relation to taxonomy and the maintenance of biological collections and databases in New Zealand (Nelson et al. 2015). Recommendations from this report focused on increasing support for the discovery, documentation, and accessibility of biodiversity information. This was followed by the decadal plan for taxonomy and biosystematics in Australia and New Zealand 2018–2028 (https://www.royalsociety.org.nz/assets/Uploads/Discovering-Biodiversity-decadal-plan.pdf) which echoed the recommendations of the New Zealand report. Increasing the capacity and capability of taxonomy in New Zealand would make significant progress towards the successful delivery of TMOTT.

18.3.7 GLOBAL MONITORING AND INDICATORS FOR MARINE BIODIVERSITY

There are numerous schemes within and between nations to monitor the marine environment, including physical, chemical, and biological components (Table 18.3). A challenge for Fisheries New Zealand, other government agencies, and for New Zealand is how to assimilate any or all of the above monitoring approaches and international datasets to assess the nature and extent of biodiversity change, and to assess the effectiveness of management measures to protect or enhance biodiversity or halt its decline.

Table 18.3: Examples of international marine global monitoring datasets.

Name	Description	Website
Global Ocean Observing System (GOOS)	Permanent global system for observations, modelling, and analysis of marine and ocean variables.	http://www.goosocean.org/
Deep Ocean Observing Strategy (DOOS)	This is the deep-sea component of GOOS.	https://deepoceanobserving.org/
Global Climate Observing System (GCOS)	United Nations-ratified programme which regularly assesses the status of global climate observations and produces guidance for its improvement.	https://gcos.wmo.int/
Southern Ocean Observing System (SOOS)	International initiative of the Scientific Committee on Antarctic Research (SCAR) and the Scientific Committee on Oceanic Research (SCOR), officially launched in 2011 and hosted by the Institute for Marine and Antarctic Studies (IMAS) at the University of Tasmania, Australia.	http://www.scar.org/soos
ARGO	International deepwater monitoring system of free-floating buoys that are part of the integrated global observation strategy. New Zealand makes a significant contribution to ARGO floats in the Pacific Ocean.	http://www.argo.net/ https://www.niwa.co.nz/coasts-and-oceans/research-projects/argo-floats
Continuous Plankton Recorder (CPR) Surveys	CPR surveys collect data about the spatio-temporal patterns of zooplankton and then use the sensitivity of plankton to environmental change as an early warning indicator of ocean health. CPR originated in the North Atlantic and the North Sea in 1931. New Zealand BRAG-funded projects contribute to the SCAR Southern Ocean CPR data collection established in 1991 by the Australian Antarctic Division to map the Southern Ocean.	https://www.cprsurvey.org/
Australia's Integrated Marine Observing System (IMOS)	IMOS, established in 2007, is designed to be a fully integrated national array of observation equipment to monitor the open oceans and coastal marine environment around Australasia, covering physical, chemical, and biological variables. All IMOS data are freely and openly available.	http://imos.org.au
Oceans 2025	This 5-year programme, now completed, was an initiative of the Natural Environment Research Council (NERC) funded Marine Research Centres to address environmental issues that require sustained long-term observations.	https://imber.info/science/endorsed-projects/completed-endorsed-prodjects/strategic-marine-research-programme-of-the-uk-natural-environment-research-council-oceans-2025/
Global Ocean Acidification Observing Network (GOA-ON)	An existing global ocean carbon observatory network of repeat hydrographic surveys, time-series stations, floats and glider observations, and volunteer observing ships, with participation from scientists from over 30 countries.	http://www.goa-on.org/
NZ Ocean Acidification monitoring Network (NZOA-ON)	Initiated in 2014 by NIWA and the University of Otago, this programme now has 11 sampling sites around New Zealand and is part of the Ocean Acidification Alliance.	https://www.niwa.co.nz/coasts-and-oceans/research-projects/new-zealand-ocean-acidification-observing-network-nzoa-on https://www.oaalliance.org/current-members

18.4 OVERALL PROGRESS IN FISHERIES NEW ZEALAND MARINE BIODIVERSITY RESEARCH

18.4.1 HISTORICAL BIODIVERSITY RESEARCH

To date, 95 research projects have been commissioned. Early studies focused primarily on reviews, identification guides, habitat and community characterisations, and revised taxonomy for certain groups of organisms. Later studies have included large collaborative ship-based surveys that have contributed to improved seabed classification in New Zealand waters and the exploration of new habitats in the region and in Antarctic waters.

Progress in the BRAG research programme has increased in conceptual complexity from simply cataloguing biodiversity to an increasingly complex understanding of environmental drivers and the functionality of biodiversity, and ultimately towards the development of standards and the protection of biodiversity. A full list of projects can be obtained from Appendix 19.6 at the back of the full document.

Greatest progress has been made in the shallower inshore parts of the marine environment, not least because of cost and ease of access. However, by leveraging from existing offshore projects, significant progress has also been made to depths of 1500 m. In recent years, biodiversity research based in Antarctica has lagged behind EEZ-based research, simply because of the difficulty in securing additional funding to access and work in such a remote marine environment.

Over time, the complexity and scale of studies has increased, with projects on the functional ecology of marine ecosystems ranging from localised experimental manipulation to broad-scale observations across hundreds of square kilometres. Such studies have also prompted the development of improved measures of biodiversity and indicators. One study reviewed genetic variation in the New Zealand marine environment and conducted field observations on several species to examine genetic variation across latitudinal gradients. Another study on changes in shelf ecosystems over the past 1000 years has yielded insights into the effects of long-term climate change, land-use effects, and fishing, on marine ecosystems; and some studies have begun to address the effects of ocean acidification on marine biodiversity.

18.4.2 ALIGNED MARINE BIODIVERSITY RESEARCH

In New Zealand, a number of marine research projects commissioned by MPI have contributed to the state of knowledge and management of marine biodiversity. The ‘Marine Futures’ programme (2012–14) investigated decision-making frameworks for ocean management and developed new tools for enabling participation of all stakeholders (public, iwi, industry, government), to facilitate economic growth, improve marine stewardship and ensure that cumulative stresses placed on the environment do not degrade the ecosystem beyond its ecological adaptive capacity (MBIE project code C01X1227). The ‘Ross Sea Climate & Ecosystem’ Programme (concluded in 2016) modelled likely future changes in the physical environment of the region and potential consequences of these changes on the ecosystem in terms of functional links between the environment and the marine food web (MBIE project code C01X1226). ‘Management of offshore mining’ (concluded in 2016) developed a clear framework that will guide appropriate and robust environmental impact assessments and the development of integrated environmental management plans for the marine mining sector, other resource users, and resource management agencies (MBIE project code C01X1228). ‘Resilience of benthic communities to the effects of sedimentation’ (2016-2021) assessed the impacts on and recovery of benthic communities to physical disturbance and sedimentation that could result from future deep-sea mining and bottom-contact fishing (MBIE project code C01X1614). ‘Juvenile fish habitat bottlenecks’ is currently assessing the restrictions on juvenile fish resulting from degraded biogenic habitats and seeks to promote actions that remove these restrictions with the ultimate aim of seeing more juvenile fish enter the adult populations (MBIE project code C01X1618).

The MBIE-funded Climate Change Impacts and Implications project had a coastal and an ocean case study and released a synthesis report upon its conclusion in 2016 (Law et al. 2016). The MBIE-funded CARIM project ‘Coastal Acidification: Rate, Impacts & Management’ that Fisheries New Zealand has contributed to was completed in 2020 (http://www.carim.nz/).

One of the largest marine research developments in recent times was the launch of the MBIE-funded National Science Challenge ‘Sustainable Seas’. Sustainable Seas aims to enhance the utilisation of New Zealand’s marine resources within environmental and biological constraints (https://sustainableseaschallenge.co.nz/). Achieving this aim will require a new way of managing the many uses of our marine resources that combines the aspirations and experience of Māori, communities, and industry with the evidence of scientific research to transform New Zealand into a world leader in sustainable marine ecosystem-based management.

Phase 2 of the challenge was launched in 2020 and this incorporates closer links between science, policy, and Treaty Partners and includes case studies for EBFM as well as end-users, particularly with regard to fisheries. Current projects are investigating cumulative effects on biodiversity and ecosystem functioning, developing ecosystem indicators in the Hauraki Gulf for EBFM, assessing shellfish habitat and potential restoration in the Marlborough Sounds, and mapping marine environmental stressors in Hawke's Bay.

International partnerships are also being leveraged to support BRAG research priorities (Table 18.4), and other research relevant or specifically linked to BRAG research workstreams are listed in Table 18.5.

Table 18.4: Current New Zealand/International partnerships.

BAYESIANMETAFLATS – Spatial organisation of species distributions: hierarchical and scale-dependent patterns and processes in coastal seascapes (completed)	https://cordis.europa.eu/result/rcn/185059_en.html
Chess – Biogeography of Deep-Water Chemosynthetic Ecosystems	http://www.coml.org/projects/biography-deep-water-chemosynthetic-ecosystems-chess.html
INDEEP – International Network for Scientific Investigation of Deepsea Ecosystems (Now incorporated into Deep Ocean Stewardship Initiative)	http://www.indeep-project.org/ https://www.dosi-project.org/
PHARMASEA – Increasing Value and Flow in the Marine Biodiscovery Pipeline	http://www.pharma-sea.eu/
UN OCEAN DECADE (2021-2030)	https://tetiniatangaroa.org.nz/

18.4.3 PROGRESS ON CHARACTERISATION AND CLASSIFICATION OF BIODIVERSITY

The characterisation and classification of biodiversity requires an assessment of the abundance and distribution of marine life. Building on earlier research to map fish and squid species (Anderson et al. 1998, Bagley et al. 2000) and the biodiversity of the New Zealand ecoregion (Arnold 2004), numerous literature reviews (e.g., Nelson et al. 2019, Twist et al. 2019), taxonomic studies, and habitat mapping surveys have been undertaken, including 'The New Zealand Inventory of Biodiversity' (Gordon 2009, Gordon 2010, Gordon 2012). Field identification guides have also been published by the Ministry of Fisheries on deep sea invertebrates (projects ENV2005-20 and ZBD2010-39, Tracey et al. 2005, 2007, 2011a), macroalgae (Nelson 2020), bryozoans (project IPA2009/14, Smith & Gordon 2011), and on fish species (IDG2006-01, McMillan et al. 2019), which further contribute to the accurate monitoring and identification of biodiversity in New Zealand waters.

Several hundred new species of marine organisms have been discovered, and the known range of species extended, through exploratory surveys such as the NORFANZ project ZBD2002-16 (Clark & Roberts 2008); the Ministry of Science and Innovation (MSI) Seamount Programme, mainly commissioned through public-good science, supplemented by Ministry of Fisheries projects ZBD2000-04 (e.g., Rowden et al. 2002, 2003), ZBD2001-10 (Rowden et al. 2004), ZBD2004-01 (Rowden & Clark 2010), and Ministry projects ENV2005-15, ENV2005-16 (Clark et al. 2010a, Rowden et al. 2008) and the Ocean Survey 20/20 programme (Clark et al. 2009); inshore surveys of bryozoans at Tasman Bay ZBD2000-03 (Grange et al. 2003); Farewell Spit ZBD2002-18 (Battley et al. 2005); Fiordland ZBD2003-04 (Wing 2005); coralline algae ZBD2001-05, ZBD2004-07 (Harvey et al. 2005, Farr et al. 2009, Opresko et al. 2014) and other deep sea invertebrates (Tracey et al. 2011a, Williams et al. 2014); soft sediment environments ZBD2003-08 (Neill et al. 2012); rhodolith community study ZBD2009-03 (Nelson et al. 2012, 2014); offshore surveys of the Chatham Rise and Challenger Plateau funded through whole-of-government Ocean Survey 20/20 Programme, ZBD2006-04 (Nodder 2008) and ZBD2007-01 (Nodder et al. 2011), ZBD2012-03 (Hewitt et al. 2011a, 2011b, Bowden 2011, Bowden & Hewitt 2012, Bowden et al. 2011a, 2011b, 2011c, 2014, Compton et al. 2012). Other national efforts included the Biosecurity New Zealand mapping projects (Beaumont et al. 2008, 2010).

Table 18.5: Other aligned research. More information on Fisheries New Zealand projects listed below can be found in Appendix 19.6 of the full document.

Fisheries New Zealand	HAB2007-01 Biogenic habitats as areas of particular significance for fisheries management (complete). ZBD2006-02 NABIS (ongoing). KAI2016-05 Rocky reef impact quantification and monitoring for the Kaikōura earthquake (complete). KAI2020-01 Monitoring of rocky reef habitats across degrees of uplift caused by the 2016 Kaikōura earthquake (underway). This is effectively a continuation of KAI2016-05. SEA2020-14 Pelorus multibeam ground-truthing and analysis (complete). BEN2018-03 Automated image analysis for habitat classification and species distribution investigation (complete). Taxonomic identification of benthic samples (DAE2018-04 completed; BEN2021-03 underway). Useful data related to defining potential vulnerable marine ecosystems (VMEs) are collected by Fisheries New Zealand scientific fisheries observers working on New Zealand authorised fishing vessels that operate on the high seas in the South Pacific.
NIWA Marine platform and MBIE funding	NIWA Strategic Science Investment Funding (SSIF) **Coasts & Estuaries Centre** programmes: Programme 1 – Sea level rise and coastal hazards Programme 2 – Ecosystem Services Programme 3 – Aquaculture environmental effects Programme 4 – Marine biosecurity Programme 5 – Catchment to Coast Connectivity NIWA Strategic Science Investment Funding (SSIF) **Oceans Centre** programmes: Programme 1 – Oceanic ecosystem structure and function Programme 2 – Oceanic physical processes Programme 3 – Ocean Atmosphere Interactions Programme 4 – Marine Geological processes and Habitat characterisation NIWA Strategic Science Investment Funding (SSIF) **Coasts, Estuaries & Oceans Cross-Centre** programme: Programme 1 – Biological Resources (Documenting biota, spatial tools, SDM, etc.) NIWA Strategic Science Investment Funding (SSIF) **Fisheries Centre** programme: Programme 1 – Stock monitoring & assessment Programme 2 – Fisheries and ecosystems Programme 3 – International Fisheries NIWA Strategic Science Investment Funding (SSIF) **Aquaculture Centre** programme: Programme 1 – Aquaculture production species MBIE funding: C01X1618 - Removing fisheries juvenile habitat bottlenecks C01X1710 - RAMPing-up protection of the Ross Sea (RossRAMP) C01X1913 - Novel high-tech underwater selection tools for environmentally and economically sustainable fishing C01X2107 Transforming coastal lowland systems threatened by sea-level-rise into prosperous communities. C01X2108 Transforming scallop fishing: Non-destructive surveying and harvesting for economic acceleration and kaitiakitanga (Smart Idea) C01X2109 Carbon sequestration via NZ's estuarine sediments: Implications for GHG budgets (Smart Idea) C01X2110 RotoTurf – seeding freshwater restoration (Smart Idea)
Central government	MEC development and application to MPAs, Regional surveys; refined habitat suitability modelling for protected coral species in the New Zealand EEZ has been undertaken along with the development of a pilot ecological risk assessment for protected corals. DOC – Delineation of Key Ecological Areas DOC – New Zealand Seafloor Community Classification DOC – Development of a thematic habitat classification DOC – Deep reef biodiversity DOC – Surfacing of geospatial biodiversity data DOC – Climate change impacts on biodiversity DOC – Refined habitat suitability modelling for protected coral species in the New Zealand EEZ DOC – Development of a pilot ecological risk assessment for protected corals DOC – Identification of protected coral hotspots using species distribution modelling DOC – Coastal Sediment Research Priorities Report DOC – New National Marine Receiving Environment Sediment Tool

	DOC – National analysis of 18 years of satellite plume imagery to detect offshore hotspot increases and reductions DOC – Marine reserve monitoring: Key species monitoring (BUV, UVC, intertidal and subtidal quadrats) or (Reef fish, pāua, kina, kōura, intertidal communities) DOC – Marine reserve monitoring: Habitat monitoring (multibeam, drop camera, UVC) or (biogenic habitats, habitat maps) DOC - Marine reserve monitoring: Ocean acidification monitoring DOC – Spyfish Aotearoa: a citizen science and machine learning project to analyse BUV data

Research in the Ross Sea Region (BioRoss projects) have also generated records of new species including MPI projects ZBD2000-02 (Page et al. 2001), ZBD2001-03 (Norkko et al. 2002), ZBD2002-02 (Sewell et al. 2006, Sewell 2005, 2006), ZBD2003-02 (Cummings et al. 2003, 2006b), ZBD2003-03 (Rowden et al. 2012a, 2013), ZBD2005-03 (MacDiarmid & Stewart 2012), ZBD2006-03 (Cummings et al. 2003, 2006a), ZBD2008-23 (Nelson et al. 2010), and IPY2007-01 (Bowden et al. 2011a, Clark et al. 2010b, Eakin et al. 2009, Hanchet 2009, 2010, Hanchet et al. 2008a, 2008b, 2008c, 2008d, 2013, Koubbi et al. 2011, Lörz et al. 2009, Mitchell 2008, O'Driscoll 2009, O'Driscoll et al. 2011, 2012, O'Loughlin et al. 2011).

The development of the Marine Environment Classification or 'MEC' (Snelder et al. 2006) was an important step in the delineation of areas with similar environmental attributes in the offshore environment. However, significant environmental drivers of variability in marine biodiversity, such as substrate type for seafloor organisms, were absent from the classification. In 2005, DOC and Ministry of Fisheries jointly commissioned a project to optimise the MEC using fish distribution data. This project (ZBD2005-02) demonstrated a substantial improvement in the MEC classification for offshore habitats (Leathwick et al. 2006a, 2006b, 2006c). In addition, the Ministry implemented spatial management tools (Benthic Protection Areas; https://www.mpi.govt.nz/protection-and-response/sustainable-fisheries/protected-areas/benthic-protection-areas), on the basis of the Marine Environment Classification (Snelder et al. 2006) to address broader statutory responsibilities on the environmental effects of fishing on biodiversity.

In 2006, three projects to map coastal biodiversity were completed in the Coromandel scallop, Foveaux Strait oyster, and southern blue whiting fisheries around the sub-Antarctic islands as part of fishery plan development for these fisheries (ZBD2005-04, ZBD2005-15 and ZBD2005-16). These projects found that the biological distribution of organisms and their habitats were not well predicted by the MEC. Ministry project (BEN2006-01) aimed to further optimise the MEC by producing a methodology for a Benthic Optimised MEC (Leathwick et al. 2012). Ecological studies to improve habitat classification and vulnerability indices have also been completed through Aquatic Environment Working Group (AEWG) projects on seamounts (ENV2005-15, ENV2005-16) (e.g., Clark et al. 2010c, 2011), and to supplement other studies funded by the Ministry and MSI (e.g., ZBD2004-01, ZBD2001-10, ZBD2000-04, and CO1X0508).

Distribution maps providing indicative abundance and characterisation of biodiversity are now emerging and have been produced through projects using predictive modelling tools, e.g., Compton et al. 2012, ZBD2010-40; the fish optimised MEC in project ZBD2005-02 (Leathwick et al. 2006a, 2006b, 2006c); the benthic optimised MEC (BOMEC; Leathwick et al. 2012); macroalgal diversity associated with soft sediment habitats ZBD2008-05 (Rowden et al. 2012b); deep sea benthic biodiversity in trench, canyon and abyssal habitats below 1500 m depth ZBD2008-27 (Lörz et al. 2012); and rhodolith distribution and associated biodiversity ZBD2009-03 (Nelson 2009, Neill et al. 2014). A joint project supported by Fisheries New Zealand, DOC, and NIWA on the state of knowledge of deep-sea corals in New Zealand has been recently published (Tracey & Hjorvarsdottir 2019).

18.4.4 WORKSTREAM 1: IWI AND COMMUNITY-BASED RESEARCH

A major theme of Te Mana o te Taiao and a focus for Fisheries New Zealand is collaboration, partnerships, and engagement between Treaty Partners, local communities, whānau, hapū, iwi, and Māori organisations to promote research efforts that connect biodiversity to local knowledge and decision-making. This workstream includes projects related to indigenous/taonga species and aims to improve understanding through multiple scientific disciplines and knowledge systems including mātauranga Māori and citizen science.

Currently, Fisheries New Zealand is in the process of developing research projects under this workstream and future editions of the AEBAR will report on the status of these projects.

18.4.5 WORKSTREAM 2: BASELINE KNOWLEDGE FOR EBFM

Since 2000, much progress has been made towards characterising and quantifying the abundance and distribution of marine biodiversity throughout New Zealand waters. Marine ecosystems are complex and changes in diversity and community composition may be driven by multiple variables. Interactions between variables are likely to be non-linear, with disturbance thresholds and the potential for multiple stable states. As a consequence, it is often difficult to distinguish 'natural' from 'anthropogenic' impacts affecting ecosystem dynamics, and research is needed to help disentangle this complexity. This is particularly so closer to shore where complexity and natural variability seems to be greatest. Furthermore, the concept of 'shifting baselines' requires researchers to be explicit about whether they are referring to a historical or shifted baseline (Soga & Gaston 2018).

Marine ecosystems influence, and are influenced by, a wide array of oceanic, climatic, and ecological processes across a broad range of spatial and temporal scales. Marine communities are generally dynamic, can occur over large areas, and have strong links to other communities through processes such as migration and long-distance physical transport (e.g., of larvae, nutrients, and biomass). Patterns observed on a small scale can interact with larger- and longer-scale processes that in turn result in large-scale patterns. Marine food webs are usually complex and dynamic over time (Link 1999). To distinguish useful descriptors of long-term ecosystem change from short-term fluctuations requires innovative approaches that integrate broad-scale correlative studies from smaller-scale manipulative experiments (Hewitt et al. 1998, 2007).

Recent theoretical and technical advances show great promise toward the goal of understanding the role of biodiversity in ecosystems. Technologies for remote sensing and deep-water surveying, combined with powerful integrative and interpretive tools such as GIS, climate modelling, qualitative ecosystem modelling, and trophic ecosystem modelling, will contribute to the development of an ecosystem-based approach to management (Thrush et al. 1997, 2000), with potential benefits for marine conservation and management. Ecosystem modelling of species distribution (and habitats) with respect to known and projected environmental parameters will improve predictability for both broad- and fine-scale biodiversity distribution. This has already resulted in improved definition of environmental classifications addressing biodiversity assessment. It is also important to make progress in establishing the links between biodiversity and the long-term viability of fish stocks under various harvesting strategies. Further, modellers should consider processes from all ecosystem function perspectives, i.e., top-down effects such as predation (e.g., trophic modelling), bottom-up effects such as the environment (e.g., habitat classification based on environmental variables), and mixed effects. Effective management of marine resources through EBFM requires sound baseline knowledge of New Zealand's marine biodiversity.

For a list of past projects and outputs, please refer to Appendix 19.6. Current and recently completed projects are as follows:

ZBD2020-08 *Role of low- and mid-trophic level fish in the Hauraki Gulf ecosystem* [Underway]

Competition between seabirds and fisheries for prey is a known issue in other parts of the world and concerns have been raised about the potential impact of fisheries on seabirds in the Hauraki Gulf. Some information suggests that there have been decreases in chick survival, reduced body condition and increases in foraging effort for some wide-foraging seabirds, and these changes may be due to changes in the abundance or availability of seabird prey, which include low- and mid-trophic level fish such as anchovy, pilchard, jack mackerel, and mullet.

This project is producing a new food-web model of the Hauraki Gulf, specifically tuned to look at interactions between small pelagic fish and their predators, including seabirds, snapper, and cetaceans. It is intended to support and contribute to Sea Change and will also inform ecosystem-based fisheries management.

ZBD2016-04 *Organic Carbon Recycling in Deepwater* [Completed]

Time incremental sediment traps from long-term moorings in subtropical and Subantarctic waters north and south of the Chatham Rise have opportunistically collected and preserved mobile nektonic crustaceans over 3- to 16-day sampling periods from 2000 to 2012. These data provide a unique opportunity to examine the ecology and functioning of deep-water communities in the 'twilight zone', a habitat that is poorly sampled globally. Throughout this time-series, amphipods have been a conspicuous component of the 'trapped' fauna, with notable variations in abundance and speciation depending on water mass and season. During the project, 43 species of amphipod were identified from the trap samples, with 28 known from previous studies and 15 either new to science or unidentifiable, mainly due to sample preservation issues (Peart & Nodder 2018). About 70–90% of the temporal patterns in amphipod abundance and biomass were accounted for statistically by the combination of several environmental factors, such as organic carbon flux, sea-surface temperature, primary productivity, and Southern Oscillation Index. It is therefore anticipated that amphipod communities in the 'twilight zone' will be impacted by future climate change-related alterations to the physical and chemical characteristics of oceanic waters and marine productivity around New Zealand.

18.4.6 WORKSTREAM 3: TOOLS AND METHODS FOR BIODIVERSITY AND ECOSYSTEM ASSESSMENT

The development of new tools and methods for biodiversity and ecosystem assessment allows us to continually understand more about marine biodiversity and the way ecosystems function. They are also useful in detecting and monitoring changes that may have been unobservable using previous techniques, thus allowing early intervention to address threats to biodiversity.

This workstream promotes the development of tools such as ID guides (hard copy or online; e.g., Identification guides | NIWA), ecological maps or atlases of seabed habitats, ecosystem and biodiversity indicators, and report cards. These tools provide an efficient means of assessing biodiversity, whether that means using ID guides to quickly identify specimens, or using machine learning methods to observe the distribution of marine macroalgae that could be ecological indicators for marine health assessments (e.g., D'Archino et al. 2019, 2021). Emerging technology (e.g., electronic monitoring, cameras on boats) and genetic tools are also included in this workstream.

Genetic biodiversity can be measured directly at the scale of genes and chromosomes or indirectly by measuring physical features at the organism scale (assuming they have a genetic basis). Genetic diversity is fundamental to the long-term survival, stability, and success of a species. Central to this is the 'metapopulation' concept where populations are sufficiently genetically distinct from each other to be identifiable as individual units. A low level of recruitment between populations counters the effects of both random genetic drift and inbreeding depression of genetic diversity.

Human activities can profoundly affect genetic diversity both within populations and between populations. For example, shipping activity (movement across the globe) and aquaculture practices (transfer of organisms to different areas) can increase population connectivity such that genetic biodiversity may decrease between populations. In extreme cases, populations can become the same genetically (homogeneous) although considerable within-population diversity may remain. In the event of increased genetic connectivity, a species may become more susceptible to extinction through biological or catastrophic stochasticity. That is, in the absence of between-population diversity there is insufficient genetic variance to adapt to the effects of climate change, disease epidemics, and other pressures.

In contrast, under the much more common scenario of habitat fragmentation caused by human activities (fishing, pollution), decreased connectivity between populations will result in greater between-population diversity, but a reduction of within-population diversity. This also results in a decrease in a species survival (fitness) because fragmented or isolated populations may become extinct through environmental and genetic stochasticity or localised depletion. Periodic fluctuations in annual temperature for example can lead to small-scale population extinction, which in the absence of recruitment between populations will result, over time, in the demise of all populations.

To reduce the risk of species loss, information about the genetic diversity both within populations (population isolation) and between populations (population connectivity) is needed. Without such information, the effects of perturbation on a species persistence and survival cannot be predicted. Furthermore, the links between genetic diversity, the dispersal capacity (mode of reproduction and life history development) of a species and the minimum viable population (MVP) size required in the marine environment to ensure population persistence, are little understood. For example, the MVP size for a species with a large dispersal capacity is likely to be quite different from that of a species with a relatively restricted dispersal capacity. Examining the connectivity between populations in the marine environment is fundamental to resolving some of the central challenges in ecology and has almost been ignored in the management of New Zealand fisheries and protection of biodiversity.

Understanding marine genetic diversity is also being enhanced through phylogenetic investigations of the relationships of the New Zealand marine biota using molecular sequence data. With some groups of the flora and fauna, genetic data are essential to understanding relationships and species identities. The research undertaken to date has important applications in both the documentation of diversity and in the recognition of foreign taxa (e.g., central to investigations of diversity of coralline algae in New Zealand: ZBD2001-05, ZBD2004-07; recognition of diversity: D'Archino et al. 2011; distinguishing native and foreign taxa: Heesch et al. 2009). Projects have also interpreted genetic population structure of five coastal species (tuatua, pipi, yellowbelly flounder and sand flounder, and scallop) (ZBD2009-10) (Gardner et al. 2010, Constable 2014, Hannan 2014, Hannan et al. 2016, Silva & Gardner 2014), created a DNA database for commercial marine fish (Smith et al. 2008), and assessed the genetic connectivity of deep-sea corals in vulnerable marine ecosystems (Holland et al. 2020).

ZBD2019-11 *Development of Electronic Automated Reporting System (EARS) to improve seabird bycatch monitoring* [Underway]

The current bycatch levels of seabirds, including the Antipodean albatross is poorly known, but international distant-water longline fishing fleets are likely to be the major cause of Antipodean albatross mortality. Other factors may play a role in seabird decline but reducing bycatch mortality will address the most direct human impact on seabirds.

The Western and Central Pacific Fisheries Commission (WCPFC) has jurisdiction over most of the Antipodean albatross foraging area and requires fishers to employ mitigation techniques to reduce seabird bycatch in certain areas. The WCPFC requires the use of at least two of the following mitigation techniques: tori (bird scaring) lines, weighted branch lines (hooks), and setting hooks at night. These mitigation techniques have been shown to reduce seabird bycatch by up to 80-90% if used appropriately (e.g., Melvin et al. 2004).

An Electronic Automated Reporting System (EARS) has been developed which monitors compliance with bycatch mitigation techniques using GPS trackers and RFID tags. The data collected from these devices is automatically sent from the vessel by satellite connection to regulatory authorities to monitor compliance. As part of project ZBD2019-11, trials are currently underway to determine the best configuration of this mitigation monitoring system prior to future widespread use of this robust sensor technology on fishing vessels in international waters.

ZBD2016-11 *Quantifying benthic biodiversity across natural gradients* [Completed]

Quantitative data on the distribution and abundance of benthic species are sparse in New Zealand waters, resulting in high levels of uncertainty in status and trends in benthic biodiversity. This project supported opportunities both for limited field validation of existing models (e.g., those developed from the Chatham-Challenger OS20/20 surveys; Bowden et al. 2021) and development of new abundance-based models. These new models (Bowden et al. 2019a, 2019b) have improved predictive ability to better inform management options, and the data and models generated by the project also informed a benthic risk assessment developed under a separate Ministry project (BEN2014-01; Mormede et al. 2021).

ZBD2019-01 *Quantifying benthic habitats Phase2* [Completed]

At the completion of the Chatham Rise work under ZBD2016-11, Fisheries New Zealand commissioned further work under the *Quantifying Benthic Biodiversity* initiative, with this subsequent project, ZBD2019-01 (Stephenson et al. 2021), the overall Objective of which was:

- To expand and develop initiatives to improve confidence in predictive models of seabed fauna and habitat distributions started under ZBD2016-11 '*Quantifying benthic biodiversity to improve benthic habitat modelling potential*'.

This was achieved via four Specific Objectives:

1. Predict gradients in benthic faunal turnover across Campbell Plateau (CP) using relationships between faunal distributions and environmental gradients developed for Chatham Rise (CR) under ZBD2016-11;
2. Run a dedicated photographic survey of seabed habitats and fauna across CP, structured on the basis of predictions from (1);
3. Use quantitative data from the CP survey to assess the utility of existing CR models when applied to a neighbouring area of the EEZ;
4. Generate updated models with a spatial domain encompassing both regions by merging data from the CP survey with existing CR dataset.

The survey of the Campbell Plateau took place in May-June 2020. While predictions for most of the individual taxa modelled were not reliable guides to actual distributions in this region, a community-level classification developed using Gradient Forests provided useful discrimination of changes in community composition. These results indicated that, using currently available methods and data, applications of habitat suitability modelling in New Zealand's marine realm should be based on sample data from within the area for which predictions are required, with sample locations spanning the full range of environmental conditions within the prediction area. While predictive species-environment modelling methods provide our best current estimates of the spatial distribution of seafloor biodiversity, their credibility remains limited by the paucity of quantitative survey data on which to train them.

18.4.7 WORKSTREAM 4: FUNCTIONAL THREATS TO BIODIVERSITY

Many marine ecosystems in New Zealand have been modified in some way through the harvesting of marine biota, the selective reduction of certain species and size/age classes, modification of food webs, including the detritus components, and habitat destruction. Epifaunal-dominated benthic communities including seamount communities, volcanic vent communities, bryozoans, corals, hydroids and sponges, and benthic communities dominated by infaunal structure (e.g., shellfish beds, tub-forming polychaetes or amphipods, large burrowing infauna) are vulnerable to human disturbance (Lundquist et al. 2017). The mechanical disturbance of marine habitats that occurs with some activities such as trawling, dredging, dumping, and oil, gas, and mineral exploration and extraction can substantially change the structure and composition of benthic communities. Indirect effects of mechanical disturbance (e.g., sediment resuspension, change in sediment grainsize) also impact on marine communities and their ecological functioning. The invasion of alien species into New Zealand waters is also a real threat, with evidence of nuisance species already well established.

A number of inshore marine ecosystems (especially estuaries and other sheltered waters) have been modified by sediment, contaminants, and nutrients derived from human land-use activities (Morrison et al. 2009). Coastal margin development has had a major impact on some inshore marine communities.

A project commissioned by the Aquatic Environment Programme, which identifies key threats to the marine environment (BEN2007-05) is complete and has listed and ranked the top threats to New Zealand's marine environment, as perceived by expert opinion. Relevant findings are that the highest-ranking threats are ocean acidification, increasing sea water temperatures, and bottom trawling (across all habitats), and that the most threatened habitats are intertidal reef systems in harbours and estuaries (MacDiarmid et al. 2012). Ecological risk assessment (ERA) methods have also been reviewed (under ENV2005-15, Rowden et al. 2008), and a trial Level 2+ risk assessment was completed for Chatham Rise seamounts to estimate the relative risk to seamount benthic habitat from bottom trawling (under ENV2005-16, Clark et al. 2011).

An existing spatially explicit patch dynamic model was expanded upon as a framework to explore effects of disturbance on functional diversity in benthic marine ecosystems, and ultimately, other elements of biodiversity and ecosystem function (such as the abundance of rare species, ecosystem productivity, and the provisioning of biogenic habitat structure) (Lundquist et al. 2013). The model was validated against available inshore (Tasman Bay and Golden Bay) and offshore (Chatham Rise and Challenger Plateau) empirical datasets, demonstrating the value of this tool for investigating disturbance and recovery dynamics in seafloor communities.

In recent times, the threat that climate change poses to marine biodiversity has been more widely recognised. Cyclical changes or trends in climate and oceanography and associated effects (such as increased ocean acidification) and how they affect the marine ecosystem as a whole have long-term implications for trophic interactions and biodiversity, as well as functional aspects of the system, e.g., biogeochemical processes. With significant improvement in remote sensing tools and global monitoring of climate change, new patterns are emerging indicating that there are long-term cycles. Examples include the Interdecadal Pacific Oscillation as well as shorter periods of change in relation to the El Niño Southern Oscillation that affect ocean ecosystems. Further, physical phenomena such as the deep subtropical gyre 'spin-up' in the South Pacific Ocean, which resulted in a warmer ocean around New Zealand from 1996 to 2002, and recent heatwaves in the Tasman Sea (see Climate Change Chapter 12 and https://www.moanaproject.org/marine-heatwave-forecast) can have flow-on effects on ecosystem functioning.

Ocean acidification trends and projections, and implications, have been reviewed extensively in the most recent IPCC report (IPCC 2018). One key projection of increasing ocean acidification is that by 2100 some 70% of cold-water corals, which provide a key refuge and feeding ground for some commercial fish species, will be exposed to corrosive waters (see also Tracey et al. 2011b, 2013). In addition, given the current greenhouse gas emission rates, it is predicted that the surface water of the highly productive Arctic Ocean will become under-saturated with respect to essential carbonate minerals by the year 2032, and the Southern Ocean by 2050 with disruptions to large components of the marine food source, in particular those calcifying species, such as foraminifera, pteropods, and coccolithophores, which rely on calcium carbonate.

Emerging research suggests that many of the effects of ocean acidification on marine organisms and ecosystems will be variable and complex and will affect different species in different ways. Evidence from naturally acidified locations confirms, however, that although some species may benefit, biological communities in acidified seawater conditions are less diverse and calcifying (calcium-reliant) species are absent, whereas algae tend to dominate. BRAG funded projects have improved understanding of the impacts of ocean acidification on deep-sea coral growth (ZBD2014-01), testing physiological responses (e.g., growth) to ocean acidification manipulations in a laboratory setting. Another project (ZBD2013-06) has examined shell generation and maintenance of important aquaculture species including pāua and flat oysters.

Many questions remain regarding the biological and biogeochemical consequences of ocean acidification for marine biodiversity and ecosystems, and the impacts of these changes on ecosystems and the services they provide, for example, in fisheries, coastal protection, tourism, carbon sequestration, and climate regulation (Tracey et al. 2013).

Studies to predict changes in biodiversity in relation to climate change in more than a rudimentary way are beyond the state of current knowledge in New Zealand. Nevertheless, surveys of biodiversity that have occurred or are planned will provide a snapshot against which future research results or trends can be compared.

Meeting the challenges of climate change and identifying crucial issues for marine biodiversity is an area of high political interest internationally and has been identified as a gap in biodiversity research in New Zealand (Green & Clarkson 2006). A revised action plan (2016–2020) to support the New Zealand Biodiversity Strategy includes a chapter on climate change (Department of Conservation 2016).

Recognising threats and changes to marine biodiversity requires effective monitoring and the establishment of suitable indicators for detecting changes. In the mid-1990s, monitoring of marine biodiversity and the marine environment was a topic of considerable discussion, yielding several reports on developing MfE indicators. However, since the publication of MfE's indicators in 2001, a reduced set of core indicators that relate to the marine environment have been reported (Ministry for the Environment 2007). A new international initiative launched in 2010: 'Biodiversity Indicators Partnership' (https://www.bipindicators.net) provides guidelines and examples of biodiversity indicators developed around the globe, however, Oceania does not appear to have any partnership identified. The link between this initiative and OECD environmental indicators is unclear.

A serious gap identified by Green & Clarkson (2006) in their review of progress on implementation of the NZBS was the lack of development of an integrated national monitoring system. Efforts to respond to this gap within the Biodiversity Programme resulted in the immediate initiation of a five-year Continuous Plankton Recorder project, and a project that convened a series of workshops to determine how best to approach monitoring on a national scale (ZBD2008-14) (Hewitt et al. 2014). One objective of monitoring would be to test the effectiveness of management measures.

Current and recently completed projects are as follows:

ZBD2014-03 *Sub-lethal effects of environment change on fish populations* [Completed]

This project (co-funded with MBIE) investigated the potential effects of ocean acidification (caused by elevated CO_2) on New Zealand's fish and fisheries, with a focus on snapper. In 2015–16 a workshop for key collaborators completed a review of existing information on ocean acidification effects on fish and how these known effects are likely to play out in New Zealand's temperate setting. Following on from the review, tank experiments were conducted to assess the response (e.g., mortality, morphology, energy utilisation, behaviour) of snapper larvae to different acidification scenarios. A range of responses were measured, with some of the most notable being a strong positive effect of elevated temperature on growth and a positive effect of elevated CO_2 on survival in the absence of other real-world factors (McMahon et al. 2020a), negative effects of elevated CO_2 on swimming ability and metabolic performance and a positive effect of temperature on swimming ability (McMahon et al. 2020b), and a negative effect of elevated CO_2 on hearing ability (Radford et al. 2021). Finally, the findings of the review and tank experiments were combined in a deterministic model to assess the effects of acidification at the broader population level (Parsons et al. 2021).

ZBD2018-02 *Climate change, fish distribution meta-analysis* [Completed]

Climate change effects on wild renewable marine resources essentially take place through changes in species distribution and their productivity. To date, changes in distribution seem to be the most pronounced and measurable response; for example, warming off southeast Australia with extension of the east Australian Current and associated fauna southwards by about 350 km. Other reports of changes include ranges of some warm-water species, temperature mediated impacts on benthic invertebrates, and localised regions of warming. Understanding the shifts in New Zealand fish species in relation to fisheries management area boundaries and stock productivity is key to adaptation and management under a changing environment. The overall objective of this project was to assess the resilience and diversity of marine living resources under regime shifts and climatic trends in New Zealand waters.

This was achieved via three Specific Objectives:

1. Explore data time series and biological data for evidence of spatial change in living marine resources that may be consistent with climate change or regime shifts.
2. Investigate novel approaches to identify ecologically relevant change by examining species and community relationships between the organisms and their environment.
3. Identify fisheries, communities, and locations that are most vulnerable or will remain stable under the response to climate change effects on the ocean.

ZBD2018-03 *Climate variability, trends, and fish population parameters* [Completed]

Ongoing climate change is predicted to have both direct and indirect effects on individual organisms, including fish during all life stages, thereby affecting populations of a species, communities, and the functioning of ecosystems. However, current methods of stock assessment assume that recruitment and productivity parameters such as natural mortality, steepness, von Bertalanffy growth curves, length-weight relationships, recruitment variability, and the mean number of recruits remain constant over time. This project investigated whether this assumption is valid as climate change related impacts on the ocean progress, how parameters may change, and what changes are likely to have a significant impact on fisheries sustainability. By identifying suitable indicators to monitor significant change for important fisheries (e.g., the top 20 species by value including finfish, rock lobster, and pāua), individual stock assessment groups will eventually be able to assess the impacts of environmental change on stock projections. A model was used to investigate the effects of climate change on different aspects of fisheries productivity. This model provided unbiased estimates of stock status on average despite annual and decadal fluctuations in all production-related parameters. This variability was much greater than that induced by climate change scenarios. The project found that for many stocks, it is difficult to tell if specific changes in productivity parameters are due to climate processes or demographic responses to fishing pressure (Neubauer et al. 2021a, b).

ZBD2018-05 *Ecosystem function and regime shifts in the Subantarctic* [Completed]

Anecdotal and scientific information from a range of sources suggest that there have been major environmental shifts in the Subantarctic over the past 40 years, but no meta-analyses or whole-of-system modelling has been conducted to determine how these shifts may be influencing different biota, including megafauna, top predators, protected species, and fish. This project examines all potential sources of data and develops a suitable modelling approach to identify synchronous trends, cycles, tipping points, and regime shifts in the Subantarctic. The project also investigates the feasibility of testing or ground-truthing aspects of the model through survey work in the area. The project develops ecosystem models to understand the effects of environmental variability and change at the scale of decades affecting ecosystem function in the Subantarctic, including effects on a protected species (New Zealand sea lion) and on ecologically and economically important fish (hoki, squid, and southern blue whiting), including potential effects on fish abundance, recruitment, and spatio-temporal movements. Preliminary results show minimal evidence for long-term shifts in the atmospheric or oceanographic climate of the Subantarctic. However, observed changes in phytoplankton productivity and predictive modelling suggest that oceanic warming will have opposing effects on primary productivity; increasingly productive in the Subantarctic, but decreasingly productive off the west coast of the South Island. The negative effects of warming off the west coast on the productivity of New Zealand fur seals and the recruitment of hoki are also identified from predictive modelling, as is the positive effect of increased productivity on the abundance of southern arrow squid along the sub-tropical front. Sea temperatures both at the surface and the seabed are identified as potential indicators of ecosystem change. Progress in this project has shown the importance of environmental cycles that occur beyond the boundaries of resident populations, particularly when a species life cycle straddles many different parts of the ocean.

ZBD2016-07 *Multiple stressors on coastal ecosystems in situ* [Completed]

Increasing acidity (lower pH) and water temperatures are two major stressors that will influence the future structure and function of coastal ecosystems. Prior research has primarily focused on the response of different faunal groups to acidified conditions in isolation. To advance understanding and the capacity to predict the future status of coastal ecosystems in New Zealand, a series of long-term mesoscale manipulation studies of coastal planktonic water column, in which pH and other parameters were altered, was carried out as part of aligned research in the MBIE funded CARIM Project. In the project described here, laboratory experiments were used to examine whether future changes in water temperature and pH projected for the New Zealand region will influence coastal phytoplankton biomass and speciation, and thus the nutritional value of plankton for green-lipped mussels (Law et al. 2021). Phytoplankton biomass was unaffected by lower pH but showed a significant increase of 10-25% in the combined low pH and elevated temperature treatment, suggesting potential for increased food availability in the lower food web in the future. However, a decrease in food quality was observed. Green-lipped mussel larvae were grown in chambers within the mesocosms during the experiment, and despite smaller maximum final shell size due to lower pH, modelling predicted future increases in food uptake rate, energy allocation, shell length, and flesh weight for green-lipped mussels resulting from increased temperatures. The results of this project provide an understanding of how a key species in New Zealand may be affected directly by future ocean climate changes, and indirectly through the influence of these changes on the lower food web.

ZBD2014-09 *Climate change risks and opportunities in the marine environment, New Zealand* [Completed]

The overall aim of this project was to identify risks and opportunities that are likely to arise for the seafood sector as a consequence of climate change effects in coastal and offshore New Zealand waters.

This study (Cummings et al. 2021), conducted in stages over four years, synthesised available information on CO_2- and climate-induced changes that affect the New Zealand region and our fisheries. Knowledge of changes to the physical and oceanographic system in the coming decades, and of how specific fisheries are influenced by these parameters, was assessed to determine potential risks to these fisheries.

The state of knowledge of climate change-associated predictions for components of New Zealand's marine environment that are most relevant to fisheries were examined. Past and future projected changes in coastal and ocean properties, including temperature, salinity, stratification, and water masses, circulation, oxygen, ocean productivity, detrital flux, ocean acidification, coastal erosion and sediment loading, and wind and waves, were reviewed.

A complete understanding of the response of key New Zealand fisheries species to climate-associated change, based on scientific data, is ambitious given the paucity of information on most New Zealand Quota Management species, the ecosystems that support them and the potential threats they face. The biological and ecological characteristics of 31 commercial fisheries species or species groups important in the New Zealand region were collated and evaluated as to how they might be affected by changes to selected properties of New Zealand's oceanic and coastal waters expected over the remainder of this century. Regardless of the data limitations, a number of species that may be affected by warming, and/or ocean acidification were identified, supporting previous findings that these are the two greatest anthropogenic CO_2-associated threats facing the New Zealand marine environment.

A range of decision support tools in use overseas were evaluated with respect to their applicability for dissemination of the state of knowledge on climate change and fisheries. Three species, for which there was a relatively large amount of information available, were chosen from the main fisheries sectors for further analysis. These were pāua, snapper, and hoki (shellfish, inshore, and middle-depths/deepwater fisheries, respectively). Evaluations of each species' sensitivity and exposure to climate change-associated threats, based on currently available published literature and expert opinion, assessed pāua vulnerability as 'very high', snapper as 'moderate', and hoki as 'low'. Potential adaptation and management options, and their effectiveness and feasibility, were also examined.

This project has summarised a large amount of cross disciplinary information, always with the fishery and effective management in a changing environment in mind. It has demonstrated the value of research into direct influences of the different parameters on the species (e.g., temperature, acidification), and the need for information across the life cycle of the species to enable robust evaluations and predictions of future impacts to be made.

ZBD2019-04 *Plastics and marine debris across the ocean floor in New Zealand waters* [Completed]

Plastic debris in the ocean is a pollution-related issue attracting worldwide concern. As well as unsightliness, the debris is on a scale that causes harm to many marine species and threatens the health of marine ecosystems across the globe. All species are affected, but attention has particularly focused on marine mammals, seabirds, other protected species, and fish. New Zealand is no exception, and plastic debris is visible along much of our coastline. Within New Zealand, beach surveys and clean-up events for visible (macro) plastics are underway and the findings are available online (https://litterintelligence.org/).

Far less is known about the occurrence of plastics in our offshore environment. Microplastics have been found in the plankton recorder time series samples between New Zealand and the Ross Sea (Antarctica) and are being quantified as part of the ongoing Continuous Plankton Recorder Programme supported by Fisheries New Zealand and Sanford Limited.

This project used NIWA's database of seafloor images collected with their Deep-Towed Imaging System (DTIS) dating back to 2006 to quantify the occurrence and distribution of macroplastics in New Zealand's sub-tidal marine environment from shallow to deep water (Behrens et al. 2021). Litter was classified according to the UNEP guidelines for monitoring and surveying marine litter (benthic) https://www.nrc.govt.nz/media/10448/unepioclittermonitoringguidelines.pdf. There were 149 occurrences of marine litter observed in seafloor images around New Zealand consisting of plastic rope (75%), glass bottles (7%), plastic sheeting (5%), and metal fishing-related litter (4%); the remaining item categories were very rare (≤ 1%). Most observations of litter were within 25 km of the coast, but litter was also recorded on remote seamounts more than 1500 km from the coast.

Only a small proportion of the seafloor in New Zealand has been surveyed photographically, leaving large data gaps and uncertainties around litter density in un-surveyed areas. Litter density across New Zealand was estimated at ~133–430 pieces per km^2 based on observed litter from DTIS imagery which is at the lower end for densities reported from other areas (e.g., Mediterranean, North Sea).

This project was exploratory in scope but there is considerable potential to increase the quality and quantity of data on marine litter by adjusting analysis protocols for photographic surveys to include the identification of marine litter according to UNEP guidelines.

ZBD2020-09 *Cumulative effects of stressors on scallops and scallop habitats in the Marlborough Sounds* [Underway]

Annual dredge surveys of scallops in the Marlborough Sounds (1997–2019) have revealed a declining trend in recruited biomass since 2009 (Williams et al. 2019). The fishery was closed in 2016 due to sustainability concerns and will remain closed until the scallop population has recovered. The most recent recruited biomass estimate, based on the 2019 dredge survey, is the lowest on record (Williams et al. 2019), indicating that despite closures, there has been no recovery of scallops in surveyed areas.

Project ZBD2020-09 has recently begun and seeks to assess the cumulative effect of key stressors on habitats and ecological processes that support scallops. The results of this work should provide insight into the recovery potential of scallops in the Marlborough Sounds, guidance as to whether the creation of refuges from bottom-fishing methods may enhance recovery, and an idea of the realistic timeframes required for recovery.

This project supports the Draft Southern Scallop Strategy (Marlborough Sounds) which proposes to “get better information” particularly around “what constitutes good scallop conditions” and has synergies with the work proposed under project 1.1 ‘Ecological responses to cumulative effects’ of Phase II of the Sustainable Seas Challenge.

ZBD2020-11 *Review of land-based effects on coastal fisheries and kaimoana and their habitats* [Underway]

Substantial changes in coastal habitats and ecosystems have occurred over the last 100 or more years and continue to occur (Morrison et al. 2009). Land-based activities are a key driver of change to coastal habitats, and both directly and indirectly impact coastal fisheries and kaimoana species. Due to the growing awareness of emerging pressures on fisheries resources and the marine environment, Fisheries New Zealand are seeking to progress New Zealand towards ecosystem-based fisheries management (EBFM). A key focus area in advancing EBFM is “improving environmental performance with a focus on protecting habitats of significance for fisheries management from the impacts of fishing and land-based effects, and ensuring the long-term viability of protected species.”

Since the last review (Morrison et al. 2009), there have been various programmes of work to assess the effects of land-based stressors on coastal ecosystems. Phase I of the Sustainable Seas Challenge included a range of projects which assessed the effects of numerous stressors and how coastal habitats and species responded to change (e.g., Blain et al. 2020, Douglas et al. 2018, Gladstone-Gallagher et al. 2018, Schiel et al. 2018). MBIE Endeavour Fund Research Programme ‘Removing juvenile fish bottlenecks’ (C01X1618) commenced in 2016 and is exploring how ‘habitat bottlenecks’ to fish species work, how fish respond, where they occur, their links to adult populations, and the stressors that affect them.

The purpose of ZBD2020-11 is to review the outputs of programmes and projects which contribute to the state of knowledge of how land-based stressors impact coastal fisheries and kaimoana, and to collate and summarise available information on the current status and trends in sediment and nutrient run-off at local, regional, and national scales in New Zealand.

ZBD2018-01 *5-year Continuous Plankton Survey (Phase 3)* [Underway]

The overall objective of the Continuous Plankton Recorder (CPR) series of projects is to map changes in the quantitative distribution of epipelagic plankton, including phytoplankton, zooplankton, and euphausiid (krill) life stages, in New Zealand’s EEZ and transit to the Ross Sea, Antarctica.

The original project was established in 2008 for a five-year period with sampling carried out annually in the Austral summer. Sanford Limited continues to provide the FV *San Aotea II* and crew to take the samples, and sample analysis is carried out by the laboratory at NIWA Christchurch. A second project further funded this long-term series (ZBD2013-03) (Robinson et al. 2014). Analyses conducted after the first two phases suggest that a changing Southern Ocean environment may benefit some zooplankton taxa (e.g., copepods, foraminifera, and *Fritillaria* spp.) but disadvantage others (e.g., pteropods) (Pinkerton et al. 2020, Robinson et al. 2021)

The current project, ZBD2018-01, continues this annual programme of CPR sampling and is funded for a further five years (2018–2023). This will enable a continuation of the data time series and provide a more robust dataset with which to make comparisons with the Southern Ocean CPR survey and potentially determine any latitudinal and/or temporal trends in the plankton community. Of interest, CPR surveys have detected microplastics, showing the broad impact of these pollutants across the global oceans. Microplastic content is being analysed retrospectively and will be part of the routine analysis from now on.

The data generated from this series of projects contributes to reports on the status and trends of zooplankton across the whole Southern Ocean for the Scientific Committee on Antarctic Research; the latest report can be viewed here.

18.4.8 WORKSTREAM 5: SUSTAINABILITY FOR FUTURE GENERATIONS

With global biodiversity declining due to multiple threats, it is now clearer than ever that sustainability must be at the forefront of decision-making about natural marine resources. There is a great need to balance extractive use of these resources with environmental protection, although this balance can sometimes be difficult to achieve. Finding the right balance is essential in ensuring that marine biodiversity is sustained and ideally thriving for future generations.

Science and research underpin this important concept and allow decision-makers to manage natural marine resources in a way that promotes sustainability. Research under this workstream includes understanding the effectiveness of protected/closed areas, preserving cultural practices as they relate to fisheries and biodiversity, maintaining a social licence to operate, and identifying creative solutions that allow for sustainable development of the marine economy with biodiversity at its core.

Current and recently started projects are as follows:

ZBD2020-06 ***Recovery of biogenic habitats: assessing the recovery potential offered by spatial planning scenarios proposed in the Sea Change Plan*** [Underway]

The Sea Change process was initiated in 2013 to develop a marine spatial plan (https://www.seachange.org.nz/read-the-plan/) that seeks to restore the health of the Hauraki Gulf Marine Park (HGMP). The plan, released in 2016, describes numerous stressors that negatively affect Hauraki Gulf and provides overall visions, objectives and proposals to address environmental decline, particularly the loss of biogenic habitats which support juvenile fish (Morrison et al. 2014). The plan proposes to use a variety of spatial management tools to mitigate key stressors (e.g., bottom fishing) and allow for the natural regeneration of biogenic habitats to increase fish population size. A number of scenarios for Marine Protected Area (MPA) placement and design are proposed in the plan.

Project ZBD2020-06 has recently begun and will assess how well the various MPA scenarios proposed in the plan may support the recovery of biogenic habitats and will identify feasible options for the restoration of habitat functions. The results of this project will inform future consultation on spatial planning and may inform an ecosystem-based approach to fisheries management in the HGMP.

ZBD2020-07 ***Recovery of seamount communities*** [Underway]

Seamounts, knolls, and hills are prominent features of underwater topography in the New Zealand region and are often sites of high biodiversity and productivity. They are the focus of important commercial fisheries for deepwater species, with about 80% of known seamount features at suitable depths for deepwater fisheries having been exploited. Benthic faunal communities on deepwater seamount features are commonly characterised by extensive growth of cold-water corals but impacts from bottom trawl gear have substantially reduced biogenic habitat formed by corals on fished seamounts (Clark et al. 2016). The overall resilience of such benthic communities, and the time frame required for recolonisation and re-growth is uncertain, yet such information is important for evaluating appropriate options for management of fishing impacts.

Project ZBD2020-07 aims to understand the nature and timescale of changes and recovery dynamics of benthic invertebrate communities on seamounts following the closure of certain seamounts around the Chatham Rise to bottom trawling in 2001. The area was surveyed using towed cameras in 2001, 2006, 2009, 2015, and again under the current project in August 2020. Analysis of the first 4 surveys showed little evidence of resilience of the benthic community, nor signs of settlement or recruitment of the main coral species (Clark et al. 2019a). However, initial observations from the most recent survey in 2020 showed very small scleractinian corals (the same species as form the large climax-community reef structures on these seamounts) estimated at between 2 and 10 years old. Images from this survey are currently being processed.

The survey links strategically to assessing the efficacy of Benthic Protection Areas and Seamount Closures (Clark et al. 2019b), and also provides an understanding of the biodiversity and recovery rates of deepwater environments and Vulnerable Marine Ecosystems (VMEs) that are part of Fisheries New Zealand international obligations.

18.4.9 PROGRESS ON BIODIVERSITY IN ANTARCTICA

Antarctic research funded by BRAG (historically known as BioRoss) aimed to improve understanding of the biodiversity and functional ecology of selected marine communities in the Ross Sea. Due to lack of resources, no research has been funded for the last 5 years and no new research is planned for the Antarctic through the Biodiversity Research Programme.

18.5 EVALUATION OF RESEARCH EFFECTIVENESS

The Fisheries New Zealand Biodiversity Programme has been running for more than 20 years and a number of new strategic documents and directions are emerging throughout government. Here the programme is evaluated with respect to its initial strategic outcomes, and its alignment with more recent strategic documents.

In 2000, five strategic outcomes were built into the Ministry of Fisheries Biodiversity Research Programme.

That by 2010:

i. *The Ministry of Fisheries Biodiversity Programme will have become an integral part of the research effort devoted to understanding New Zealand's marine environment.*
ii. *Research planning will benefit from close cooperative relationships within the Ministry of Fisheries, with other government agencies, and with external stakeholders.*
iii. *Mutually beneficial collaborative research projects will be carried out alongside other New Zealand and international research providers, especially for vessel-based research.*
iv. *Ministry of Fisheries Biodiversity projects will have contributed substantially to an improved understanding of New Zealand's marine biodiversity and its role in marine ecosystem function, yielding scientifically rigorous outputs for a national and international professional audience.*
v. *Results generated by MFish Biodiversity projects will be incorporated into management policy, with clear benefits for the New Zealand marine environment.*

The Biodiversity Programme has been highly effective in delivering the first four and part of the fifth of these five outcomes, though currently there is no clear metric to evaluate the measure of whether the Programme is providing *"clear benefits for the New Zealand marine environment"*. In recent years, significant whole-of-government projects have been administered through the programme, and one-off funding applications made jointly with other stakeholders have been successful. The programme has made a significant contribution to increasing understanding about biodiversity in the marine environment. Achievements in each outcome are addressed below.

i. *Has the Biodiversity Research Programme become integrated with New Zealand's research effort to understand the marine environment?*

Seven science objectives were developed by multiple stakeholders through the Biodiversity Research Advisory Group. The original objectives included: 1) ecosystem-scale studies in the New Zealand marine environment, 2) the classification and characterisation of the biodiversity of near-shore and offshore marine habitats, 3) the role of biodiversity in the functional ecology of marine communities, 4) connectivity and genetic marine biodiversity, 5) the assessment of the effects of climate change and increased ocean acidification, 6) identification of indicators of biodiversity that can be used to monitor change, and 7) identification of key threats to biodiversity, identification of threats and impacts to biodiversity and ecosystem functioning beyond natural environmental variation.

Projects ranged from localised experiments on seabed communities of shellfish and echinoderms, to integrated studies of rocky reef systems (Schwarz et al. 2006) and offshore fishery-scale trophic studies. The effects of ocean climate change (temperature, acidification) have been explored on shellfish, rhodolith communities, plankton productivity, and the microbial productivity engines of polar waters. A major project to investigate shelf communities in relation to climate over the past 1000 years has resulted in the development of new methods and insights into past changes and the human impact on New Zealand's marine environment.

A total of 95 projects were commissioned and managed within this 20-year period, yielding over 100 final research reports, most of which have been published through

Ministry publications (Marine Biosecurity and Biodiversity Reports and Aquatic Environment and Biodiversity Reports), books, identification guides, and mainstream scientific literature. Additional publications continue to be added to the scientific literature. In addition, several workshops have been run through the Programme, including on qualitative modelling techniques, how to set up a marine monitoring programme, and predictive modelling. A large number of science providers, including NIWA, Cawthron Institute, University of Auckland, Auckland University of Technology (AUT), University of Waikato, Victoria University of Wellington, University of Otago, University of Canterbury, and Massey University have been directly commissioned or sub-contracted to take part in or conduct research projects through the Programme during the 20-year period. For some, the projects have provided critical synergies with aligned research, whereas others have provided one-off opportunities for marine biodiversity investigation or opportunistic leveraging for research voyages.

Research into the biodiversity of habitats such as seamounts has been completed and new methods to assess the vulnerability of seabed habitats have been developed. The land-sea interface is being investigated and projects have shown how land use in a given catchment can affect nutrient transfer and living conditions, and impact the diversity and functioning, of estuarine and coastal organisms. Publication and presentation of the results from these projects has resulted in widespread contribution to the development of marine science in New Zealand. Partnership with overseas researchers and presentations to international meetings and conferences has added to the growing global initiatives on marine biodiversity research questions.

Feedback from stakeholders has indicated that the move to a five-year research planning horizon was welcomed by research providers, but some stakeholders felt that Requests for Proposals should be at a higher level than individual projects to safeguard intellectual property on new ideas and methods.

ii. *Does research planning now benefit from close cooperative relationships within the Ministry of Fisheries, with other government agencies, and with external stakeholders?*

The Biodiversity Programme is highly cooperative. Of 38 projects underway in the last five years, 14 have formal collaborative components across government departments, with other stakeholders or multiple research providers and 10 have formal linkages to international research programmes. Within Fisheries New Zealand and with other stakeholders (NGOs, industry, other government departments), the Biodiversity Projects have contributed to Marine Stewardship Council certification processes, decision papers on aspects of Antarctic management under CCAMLR, and fulfilling Ministry commitments to the New Zealand Biodiversity Strategy. There are many other examples, e.g., the Programme has contributed towards DOC and Ministry decisions on marine protected areas. The interaction at the research and policy advice stages of resource management feeds back into the BRAG planning for future research.

There are close links with the Fisheries New Zealand Aquatic Environment research programme, the National Aquatic Biodiversity Information System (NABIS), a web-based interactive data access and mapping tool, and the Fisheries New Zealand Antarctic Research programme. These and other links have enabled contributions resulting from progress on land-sea interface research, habitats of significance to fisheries management, trophic studies (Marine Stewardship Council Certification), climate change (effects on shellfish), and habitat classification (fish optimised MEC, testing of MEC and BOMEC). The successful involvement of the Biodiversity Programme in major whole-of-government projects such as Ocean Survey 20/20 and IPY-CAML has also raised the profile of Fisheries New Zealand and the research it has commissioned both within New Zealand and internationally.

Datasets, voucher specimens, and samples from all biodiversity research projects have resulted in a substantial amount of material that has been physically preserved and housed in the Te Papa National Fish Collection, the NIWA Invertebrate Collection, and the Te Papa Herbarium (macroalgae). All data are held in databases either at Fisheries New Zealand, NIWA, or Te Papa, and accessibility is being improved. The recent Bay of Islands Ocean Survey 20/20 Portal was very well received and nominated for New Zealand Government Open Source awards. It also incorporates data access from Chatham Challenger and IPY projects. Data from a number of Ministry biodiversity projects have also been entered into international biodiversity databases such as OBIS and from there into the Global Biodiversity Information Facility (GBIF).

Biodiversity Research planning receives regular input from DOC, Seafood New Zealand, MfE, Cawthron Institute, NIWA, GNS, LINZ, Fisheries New Zealand, Te Papa Tongarewa, University of Auckland, AUT, University of Otago, MBIE, MFAT, regional councils, civil society, and others. Research planning for 2018–19 and beyond has included a re-alignment of the current research programme to take account of new developments such as The Future of Our Fisheries, MfE's environmental reporting programme, DOC's integrated marine protected area monitoring programme, and international commitments such as the Convention for Biological Diversity (CBD) COP10 Aichi-Nagoya Agreement.

Feedback and support for projects by external stakeholders has shown that the Programme has been effective in promoting inter-agency collaboration.

iii. *Have mutually beneficial collaborative research projects been carried out alongside other New Zealand and international research providers, especially for vessel-based research?*

As discussed above, collaborative research projects throughout government and among research providers have resulted in many mutually beneficial data and specimen collections; surveys of New Zealand marine biodiversity in New Zealand Territorial Sea, the EEZ, and the Ross Sea; ground-breaking research into seamount biodiversity and the identification of VMEs; and research for international collaboration, particularly vessel-based studies. Large-scale vessel-dependent oceanic research projects have made significant gains in baseline knowledge about the distribution and abundance of biodiversity in the EEZ/Ross Sea region. Vessel-based projects include: NORFANZ (Norfolk Island-Australia-New Zealand survey of biodiversity on Norfolk Ridge and Lord Howe Rise); BioRoss (MFish-LINZ, first New Zealand survey of biodiversity in the Ross Sea); Chatham-Challenger (LINZ-MFish-NIWA-DOC first Ocean Survey 20/20 project); NZ IPY-CAML (MFish-LINZ-NIWA; with international and New Zealand-wide collaboration) survey of the Ross Sea as part of International Polar Year; and biodiversity of seamounts (MFish-NIWA-LINZ-MBIE voyages to the Kermadec Arc and the Chatham Rise). These projects have generated huge geo-referenced datasets and thousands of specimens for Te Papa Tongarewa and the NIWA Invertebrate Collection. They have also resulted in the identification of new species, new genera, and new families, as well as new records extending the known distribution of species. These surveys have contributed to habitat classification, identified areas of high biodiversity, and challenged paradigms on the environmental drivers that determine biodiversity. More recently they have provided new information on the effects of ocean acidification on the productivity of polar seas, and in New Zealand waters.

Vessel-dependent coastal projects have also generated significant new understanding about the distribution of inshore biota, and the role they play in maintaining a healthy ecosystem. Experimental field work on the productivity of the seabed has been carried out in New Zealand waters (Fiordland, Otago, Bay of Islands, Hauraki Gulf, and Kaipara and Manukau harbours) and along the coast of the western Ross Sea. The impact of land practices on the land-sea interface has also highlighted real downstream effects on the productivity of the coastal environment. These projects have provided new insights into the connectivity between different species groups, and data are being used in a number of ways to assist with spatial planning by RMAs.

Feedback from stakeholders has indicated that the collaborative voyages administered through the Programme have successfully created synergy and opportunity for New Zealand scientists as well as facilitating new international collaborations.

iv. *Have MFish Biodiversity projects contributed substantially to an improved understanding of New Zealand's marine biodiversity and its role in marine ecosystem function, yielding scientifically rigorous outputs for a national and international professional audience?*

In the early years, the Programme focused primarily on taxonomy and the description of marine biodiversity. As the Programme matured, projects to address biodiversity roles in ecosystem function were introduced. Some were experimental and on a local scale whereas others were on a regional scale. Recent projects have addressed patterns of marine biodiversity in relation to environmental drivers and ecosystem function. These projects enabled modelling to predict the distribution of biodiversity in un-surveyed areas of ocean, evaluate the vulnerability of biodiversity to perturbations such as climate change, as well as model the trophic interactions among key fish species. Presentations of research results have been made to numerous overseas and New Zealand science audiences, and publications in the mainstream literature have been encouraged.

v. *Have results generated by Ministry of Fisheries Biodiversity projects been incorporated into management policy, with clear benefits for the New Zealand marine environment?*

Examples of incorporation into management policy with clear benefits for the marine environment include the increased awareness of research topics initiated in the biodiversity programme by policy analysts to core Aquatic Environment research projects and Fishery Plans (land-use effects, climate change in the ocean, habitat classification); links to the Antarctic research programme and uptake into CCAMLR (ecotrophic studies, ecosystem baselines, VME risk assessment, bioregionalisation), spatial management (seamount closures, BPAs, MPAs, RMAs); and the need by MfE to report on the marine environment at a national scale (plankton recording programme, Marine Environmental Monitoring Programme). Fisheries New Zealand biodiversity advice is frequently requested to contribute to cross-government initiatives including Ocean Survey 20/20, DOC Sub-Antarctic Islands Forum National Monitoring, Statistics New Zealand Tier 1 statistic review and Environmental Domain Stocktake, International Year of Biodiversity, OECD and CBD reports, International Oceans Issues, SPRFMO, NRS marine issues paper, the Antarctic Science Framework, Ocean Fertilisation, and IPCC. Finally, the Programme has contributed to New Zealand's efforts in the international Census of Marine Life and an ongoing assessment of New Zealand's progress in Marine Biodiversity has been proposed as a new Tier 1 Environmental Statistic.

However, the benefits to the marine environment are more inferred than demonstrated. There is increased awareness within Fisheries New Zealand and throughout government, that the health of fisheries and other valued uses of the sea depend on intact ecosystem services provided by the diversity of organisms, the diversity of habitats, and the genetic diversity found in the marine environment. Future biodiversity research will need to be tuned to Fisheries New Zealand Strategic Directions as well as Te Mana o te Taiao.

18.6 CONCLUDING REMARKS

Since its inception 21 years ago, the research in this Biodiversity Programme has evolved from defining and mapping the biological diversity of the sea, to elucidating the role of biodiversity in marine ecosystem function, threats to these roles, and how best biodiversity and its successful protection can be measured. Advances have been made in the provision of new identification tools for major groups (e.g., coralline algae), and progress has been made in raising the profile of biodiversity in coastal and ocean environmental management, in particular within new environmental legislation such as the Exclusive Economic Zone and Continental Shelf (Environmental Effects) Act (2012) and the Environmental Reporting Act (2015). As fisheries management moves towards EBFM, the role of biodiversity information will inevitably change towards informing EBFM.

As the CBD updates the Aichi-Nagoya Agreement with new post-2020 Biodiversity strategy and associated biodiversity and environmental targets, New Zealand must continue to progress our understanding of the marine biodiversity and the ecosystem services provided by the marine environment to inform national and international decision-making. This will continue to require a coordinated cross-government approach to link science to policy needs. Essentially, we need to know four things: what is out there in the marine environment to use, protect, or manage; how does the ecosystem function; what are the impacts of natural- and human-induced changes; and what tools will allow for effective monitoring and management of environmental change? For example, there is a compelling need for large-scale projects such as mapping seafloor habitats and establishing long-term nationwide monitoring and reporting schemes to measure the effects of ocean climate change, and regular assessment of the cumulative effects of anthropogenic activities and multiple stressors in the ocean and the effectiveness of their management. Without these, we face the risks that New Zealand's 'green' branding will be increasingly challenged, and that tipping points in the health of the aquatic environment may be reached before evasive action can be taken.

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19 APPENDICES

Status of chapter	This chapter has been updated for AEBAR 2021

19.1 GLOSSARY OF TERMS AND ACRONYMS

19.1.1 GLOSSARY

Term	Definition
Aquaculture	The practice of farming aquatic organisms such as salmon, freshwater prawns, or shellfish in marine and freshwater environments.
Benthic Protection Area (BPA)	Any area established by the Fisheries (Benthic Protection Areas) Regulations 2007 as being a BPA.
Biodiversity	The variety of species, and the ecological systems in which they reside. Includes diversity within and between species, and diversity of habitat. This term is defined in the Resource Management Act 1991 and the Fisheries Act 1996.
Biomass	The total quantity or weight of organisms in a given area or volume.
Bycatch	Species not targeted by a fishery but caught incidentally during fishing operations. Once caught, they are either landed, discarded or released.
Chondrichthyan	A member of the Chondrichthyes class, often referred to as cartilaginous fishes. Members of this class include sharks, rays, skates and chimaeras.
Continental Shelf	The submerged border of a continent that slopes gradually and extends to a point of steeper descent to the ocean bottom.
Continental Slope	A region on the outer edge of a continent between the generally shallow continental shelf and the deep-ocean floor, usually demarcated from the shelf by a 200m isobath.
Cryptic Mortality	Interactions that result in mortality which are unobserved or unobservable.
Ecosystem	An interacting system of living and non-living parts such as sunlight, oxygen, water and nutrients. Ecosystems can vary in size and longevity (e.g., a raft of detached seaweed floating in the open ocean that supports a small community, or a seamount and its associated flora and fauna).
El Niño-Southern Oscillation (ENSO)	An irregularly recurring climate pattern that results in changes in the temperature of waters in the central and eastern tropical Pacific Ocean. The warming phase of sea temperature is known as El Niño and the cooling phase as La Niña, with each period lasting up to several years.
Endemic	An organism that occurs naturally only in one place or region.
Exclusive Economic Zone (EEZ)	An area of the ocean extending from 12 to 200 nautical miles from shore, including the seabed and subsoil. 1982 United Nations Convention on the Law of the Sea (UNCLOS).
Habitat	The place or type of area in which an organism resides. Can be biogenic/living (e.g., kelp forest) or abiotic/non-living (e.g., a rocky reef).
Indicator	A measure against which some aspects of performance can be assessed.
Indicator Species	A species whose presence or absence is indicative of a particular habitat, community or set of environmental conditions. The presence/absence of such a species is sometimes used as a proxy for characteristics of the environment.
Interdecadal Pacific Oscillation (IPO)	A pattern of climate variability in the Pacific Ocean with phases lasting 20-30 years.

Term	Definition
Intergovernmental Panel on Climate Change (IPCC)	An international body that assesses the science related to climate change. IPCC assessments are written by hundreds of leading scientists from around the world.
Invertebrate	An animal without a backbone or spinal column. Corals, sponges and jellyfish are common marine examples.
Kaitiakitanga	The act of stewardship, guardianship or trusteeship; usually in reference to species and the environment.
Longline Fishing	A method of fishing using lines that trail back behind a boat for a set distance, with baited hooks set at fixed intervals.
Marine Protected Area (MPA)	Protected areas of seas, oceans, or estuaries that restrict human activity for a conservation purpose, typically to protect natural or cultural resources.
Mātauranga Māori	The knowledge, comprehension or understanding of everything visible and invisible; often used to mean 'wisdom'. Includes contemporary, historic, local and traditional knowledge.
Monitoring	The act of measuring change in the state, characteristics, number or presence of something.
Overfishing	A situation where observed fishing mortality (or exploitation) rates are higher than target or threshold levels.
Productivity	Productivity is a function of the biology of a species and the environment in which it lives. Generally, species with a high productivity are more resilient and take less time to rebuild from a depleted state.
Quota Management System (QMS)	A system established in 1986 to control the total commercial catch by allocating set amounts of catch or 'quota' for most of the main fish stocks in New Zealand's Exclusive Economic Zone.
Seamount	An isolated rise in elevation of 1,000 metres or more from the surrounding seafloor.
Seine Fishing	A method of fishing that uses a net which hangs vertically in the water with its bottom edge held down by weights and its top edge buoyed by floats. Purse-seines have a number of rings on their bottom edge with a line running through.
Species	A group of genetically isolated interbreeding populations, that typically share an ecological niche
Territorial Sea	A belt of coastal waters extending at most 12 nautical miles from the baseline (usually the mean low-water mark) of a coastal state.
Threat Classification Sytem	The New Zealand Threat Classification System classifies species as Not Threatened, Not Assessed or Threatened. Threatened includes three subcategories: Nationally Critical, Nationally Endangered and Nationally Vulnerable.
Trawling	A method of fishing that involves towing a large net through the water behind a boat for a set period before hauling on deck. Some operate in midwater, others touch the seafloor.

19.1.2 MĀORI NAMES MARINE SPECIES

Table 2. List of Aotearoa moana taxa in te reo Māori, English common names, and recognised scientific names to lowest level of taxonomic resolution.

Māori Name	English Common Name	Scientific Name
Aihe	Common dolphin	*Delphinus delphis*
Arokehe/kirirua/orea	Longfin eel	*Anguilla dieffenbachii*

Māori Name	English Common Name	Scientific Name
Ehouhounamu/ngutere/nanua/manua/maratea/ngutere	Red moki	*Cheiliodactylus spectabilis*
Hānea	Black mussel	*Xenostrobus pulex*
Hangenge/Ihe/takeke	Piperfish/garfish	*Hyporhamphus ihi*
Hanikura	Wedge clam	*Macomona liliana*
Hao/matamoe	Shortfin eel	*Anguilla australis*
Hāpuka/hāpuku/whapuku	Groper/wreckfish	*Polyprion oxygeneios*
Hoiho	Yellow-eyed penguin	*Megadyptes antipodes*
Hoka	Red cod	*Pseudophycis bachus*
Honu	Loggerhead sea turtle	*Caretta caretta*
Honu	Leatherback sea turtle	*Dermochelys coriacea*
Hoka/Hokorari	Ling	*Molva molva*
Hoki	Blue grenadier/blue hake/whiptail /hoki	*Macruronus novaezelandiae*
Hopuhopu/Kanae	Grey mullet	*Mugil cephalus*
Huangi/tuangi	New Zealand cockle	*Austrovenus stutchburyi*
Kakau moana	Gummy weed	*Splachnidum rugosum*
Kāeo	Cook's turban shell	*Cookia sulcata*
Kāeo	Snail whelk	*Haustrum haustorium*
Kahawai (hapukupuku)	Kahawai (juvenile form)	*Arripis trutta*
Kaikaikaroro	Triangle Shell	*Spisula aequilatera*
Kaitangata	Cat's eye	*Turbo smargdus*
Kakere/mangō pare	Hammerhead shark	*Sphyrna zygaene*
Kapetā	School shark	*Galeorhinus galeus*
Karahiwa/mararіwha	Queen pāua/austral abalone/yellow-footed pāua	*Haliotis australis*
Karengo/parengo/tupata	Southern laver	*Porphyra columbina*
Kataha/makawhiti/maraua/mokowhiti	Yellow-eye mullet	*Aldrichetta forsteri*
Kāunga	Hermit crab	Superfamily: Paguroidea
Kawikawi/Kehe/Koeae/ngehe	Marblefish	*Aplodactylus arctidens*
Karoro	Black-backed gull	*Larus dominicanus*
Kehe	Hake	*Merluccius australis*
Kekeno	New Zealand fur seal	*Arctocephalus forsteri*
Kina	Common sea urchin	*Evechinus chloroticus*
Koakoa/tītī/totorore/hakoko	Muttonbird/sooty shearwater	*Puffinsus griseus*
Koinga/mango tara/pioke	Spiny dogfish	*Squalus acanthias*
Koio/marapeka	Virgin/white foot pāua	*Haliotis virginea*
Kōiro/ kōriro/ngōiro	Southern conger eel	*Conger verreauxi*
Kōkiri	Leatherjacket	*Parika scaber*
Kōrama	Winkle/cat's eye	*Turbo smaragdus*
Korokoro/piharau/puhikorokoro/tuna korokoro	Lamprey	*Geotria australis*
Kororā	Little blue penguin/fairy penguin	*Eudyptula minor*
Kōtore moana	Red sea anemone	*Actinia tenebrosa*
Kōura papatea/matapara	Red rock lobster/spiny craysfish	*Jasus edwardsii*

Māori Name	English Common Name	Scientific Name
Kuku/kūtai/pōrohe	Green-lipped mussel	*Perna canaliculus*
Kuku-mau-toka	Banded mussel	*Modiolarca impacta*
Kumukumu/pūwhaiau	Red gurnard	*Chelidonichthys kumu*
Kūpā	Horse mussel	*Atrina pectinata zelandica*
Kupae	Sprat	*Sprattus* spp.
Kuparu	John Dory	*Zeus faber*
Kuwha	Elegant venus shell	*Irus elegans*
Kuwharu	Purple sunset shell	*Gari stangeri*
Kōpūpūtai roa	Long finger sponge	*Callyspongia ramosa*
Kōpūpūtai nui	Large cup sponge	*Gellius imperialis*
Kūtoro/moamoa/kourepoua	Spotted stargazer	*Genyagnus monopterygius*
Maki	Orca/killer whale	*Orcinus orca*
Mako	Shortfin mako	*Isurus oxyrinchus*
Makō/pioke	Rig/spotted dogfish/gummy shark	*Mustelus lenticulatus*
Mangō pounamu	Blue shark	*Prionace glauca*
Mangō ripi	Thresher shark	*Alopias vulpinus*
Mangō ururoa/mango tuatini	Great white shark/white shark/white pointer shark	*Carcharodon carcharias*
Mararī/rarī	Greenbone butterfish	*Odax pullus*
Moamoa/kourepoua	Giant stargazer/monkfish	*Kathetostoma giganteum*
Mohimohi	Pilchard	*Sardinops sagax*
Ngaeti	Common periwinkle	*Nodilittorina antipodum*
Ngerungeru/reremai	Bronze whaler	*Carcharhinus brachyurus*
Ngutoro/moeone	Bass	*Polyprion americanus*
Paea	Swordfish	*Xiphias gladius*
Pahiwihiwi	Kelpfish	*Chironemus marmoratus*
Paikea	Humpback whale	*Megaptera novaeangliae*
Pākirikiri/pātutuki/rāwaru	Blue cod/rock cod	*Parapercis colias*
Pākirikiri	Spotty	*Notolabrus celidotus*
Pāmu wēra/parāoa	Sperm whale	*Physeter macrocephalus*
Pāpaka	Paddle crab	*Ovalipes catharus*
Parakipīhi/tukuperu/upokohue/whāngai mokopuna	Pilot whale	*Globicephala melas*
Pātangaroa	Sun star	*Stichaster australis*
Pātiki	Flounder	*Rhombosolea* spp.,
Pātiki	Lemon sole	*Pelotretis flavilatus*
Pātiki	New Zealand turbot	*Colistium nudipinnis*
Pātiki tōtara/pātōtara	Yellowbelly flounder	*Rhombosolea leporina*
Pātiki mohoao/mohoao	Black flounder	*Rhombosolea retiaria*
Pawharu	Packhorse rock lobster	*Jasus verreauxi*
Pekapeka rau	Medusa-headed starfish	*Gorgonocephalus novaezelandiae*
Pōrohe	Blue mussel	*Mytilus edulis*

Māori Name	English Common Name	Scientific Name
Popoto	Māui dolphin	*Cephalorhynchus hectori maui*
Porotaka moana	Globe sponge	*Tethya aurantium*
Puhikorokoro	Yellow moray	*Gymnothorax prasinus*
Pūngorungoru/kōpūpūtai	Sponge	Phylum: Porifera
Purewha	Hairy mussel	*Modiolus areolatus*
Pūpū terakihi/muheke	Paper nautiluses	*Argonauta* spp.
Rātāhuihui	Sunfish	*Mola ramsayi*
Raiti wēra/tohora/kewa	Southern right whale	*Eubalaen australis*
Rāpoka	New Zealand Sea Lion	*Phocarctos hookeri*
Rau wheke	Sea lily	*Comanthus novaezelandiae*
Repe/reperepe	Elephantfish	*Callorrhinchus milii*
Rimu/rimurimu	Seaweed	Divisions: Rhodophyta, Phaeophyta, Chlorophyta
Rimurapa/kōauau	Bull kelp	*Durvillaea antarctica*
Rimurehia	Eelgrass/seagrass	*Zostera* spp.
Rori whiore	Tailed sea cucumber	*Caudina coriacea*
Taketake/tarāpunga	Red-billed seagull	*Larus noaehollandiae scopulius*
Tāmure	Snapper	*Chrysophrys auratus*
Tāngahangaha/tāngāngā	Banded wrasse	*Notolabrus fucicola*
Tarakihi	Ocean bream	*Nemadactylus macropterus*
Taranui	Caspian tern	*Hydroprogne caspia*
Tawaki/tawhaki/pokotiwha	Firodland crested penguin	*Eudpytes pachyrhynchus*
Tiikati	Hake	*Merluccius australis*
Tio reperepe/karauria ngakihi	Rock oyster	*Saccostrea glomerata*
Tītī	Cook's petrel	*Pterodroma cooki*
Totorore/tōtōrore	Antarctic prion	*Pachyptila desolata*
Toroa	Black-browed mollymawk	*Diomeda melanophrys*
Toroa	Northern royal albatross	*Diomeda sanfordi*
Toroa/toroa ingoingo/toroa whakaingo	Southern royal albatross	*Diomeda epomophora*
Toroa	Wandering albatross	*Diomedea exulans*
Toroa pango/pēō/kōputu	Light-mantled sooty albatross	*Phoebetria palpebrata*
Toheroa/moeone	-	*Paphies ventricosa*
Tuare/tuere	Common hagfish	*Eptatretus cirrhatus*
Tuatini	Sevengill shark	*Notorynchus cepedianus*
Tuatini	Sixgill shark	*Hexanchus griseus*
Tuatua/kahitua	-	*Paphies subtriangulata*
Tutumairekurai/tūpoupou/upokohue/pehipehi/ahoaho/waiaua	Hector's dolphin	*Cephalorhynchus hectori*
Warehenga	Kingfish	*Seriola lalandi*

Māori Name	English Common Name	Scientific Name
Whai	Stingray/skate	Order: Myliobatiformes, rajiformes
Whai keo/whai repo	Eagle ray	*Myliobatis tenuicaudatus*
Whai repo	Short-tail stingray	*Dasyatis brevicaudata*
Whai repo	Long-tail stingray	*Dasyatis thetidis*
Wheke	Octopus/squid	Subclass: Coleodiea

19.2 MEMBERSHIP AND PROTOCOLS FOR ALL SCIENCE WORKING GROUPS

Terms of Reference for Fisheries Assessment Working Groups (FAWGs) in 2021

Overall purpose

The purpose of the FAWGs is to assess the status of fish stocks managed within the Quota Management System, as well as other important species of interest to New Zealand. Based on scientific information the FAWGs assess the current status of fish stocks or species relative to MSY-compatible reference points and other relevant indicators of stock status, conduct projections of stock size and status under alternative management scenarios, and review results from relevant research projects. They do not make management recommendations or decisions (this responsibility lies with Fisheries New Zealand fisheries managers and the Minister responsible for fisheries).

Preparatory tasks

1. Prior to the beginning of the main sessions of FAWG meetings (January to May and September to November), Fisheries New Zealand fisheries scientists will produce a list of stocks and issues for which new stock assessments or evaluations are likely to become available prior to the next scheduled sustainability rounds. This list will include stocks for which the fishing industry and others intend to directly purchase scientific analyses. It is therefore incumbent on those purchasing research to inform the relevant FAWG chair of their intentions at least three months prior to the start of the sustainability round. FAWG Chairs will determine the final timetables and agendas for each Working Group.

2. At least six months prior to the main sessions of FAWG meetings, Fisheries New Zealand fisheries managers will alert Fisheries New Zealand science managers and the Fisheries New Zealand Principal Science Advisor to unscheduled special cases for which assessments or evaluations are urgently needed.

Technical objectives

3. To review new research information on stock structure, productivity, abundance, and related topics for each fish stock/issue under the purview of individual FAWGs.

4. Where possible, to derive appropriate MSY-compatible reference points[1] for use as reference points for determining stock status, based on the Harvest Strategy Standard for New Zealand Fisheries[2] (the Harvest Strategy Standard).

5. To conduct stock assessments or evaluations for selected fish stocks to determine the status of the stocks relative to MSY-compatible reference points[1] and associated limits, based on the "Guide to Biological Reference Points for Fisheries Assessment Meetings", the Harvest Strategy Standard, and relevant management reference points and performance measures set by fisheries managers.

6. For stocks where the status is unknown, FAWGs should use existing data and analyses to draw logical conclusions about likely future trends in biomass levels and/or fishing mortality (or exploitation) rates if current catches and/or TACs/TACCs are maintained, or if fishers or fisheries managers are considering modifying them in other ways.

7. Where appropriate and practical, to conduct projections of likely future stock status using alternative fishing mortality (or exploitation) rates, or catches, or other relevant management actions, based on the Harvest Strategy Standard and input from the FAWG and fisheries managers.

8. For stocks that are deemed to be depleted or collapsed, to develop alternative rebuilding scenarios based on the Harvest Strategy Standard and input from the FAWG and fisheries managers.

9. For fish stocks for which new stock assessments or analyses are not conducted in the current year, to review the existing Fisheries Assessment Plenary report text on the “Status of the Stocks” to determine whether the latest reported stock status summary is still relevant; else to revise the evaluations of stock status based on new data or analyses, or other relevant information.

Working Group reports

10. To include in the Working Group report information on commercial, Māori customary, non-commercial and recreational interests in the stock; as well as all other mortality to that stock caused by fishing, which might need to be allowed for in setting a TAC or TACC. Estimates of recreational harvest will normally be provided by the Marine Amateur Fisheries Working Group (MAFWG).

11. To provide information and advice on other management considerations (e.g., area boundaries, bycatch issues, effects of fishing on habitat, other sources of mortality, and input controls such as mesh sizes and minimum legal sizes) required for specifying sustainability measures. Sections of the Working Group reports related to bycatch and other environmental effects of fishing will be reviewed by the Aquatic Environment Working Group (AEWG) although the relevant FAWG is encouraged to identify to the AEWG Chair any major discrepancies between these sections and their understanding of the operation of relevant fisheries.

12. To summarise the stock assessment methods and results, along with estimates of MSY-compatible references points and other metrics that may be used as benchmarks for assessing stock status.

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[1] MSY-compatible reference points include those related to stock biomass (i.e., B_{MSY}), fishing mortality (i.e., F_{MSY}) and catch (i.e., MSY itself), as well as analytical and conceptual proxies for each of these three quantities.

[2] Link to the Harvest Strategy Standard: https://www.mpi.govt.nz/dmsdocument/728-harvest-strategy-standard-for-new-zealand-fisheries

13. To review, and update if necessary, the "Status of the Stocks" tables in the Fisheries Assessment Plenary report for all stocks under the purview of individual FAWGs (including those for which a full assessment has not been conducted in the current year) based on new data or analyses, or other relevant information.

14. For all important stocks, to complete (and/or update) the Status of Stocks tables using the template provided in the Introductory chapter of the most recent May and November Plenary reports.

15. It is desirable that full agreement amongst technical experts is achieved on the text of the FAWG reports, particularly the "Status of the Stocks" sections, noting that the AEWG will review sections on bycatch and other environmental effects of fishing, and the MAFWG will provide text on recreational harvests. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the FAWG report, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

Working Group input to the Plenary

16. To advise the Fisheries New Zealand Principal Science Advisor about stocks requiring review by the Fisheries Assessment Plenary and those stocks that are not believed to warrant review by the Plenary. The general criteria for determining which stocks should be discussed by the Plenary are that (i) the assessment is controversial and Working Group members have had difficulty reaching consensus on one or more base cases, or (ii) the assessment is the first for a particular stock or the methodology has been substantially altered since the last assessment, or (iii) new data or analyses have become available that alter the previous assessment, particularly assessments of recent or current stock status, or projections of likely future stock status. Such information could include:

    - new or revised estimates of MSY-compatible reference points, recent or current biomass, productivity or yield projections;
    - the development of a major trend in the catch or catch per unit effort; or
    - any new studies or data that extend understanding of stock structure, fishing patterns, or non-commercial activities, and result in a substantial effect on assessments of stock status.

Membership and Protocols for all Science Working Groups

17. FAWG members are bound by the Membership and Protocols required for all Science Working Group members (see separate document).

19.3 TERMS OF REFERENCES FOR THE AQUATIC ENVIRONMENT WORKING GROUPS (AEWG) FOR 2021

Overall purpose

For all New Zealand fisheries in the New Zealand TS and EEZ as well as other important fisheries in which New Zealand engages to assess, based on scientific information, the effects of (and risks posed by) fishing on the aquatic environment, include:

- bycatch and unobserved mortality of protected species (e.g., seabirds and marine mammals), fish, and other marine life, and consequent impacts on populations;
- effects on benthic ecosystems, species, and habitat;
- effects on biodiversity, including genetic diversity; and

- changes to ecosystem structure and function from fishing, including trophic effects.

Where appropriate and feasible, such assessments should explore the implications of the effect, including with respect to government standards, other agreed reference points, or other relevant indicators of population or environmental status. Where possible, projections of future status under alternative management scenarios should be made.

AEWG does not make management recommendations or decisions (this responsibility lies with Fisheries New Zealand fisheries managers and the Minister responsible for Fisheries).

Fisheries New Zealand also convenes a Biodiversity Research Advisory Group (BRAG) which has a similar review function to the AEWG. Projects reviewed by BRAG and AEWG have some commonalities in that they relate to aspects of the marine environment. However, the key focus of projects considered by BRAG is on the functionality of the marine ecosystem and its productivity, whereas projects considered by AEWG more commonly focus on the direct effects of fishing.

Preparatory tasks

1. Prior to the beginning of AEWG meetings each year, Fisheries New Zealand fisheries scientists will produce a list of issues for which new assessments or evaluations are likely to become available that year.

2. The Ministry’s research planning processes should identify most information needs well in advance but, if urgent issues arise, Fisheries New Zealand staff will alert the relevant AEWG Chair prior to the required meeting of items that could be added to the agenda. AEWG Chairs will determine the final timetables and agendas for meetings.

Technical objectives

3. To review any new research information on fisheries, including risks of impacts, and the relative or absolute sensitivity or susceptibility of potentially affected species, populations, habitats, and systems.

4. To estimate appropriate reference points for determining population, system, or environmental status, noting any draft or published Standards.

5. To conduct environmental assessments or evaluations for selected species, populations, habitats, or systems in order to determine their status relative to appropriate reference points and Standards, where such exist.

6. In addition to determining the status of the species, populations, habitats, and systems relative to reference points, and particularly where the status is unknown, AEWG should explore the potential for using existing data and analyses to draw conclusions about likely future trends in fishing effects or status if current fishing methods, effort, catches, and catch limits are maintained, or if fishers or fisheries managers are considering modifying them in other ways.

7. Where appropriate and practical, to conduct or request projections of likely future status using alternative management actions, based on input from AEWG, fisheries plan advisers, and fisheries and standards managers, noting any draft or published Standards.

8. For species or populations deemed to be depleted or endangered, to develop ideas for alternative rebuilding scenarios to levels that are likely to ensure long-term viability based on input from AEWG, fisheries managers, noting any draft or published Standards.

9. To review and revise existing environmental and ecosystem consideration sections of Fisheries Assessment Plenary report text based on new data or analyses, or other relevant information.

Working Group input to the Aquatic Environment and Biodiversity Annual Review

10. To include in contributions to the Aquatic Environment and Biodiversity Annual Review (AEBAR) summaries of information on selected issues that may relate to species, populations, habitats, or systems that may be affected by fishing. These contributions are analogous to Working Group reports from the Fisheries Assessment Working Groups.

11. To provide information and scientific advice on management considerations (e.g., area boundaries, bycatch issues, effects of fishing on habitat, other sources of mortality, and input controls such as mesh sizes and minimum legal sizes) that may be relevant for setting sustainability measures.

12. To summarise the assessment methods and results, along with estimates of relevant standards, references points, or other metrics that may be used as benchmarks or to identify risks to the aquatic environment.

13. It is desirable that full agreement among technical experts is achieved on the text of contributions to the AEBAR. If full agreement among technical experts cannot be reached, the Chair will determine how this will be depicted in the AEBAR, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

14. To advise the Fisheries New Zealand Principal Science Advisor and Aquatic Environment manager about issues of particular importance that may require independent review or updating in the AEBAR. The general criterion for determining which issues should be discussed by a wider group or text changed in the AEBAR is that new data or analyses have become available that alter the previous assessment of an issue, particularly assessments of population status or projection results. Such information could include:

    - New or revised estimates of environmental reference points, recent or current population status, trend, or projections;
    - The development of a major trend in bycatch rates or amount;
    - Any new studies or data that extend understanding of population, system, or environmental susceptibility to an effect or its recoverability, fishing patterns, or mitigation measures that have substantial implications for a population, system, or environment, or identify risks associated with fishing activity; and
    - Consistent performance outside accepted reference points or Standards.

Membership and Protocols for all Science Working Groups

15. The AEWG is bound by the same membership and protocols as are other Science Working Groups (see separate document).

19.4 TERMS OF REFERENCE FOR THE BIODIVERSITY RESEARCH AND ADVISORY GROUP (BRAG) IN 2021

Overall purpose

Since 2000, the objectives of the Biodiversity Research Programme had been drawn directly from Fisheries New Zealand commitments to Theme 3 of the New Zealand Biodiversity Strategy (NZBS) 2000. The refresh of the NZBS (Te Mana o te Taiao – The Aotearoa New Zealand Biodiversity Strategy 2020) was released in 2020 and Fisheries New Zealand subsequently developed new workstreams within the Biodiversity Research Programme to align with the objectives and goals of this strategy. Within this framework, the workstreams will adapt over time as new issues emerge, to build on synergies with other research programmes and work where biodiversity is under greatest threat from fishing or other anthropogenic activities, within the constraints of the overall purpose of the programme which is:

> "To improve our understanding of New Zealand marine ecosystems in terms of species diversity, marine habitat diversity, and the processes that lead to healthy ecosystem functioning, and the role that biodiversity has for such key processes"

The science currently commissioned broadly aims to:

- Describe and characterise the distribution and abundance of fauna and flora, as expressed through measures of biodiversity, and improving understanding about the drivers of the spatial and temporal patterns observed;

- Determine the functional role of different organisms or groups of organisms in marine ecosystems, and assess the role of marine biodiversity in mitigating the impacts of anthropogenic disturbance on healthy ecosystem functioning; and

- Identify which components of biodiversity must be protected to ensure the sustainability of a healthy marine ecosystem as well as to meet societal values on biodiversity.

Fisheries New Zealand also convenes an Aquatic Environment Working Group (AEWG) which has a similar review function to BRAG. Projects reviewed by BRAG and AEWG have some commonalities in that they relate to aspects of the marine environment. However, the key focus of projects considered by BRAG is on marine issues related to the functionality of the marine ecosystem and its productivity, whereas projects considered by AEWG are more commonly focused on the direct effects of fishing.

BRAG may identify natural resource management issues that extend beyond fisheries management and make recommendations on priority areas of research that will inform Fisheries New Zealand or other government departments of emerging science results that require the attention of managers, policymakers, and decision-makers in the marine sector. BRAG does not make management recommendations or decisions (this responsibility lies with Fisheries New Zealand fisheries managers and the Minister responsible for Fisheries).

Preparatory tasks

1. Prior to the beginning of BRAG meetings each year, Fisheries New Zealand fisheries scientists will produce a list of issues for which new research projects are likely to be required in the forthcoming financial year. The BRAG Chair will determine the final timetables and agendas.

2. The Ministry's research planning processes should identify most information needs well in advance but, if urgent issues arise, Fisheries New Zealand fisheries managers will alert the Aquatic Environment and Biodiversity Science Manager and the Principal Advisor Fisheries Science at least three months prior to the required meetings where possible.

BRAG technical objectives

3. It is the responsibility of the BRAG to review, discuss, and convey views on the results of marine biodiversity research projects contracted by Fisheries New Zealand. The review process is an evaluation of how existing research results can be built upon to address emerging research issues and needs. It is essentially an evaluation of "what we already know" and how this can be used to obtain "what we need to know". This information should be used by BRAG to identify gaps in our knowledge and for developing research plans to address these gaps.

4. It is the responsibility of BRAG participants to discuss, evaluate, make recommendations, and convey views on particular research areas as required. Individual related projects on a species or fishery or research topic need to be aligned with relevant strategic and policy directions.

5. The recommendations on project proposals for the next financial year will be submitted via the Chair of BRAG to the Principal Science Advisor Fisheries.

6. The Biodiversity Research Programme includes research in New Zealand's TS, EEZ, Extended Continental Shelf, the South Pacific Region, and the Ross Sea region. There are five scientific work streams as follows:

- Iwi and community-based biodiversity projects
- Baseline knowledge for Ecosystem-Based Fisheries Management (EBFM)
- Tools and methods for biodiversity and ecosystem assessment
- Functional threats to biodiversity
- Sustainability for future generations

BRAG input to the Fisheries Assessment Plenary and the Aquatic Environment and Biodiversity Annual Review

7. To contribute to and summarise progress on biodiversity research in the Aquatic Environment and Biodiversity Annual Review. This contribution is analogous to Working Group Reports from the Fisheries Assessment Working Groups.

8. To summarise the assessment methods and results, along with estimates of relevant standards, references points, or other metrics that may be relevant to biodiversity objectives, the Biodiversity Strategy, and international obligations.

9. It is desirable that full agreement among technical experts is achieved on the text of these contributions. If full agreement among technical experts cannot be reached, the Chair will determine how this will be depicted in the Aquatic Environment and Biodiversity Annual Review, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

10. To advise the Principal Science Advisor Fisheries about issues of particular importance that may require review by a Plenary meeting or summarising in the Aquatic Environment and Biodiversity Annual Review. The general criterion for determining which issues should be discussed by a wider group include:
    - Emerging issues, recent or current biodiversity status assessments, trends, or projections;
    - The development of a major trend in the marine environment that will impact on marine productivity or ecosystem resilience to stressors; and
    - Any new studies or data that impact on international obligations.

Membership and Protocols for all Science Working Groups

11. The BRAG is bound by the same membership and protocols as are other Science Working Groups (see separate document).

19.5 TERMS OF REFERENCE FOR THE ANTARCTIC WORKING GROUP (ANTWG) IN 2021

Overall purpose

The purpose of the ANTWG is to review science and research information intended for submission to or use by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). CCAMLR is an inter-governmental organisation that is committed to conserving the marine life of the Southern Ocean while allowing rational use of marine resources, including commercial fishing. The CCAMLR Convention requires that management considers the effects of fishing on dependent and associated species as well as on the target species. The area of jurisdiction of the CCAMLR Convention is approximately south of the circumpolar Antarctic Polar Front in the Southern Ocean. Science and research requested or used by CCAMLR may include, *inter alia*, fisheries characterisations, abundance indices, catch-at-age or catch-at-length data, and stock assessment modelling to assess the status of fish stocks managed by CCAMLR; bycatch and unobserved mortality of protected species, fish, and other marine life; effects on biodiversity and benthic biodiversity, species, and habitat; and changes to ecosystem structure and function as a result of fishing, including trophic effects. The ANTWG also undertakes scientific review of documents and papers that may be submitted to the scientific working groups of CCAMLR to aid and inform its management. The ANTWG does not make management recommendations or decisions; these responsibilities lie with CCAMLR's Scientific Committee and the Commission.

Preparatory tasks

1. Prior to the first meeting of the ANTWG each year, the ANTWG Chair will produce a list of stocks/issues for which new stock assessments, evaluations, impact assessments, risk assessments, or other scientific analyses have been requested by the CCAMLR Scientific Committee or the Commission (including its contributing bodies), fishing industry, or other stakeholders. The ANTWG Chair will determine the final timetables and agendas of the working group each year, taking account of the available time and resources.

Technical objectives

2. To review new research information on stock structure, productivity, abundance, and related topics for each fish stock or environmental issue under the purview of the ANTWG.
3. Where possible, to derive yields or reference points requested by CCAMLR's Scientific Committee or Commission related to fish stocks or environmental issues relevant to CCAMLR fisheries.

4. To conduct stock assessments or evaluations for selected stocks to determine the precautionary yields and status of the stocks relative to the requested reference points or, if no such reference points are specified by CCAMLR, MSY-compatible reference points and associated limits, based on the "Guide to Biological Reference Points for Fisheries Assessment Meetings" and New Zealand's Harvest Strategy Standard.
5. For stocks where the status is unknown, the ANTWG should, where possible, use any existing data and analyses to draw conclusions about likely future trends in biomass levels and/or fishing mortality (or exploitation) rates if current catches and/or TACs are maintained, or if fishers or CCAMLR are considering modifying them in other ways.
6. Where requested by the CCAMLR Scientific Committee or Commission, to conduct projections of likely future stock status using alternative fishing mortality (or exploitation) rates or catches and other relevant management actions, based on input from the ANTWG and any guidance from the CCAMLR Scientific Committee or Commission.
7. Where requested by the CCAMLR Scientific Committee or Commission, in relation to specified stocks, to develop and report on alternative rebuilding scenarios.
8. To conduct environmental impact assessments and qualitative or quantitative risk assessments in relation to bycatch species, other species of concern, benthic systems, or vulnerable marine ecosystems to support the work of the CCAMLR Scientific Committee and Commission.

Working Group reports

9. To review, and update if necessary, the "Status of the Stocks" tables in the Fisheries Assessment Plenary report based on new data or analyses, or other relevant information.
10. To complete (and/or update) the Status of Stocks tables using the template provided in the Introductory chapter of the most recent May Plenary report.
11. To review, and update if necessary, the "Antarctic Science" chapter of the Aquatic Environment and Biodiversity Review (AEBAR) based on new data or analyses, or other relevant information.
12. It is desirable that full agreement amongst technical experts is achieved on the text of the ANTWG reports. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the ANTWG report, will document the extent to which agreement or consensus was achieved, and record and attribute any residual disagreement in the meeting notes.

Papers and reports to CCAMLR

13. Papers and reports summarising work reviewed by the ANTWG are generally submitted to CCAMLR's Scientific Committee, and their content varies widely. It is desirable that full agreement amongst technical experts is achieved on the content of such papers or reports, noting that deadlines for submission to CCAMLR may require the Chair to finalise text after a meeting of the ANTWG has considered and resolved scientific issues. If full agreement amongst technical experts cannot be reached, the Chair will determine how this will be depicted in the paper or report to be submitted to CCAMLR. In such cases, the Chair will also document the extent to which agreement or consensus was achieved and record and attribute any residual disagreement in the meeting notes.

Membership and Protocols for all Science Working Groups

14. ANTWG members are bound by the Membership and Protocols required for all Science Working Group members (see separate document).

FISHERIES ASSESSMENT WORKING GROUPS: MEMBERSHIP 2021

Antarctic Working Group

Convenors: Marine Pomarède and Nathan Walker

Members: Matthew Baird, Stephanie Brown, Jennifer Devine, Alistair Dunn, Jack Fenaughty, Greig Funnell, Simon Hoyle, Leyla Knittweis-Mifsud, Dan MacGibbon, Bradley Moore, Monique Messina, Phillip Neubauer, Richard O'Driscoll, Steve Parker, Matt Pinkerton, Brodie Plum, Darryn Shaw, Andy Smith, Perry Smith, Josh Van Lier, Tim Vaughan-Sanders, Barry Weeber, D'arcy Webber.

Species: Antarctic toothfish

Aquatic Environment Working Group – Protected Species

Convenors: William Gibson and Ben Sharp

Members: Ed Abraham, Carolyn Aguilar, Owen Anderson, Sonja Austin, Hilary Ayrton, Karen Baird, Barry Baker, Scott Baker, Joshua Baller, Josh Barclay, Steve Beatson, Erik Behrens, Elizabeth Bell, Mike Bell, Katrin Berkenbusch, Tiffany Bock, Laura Boren, Christine Bowden, David Bowden, Erin Breen, Paul Breen, Anthony Brett, Tom Brough, Curly Brown, Ian Brown, Sarah Bury, Glen Carbines, Susan Chalmers, Mark Chambers, Simon Childerhouse, Malcolm Clark, Tom Clark, Katie Clemens-Seely, Deanna Clement, George Clement, Damian Cloeter, Rochelle Constantine, Justin Cooke, Vonda Cummings, Roberta D'Archino, Steve Dawson, Igor Debski, Jessica Desmond, Jennifer Devine, Christopher Dick, Peter Dillingham, Clinton Duffy, Alistair Dunn, Matt Dunn, Charles Edwards, Mark Edwards, Pablo Esobar-Flores, Jack Fenaughty, Brit Finucci, David Foster, Allen Frazer, Debbie Freeman, Richa Garg, Sharleen Gargiulo, Shane Geange, Mark Geytenbeek, Dave Goad, Bruce Hartill, Barb Hayden, Jeremy Helson, Hannah Hendriks, Kristina Hillock, Freyda Hjorvarsdottir, Lyndsey Holland, Steven Holmes, Simon Hoyle, Lucy Jacob, Emma Jones, Daniel Kerrigan, Brianna King, Kirstie Knowles, Jo Lambie, Todd Landers, Kath Large, Laws Lawson, Mary Livingston, Carolyn Lundquist, Dave Lundquist, Greg Lydon, Darryl MacKenzie, Lucy Manning, Thomas Mattern, Sue Maturin, Gemma McGrath, Andy McKenzie, Stefan Meyer, Karen Middlemiss, David Middleton, Jodi Milne, Janice Molloy, Kiri Morgan, Mark Morrison, Rikki Mules, Philip Neubauer, Richard O'Driscoll, Enrique Pardo, Graham Parker, Steve Parker, Darren Parsons, Michael Patrick, Heiko Philippi, Johanna Pierre, Matt Pinkerton, Tiffany

Plencner, Will Rayment, Trish Rea, Nathan Reid, Yvan Richard, Jesse Rihia, Peter Ritchie, Jim Roberts, Ashley Rowden, Richard Saunders, Carol Scott, Katherine Short, Liz Slooten, Andy Smith, Paul Starr, Kevin Sullivan, Darryl Sykes, John Taunton-Clark, Graeme Taylor, David Thompson, Finlay Thompson, Hamish Tijsen, Rob Tilney, Geoff Tingley, Rob Tinkler, Di Tracey, Ian Tuck, Dominic Vallieres, Anton Van Helden, Josh van Lier, Adam Watson, Shannon Weaver, D'Arcy Webber, Trudi Webster, Barry Weeber, Richard Wells, Tamar Wells, James Williams, Oliver Wilson, Inge Wisselink, Jeanne Wissing, Andrew Wright, Jingjing Zhang.

Aquatic Environment Working Group – Benthic and Habitats

Convenors: Karen Tunley

Members: Ed Abraham, Carolyn Aguilar, Owen Anderson, Sonja Austin, Hilary Ayrton, Karen Baird, Barry Baker, Scott Baker, Joshua Baller, Josh Barclay, Steve Beatson, Erik Behrens, Elizabeth Bell, Mike Bell, Katrin Berkenbusch, Tiffany Bock, Laura Boren, Christine Bowden, David Bowden, Erin Breen, Paul Breen, Anthony Brett, Tom Brough, Curly Brown, Ian Brown, Sarah Bury, Glen Carbines, Susan Chalmers, Mark Chambers, Simon Childerhouse, Malcolm Clark, Tom Clark, Katie Clemens-Seely, Deanna Clement, George Clement, Damian Cloeter, Rochelle Constantine, Justin Cooke, Vonda Cummings, Roberta D'Archino, Steve Dawson, Igor Debski, Jessica Desmond, Jennifer Devine, Christopher Dick, Peter Dillingham, Clinton Duffy, Alistair Dunn, Matt Dunn, Charles Edwards, Mark Edwards, Pablo Esobar-Flores, Jack Fenaughty, Brit Finucci, David Foster, Allen Frazer, Debbie Freeman, Richa Garg, Sharleen Gargiulo, Shane Geange, Mark Geytenbeek, Dave Goad, Bruce Hartill, Barb Hayden, Jeremy Helson, Hannah Hendriks, Kristina Hillock, Freyda Hjorvarsdottir, Lyndsey Holland, Steven Holmes, Simon Hoyle, Lucy Jacob, Emma Jones, Daniel Kerrigan, Brianna King, Kirstie Knowles, Jo Lambie, Todd Landers, Kath Large, Laws Lawson, Mary Livingston, Carolyn Lundquist, Dave Lundquist, Greg Lydon, Darryl MacKenzie, Lucy Manning, Thomas Mattern, Sue Maturin, Gemma McGrath, Andy McKenzie, Stefan Meyer, Karen Middlemiss, David Middleton, Jodi Milne, Janice Molloy, Kiri Morgan, Mark Morrison, Rikki Mules, Philip Neubauer, Richard O'Driscoll, Enrique Pardo, Graham Parker, Steve Parker, Darren Parsons, Michael Patrick, Heiko Philippi, Johanna Pierre, Matt Pinkerton, Tiffany Plencner, Will Rayment, Trish Rea, Nathan Reid, Yvan Richard, Jesse Rihia, Peter Ritchie, Jim Roberts, Ashley Rowden, Richard Saunders, Carol Scott, Katherine Short, Liz Slooten, Andy Smith, Paul Starr, Kevin Sullivan, Darryl Sykes, John Taunton-Clark, Graeme Taylor, David Thompson, Finlay Thompson, Hamish Tijsen, Rob Tilney, Geoff Tingley, Rob Tinkler, Di Tracey, Ian Tuck, Dominic Vallieres, Anton Van Helden, Josh van Lier, Adam Watson, Shannon Weaver, D'Arcy Webber, Trudi Webster, Barry Weeber, Richard Wells, Tamar Wells, James Williams, Oliver Wilson, Inge Wisselink, Jeanne Wissing, Andrew Wright, Jingjing Zhang.

Aquatic Environment Working Group – Non-Target Fish and Invertebrate Catch

Convenors: Marco Milardi and Josh van Lier

Members: Owen Anderson, Hilary Ayrton, Ian Brow, Tiff Bock, Glen Carbines, Mark Chambers, Simon Childerhouse, Tom Clark, Damien Cloeter, Rochelle Constantine, Jean Davis, Jennifer Devine, Clinton Duffy, Charles Edwards, Rosa Edwards, Malene Felsing, Jack Fenaughty, Brit Finucci, David Foster, Dave Goad, Kat Goddard, Phil Heath, Lyndsey Holland, Aaron Irving, Emma Jones, Brianna King, Todd Landers, Kath Large, Mary Livingston, David Lundquist, Greg Lydon, Stefan Meyer, Karen Middlemiss, David Middleton, Sophie Mormede, Mark Morrison, Campbell Murray, Richard O'Driscoll, Enrique Pardo, Darren Parsons, Trent Rasmussen, Nathan Reid, Jesse Rihia, Carolyn Scott, Fabrice Stephenson, Karli Thomas, Rob Tilney, Rob Tinkler, Ian Tuck, Karen Tunley Te Aomihia Walker, Cath Wallace, Barry Weeber, Richard Wells, Tamar Wells, John Willmer, Jeanne Wissing

Biodiversity Research and Advisory Group (BRAG)

Convenor: Mary Livingston

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19.6 AQUATIC ENVIRONMENT AND BIODIVERSITY FUNDED AND RELATED PROJECTS

The AEBR citations in the following table can be found in an online repostitory here, and FRR and other reports may be found here.

Theme	Project code	Project title	Specific objectives	Status	Citation/s
BEN	BEN2021-03	Taxonomic identification of benthic invertebrate samples	1. To taxonomically identify deepwater benthic invertebrates to species level where possible in samples taken during research trawls and by observers on fishing vessels targeting deepwater stocks. 2. To update relevant databases recording the catch of invertebrates in research trawls and commercial fishing.	In progress	
BYC	BYC2021-02	Protected Chondrichthyan captures characterisation	1. Describe and characterise captures of all protected Chondrichthyans in New Zealand waters, using all available data. 2. Analyse gears/fisheries, locations, status at release, and capture trends, and identify factors that appear to lead to captures.	In progress	
BYC	BYC2021-03	Bycatch monitoring and quantification in deepwater fisheries	Estimate catch composition in target deepwater fisheries (various fisheries to be addressed cyclically). This should include an estimation of the quantity of non-target fish species caught, and the target and non-target fish species discarded, in a format that meets management needs. Estimated rates and amounts of bycatch and discards in the different deepwater fisheries are to be compared with previous estimates to identify trends over time.	In progress	
BYC	ENV2020-20	Temporal and spatial distribution of non-target catch, and non-target species, in deepwater fisheries	1. A spatially- and temporally-detailed representation of non-target catch in deepwater fisheries around New Zealand, coupled with catch data and fishery information on the location of non-target catch hotspots; the influence of seasonal factors on non-target catch hotspots; and the species and fishing methods that may be driving these hotspots. 2. A spatial distribution of some of the most representative non-target species, modelled from catch and survey data, as well as environmental data.	In progress	

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BEN	KAI2020-01	Monitoring of rocky reef habitats across degrees of uplift caused by the 2016 Kaikoura earthquake	1. Repeat three annual surveys of 16 intertidal areas across degrees of uplift. 2. Repeat three annual surveys of 6 subtidal sites across degrees of uplift. 2. Assess the recovery of nearshore rocky reef communities.	In progress	Falconer et al. (2021) FRR 4243
BEN	SEA2020-14	Pelorus multibeam groundtruthing and analysis	1. Generate BTM classifications from attributes, including depth, depth statistics (standard deviation), depth range (difference between minimum and maximum depth), slope (angular units from the horizontal), slope statistics (standard deviation), curvature (a measure of the change of slope), aspect (direction of the downslope dip) and rugosity (ratio of surface area to planar area, roughness) or a measure of terrain complexity. 2. Data compilation of marine mammal sighting into the integrated GIS project. 3. Production of GIS layers for MPI and compilation of a digital summary document that provides an overview of the BTM results and appropriate examples.	Complete	Maier et al. (2021) FRR 4202
ZBD	ZBD2020-06	Recovery of biogenic habitats	1. Assess the potential for the recovery of habitats, offered by various spatial planning scenarios, taking into account meta-community dynamics, habitat condition, and ongoing stressors at local and regional scales and the interactive effects thereof. 2. Identify feasible strategies for the recovery and maintenance of habitats in the Hauraki Gulf Marine Park and suggest principles for future spatial planning processes to enhance recovery potential.	In progress	
ZBD	ZBD2020-07	Recovery of Seamount Communities	1. To repeat the quantitative photographic survey of benthic invertebrate communities on features of the Graveyard Knolls complex. 2. To assess changes in benthic communities since the first survey in 2001.	In progress	
ZBD	ZBD2020-08	Role of low and mid trophic fish in the Hauraki Gulf	1. The project should deliver a model describing the role of low- and mid-trophic level fish in terms of energy transfer through the food web of the Hauraki Gulf.	In progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2020-09	Cumulative effects of stressors on scallops and scallop habitats in the Marlborough Sounds	1. Use experimental and modelling techniques to assess the cumulative effect of a range of physical, biological, and ecological stressors (including fishing) on scallops and scallop habitat in the Marlborough Sounds.	In progress	
ZBD	ZBD2020-11	Review of land-based effects on coastal fisheries and kaimoana and their habitats	1. To review current state of knowledge of land-based effects on coastal fisheries and kaimoana species in New Zealand. 2. To collate information on land-based stressors to coastal ecosystems, mitigation measures taken, and to provide a detailed list of all available datasets and spatial layers at regional and national scales	In progress	
BEN	BEN2019-01	Monitor the extent and intensity of bottom contact by trawl and dredge fishing in the Territorial Sea and Exclusive Economic Zone	1. To help MPI groom data, develop and compile summary statistics for all deepwater and inshore trawl and dredge fishing by year, depth zone, sediment categories, fishable area, and any other agreed habitat classifications or proxies, and to identify any trends or changes and meet management needs. 2. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results.	Complete	Baird & Mules (2021b) AEBR 260
BEN	BEN2019-05	Spatial decision support tool development for managing the impacts of bottom fishing on in-zone, particularly vulnerable or sensitive habitats.	1. Compile relevant inputs to be used in a spatial planning tool. This should include benthic biodiversity inputs, naturalness and the value to resource users. 2. Apply the outputs of objective 1 to a spatial decision support tool to be used by fisheries management that will enable scenario testing and allow for the cost to fishing to be determined. 3. Update any relevant sections in the Aquatic Environment and Biodiversity Annual Review with new results.	In progress	
PRO	PMM2019-10	Update Campbell Island NZSL (Population Sustainability Threshold) estimation.	1. Compile and update population demographic data indicative of population size and trend, and indicators of relevant demographic rates, from the Campbell Islands New Zealand Sea Lion population 2. Update population models available from previous and ongoing research projects (i.e.,	In progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			Roberts et al. 2014 and PRO2018-01) to approximate the current population consistent with Objective 1 and simulate future population trajectories under alternate management and climatic scenarios, applying a range of sensitivities chosen in consultation with FNZ. 3. Apply the model(s) and sensitivities in Objective 2 to estimate Population Sustainability Threshold (PST) values consistent with a range of relevant population outcomes (selected in consultation with FNZ).		
PRO	PRO2019-01	Preparation and documentation of standardized linked database – including commercial fisheries effort data, fisheries observer data, and protected species captures.	1. Describe and catalog existing MPI databases (including their history and custody chain) containing data relevant to understanding protected species/ commercial fisheries interactions. 2. Host a data linking/grooming workshop (including FNZ, industry, research providers, and technical/database experts) to discuss, document, and modify existing data linking and grooming algorithms used to combine and modify these databases (especially regarding the linking of trip level data from COD and WAREHOU). 3. Utilising existing databases, prepare a comprehensive database linking data from commercial catch/effort databases (e.g., WAREHOU), fisheries observer databases (i.e., COD), and protected species captures databases owned by MPI (e.g., OREO, sea lion SLED database, etc). 4. Audit, document, and catalog in a standardised way the error-trapping and grooming code that are applied to modify these data before they are used in protected species research, both for the standard 'root' database and at the 'branch' level, i.e., as applied in the course of delivering individual research projects. 5. Build and update a standardised and easily queried repository for metadata documenting the database, its genesis and structure, version control, and all grooming/linking/error-trapping code, including those arising from future protected species research projects.	In progress	
PRO	PRO2019-02	Maintenance of PSC (protected species captures) website displaying updated	To maintain a public website displaying a time series of fishing effort, observer effort, observed captures and available estimates of total captures/mortalities	In progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		observer commercial fisheries captures, and total estimated captures for selected species.	in trawl, longline, set net and purse seine fisheries within the New Zealand EEZ.		
PRO	PRO2019-09	Spatial distribution modelling of at-risk seabirds in New Zealand commercial fisheries.	1. Secure access to existing satellite telemetry for New Zealand seabird populations, including via the Global Procellariform Tracking Database. 2. Compile available data indicative of seabird distributions -- including satellite telemetry, sightings data, fisheries captures, and colony-specific population estimates -- for up to ten seabird species chosen in consultation with FNZ. 3. Apply spatial modelling techniques to estimate the spatial distribution and density of selected seabird species (including seasonal variation, and estimated separately for breeding and non-breeding birds), using seabird data from Objective 2 and spatially comprehensive environmental data layers available from other FNZ funded projects. 4. In collaboration with researchers delivering updated parameterisation of the seabird SEFRA model, examine the spatial goodness-of-fit of captures estimates using the updated spatial distribution layers from Objective 3, and modify these layers as appropriate where poorly fitted patterns seem to reflect poorly estimated spatial distribution layers.	In progress	
PRO	PRO2019-10	Refine SEFRA model parameterization for at-risk protected species (seabirds).	1. Use available published and unpublished data to test and refine biological input parameterisation (i.e., definition of priors) in the multi-species seabird risk assessment model produced under project PRO2016-06, with an emphasis on at-risk and high-capture species. 2. Within the existing Risk Atlas platform, test and refine the underlying structural assumptions affecting the estimation of vulnerability in the multi-species seabird risk assessment model (i.e., the definition of fisheries vulnerability groups, species vulnerability groups, and time periods over which vulnerability is estimated). 3. Illustrate the effects of input and structural changes applied under Objectives 1 and 2 in a transparent way, using sensitivities, and document their scientific justification.	In progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			4. Using the Risk Atlas platform, query, disaggregate and summarise the outputs of the updated multi-species seabird risk assessment in consultation with FNZ scientists and managers, to inform relevant questions at the scale of particular fisheries or to address the particular information needs of fisheries managers at smaller scales, rather than only at the species scale. 5. Create a publicly accessible online repository of risk assessment inputs (biological parameters, spatial data layers, and vulnerability group definitions under objectives 1 and 2) and summarised risk assessment outputs, with a simple user interface to facilitate transparency.		
PRO	PRO2019-11	Historical reconstruction and characterization of spatially explicit historical set-net fishing effort data.	1. Compile and summarise available fisheries catch/effort data, landings data, and other historical data indicative of spatial patterns of setnet fishing (including subsistence, recreational, and commercial setnetting) in the period before there were requirements to report fisheries catch and effort data at finer spatial scales (and as early as 1900). 2. Compile and summarise available data indicative of historical protected species capture rates in setnet fisheries, especially in the period before there were requirements to report protected species captures in a standardised way. 3. Host a workshop including fisheries managers, scientists, fishing industry experts and historians to identify relevant data under Objectives 1 and 2 and to discuss its interpretation, with the aim of turning qualitative or summarised data into spatially and temporally resolved estimates of historical setnet effort and protected species captures. 4. Using spatial modelling approaches, generate spatially explicit estimates of setnet fishing effort intensity at decadal scales during the 20th and 21st centuries, within fishery groups corresponding to gear or deployment characteristics likely to affect protected species capture rates (e.g., net material, mesh size, target species). 5. Consistent with the SEFRA approach, intersect spatial effort characterisations under Objective 4 with protected species and fish distribution layers available from other projects, to estimate spatially resolved estimates of historical encounter rates (i.e., spatial overlap) at a decadal scale.	In progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			6. Combining estimated encounter rates from Objective 5 with evidence of historical captures from Objectives 1 and 2, estimate catchability, total captures, and population-level risk in historical setnet fisheries for up to four species of interest, chosen in consultation with FNZ (including Hector's – Māui dolphins).		
PRO	PSB2019-01	Estimation of total seabird captures using standardized estimation methods.	To estimate capture rates and total captures of seabirds (plus fur seals) by method, area, and target fishery, and where possible, by species for the 2019/20, 2020/21 and 2021/22 fishing years.	In progress	
PRO	PSB2019-02	Distributional study of Antipodean albatross using satellite reporting GPS tags.	To collect spatial data to allow refinement of the spatial overlap with fishing throughout New Zealand's EEZ and the South Pacific, with tracking devices to be deployed on a wider range of ages/breeding stages.	In progress	
PRO	PSB2019-04	Black petrel population monitoring and distributional study.	1. To collect information on population size, adult survival, age at first reproduction and key demographic parameters for black petrel to reduce uncertainty or bias in estimates of risk. 2. To collect spatial data to allow refinement of the spatial overlap with fishing, with tracking devices to be deployed on a wider range of ages/breeding stages.	In progress	
PRO	PSB2019-06	Review footage collected from the 2018/19 Black Petrel Electronic Monitoring project.	1. To conduct footage review of electronic monitoring footage collected from the snapper and bluenose bottom longline fisheries in FMA1. 2. To conduct a comparative analysis of the various datasets collected regarding seabird captures.	In progress	
PRO	PSB2019-07	Continuation of the Black Petrel Electronic Monitoring Project for the 2019/20 summer.	1. To collect footage via an electronic monitoring programme on snapper and bluenose bottom longline fisheries in FMA1. 2. To review footage. 3. To conduct a comparative analysis of the various datasets collected regarding seabird captures.	In progress	

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PRO	PSB2019-08	Feasibility trial of underwater baitsetter.	1. Feasibility trial of the underwater baitsetter in New Zealand surface longline. 2. Assessment of the behavioural response of seabirds to the underwater baitsetter.	In progress	
PRO	PSB2019-09	Aerial survey of white-capped albatross on the Auckland Islands.	1. To undertake aerial surveys of the white-capped albatross on Auckland Islands in 2019/20 and 2020/21, and undertaking analysis of total counts for each field season. 2. Based on the counts in Objective 1 and previous counts on the resulting dataset covering 14 years, assess the population trend, taking into account the proportion of loafers identified in the photo montages and by ground counts.	In progress	
PRO	SEA2019-06	Increased fisher-reporting of seabird captures during an electronic monitoring trial.	1. To conduct comparative analysis of fisher reporting of protected species captures for vessels in the SNA BLL fishery in FMA1 for vessels in the electronic monitoring pilot fleet	Complete	Tremblay-Boyer & Abraham (2020) AEBR 238
ZBD	ZBD2019-01	Quantifying Benthic Biodiversity – Phase 2	1. Predict gradients in benthic faunal turnover across Campbell Plateau (CP) using relationships between faunal distributions and environmental gradients developed for Chatham Rise (CR) under ZBD2016-11; 2. Run a dedicated photographic survey of seabed habitats and fauna across CP, structured on the basis of predictions from (1); 3. Use quantitative data from the CP survey to assess the utility of predictions from the existing CR models when applied to a neighbouring area of the EEZ; 4. Generate updated models with a spatial domain encompassing both regions by merging data from the CP survey with the existing CR dataset.	Complete	Stephenson et al. (2021) AEBR 276
ZBD	ZBD2019-04	Plastics and marine debris across the ocean floor in New Zealand waters.	1. Create a metadatabase of all potential sources of data on plastics and marine debris in New Zealand waters. 2. Select and analyse appropriate sets and subsets of data that will provide density estimates of plastics and map debris on the seabed.	Complete	Behrens et al. (2021) AEBR 267

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			3. Provide summary statistics of benthic debris density, types of debris, and heat maps of occurrence throughout the TS and EEZ.		
ZBD	ZBD2019-08	Sources of suspended sediment load and impact on coastal kaimoana.	Compare and contrast sedimentation datasets from different estuaries and determine how different sedimentation histories have impacted on a range of kaimoana species.	Procurement not initiated	
ZBD	ZBD2019-11	Development of Electronic Automated Reporting System (EARS) to improve seabird bycatch monitoring.	1. Develop and ruggedize technology and test on NZ domestic vessel. 2. At sea trials in New Zealand and Japan.	In progress	
BEN	BEN2018-01	Monitoring of trawl footprint (including coastal).	1. To help MPI groom data, develop and compile summary statistics for all deepwater and inshore trawl fishing by year, depth zone, fishable area, and any other agreed habitat classifications or proxies, and to identify any trends or changes to meet management needs. 2. To update any relevant sections in the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results.	Complete	Baird & Mules (2021a) AEBR 259
BEN	BEN2018-03	Best practice in marine image analysis for determining taxa, habitat or substrate distribution.	To complete a desktop analysis to determine how high-quality data on taxonomic, habitat or substrate occurrence can be best collected or extracted from deep sea video imagery.	Complete	Bowden et al. (2020) AEBR 239
BYC	DAE2018-01	Catch composition in deepwater fisheries.	1. To estimate the catch composition in specified fisheries. This should include the quantity of non-target fish species caught, and the target and non-target fish species discarded, using data from MPI Observers to the end of the most recent complete fishing year in a format that meets management needs. 2. To compare estimated rates, amounts, and trends of bycatch and discards over time in specified fisheries. 3. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Complete	Finucci et al. (in press)

Theme	Project code	Project title	Specific objectives	Status	Citation/s
NPB	DAE2018-04	Taxonomic identification of benthic samples.	1. To identify deepwater benthic invertebrates in samples taken during research trawls and by observers on fishing vessels targeting deepwater stocks. 2. To update relevant databases recording the catch of invertebrates in research trawls and commercial fishing.	Complete	Schnabel et al. (2021) AEBR 269
NPB	ENV2018-06	Improved distribution information for higher risk non-QMS shark species.	1. To update, using new analytical techniques where applicable, predictive layers of distribution from Leathwick et al. (2006) for the seven demersal shark species listed below. 2. To analyse and discuss any patterns in distribution in regards to depth, sex and stage (where possible) for the seven shark species below. 3. To compare the patterns from specific objectives 1 and 2 above with observer records and patterns observed from overseas in order to better understand the reliability of these predictions.	In progress	Finucci et al. (2021) AEBR 271
PRO	PMM2018-04A	Estimate spatial distribution for at-risk mammals to assess fisheries overlap and risk: New Zealand fur seals.	1. Summarise available information indicative of population structure for New Zealand fur seals, and population trend (i.e., stable, increasing, decreasing) at different colonies. 2. Characterise the spatio-temporal foraging distribution of New Zealand fur seals (including by sex and by season) using available telemetry, sightings, captures, and other data to parameterise spatially comprehensive habitat utilisation models. 3. Apply the Spatially Explicit Fisheries Risk Assessment (SEFRA) method to estimate encounter rate with fisheries, fishery related deaths, and population risk to New Zealand fur seals at biologically relevant scales.	Complete	Large et al. (2019) AEBR 224
PRO	PMM2018-04B	Estimate spatial distributions for at-risk marine mammals to assess potential fisheries overlap and risk: Mainland and Stewart Island New Zealand Sea Lions.	1. Characterise and estimate the spatio-temporal foraging distributions of New Zealand sea lions at mainland and Stewart Island colonies, using satellite telemetry and other available data to parameterise spatially comprehensive habitat utilisation models, including variation by sex and season. 2. Apply the Spatially Explicit Fisheries Risk Assessment (SEFRA) method to estimate overlap with commercial fisheries, fishery related deaths,	In progress	

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			and population risk to mainland sea lion subpopulations. 3. Characterise current mainland and Stewart Island haul-out locations for New Zealand sea lions, and identify likely future haul-out and colony locations based on observed or known habitat preference functions 4. Apply the analyses in Objectives 1 and 2 to potential future haul out locations identified in Objective 3, to estimate fisheries overlap and risk associated with potential future haul out locations as the mainland and Stewart Island sea lion population expands.		
PRO	PMM2018-07	Updated spatially explicit fisheries risk assessment for New Zealand marine mammal populations.	1. Assemble and prepare new spatial distributions for marine mammal species (including on a season and/or sex-specific basis where appropriate), and updated demographic parameterisation (e.g., population size, adult survival), as inputs to an updated Marine Mammal Risk Assessment (MMRA) using the SEFRA method. 2. Produce and fit a multi-species multi-fishery MMRA model using the updated inputs, including model diagnostics to evaluate fishery group definitions, species group definitions, structural assumptions, and spatial goodness of fit. 3. Produce a model interface to facilitate automated/routine update of the model as new fisheries data and updated inputs become available, to allow interrogation of the model to produce user-defined outputs and diagnostics, and to facilitate evaluation of alternate management scenarios.	In progress	
PRO	PMM2018-08	Update SEFRA risk assessment tool – build observer coverage optimization function.	1. Update the SEFRA risk assessment tool 'Risk Atlas' to include a fisheries observer coverage simulation tool and power analysis, to evaluate the consequences of assigning different levels of fisheries observer coverage or digital monitoring for the estimation of non-target species captures and risk under different user-defined scenarios 2. In close collaboration with MPI, run the working model to evaluate particular observer scenarios as requested by MPI fishery managers.	Procurement not initiated	
PRO	PMM2018-09	Desktop estimation of New Zealand sea lion cryptic	1. Summarise existing data to assign priors for various parameters within a simple transition matrix model to estimate cryptic mortality of sea lions in	Complete	Meyer (2019) AEBR 222

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		mortality in trawls using SLEDs.	trawls using SLEDs, including rates of: live escape survival; death by drowning in the net; death by post-escape drowning; and death by mild traumatic brain injury; as well as rates of body retention vs. body non-retention in the net. 2. Construct a simple Bayesian model combining the parameter estimates from Objective 1, and fit to available data (e.g., pre- vs. post-SLED observed capture rates; results of deployed net camera trials) to estimate cryptic mortality.		
PRO	PMM2018-11	Update Auckland Islands New Zealand sea lion demographic population model.	1. Update the existing Auckland Islands New Zealand sea lion population demographics model including new data up to and including from the 2018-19 summer field season. 2. Project future population trajectories and estimate Population Sustainability Threshold values (PSTs) to meet a range of specified population reference outcomes, including for a best estimate base case and sensitivities.	Complete	Roberts (2019) AEBR 223
PRO	PRO2018-01	Protected species population dynamics model and simulations to estimate Population Sustainability Threshold (PST).	1. Construct a generalised population dynamics model, consistent with the SEFRA framework, to estimate likely population recovery or stabilisation outcomes associated with different levels of anthropogenic impact and estimated risk. 2. Run simulations to test the sensitivity of expected population outcomes to variation across the full range of plausible life history parameters for different taxonomic or life history groups (e.g., seabirds, marine mammals, protected fish, harvested fish, benthic organisms). 3. Develop the means to generate comparable risk ratios reflecting non-lethal impacts (i.e., impacts affecting demographic parameters other than adult survival), including across different life history groups. 4. Test the sensitivity of expected population outcomes (i.e., also Risk Ratios and PSTs) to environmental stochasticity at different time scales, and investigate what range of environmental stochasticity is plausible for different life history groups and in different environments.	In progress	

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PRO	PSB2018-01A	Habitat use and spatial distribution of Antipodean albatross.	1. Design a sampling plan to achieve a statistically robust spatial and temporal distribution across age/breeding stages. 2. To collect spatial data on a wider range of ages/breeding stages of Antipodean albatross to allow refinement of the spatial overlap with fishing throughout New Zealand's EEZ and the South Pacific. 3. To produce an updated species distribution suitable for use in spatial risk assessment.	In progress	
PRO	PSB2018-09	Monitoring trial: comparing obersevers v Electronic Monitoring (EM) for seabirds on FMA1 bottom longlinge vessels.	1. To conduct an audit of electronic monitoring footage collected from the bottom longline fishery. 2. To conduct a comparative analysis of the observer and electronic monitoring review datasets of seabird captures.	Complete	McKenzie (2021) AEBR 251
PRO	PSB2018-10	Deepwater net capture analysis.	To complete model-based analyses to examine the influence of various factors that could potentially lead to the trawl net captures of seabirds on larger trawl vessels (>28m).	Complete	Edwards & Dunn (2021) AEBR 266
PRO	PSB2018-13	Multi-threat risk assessment for yellow-eyed penguin.	1. Construct population models for yellow-eyed penguin at the appropriate colony/sub-population scale (as possible given data availability). 2. Map fishery and non-fishery threats to yellow-eyed penguins and estimate the overlap between penguin distributions and threats. 3. Apply the SEFRA method to estimate fisheries impact and risk to yellow-eyed penguins, using the new information from specific objectives 1 and 2, including at a regional sub-population level. This analysis should include estimation and partition of total mortalities attributable to different threats (with uncertainty) at a regional sub-population level. 4. In consultation with government scientists and managers examine alternate candidate spatial management scenarios through both modelling and participation in a multi-threat risk assessment workshop.	On hold until 2021, under de velopment as PRO2022-01	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PSB2018-14	Development and testing of mitigation techniques to reduce penguin captures in setnets.	To develop and test mitigation techniques to reduce penguin captures in setnets, including the assessment of any impact on target species catch.	On hold until 2022	
ZBD	ZBD2018-01	Five-year continuous plankton survey (Phase 3).	1. Conduct annual return transits with the CPR to the Ross Sea during the austral summer in collaboration with Sanford Limited, and contribute the data collected to the SCAR Southern Ocean CPR Survey (SO-CPR). 2. Combine the data collected during phase 1 and phase 2(years 1–10) with the data collected during phase 3 (years 11–15) to: (i) determine zooplankton and phytoplankton trends in the full dataset; and (ii) compare results with datasets available through the SCAR SO-CP Survey. 3. Characterise latitudinal and temporal changes in phytoplankton and zooplankton populations in New Zealand's EEZ and transit to the Ross Sea. 4. Complete couple analysis of phytoplankton and zooplankton with environmental data to investigate causes of variations between the Ross Sea and other Southern Ocean zones. 5. Describe broad similarities and differences observed between NZ, and other southern hemisphere data with patterns or trends observed in the northern hemisphere.	In progress	
ZBD	ZBD2018-02	Climate change, fish distribution meta-analysis.	1. Explore data time series and biological data for evidence of spatial change in living marine resources that may be consistent with climate change or regime shifts. 2. Investigate novel approaches to identify ecologically relevant change by examining species and community relationships between the organisms and their environment. 3. Identify fisheries, communities and locations that are most vulnerable or will remain stable under the response to climate change effects on the ocean.	In progress	

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ZBD	ZBD2018-03	Climate variability, trends and fish population parameters.	1. Determine and review how productivity parameters commonly used in New Zealand stock assessments may be affected by climate change, or long-term environmental variability. 2. Review how these productivity parameters are likely to change, and determine what level of change is likely to be important for stock assessment advice. 3. Identify methods of monitoring to detect important levels of change using existing data sources.	In progress	
ZBD	ZBD2018-05	Environmental variability, regime shifts, and ecosystem function in the Sub-Antarctic.	1. Explore and identify datasets that provide environmental and biological signals about environmental change affecting the state of the marine environment in the Subantarctic. Known datasets of interest include from population monitoring programmes (e.g., New Zealand sea lions; Antipodean albatrosses); trawl survey time series; commercial catch and stock assessment time series of important species (e.g., squid, southern blue whiting) and remotely-sensed environmental and oceanographic data. 2. Explore the various data for synchronous changes and analyse these signals to identify likely environmental drivers or mechanisms affecting ecosystem function and change 3. Construct one or more ecosystem models incorporating change mechanisms identified in Objective 2 4. Design and conduct RV Tangaroa survey to test structural assumptions and inform parameterisation of ecosystem models produced under Objective 3. 5. Identify likely indicators of important environmental change, such as regime shifts, environmental tipping points or persistent trends affecting the distribution and productivity of marine living resources in the area, to inform the design of environmental monitoring.	In progress	
PRO	PRO2017-01A	Research into the demographic parameters for at-risk seabirds as identified	1. To collect information on population size, adult survival, age at first reproduction and key demographic parameters for black petrel to reduce uncertainty or bias in estimates of risk.	Complete	Bell et al. (2020) AEBR 246

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		by the Risk Assessment (black petrels)	2. To collect spatial data to allow refinement of the spatial overlap with fishing, with tracking devices to be deployed on a wider range of ages/breeding stages.		
PRO	PRO2017 -01B	Research into the demographic parameters for at-risk seabirds as identified by the Risk Assessment (Southern Buller's/Snares)	To collect information on population size, adult survival, age at first reproduction and key demographic parameters for southern Buller's on The Snares to reduce uncertainty or bias in estimates of risk.	Complete	Thompson & Sagar (2019)
PRO	PRO2017 -04	Risk Assessment to support the development of revised NPOA seabirds	1. To explore the recommendations made by the expert review of the risk assessment framework due to occur in June 2017, via trialling any suggested changes to the methodology or undertaking sensitivity runs. 2. Following the methods as described in the AEBAR and agreed changes from the expert review, construct a spatially explicit fisheries risk assessment for seabirds.	Withdrawn	
PRO	PRO2017 -05A	Population specific modelling of adult survival of black petrels	To update previous population modelling of black petrels to produce an updated population trend and estimate of adult survival.	Complete	Zhang et al. (2020) AEBR 244
PRO	PRO2017 -05B	Population specific modelling of adult survival of Chatham island albatross	To compile all mark-recapture data collected for the Chatham Island albatross and produce an adult survival estimate for the time period corresponding to that used by the risk assessment.	Contracted, in progress	
PRO	PRO2017 -06	Characterisation of yellow eyed penguin / fishery interactions	To undertake a review and characterisation of all available information to better understand when, where and how yellow-eyed penguins become caught in set nets, and to the extent possible, the frequency of occurrence.	Complete	Mattern (2020) AEBR 243
PRO	PRO2017 -08A	Research into the demographic parameters for at-risk marine mammals as identified by the marine	Characterise population structure and estimate population size for New Zealand common dolphin population(s), with an emphasis on populations that are most exposed to fisheries risk.	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		mammal risk assessment (common dolphins)			
PRO	PRO2017-08C	Research into the demographic parameters for at-risk marine mammals as identified by the marine mammal risk assessment (sea lions)	1. To investigate the likely causes and consequences of changing New Zealand sea lion pup mass and pup survival, using biological and population monitoring data collected at the Auckland islands both before and after the advent of population decline (i.e., pre- and post-2000). 2. To investigate the extent to which indices derived from the analyses in Objective 1 can be used to predict sea lion population trends, by incorporation into the existing sea lion population model and/or as a relevant indicator for future population monitoring efforts.	Complete	Roberts et al. (2021) AEBR 255 Edwards & Roberts (2021) AEBR 256
PRO	PRO2017-10	Analysis of New Zealand sea lion tracking data to estimate overlap with fisheries	1. Characterise the foraging behaviour of Auckland Islands sea lions in a spatially and temporally explicit manner using available satellite telemetry data. 2. Apply spatial overlap methods to inform improved estimation of encounter rate, strike rate, and cryptic mortality rate of Auckland Islands sea lions with commercial fisheries over time, including for fishing effort with and without the use of Sea Lion Exclusion Devices (SLED). 3. Apply estimates from Objective 2 (with uncertainty) to inform spatially explicit estimates of fishery related deaths in association with current fishing effort patterns.	Contracted, in progress	
PRO	PRO2017-12	Hector's and Māui dolphin multi-threat risk assessment to support review of the TMP	1. Construct population models for Māui dolphins, and for Hector's dolphins in those regional sub-populations where data are sufficient. 2. Map potential non-fishery threats to Māui and Hector's dolphins and estimate the overlap between dolphin distributions and both fishery and non-fishery threats. 3. Apply the Spatially Explicit Fisheries Risk Assessment (SEFRA) method to estimate fisheries impact and risk to Māui and Hector's dolphins, using the new information in the objectives above, including at a regional sub-population level. This analysis should include estimation and partition of total mortalities attributable to different threats (with uncertainty) at a regional sub-population level.	Complete	Roberts et al (2019) – AEBR 214

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			4. In consultation with government scientists and managers examine alternate spatial management scenarios through both modelling and participation in a multi-threat risk assessment workshop.		
PRO	PRO2017-15	Use of innovative tag technology to examine foraging patterns of seabirds and association with fishing vessels	To undertake tagging programs alongside other field programs monitoring populations of relevant seabird species using tags that can detect radar strength and potentially depth of dives to examine the relative occurrence of seabird foraging close to fishing vessels.	Contracted, in progress	
PRO	PRO2017-19	Factors affecting capture rate of black petrels and flesh-footed shearwaters	Build a spatially and temporally explicit commercial fisheries risk model estimating capture/ rates of black petrels and flesh-footed shearwaters as a function of multiple spatial, temporal, and vessel- or effort-specific variables potentially affecting capture rates, for fishery groups generating considerable risk to these species. Identify what factors most strongly drive fisheries risk, and evaluate risk reduction options.	Contracted, in progress	
PRO	SEA2017-10	Black petrel Electronic Monitoring; Audit and Analysis	Audit, data analysis, and report of the footage of seabird captures recorded electronically and by on-board observers collected as a result of the collaborative Black Petrel Electronci Monitoring trial.	Complete	McKenzie (2019) – FRR held by FNZ
PRO	SEA2017-08	A synthesis of the population work carried out as part of PRO2006-01	Preparation of a report which summarises work under the PRO2006-01 A to E Mfish contracts including White-chinned petrel and grey petrel on Antipodes Island, Salvin's albatross on Snares, Northern buller's albatross, northern royal albatross and northern giant petrel on Fourty-Fours and Sisters, and Chatham Island albatross on the Pyramid.	Complete	Thompson (2019) – AEBR 228
PRO	SEA2017-03	Shark qualitative analysis for risk assessment rerun	Collection from post November 2014 for each shark species assessed in Ford et al. (2015) of plenary chapters, data files, summaries and maps of reported captures over last 5 complete fishing years up to 30 September 2016, heat maps as generated for NABIS layers, trawl survey information on distribution and trends and papers or summaries of	Complete	Ford et al. (2018) AEBR 201

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			biology, age, growth, fecundity and general productivity.		
BEN	KAI2016-05	Rocky reef impact quantification and monitoring for the Kaikoura earthquake	1. Determine the impact of the Kaikōura earthquake on rocky reef systems, this may also include sub-lethal responses where methodologies to test this exist. 2. Assess long-term monitoring sites to quantify the recovery from the earthquake in order to inform future marine management decisions. 3. Compare impacts across the range of uplift and habitats impacted on the rocky shore. 4. Continue monitoring sediment cover to suggest causation between short-term uplift and potentially longer-term increased sedimentation as a result of the Kaikōura earthquake. 5. Where possible include local participation in the recovery package work and specifically refer to relevant South Island iwi (Te Rūnanga o Kaikōura and Te Tau Ihu), and local community.	Complete	Alestra et al. (2019) AEBR 212 Alestra et al. (2020) FAR2020/01 Alestra et al. (2021) AEBR 253
PRO	KAI2016-6	Sperm Whale impact quantification and monitoring for the Kaikoura earthquake	1. Determine the impact of the Kaikōura earthquake on the local sperm whale population, through surveys that include resighting of known whales, their spatial distribution, behaviour and collection of samples to investigate potential trophic impacts. 2. Process samples to investigate potential trophic impacts of the Kaikōura earthquake on the local sperm whale population.	Complete	Guerra et al. (in prep)
PRO	KAI2016-08	Hutton's Shearwater Survey after the Kaikoura Earthquake	1. To survey Hutton's shearwater populations in colonies around Kaikoura 2. To assess ongoing threats and limits to recovery from recent earthquake impacts of the colonies of Hutton's shearwater around Kaikoura	Complete	Cuthbert (2018) AEBR 209 Scofield et al. (in prep)
PRO	KAI2016-10	Hector's Dolphin Survey after the Kaikoura Earthquake	1. To survey Hector's dolphins along Kaikoura coast in 2017/18 in a manner consistent with previous surveys (Weir and Sagnol 2015) 2. To analyse historical and 2017/18 Hector's dolphin survey information to characterise any impact of the 2016 Kaikoura earthquakes.	Complete	Weir & MacKenzie (2021) AEBR 252

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	PRO2016-01A	Demographic parameters of black petrels	To collect or analyse information on population size, distribution, or key demographic parameters to reduce uncertainty or bias in estimates of risk for selected at-risk seabirds.	Complete	Bell (2018) AEBR 199
PRO	PRO2016-02	Factors affecting capture rate of black petrels and flesh-footed shearwaters	Build a spatially and temporally explicit commercial fisheries risk model estimating capture/kill rates of black petrels and flesh-footed shearwaters as a function of multiple spatial, temporal, and vessel- or effort-specific variables potentially affecting capture rates, for fishery groups generating considerable risk to these species. Identify what factors most strongly drive fisheries risk, and evaluate alternate risk reduction options.	Withdrawn	
PRO	PRO2016-03	Estimation of captures of protected species in New Zealand Fisheries	To summarise fishing effort, observer effort, and observer reported captures in trawl, longline, set net and purse seine fisheries within the New Zealand EEZ, for the 2016/17, 2016/17 and 2017/18 fishing years. To estimate capture rates and total captures of protected species by method, area, and target fishery, and where possible, by species for the 2016/17, 2016/17 and 2017/18 fishing years.	Complete	Abraham & Berkenbusch 2020 AEBR 237, Abraham & Berkenbusch 2019 AEBR 234, Abraham & Berkenbusch 2019 AEBR 233, Abraham & Richard 2019 AEBR 226, Abraham & Richard (2019a) – AEBR 211; Abraham & Richard (2019b) – AEBR 226, Abraham & Berkenbusch (2019), Abraham et al. (2019)
PRO	PRO2016-04	Characterisation and quantification of non-fishing threats on seabirds	To characterise and quantify the non-fishing threats to seabirds.	Withdrawn	
PRO	PRO2016-06	Spatially explicit risk assessment query and simulation tool	Build an interactive user-driven query and simulation tool to enable MPI fisheries managers and government scientists to: i) access, query, display, and disaggregate spatially explicit data layers and outputs of the L2 seabird risk assessment for user-defined combinations of fishery groups, species, and/or areas; ii) define and examine the	Contracted, in progress	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			consequences of alternate assumptions and alternate risk management scenarios, including spatial and temporal effort controls, mitigation uptake, and/or new research to reduce biological and statistical uncertainty -- and iii) for each alternate scenario, estimate seabird captures, fatalities, uncertainty, and corresponding risk; iv) simulate the assignment of fisheries observer coverage within user defined scenarios and estimate the power to accurately estimate seabird captures and risk under different observer coverage levels.		
PRO	PRO2016-09	Abundance and distribution of Hector's and Māui dolphins	1. To develop and refine designs and methods for summer and winter surveys for Hector's dolphins along the SCSI. 2. To estimate the abundance of Hector's dolphins along the SCSI applying an agreed survey and analysis methodology. 3. To estimate the distribution of Hector's dolphins along the SCSI applying an agreed survey and analysis methodology.	Complete	MacKenzie & Clement (2019)
PRO	SEA2016-29	Analysis for the White-capped albatross Aerial Survey	1. Prepare photo montages and count nesting and loafing albatross according to existing methodology (as presented to CSP TWG and AEWG previously) for the aerial surveys undertaken during the summers of 2015/16 and 2016/17 (undertaken under contract to DOC, Deepwater Group and MPI), 2. Based on the counts in objective 1 and previous counts, assess the population trend, taking into account the proportion of loafers identified in the photo montages and by ground counts. 3. Analyse the trends shown by sub-areas for the entire time series of aerial surveys to assess whether selected sub-areas could be monitored and represent the trend of the wider population.	Contracted, in progress	
PRO	SEA2016-19	Spatial methods development to support risk assessment (part II). Estimation of capture and retention efficiency for non-target fish species in commercial trawl fisheries	1. Exploration and testing of alternative methods for species density estimation. 2. Incorporation and propagation of uncertainty in species density estimates. 3. Application of the model with environmental attribute data assigned to individual fishing events.	Contracted	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2016-20	Helicopter based aerial surveys of the Auckland Islands	Helicopter based aerial surveys of the Auckland Islands.	Complete	
PRO	SEA2016-21	Stocktake of Status of development of Mitigation measures applicable to New Zealand commercial fisheries	Stocktake of Status of development of Mitigation measures applicable to New Zealand commercial fisheries.	Complete	
PRO	SEA2016-21	Stocktake of Status of development of Mitigation Measures applicable to New Zealand commercial fisheries	Report summarising (by fishing method) the bycatch mitigation measures and hurdles to uptake in New Zealand, including development, testing and cost.	Complete	
PRO	SEA2016-24	SEA2016-24 Supplemental sea lion population modelling to support an updated Squid Trawl Fishery Operational Plan	Update the existing Auckland Islands sea lion population model with two additional years of pup count data and estimate population trajectories corresponding to alternate management and hypothetical mortality scenarios.	Contracted	
PRO	SEA2016-26	SEA2016-26 SPRFMO bottom fishing impact assessment	1. Use NZ bottom trawl data from the entire SPRFMO area and apply the plotImpact method developed for CCAMLR to the SPRFMO area using NZ bottom trawl data. 2. The overall dataset will be divided into the type of fishing (slope, seamount, or mixed) and the impact summaries and histograms of percent impact will be generated for each fishing type at the four spatial scales. 3. The relationship between the cell size and estimated percentage impact will be evaluated for two selected habitats (slope and seamount). For this exercise, the data for a given habitat will be summarised as a distribution of impact percentages for the cells included, and profiled across cell sizes starting at 100m and with increasing cell size to show the relationship between estimated percent impact and cell size. 4. If time allows, illustrate the potential impact and recovery dynamics of an example VME taxon using an assumed spatial distribution and demographic parameters in one area of interest, and an assumed single move on rule (trigger and distance) to redistribute fishing effort to the remaining areas.	Complete	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2016-30	Hector's and Māui dolphin risk disaggregation tool	1. Expand the custom risk assessment disaggregation and query tool (contract PRO2016-06) to include Hector's and Māui dolphins, incorporating all data inputs for Hector's and Māui dolphins utilised in the Marine Mammal Risk Assessment. Risk estimation will be carried out via a single-species application of the SEFRA method (2017 MPI AEBAR, Chapter 3) for both the setnet and inshore trawl fishery groups. 2. Expand the capability of the risk query tool to include a sub-population definition function that subdivides a population according to user-defined boundaries, and automatically generates separate outputs for each. 3. Expand the capability of the risk query tool to include analysis of hypothetical scenarios using alternate species spatial distribution layers and/or alternate fishing effort distribution layers (provided as user-defined inputs, e.g., as GIS layers) with standardised diagnostic outputs comparing alternate scenarios with the base case scenario. 4. Expand the capability of the risk query tool to allow the fishing effort and observed captures database query and display function to be applied to any protected species in the groomed and linked effort and captures database. This objective is for the visual display outputs only; it does not include risk assessment modelling for species other than Hectors and Māui dolphins.	Contracted	
ZBD	ZBD2016-04	Organic Carbon Recycling in Deepwater	1. To describe the biodiversity of large nektonic crustaceans in subtropical and subantarctic waters. 2. To assess the environmental drivers of nektonic plankton crustacean abundance and diversity over weekly to sub-decadal time scales in subtropical and subantarctic waters	Complete	
PRO	DAE2015-01	Characterisation of seabird capture data	To collate and characterise the seabird capture information from deepwater trawl fisheries to improve understanding of potential risk factors for captures of seabirds, with a focus on net captures.	Complete	
PRO	PRO2015-01	Improving estimates of cryptic mortality for use in seabird risk assessments	1. To develop guidelines for the production of estimates of total seabird captures from observer data, with methods varying based on the level and quality of data.	Complete	Baker et al. (2021) AEBR 268

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			2. To increase the capability of other countries to produce robust estimates of seabird captures.		
PRO	PRO2015-04	Addressing key information gaps for Māui dolphins	1. To collect information on spatial distribution and overlap with fisheries to decrease uncertainty in our understanding and estimates of risk to Māui dolphins.	Contracted, in progress	
PRO	SEA2015-06	Additional aerial survey effort for Hector's dolphins on the West Coast South Island	Following increased Hector's dolphin sightings over the summer survey this project allows for extra effort in the Grey (0-4nm), Hector (4-12nm) and Okarito (4-12nm) strata (Plan A). If time and weather permit, this will also allow for extra effort in the 4-12nm strata off Whanganui, Jackson Bay and Milford.	Completed as part of PRO2013-06	Clement & MacKenzie (2016)
PRO	SEA2015-10	Sea lion prey survey	1. Undertake a demersal trawl survey of the Auckland Islands and Stewart/Snares shelf to determine the spatial and bathymetric distribution and abundance of the main prey species of NZ sea lions in the areas used by benthic and pelagic foraging lactating females. 2. Conduct a potting feasibility study to determine the distribution, abundance and biology of yellow octopus (Enteroctopus zealandicus). 3. Conduct a benthic habitat characterisation based on acoustic swath mapping of the seafloor in the area immediately surrounding demersal trawl stations. 4. Deploy underwater cameras to visually survey seafloor habitat and sea lion prey species at a representative subsample of habitat types, identified from the acoustic swath habitat characterisation. 5. Make oceanographic observations to quantify physical characteristics of sea lion foraging habitat.	Complete	Roberts et al. (2018) AEBR 204
PRO	SEA2015-12	Potential impacts of fisheries restrictions for the NZ sea lion TMP	1. To estimate the likely impact on catch rates and total catches of squid, scampi, and hoki of a range of specified potential fishing restrictions. 2. To estimate the likely impact on sea lion interactions and captures of the specified fishing restrictions.	Complete	Roux & Roberts (2016) FRR

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2015-15	Stewart Island sea lion survey	1. To determine the feasibility of monitoring pup production in January, concurrent with the monitoring of other colonies. 2. While testing the feasibility of using thermal technology in finding sea lion pupping locations. 3. To screen for Klebsiella pneumonia. 4. Determine common causes of death for Stewart Island pups.	Complete	Boren et al. (2016)
PRO	PRO2014-06	Update of level-2 seabird risk assessment	1. To update the level-2 seabird risk assessment using all new information on bird population size, productivity, and distribution, and all relevant fishing effort and observer data for the 2009/10 to 2013/14 fishing years. 2. To identify key drivers of uncertainty and opportunities to reduce uncertainty in the risk ratios for species at high or very high risk. 3. To participate in, and provide data for, a workshop to review the findings relative to other available data and results. 4. To update the level-2 seabird risk assessment using all new information on bird population size, productivity, and distribution, and all relevant fishing effort and observer data for the 2010/11 to 2014/15 fishing years. 5. To identify key drivers of uncertainty and opportunities to reduce uncertainty in the risk ratios for species at high or very high risk.	Complete	Richard et al. (2017)
PRO	PRO2014-03	Research in response to advice from the Māui's dolphin research advisory group	1. To be developed through the MRAG process: agreed project was genetic mark recapture estimation of Māui dolphin abundance with field effort in 2014/15 and 2015/16.	Complete	
PRO	PRO2014-02	Risk assessment modelling for fishing-related mortality of sea lions to underpin the TMP	1. To review existing models of New Zealand sea lions that have been used to estimate key demographic rates and their variability. 2. Based on the results of Objective 1, develop an operating model of the Auckland Island population of New Zealand sea lions suitable for use in management strategy evaluation. 3. To use a management strategy evaluation to assess the risk posed by commercial fishing to New	Complete	Roberts & Doonan (2016) AEBR 166

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			Zealand sea lions, including assessing the likely performance of candidate management approaches against current or agreed performance criteria. 4. To extend the modelling to other populations and risks as information permits.		
PRO	PRO2014-05 Co-funded with DOC POP2015-01	Reducing uncertainty in biological components of the risk assessments for at-risk seabird species	1. Species, population, and information requirements to be determined based on the prioritisation procedures in the NPOA-seabirds and the table of priorities from the outputs of the review workshop.	Complete	Bell et al. 2016
PRO	PRO2014-01	Improving information on the distribution of key protected species	1. To produce an agreed list of seabird and marine mammal species for inclusion and compile all available spatial data for these species. 2. To model and map the distribution of the species identified in objective 1 from available spatial data, reflecting any temporal changes (seasonality or trends). 3. To refine the results of the mapping for priority species by developing and implementing predictive habitat distribution models.	Ongoing	
PRO	SEA2014-12	NZ sea lion stable isotope analysis	1. Locate the ideal NZ sea lion teeth for stable isotope analysis from that will provide the best temporal coverage. 2. Prepare and micro-drill the NZ sea lion teeth within the annual bands to document changes in their foraging and changes in ocean conditions through time using stable isotope analysis. 3. Assess the stable isotope datasets in combination with existing diet studies, prey abundance estimates and climate indices to best examine temporal patterns.	Ongoing	
PRO	SEA2014–15	Sensitivity of the Seabird Risk Assessment to selected scenarios	1. Assess the sensitivity of the Seabird Risk Assessment to assumptions about Buller's albatrosses. 2. Assess the ability of the Seabird Risk Assessment to detect changes in the capture rates.	Complete	Abraham & Richard 2017

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			3. Assess the sensitivity of the risk assessment to live captures.		
PRO	SEA2014-16	Observer coverage power analysis for NPOA	1. Assess the level of observer coverage required to detect a change in the estimated risk of fisheries to New Zealand seabirds, for varying levels of decrease in fishing-related fatalities, for selected seabird species and fisheries.	Complete	
PRO	SEA2014-19	Development and production of smaller hook pods for trial in NZ	To modify the current Hook Pod to New Zealand version without the LED incorporated. This will result in a smaller and more robust Hook Pod that will be equally effective at reducing seabird bycatch in New Zealand's surface longline fisheries.	Complete	
PRO	SEA2014-21	Additional analyses to support the New Zealand sea lion risk assessment	1. The effect of past mortality resulting from key threats for which data are available (such as disease and fishing mortality), or for which plausible estimates are available (such as cryptic mortality), will be explored by fitting the historical demographic model including data on mortality arising from known threats to estimate starting (1960) and current population structure. Threat-derived mortality will then be excluded from the model and re-run from the estimated starting population to predict population structure in the absence of such mortality. 2. Questions were raised about the most appropriate way to deal with animals of unknown pupping status in the model. At present, decision rules are used to determine pupping status from observations (observed suckling, at least 3 sightings with a pup or 3 sightings without a pup) to determine pupping status, with the remaining animals classified as unknown and divided in the proportion of known pupping/non-pupping. Exclusion of animals of unknown status results in increased estimates of pupping rate. Alternative approaches should be considered and the sensitivity of pupping rate to relaxing the decision rules should be explored, such as relaxing the decision rules used to determine pupping status to 2 or 1 observations with or without a pup, or use of other information such as females calling to pups. (Linked to the following item). 3. Similar questions were raised about determining pupping status before an animal that has moved between colonies is used to estimate migration	Complete PRO2014-02.	Roberts & Doonan (2016) AEBR 166

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			(translocation) rates. As an alternative, these requirements could be relaxed to include animals simply observed (but not confirmed to be pupping) at another colony to be included in migration rate estimation. 4. The assumption of a CV of 0.06 for pup census indices, as the only way of specifying a relative weighting between census and tag-recapture data, was questioned. Alternative CVs and weighting approaches should be determined using something like standard deviations of Pearson residuals. 5. Incorporation of time-varying re-sighting probability was noted to improve model fits, indicating that re-sighting probabilities did vary over time. One could explore whether the number of days on which re-sightings were conducted each year are correlated with effort days, in which case effort days could be used to estimate re-sight probabilities for recent years that have not been back-corrected. 6. It was recommended that the effect of incorporation of 'phantom tags' on parameters such as re-sighting probability should be explored. An alternative approach would be to simply multiply the survival rate from tagging to age 1 yr by the directly estimated proportion of pups that die prior to tagging. The latter is, after all, the basis for how many phantom tags are added.		
PRO	SEA2014-23	An assessment of thermal aerial survey techniques on fur seals	1. Undertake field work component. 2. Submit draft report and present to the Aquatic Environment Working Group.	Complete	
PRO	SEA2014-25	Black petrel distribution and fisheries overlap	1. Geographical Information Systems (GIS) data, giving the distribution of black petrel. 2. GIS data giving the overlap of black petrel with bottom longline, surface longline, set net, and trawl fisheries.	Complete	
PRO	SEA2013-06	Black Petrel Distribution Modelling	1. To use the best available information to develop a spatial and seasonal distribution of black petrel, in New Zealand waters.	Complete	Abraham et al. (2015) AEBR 161

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2013 –14	Re-Run of Level-2 Seabird Risk Assessment 2014	1. To provide an update of the Seabird Risk Assessment, including observer and fisheries data to the end of the 2012/13 fishing year.	Complete	Richard & Abraham (2015) AEBR 162
PRO	SEA2013-08	Data preparation for protected species bycatch estimation	1. Groom catch effort, observer, and protected species capture data. 2. Provide web-based interface to allow exploration, display, and reporting on the data.	Completed: preparation for PRO2013-01	
PRO	PRO2013 -01	Protected species capture estimation	1. To estimate capture rates and total captures of seabirds, marine mammals, turtles, and protected fish species by method, area, and target fishery, and where possible, by species for the fishing years 2012/13, 2013/14 and 2014/15. 2. To estimate factors associated with the capture of seabirds and marine mammals. 3. To estimate, where possible, the nature and rate of warp strike incidents and total number of seabirds affected.	Complete	Abraham et al. (2016) AEBR 169 Abraham & Richard (2017) AEBR 184 Thompson et al. (2017) AEBR 192 Abraham & Richard (2018) AEBR 197 Abraham & Berkenbusch (2017) AEBR 188
PRO	PRO2013 -06	Abundance and distribution of WCSI Hector's dolphins	1. To develop and refine designs and methods for summer and winter aerial surveys for Hector's dolphins along the WCSI consistent with the recent ECSI surveys. 2. To estimate the abundance of Hector's dolphins along the WCSI in summer 2013/14 applying an agreed aerial survey methodology. 3. To estimate the distribution of Hector's dolphins along the WCSI in summer 2013/14 applying an agreed aerial survey methodology. 4. To estimate the abundance of Hector's dolphins along the WCSI in winter 2014 applying an agreed aerial survey methodology. 5. To estimate the distribution of Hector's dolphins along the WCSI in winter 2014 applying an agreed aerial survey methodology.	Complete	MacKenzie & Clement (2016) AEBR 168
PRO	PRO2013 -08	Reanalysis of Hector's dolphin line transect aerial survey data	1. To collate sightings and effort data for all Hector's dolphin aerial surveys that applied different approaches to estimating the detection function. 2. To assess the impact of different approaches to estimating the detection function on estimates of	Included in PRO2013-06	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			abundance and distribution and develop correction factors. 3. To reanalyse all relevant survey data to estimate Hector's dolphin abundance and distribution applying the agreed approach to estimating the detection function.		
PRO	PRO2013 -13	Global seabird risk assessment (for New Zealand species)	1. Evaluate relative exposure to commercial fisheries at a global scale for New Zealand seabird populations applying a seasonally-disaggregated spatial overlap approach (i.e., accessing global seabird spatio-temporal distribution data and compiling comprehensive global fisheries effort databases) for different categories of fishing effort. 2. Apply estimates of population PBR (from the updated NZ-EEZ seabird risk assessment (including uncertainty), and species- or guild-specific estimates of seabird Vulnerability (i.e., as estimated in the updated NZ-EEZ seabird risk assessment, modified to the extent possible by data indicative of relative seabird bycatch rates in comparable fishing effort inside vs. outside the New Zealand EEZ (including uncertainty), to estimate global fisheries risk for New Zealand seabird populations. 3. For each New Zealand seabird population estimate what proportion of global fisheries risk is attributable to mortalities occurring inside vs. outside the NZ-EEZ, and what proportion is likely to be unaccounted for in the analysis (e.g., due to incomplete global fisheries data or risk from IUU fishing). 4. For that portion of species risk outside the NZ-EEZ, summarise the source of that risk to the extent possible, for example by RFMO (or other relevant management agency), and by fishery group, geographic area, season, vessel size, and other relevant categories.	Contracted, ongoing	Abraham et al. 2017 (CCBST ERS paper) Francis & Hoyle (2019) AEBR 213
PRO	PRO2013 -17	Repeat quantitative modelling of southern Buller's albatross	1. To update the fully quantitative population model of southern Buller's albatross to assess population trend and key demographic rates for this population. 2. To use the model to predict future trends assuming recent average demographic rates.	Complete	Fu & Sagar (2016) AEBR 165
PRO	PRO2013 -18	Authoritative Sea Lion Capture List	To produce a definitive data set of New Zealand sea lion captures and to reconcile data from the different sources, and resolve any discrepancies.	Complete	Thompson et al. (2015) AEBR 145

Theme	Project code	Project title	Specific objectives	Status	Citation/s
PRO	SEA2013-08	Data preparation for protected species bycatch estimation	1. Groom catch effort, observer, and protected species capture data 2. Provide web-based interface to allow exploration, display, and reporting on the data	Completed: preparation for PRO2013-01	
PRO	No project number	A risk assessment of threats to Māui's dolphins	To evaluate of the risks posed to Māui's dolphin to support the review of the TMP.	Complete	Currey et al. 2012
PRO	PRO2012-02	Assessment of the risk to marine mammal populations from New Zealand commercial fisheries	1. To scope the risk assessment, including producing an agreed list of marine mammal populations (in concert with MAF and DOC). 2. To review the literature, compile the required information and evaluate the appropriate level of risk assessment for the marine mammal populations identified in objective 1. 3. To conduct a risk assessment for the marine mammal populations identified in objective 1 using, where possible, a risk index reflecting the ratio of fisheries-related mortality to the level of potential biological removal. 4. To refine the results of the risk assessment for priority marine mammal populations by incorporating spatially and temporally-explicit abundance, distribution and capture information.	Complete	Berkenbusch et al. (2013) AEBR 119 Abraham et al. (2017) AEBR 189
PRO	PRO2012-07	Cryptic mortality of seabirds in trawl and longline fisheries	1. To review available information from international literature and unpublished sources to characterise and inform estimation of cryptic mortality and live releases for at-risk seabirds in New Zealand trawl and longline fisheries 2. To review the extent to which fisheries observer data informing current estimates of seabird captures may be used to also estimate cryptic mortalities in different fishery groups in the seabird risk assessment, and identify key assumptions and associated uncertainty in the estimation of cryptic mortalities. 3. To identify those species and/or fishery groups for which current uncertainty regarding cryptic mortality contributes most strongly to high-risk scores for at-risk seabird species, and recommend options to improve estimation of cryptic mortality for those species / fishery group combinations.	Complete	Pierre et al. 2015
PRO	PRO2012-10	Level 3 risk assessment for Antipodean albatross	1. Develop an Antipodean albatross population model. 2. Assess the effect of fisheries mortality on	Complete	Edwards et al. (2017) AEBR 180

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			population viability. 3. As information permits, assess the effect of alternative management strategies.		
PRO	ENV2011-01	NPOA-sharks science review	1. To collate and summarise information in support of a review of the National Plan of Action for the Conservation and Management of Sharks (NPOA-sharks). 2. To identify research gaps from objective 1 and suggest cost-effective ways these could be addressed.	Complete	Francis & Lyon (2012) AEBR 102 Francis & Lyon (2013) AEBR 107
PRO	SEA2011-14	CCSBT Seabird risk assessment	To undertake an Ecological Risk Assessment for seabird interactions in surface longline fisheries managed under the Convention for the Conservation of Southern Bluefin Tuna.	Complete	Waugh et al. 2012
PRO	SRP2011-03	Probabilistic modelling of sea lion interactions	1. Estimate the probability that a sea lion suffers mild head trauma following a collision with a SLED grid.	Complete	Abraham (2011) FRR
PRO	SRP2011-04	HSL Modelling	1. Revise Breen-Fu-Gilbert sea lion model.	Complete	Breen et al. 2010
PRO	PRO2010-01	Estimating the nature and extent of incidental captures of seabirds, marine mammals and turtles in New Zealand commercial fisheries	1. To estimate the nature and extent of captures of seabirds, marine mammals and turtles, and the warp strikes of seabirds in New Zealand fisheries for the fishing years 2009/10, 2010/11 and 2011/12.	Complete	Thompson et al. (2012) FRR Thompson et al. (2013) AEBR 105 Thompson et al. (2016) AEBR 167 Richard & Abraham (2015)
PRO	PRO2010-02	Research into key areas of uncertainty or development of mitigation techniques for the revised NPOA-seabirds	1. To provide the information necessary to underpin the revised NPOA-seabirds or develop mitigation techniques to reduce risk identified via the revised NPOA-seabirds.	Complete	Richard & Abraham (2013a) AEBR 108 Richard & Abraham (2013b) AEBR 109 & 109s Richard & Abraham (2013c)

Theme	Project code	Project title	Specific objectives	Status	Citation/s
					Berkenbusch et al. 2013
PRO	No project number	A risk assessment framework for incidental seabird mortality associated with New Zealand fishing in the New Zealand EEZ	To describe the conceptual and methodological framework of this risk assessment approach to guide the completion of similar risk assessments elsewhere.	Complete	Sharp et al. 2011
PRO	SRP2010-03	Fur Seal interactions with a SED excluder device	1. Fur seal interactions with SED excluder device (Dr J Lyle).	Complete	Lyle 2011
PRO	SRP2010-05	Fur seal interaction with an SLED excluder device	1. Using a series of 10-15 impact tests at a maximum collision speed of 5 or 6 ms-1, develop a 'HIC map' for the SLED grid to enable the consequences of collisions with different parts of the grid by sea lions of different head masses to be predicted (scaling values (for eq 3) will include -1/3, -2/3, and -3/4). 2. Using a small number of collision tests, verify that the HIC for a glancing blow can be predicted with sufficient accuracy by resolving vectors. 3. Calculate the maximum possible sensitivity to different boundary conditions using the relative masses of the SLED grid and sea lion heads. 4. Clarify in the final research report that undertaking tests in air (as opposed to underwater) should not affect the results.	Complete	Ponte et al. 2011
PRO	IPA2009-09	Sea Lion bioenergetics modelling	1. To review and collate data on growth, metabolism, diet and reproductive parameters of NZ sea lions or, if data are inexistent, of other sea lion species. 2. To analyse the energy density of various NZ sea lion prey items. 3. To incorporate the data acquired in Objectives 1. and 2. into a bioenergetics model to estimate the energy and food requirements of NZ sea lions.	Complete	Meynier 2010
PRO	IPA2009-16	Preliminary impact assessment of NZ sea lion interaction with SLEDS	1. Preliminary impact assessment of New Zealand sea lion interactions with SLEDs.	Complete	Ponte et al. 2010
PRO	IPA2009-19/20	Level 2 seabird risk assessment rerun	1. To examine the risk of incidental mortality from commercial fishing for 64 seabird species in New Zealand trawl and longline fisheries.	Complete	Richard et al. 2011

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PRO	No project number	External review of NZ sea lion bycatch necropsy data and methods	The primary purposes of this review were to determine whether, in the opinion of a group of independent experts: - the interpretation of necropsy findings and trauma classification system used by Dr Wendi Roe are valid - sea lions recovered from trawl nets have sustained clinically significant trauma - some or all of the sea lions exiting through SLEDs are likely to survive.	Complete	Roe 2010a
PRO	PRO2009-01A	Abundance & distribution of Hector's & Māui's dolphins (5-year project)	1. To estimate the distribution of the South Coast South Island Hector's dolphin sub-population in both winter and summer. 2. The work for this sub-project was subsequently extended to include data collection necessary to estimate abundance.	Complete	Clement & Mattlin (2010) FRR
PRO	PRO2009-01B	Abundance, distribution, and productivity of Hector's (and Māui's) dolphins	1. To estimate the likely precision of abundance estimates from summer aerial surveys for Hector's dolphins along the East Coast South Island (ECSI; from Farewell Spit to Nugget Point) under different levels of sampling intensity and stratification. 2. To estimate the likely precision of abundance estimates and the likely quality of distribution information from winter aerial surveys for Hector's dolphins along the ECSI under different levels of sampling intensity and stratification. 3. To identify and quantify trade-offs between the precision of abundance estimates and the quality of distribution information as well as between overall precision and likely cost (e.g., based on the number of flying hours required). 4. To identify key areas and times for which it would be particularly useful to have information on Hector's dolphin distribution (e.g., where risk may come from overlap with particular fisheries) and quantify trade-offs between the precision of ECSI-wide surveys and collecting such fine-scale information. 5. Assess the extent to which two-phase or adaptive approaches would be useful to improve the surveys' utility for assessing dolphin distribution, particularly the seaward limit.	Complete	MacKenzie et al. (2013) FRR
PRO	PRO2009-01C	Abundance, distribution and productivity of Hector's (and Māui) dolphins	1. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the abundance of Hector's dolphins along the ECSI in	Complete	MacKenzie & Clement (2014) AEBR 123 & 123s

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			summer 2012/13 applying an agreed aerial survey methodology. 2. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the distribution of Hector's dolphins along the ECSI in summer 2012/13 applying an agreed aerial survey methodology. 3. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the abundance of Hector's dolphins along the ECSI in winter 2013 applying an agreed aerial survey methodology. 4. To estimate critical aspects of the biology, abundance and distribution of Hector's and Māui's dolphin populations to assess the effects of fishing-related mortality on these populations including the distribution of Hector's dolphins along the ECSI in winter 2013 applying an agreed aerial survey methodology.		
PRO	PRO2009-04	Development and efficacy of seabird mitigation measures	1. To test the efficacy of a variety of configurations of mitigation techniques at reducing seabird mortality (or appropriate proxies for mortality) in longline fisheries.	Complete	No reports specified as required output
PRO	ENV2008-03	Bycatch of basking sharks in New Zealand fisheries	1. To review the productivity of basking sharks. 2. To describe the nature and extent of fishery-induced mortality of basking sharks in New Zealand waters and recommend methods of reducing the overall catch.	Complete	Francis & Smith (2010) AEBR 49
PRO	PRO2008-01	Risk assessment of protected species bycatch in NZ fisheries	1. To provide an assessment of the risk posed by different fisheries to the viability of New Zealand protected species, and to assign a risk category to all New Zealand fishing operations.	Complete	Waugh et al. 2009
PRO	PRO2008-03	Necropsy of marine mammals captured in New Zealand	1. To necropsy marine mammals captured incidentally to New Zealand fishing operations in the SQU6T fishery during the 2008/09 fishing year to determine life-history characteristics such as sex-reproductive status and the likely cause of mortality- and to determine the species- and sex of captured animals returned for necropsy. 2. To determine, through examination of returned	Complete	Roe (2010) AEBR 47 Roe & Meynier (2012) AEBR 98

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			carcasses, the species, sex, reproductive status, and age-class of sea lions and fur seals captured in the SQU6T New Zealand fishery. 3. To detail any injuries and, where possible, the cause of mortality of sea lions and fur seals returned from New Zealand fisheries, and examine relationships between injuries and body condition, breeding status, and other associated demographic characteristics. 4. To review and collate data from previous NZ sea lion autopsy programmes.		
PRO	SAP2008-14	Sea lion population modelling, additional	1. To assess the likely performance of different bycatch control rules for the SQU6T fishery. 2. To correct and update the Breen-Fu-Gilbert (2008) sea lion model, including assessment of the performance of 200-series and 300-series management control rules. 3. To document the development of the model, including all four objectives of project IPA2006/09 and objective 1 of this project, in a single report suitable for an international review.	Complete	Breen et al. 2010
PRO	Deepwater Group	Necropsy of marine mammals captured in New Zealand fisheries in the 2007–08 fishing year	Necropsy of marine mammals captured in New Zealand fisheries in the 2007–08 fishing year.	Complete	Roe 2009a
PRO	IPA2007-09	Protected species risk assessment	To provide an assessment of the risk posed by different fisheries to the viability of NZ protected species, and to assign a risk category to all NZ fishing operations.	Complete	Waugh et al. 2008
PRO	PRO2007-01	Estimating the nature and extent of incidental captures of seabirds in New Zealand commercial fisheries	1. Estimate capture rates per unit effort and total captures of seabirds for the New Zealand EEZ and in selected fisheries by method, area, target fishery, in relation to mitigation methods in use, and, where possible, by seabird species for the fishing year 2006/07, 2007/08 and 2008/09. 2. Examine the incidence of seabird warp strike in trawl fisheries where these data are available from fisheries observers, and estimate the rate of incidents (birds affected per hour) and total number of seabirds affected by fishery, area and method. Examine the factors (fishery, environmental, seasonal, mitigation, area) that influence the probability of warp-strike occurring.	Complete	Abraham (2010) AEBR 60 Abraham & Thompson (2009) AEBR 33 Abraham & Thompson (2010) FRR Abraham & Thompson (2010) AEBR 80

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					Abraham & Thompson (2011a) AEBR 79 Thompson & Abraham (2009) Abraham et al. (2010) AEBR 45
PRO	PRO2007-02	Estimating the nature and extent of incidental captures of mrine mammals in New Zealand commercial fisheries	1. Describe the nature and extent of marine mammal captures in New Zealand commercial fisheries. 2. Estimate the total numbers, releases, and deaths of selected marine mammals, where possible by species, fishery, and fishing method, caught in commercial fisheries for the fishing years 2006–07, 2007–08, and 2008–09.	Complete	Abraham et al. (2010) AEBR 45 Thompson & Abraham (2009a) AEBR 36 Thompson & Abraham (2009b) AEBR 41 Thompson & Abraham (2010a) AEBR 61 Thompson & Abraham (2010b) AEBR 66 Thompson et al. (2010a) Was AEBR 63, became FRR 2014/02 Thompson et al. (2010b) AEBR 52 Thompson et al. (2010c) AEBR 56
PRO	ENV2006-05	The use of electronic monitoring technology in New Zealand longline fisheries	1. Trial the deployment of electronic monitoring systems in selected longline fisheries, monitoring incidental take of protected species. 2. Evaluate the efficacy of electronic monitoring in allowing enumeration and identification of protected species captures. 3. Recommend options for data management and information transfer arising from the deployment of electronic monitoring in selected fisheries.	Complete	McElderry et al. 2008

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PRO	IPA2006-02	The efficacy of warp strike mitigation devices: trials in the 2006 squid fishery	1. Groom the mitigation trial data and produce a summary of the data. 2. Examine strike rates and capture rates on warps and mitigation devices. 3. Determine the relative efficacy of mitigation devices tested in the trial. 4. Make recommendations regarding future trials. 5. Compare seabird warp strike data for 2005 and 2006. 6. Work with SeaFIC and the mitigation trials TAG to produce analyses and outputs.	Complete	Middleton & Abraham 2007
PRO	IPA2006-09	Modelling interactions between trawl fisheries and New Zealand Sea lion interactions	1. Model the New Zealand sea lion population and explore alternative management procedures for controlling New Zealand sea lion bycatch in the SQU 6T fishery. 2. Collate and review all available sea lion biological data, fisheries data, and sea lion bycatch data relevant to a population model and management strategy evaluation for the Auckland Islands sea lion population. 3. Update and improve the existing Breen and Kim sea lion population model (2003) to incorporate all relevant data and address model uncertainties including but not necessarily limited to those identified by the AEWG. 4. Fit the revised model to all available data and test sensitivity including but not necessarily limited to runs identified by the AEWG. 5. Test a range of management procedures (rules) with the model to determine if they meet agreed management criteria.	Complete	Breen 2008
PRO	IPA2006-13	Identification of Marine Mammals Captured in New Zealand Fisheries	1. To determine, through examination of returned marine mammal carcasses, the species, sex, reproductive status, and age-class of marine mammals returned from New Zealand fisheries. 2. To detail any injuries and, where possible, the cause of mortality of marine mammals returned from New Zealand fisheries, and examine relationships between injuries and body condition, breeding status, and other associated demographic characteristics.	Complete	Roe 2009b
PRO	PRO2006-01	Data collection of demographic, distributional and trophic information on selected seabird species to allow estimation of effects of fishing on population viability	1 To gather demographic, distributional and dietary information on selected seabird species to allow assessment of effects of fishing on population viability.	Complete	Sagar & Thompson 2008 Sagar et al. 2009a, 2009b,

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					2010a, 2010b, 2010c Baker et al. 2008, 2009, 2010 Baker et al. (2020) AEBR 242
PRO	PRO2006-02	Modelling of the effects of fishing on the population viability of selected seabirds	1. Model the effects of fisheries mortalities on population viability compared with other sources of mortality or trophic effects of fishing. 2. Examine the overlap of fishing activity with species distribution at sea for different stages of the breeding and life-cycle and for different sexes, and assess the likely risk to species or populations from fisheries (by target species fisheries, fishing methods, area and season) in the New Zealand EEZ.	Complete	Francis & Bell (2010) AEBR 51 Francis (2012) AEBR 104 Francis et al. (2015) AEBR 152
PRO	PRO2006-04	Estimation of the nature and extent of incidental captures of seabirds in New Zealand commercial fisheries	1. To estimate the nature and extent of captures and warp-strikes of seabirds in New Zealand fisheries for the fishing year 2005/06.	Complete	Baird & Smith 2008
PRO	PRO2006-05	Estimating the nature and extent of marine mammal captures in New Zealand commercial fisheries	1. To estimate and report the total numbers, releases and deaths of marine mammals where possible by species, fishery and fishing method, caught in commercial fisheries for the years 1990 to the end of the fishing year 2005/06. 2. To analyse factors affecting the probability of fur seal captures for the years 1990 to the end of the fishing year 2005/06. 3. To classify fishing areas, seasons and fishing methods into different risk categories in relation to the probability of marine mammal incidental captures for the years from 1990 through to the end of the fishing year 2005/06.	Complete	Mormede et al. 2008 Baird 2008a, 2008b Baird (2011) AEBR 72 Smith & Baird (2009) AEBR 40 Smith & Baird (2011) AEBR 71
PRO	PRO2006–07	Characterise non-commercial fisheries interactions	1. To characterise non-commercial fisheries interactions with seabirds and marine mammals. 2. Characterise non-commercial fisheries risk to seabirds and marine mammals by area and method. Recommend mitigation measures appropriate for uptake in non-commercial fisheries in which seabird or marine mammal captures occur.	Complete	Abraham et al. (2010) AEBR 64 Thompson & Abraham 2009a, 2009b, 2009c, 2010, 2011 Thompson et al. 2010a, 2010b, 2010c

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PRO	ENV2005 -01	Estimation of the nature and extent of incidental captures of seabirds in New Zealand fisheries	1. To estimate the nature and extent of captures of seabirds in selected New Zealand fisheries for the fishing year 2004/05.	Complete	Baird & Smith (2007) Baird & Gilbert (2010) AEBR 50
PRO	ENV2005 -02	Estimation of the nature and extent of marine mammal captures in New Zealand fisheries	To examine the nature and extent of the captures of marine mammals in New Zealand fisheries, for the whole New Zealand EEZ, by Fishery Management Area and fishing season, and by smaller metrics as appropriate for the fishing year 2004/05. 2. Examine alternative methods for estimating sea lion captures and recommend one or more alternative standardised methods for describing and estimating sea lion captures in the SQU 6T fishery.	Complete	Abraham 2008 Baird 2007 Smith & Baird 2007 Baird & Smith 2007
PRO	ENV2005 -04	Identification of marine mammals captured in New Zealand	1. To determine the species, sex, and where possible, age and reproductive status of marine mammals captured in New Zealand fisheries. 2. To necropsy marine mammals captured incidentally in New Zealand fishing operations to determine life-history characteristics and the likely cause of mortality. 3. To determine, through examination of returned marine mammal carcasses, the taxon to species level, sex, and reproductive status, and age-class of marine mammals captured in New Zealand fisheries. 4. To detail the injuries and where possible the cause of mortality of marine mammals returned from New Zealand fisheries, along with their body condition and breeding status, and other associated demographic characteristics. 5. To detail the protocol used for the necropsy of marine mammals, to provide a standardised procedure for autopsy to determine species, age, sex and associated demographic characteristics for fishery-killed specimens.	Complete	Roe 2007
PRO	ENV2005 –06	Estimation of protected species captures in longline fisheries using electronic monitoring	1. To provide estimates of seabird and marine mammal mortalities from longline fisheries in New Zealand using electronic monitoring systems and to recommend deployment and data management options for ongoing use of these systems for estimation of protected species incidental take.	Complete	McElderry et al. 2007

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PRO	ENV2005-09	Data collection to estimate key performance indicators in the Chatham albatross, *Diomedea eremita*.	1. To gather data on key population parameters for Chatham albatross *Diomedea eremita,* to enable population viability to be assessed, and the responses of key parameters to fisheries mortality and fisheries management activities to mitigate fisheries related risk. 2. To undertake field research to collect data on population growth rates, adult survival, inter-breeding season survival, mortality due to predation at the colony, fecundity, and associated parameters for Chatham Albatross following the study design project. 3. To undertake field research to determine the range and extent foraging movements of Chatham albatrosses within New Zealand fishing waters, and examine the nature and extent of any association between Chatham albatrosses and fishing activities.	Complete	No reports specified as required output
PRO	ENV2005-13	Assessment of risk to yellow-eyed penguin *Megadyptes antipodes* from fisheries incidental mortality	1. To review existing data on yellow-eyed penguin *M. antipodes* population performance and fisheries information and provide an analysis of the potential effect of fishing mortality and other factors on population viability. 2. To recommend data collection requirements and protocols for the assessment of the effects of fishing on yellow-eyed penguins.	Complete	Maunder 2007
PRO	ENV2004-02	Estimation of New Zealand sea lion incidental captures in New Zealand Fisheries	1. To estimate the level of New Zealand sea lion (*Phocartos hookeri*) incidental capture in New Zealand fisheries	Complete	Smith & Baird 2007a
PRO	ENV2004-04	Characterisation of seabird captures in New Zealand fisheries	1. Characterisation of seabird captures in New Zealand fisheries.	Complete	MacKenzie & Fletcher 2006
PRO	ENV2004–05	Modelling of impacts of fishing-related mortality on New Zealand seabird populations	1. To examine and identify modelling approaches to analyse seabird demographic impacts that may be occurring as a result of fisheries mortality. 2. To compile databases of available demographic and distributional data on selected seabirds affected by fisheries mortality and New Zealand fisheries and estimate key population parameters and seasonal distribution for each species. 3. To estimate rates of removals related to fishing activities in New Zealand for selected seabird species, where possible by age class and sex. 4. To describe the spatial overlap of seabird distributions at sea, with fisheries where the risk of incidental mortality has been demonstrated to be	Complete	Fletcher et al. 2008

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			moderate to high. 5. To examine the potential for factors other than fisheries removals within the New Zealand zone to influence the population dynamics of the selected study species. 6. To characterise selected seabird populations' abilities to sustain removals related to fishing operations within the New Zealand EEZ, and to recommend, where possible environmental standards for assessing the sustainability of selected fishing operations in relation to impacts on seabird populations.		
PRO	ENV2004-06	Māui's dolphin study	1. To quantify and compare summer and winter distribution of Māui's dolphin.	Complete	Slooten et al. 2005
PRO	IPA2004-14	Seabird warp strike in the southern squid trawl fishery	1. To document seabird warp strike in the southern squid trawl fishery, 2004–05.	Complete	Abraham & Kennedy 2008
PRO	ENV2003-05	Review of the Current Threat Status of Associated or Dependent Species	1. To assess the current threat status of selected associated or dependent species.	Complete	Baird et al. 2010
PRO	No project number	QMA SQU6T New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03	Report on New Zealand sea lion incidental catch and necropsy data for the fishing years 2000–01, 2001–02 and 2002–03	Complete	Mattlin 2004
PRO	MOF2002–03L	Exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery	Report on exploring alternative management procedures for controlling bycatch of Hooker's sea lions in the SQU 6T squid fishery.	Complete	Breen & Kim 2006
PRO	ENV2001-01	Estimation of seabird incidental captures in New Zealand fisheries	1. To estimate the level of seabird incidental capture in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of seabird incidental capture in New Zealand fisheries.	Complete	Baird 2004a, 2004b, 2004c Smith & Baird 2008b
PRO	ENV2001–02	Incidental capture of *Phocarctos hookeri* (New Zealand sea lions) in New Zealand commercial fisheries, 2001–02.	1. To estimate and report the total numbers of captures, releases, and deaths of *Phocarctos hookeri* caught in fishing operations, including separate estimates for SQU 6T and other areas, as appropriate, during the 2001-02 fishing year,	Complete	Baird 2005a, 2005b Baird & Doonan 2005

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			including confidence limits and an investigation of any statistical bias in the estimate.		
PRO	ENV2001 -03	Estimation of *Arctocephalus forsteri* (New Zealand fur seal) incidental captures in New Zealand fisheries	1. To estimate the level of *Arctocephalus forsteri* incidental capture in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of *Arctocephalus forsteri* incidental capture in New Zealand fisheries.	Complete	Smith & Baird 2008a Baird 2005c, 2005d, 2005e
PRO	ENV2000 –01	Protected species bycatch	1. To estimate the total numbers of captures, releases, and deaths of seabirds and marine mammals by species caught in fishing operations during the 1999–2000 fishing year.	Complete	Baird 2003
PRO	ENV2000 -02	Estimation of incidental mortality of New Zealand sea lions in New Zealand fisheries	1. To examine the factors that may influence the level of incidental mortality of New Zealand sea lion in New Zealand fisheries. 2. To recommend appropriate levels of observer coverage for estimation of incidental mortality of New Zealand sea lion in New Zealand fisheries.	Complete	Doonan 2001 Bradford 2002 Smith & Baird 2005a, 2005b
PRO	ENV2000 -03	ENV 2000-A Estimation of seabird and marine mammal incidental capture in New Zealand fisheries	1. To estimate the level of seabird and marine mammal incidental capture in New Zealand fisheries. 2. To determine the factors that influence the level of seabird and marine mammal incidental capture in New Zealand fisheries. 3. To recommend appropriate levels of observer coverage for estimation of seabird and marine mammal incidental capture in New Zealand fisheries.	Complete	Bradford 2002, 2003 Francis et al. 2004
PRO	ENV99-01	Incidental capture of seabirds, marine mammals and sealions in commercial fisheries in New Zealand waters	To estimate the level of seabird and marine mammal incidental captures in New Zealand fisheries.	Complete	Baird 2001 Doonan 2000
PRO	No project number	Factors influencing bycatch of protected species	To determine the factors that influence the level of seabird and marine mammal incidental capture in New Zealand fisheries.	Complete	Baird & Bradford 2000a, 2000b
PRO	ENV98-01	Estimation of non-fish bycatch in commercial fisheries in New Zealand waters, 1997–98	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1999b Baird & Bradford 1999

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PRO	No project number	Annual review of bycatch in southern bluefin and related tuna longline fisheries in the New Zealand 200 n. mile Exclusive Economic Zone	Review bycatch in New Zealand’s southern bluefin and related tuna longline fisheries.	Complete	Baird et al. 1998
PRO	SANF01	Report on the incidental capture of nonfish species during fishing operations in New Zealand waters	To report on incidental captures of non-fish species in New Zealand fisheries.	Complete	Baird 1997
PRO	No project number	Non-fish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1996
PRO	No project number	Analyses of factors which influence seabird bycatch in the Japanese southern bluefin tuna longline fishery in New Zealand waters, 1989-93	1. To assess the influence that 15 monitored environmental and fishery related factors had on seabird bycatch rates, and to gauge the effectiveness of various mitigation measures.	Complete	Duckworth 1995
PRO	No project number	Incidental catch of Hooker's sea lion in the southern trawl fishery for squid, summer 1994	Report on the incidental catch of Hooker’s sea lion in the southern trawl fishery for squid, summer 1994.	Complete	Doonan 1995
PRO	No project number	Nonfish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1995
PRO	No project number	Nonfish Species and Fisheries Interactions	To estimate the level of non-fish bycatch in New Zealand fisheries.	Complete	Baird 1994
PRO	No project number	Incidental catch of fur seals in the west coast South Island hoki trawl fishery, 1989-92	To report on incidental captures of fur seals in the west coast South Island hoki trawl fishery 1989–92.	Complete	Mattlin 1993
PRO	No project number	Incidental catch of non-fish species by setnets in New Zealand waters	To report on incidental captures of non-fish species in New Zealand setnet fisheries.	Complete	Taylor 1992

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PRO	No project number	Seabird bycatch by Southern Fishery longline vessels in New Zealand waters	1. To describe the tuna longline fishery in the New Zealand EEZ and how seabirds are caught by longline vessels. 2. To summarise information available on seabird population trends, and estimate the scale of the incidental capture of seabirds in the larger of two tuna longline fisheries in the EEZ. 3. To describe measures which could reduce the number of seabirds caught by tuna longlines.	Complete	Murray et al. 1992
NPB	DAE2017-01	Bycatch monitoring and quantification in deepwater fisheries (HOK/HAK/LIN)	1. To estimate the catch composition in target trawl fisheries for hoki, hake and ling. This should include the quantity of non-target fish species caught, and the target and non-target fish species discarded, using data from MPI Observers to the end of the most recent complete fishing year in a format that meets management needs. 2. To compare estimated rates, amounts, and trends of bycatch and discards over time in the hoki, hake, and ling trawl fisheries. 3. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Complete	Anderson et al. (2019) AEBR 220
NPB	DAE2016-01	Total catch composition in deepwater fisheries (squid & scampi)	1. To estimate the catch composition in the target fisheries for squid and scampi. This should include the quantity of non-target fish species caught, and the target and non-target fish species discarded, using data from MPI Observers and commercial fishing returns to the end of the most recent complete fishing year in a format that meets management needs. 2. To compare estimated rates, amounts, and trends of bycatch and discards from this study with previous projects on bycatch in the squid and scampi fisheries. 3. To update any relevant sections of the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Complete	Anderson & Edwards (2018) AEBR 199
NPB	SEA2016-19	Spatial methods for development to support risk assessment (part II). Estimation of capture and retention efficiency for non-	1. To implement and test a spatially-explicit two-part delta-gamma statistical model (e.g., Thorson et al. 2015) for estimating species density and capture and retention efficiency in the commercial fishing gear. 2. To estimate relative densities and fishery groups catchability (with uncertainty) in a number of non-	Contracted, in progress	

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		target fish species in commercial trawl fisheries	target fish species in Chatham Rise trawl fisheries. 3. To perform a simulation self-test of the model. 4. To submit the results for publication in the primary literature.		
NPB	ENV2015-01	Updating tools for at-sea fish identification	1. To review the level of information required by the seafood sector and other users of fish identification guides in New Zealand. 2. To evaluate the most beneficial and cost-effective methods of delivery that are practicable and consistent with MPI policy directions. 3. To review, revise and produce the appropriate information tools on fish identification.	Ongoing	
NPB	ENV2015-03	Addressing key information gaps identified by the shark qualitative risk assessment	1. To collect and analyse biological information to improve estimates of risk for inshore and deepwater shark species identified as being at relatively high risk.	Complete	McMillan et al. (2018) AEBR 203
NPB	No Project Code	Qualitative Shark risk assessment	To produce a qualitative risk assessment for all shark species possible within the New Zealand EEZ.	Complete	Ford et al. 2015
NPB	ENV2014-02	NPOA-sharks: age and growth of selected at-risk species	1. To estimate basic biological parameters for high risk, high uncertainty chondrichthyans.	Ongoing	Francis et al. (2018) AEBR 195 Francis et al. (2018) AEBR 196
NPB	No project code	Mitigation options for shark bycatch in longline fisheries	Conduct a literature review and assess the options for improvements in the practice of fisheries to mitigate shark bycatch.	Complete	Howard 2015
NPB	SEA2013-16	Data collation for shark risk assessments	1. To assemble and collate all available information on the distribution and intensity of all fishing methods for the most recent five full fishing years that potentially cause fishing-related mortality of chondrichthyans. 2. To assemble and collate all available information on the distribution, abundance, demographics and productivity of all New Zealand chondrichthyans.	Complete	Francis (2015) AEBR 156

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NPB	ENV2013-01	Development of model-based estimates of fish bycatch	1. To develop a statistical modelling approach to estimate total captures of fish and invertebrates using observer and catch-effort information from selected fisheries. 2. To compare estimates of total captures, confidence limits, and trends for selected species, species groups, and fisheries made using existing ratio-based methods and statistical models. 3. To estimate, within a simulation framework, the potential for bias in ratio-based and model-based methods, the sizes of confidence limits for estimates from the two approaches in comparable situations, and identify the factors associated with good and poor performance.	Complete	Edwards et al. (2015) AEBR 154
NPB	DAE2010-02	Bycatch monitoring & quantification for scampi bottom trawl	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded in the specified fishery, for the fishing years since the last review, using data from Ministry of Fisheries Observers and commercial fishing returns. 2. To compare estimated rates and amounts of bycatch and discards from this study with previous projects on bycatch in the specified fishery. 3. To compare any trends apparent in bycatch rates in the specified fishery with relevant fishery independent trawl surveys. 4. To provide annual estimates of bycatch for nine Tier 1 species fisheries and incorporate into the Aquatic Environment and Biodiversity Report specified in Objective 3 for SQU, SCI, HAK, HOK, JMA, ORH, OEO, LIN, SBW.	Complete	Anderson (2017) AEBR 181 Anderson (2012) AEBR 100 Anderson (2013a) AEBR 112 Anderson (2013b) AEBR 113 Anderson (2014a) AEBR 138 Anderson (2014b) AEBR 139 Ballara (2015) AEBR 158 Ballara & O'Driscoll (2015) AEBR 163
NPB	ENV2009-02	Bycatch and discards in oreo and orange roughy trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for oreos for the fishing years 2002/03 to 2008/09 using data from Scientific Observers and commercial fishing returns. 2. To estimate the quantity of non-target fish species caught, and the target and non-target fish	Complete	Anderson (2011) AEBR 67

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			species discarded, in the trawl fisheries for orange roughy for the fishing years 2004/05 to 2008/09 using data from Scientific Observers and commercial fishing returns.		
NPB	IDG2009-01	Finfish field identification guide	1. To complement the field identification guide under IDG2006/01 with the remaining 120 fish species caught by commercial fishers in New Zealand waters.	Complete	McMillan et al. (2011a) AEBR 78, 2011b, 2011c
NPB	ENV2008-01	Fish and invertebrate bycatch and discards in southern blue whiting fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for southern blue whiting for the fishing years 2002/03 to 2006/07 using data from Scientific Observers and commercial fishing returns.	Complete	Anderson (2009b) AEBR 43
NPB	ENV2008-02	Estimation of non-target fish catch and both target and non-target fish discards in hoki, hake and ling trawl fisheries	Estimates of the catch of non-target fish species, and the discards of target and non-target fish species in the hoki (*Macruronus novaezelandiae*), hake (*Merluccius australis*), and ling (*Genypterus blacodes*) trawl fisheries for the fishing years 2003–04 to 2006–07 using data from Scientific Observers and commercial fishing returns.	Complete	Ballara et al. (2010) AEBR 48
NPB	ENV2008-04	Productivity of deepwater sharks	1. To determine the growth rate, age at maturity, longevity and natural mortality rate of shovelnose dogfish (*Deania calcea*) and leafscale gulper shark (*Centrophorus squamosus*).	Complete	Parker & Francis (2012) AEBR 103
NPB	ENV2007-01 & ENV2007-02	Bycatch and Discards in Squid Trawl Fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for squid for the fishing years 2001/02 to 2005/06 using data from MFish Observers and commercial fishing returns.	Complete	Ballara & Anderson (2009) AEBR 38
NPB	ENV2007-03	Productivity and Trends in Rattail Bycatch Species	1. To estimate growth, longevity, rate of natural mortality, and length at maturity of four key rattail bycatch species in New Zealand trawl fisheries. 2. To examine data from trawl surveys and other data sources for trends in catch rates or indices of relative abundance for species in Objective 1.	Complete	Stevens et al. (2010) AEBR 59
NPB	DEE2006-03	Monitoring the abundance of deepwater sharks	1. To monitor the abundance of deepwater sharks taken by commercial trawl fisheries.	Complete	Blackwell 2010

Theme	Project code	Project title	Specific objectives	Status	Citation/s
NPB	ENV2006-01	Bycatch and discards in ling longline fisheries	To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the longline fisheries for ling for the fishing years 1998/99 to 2005/06 using data from MFish Observers and commercial fishing returns.	Complete	Anderson 2008
NPB	IDG2006-01	Finfish field identification guide	1. To produce a field guide for fish species in New Zealand. 2. To produce a field identification guide for all QMS and other fish species commonly caught in commercial and non-commercial fisheries.	Complete	McMillan et al. (2011a) AEBR 78, 2011b, 2011c
NPB	TUN2006-02	Estimation of non-target fish catches in the tuna longline fishery	1. To estimate the catches, catch rates, and discards of non-target fish in tuna longline fisheries data from the Observer Programme and commercial fishing returns for the 2005/06 fishing year. 2. To describe bycatch trends in tuna longline fisheries using data from this project and the results of previous similar projects.	Complete	Griggs et al. 2008
NPB	ENV2005-17	Estimation of non-target fish catch and both target and non-target fish discards in jack mackerel trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for jack mackerel for the fishing years 2001/2002 to 2004/05 using data from Mfish observers and commercial fishing returns.	Complete	Anderson 2007a
NPB	ENV2005-18	Estimation of non-target fish catch and both target and non-target fish discards in orange roughy trawl fisheries	1. To estimate the quantity of non-target fish species caught, and the target and non-target fish species discarded, in the trawl fisheries for orange roughy for the fishing years 1999/2000 to 2003/04 using data from Scientific Observers and commercial fishing returns.	Complete	Anderson 2009a
NPB	TUN2004-01	Estimation of non-target fish catches in the tuna longline fisheries	1. To estimate the catch rates of non-target fish in the longline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05 fishing years. 2. To estimate the quantities of non-target fish caught in the longline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05 fishing years. 3. To estimate the discards of non-target fish caught in the longline fisheries for tuna using data from the Observer Programme and commercial fishing returns for the 2002/03, 2003/04 and 2004/05	Complete	Griggs et al. 2007

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			fishing years. 4. To describe trends in the non-target fish catches in the tuna longline fisheries using data from this project and the results of previous similar projects.		
NPB	ENV2003 -01	Estimation of non-target catches in the hoki fishery	1. To estimate the catch rates, quantity and discards of non-target fish catches and the discards of target fish catches in trawl fisheries for hoki, using data from the Observer Programme and commercial fishing returns for the 1999/00 to 2002/03 fishing years. 2. To compare and contrast the estimates from the four years of data in Specific Objective 1 above with the 1990/91 through 1998/99 series previously reported.	Complete	Anderson & Smith 2005
NPB	ENV2002 -01	Estimation of non-target fish catch and both target and non-target fish discards for the tuna longline fishery	1. To estimate the catch rates, quantity and discards of non-target fish, particularly oceanic shark species, broadbill swordfish and marlin species, caught in the longline fisheries for tuna, using data from Scientific Observers and commercial fishing returns for the 2000/01 and 2001/02 fishing years.	Complete	Ayers et al. 2004
NPB	ENV2001 -04	Non-target fish catch and discards in selected New Zealand fisheries	To generate estimates of the catch of non-target fish species, and the discards of target and non-target fish species in three important New Zealand trawl fisheries: arrow squid (*Nototodarus sloani* & *N. gouldi*), jack mackerel (*Trachurus declivis*, *T. novaezelandiae*, & *T. symmetricus murphyi*) and scampi (*Metanephrops challengeri*).	Complete	Anderson 2004
NPB	ENV2001 -05	To assess the productivity and relative abundance of deepwater sharks	1. To review the relative abundance, distribution and catch composition of the most commonly caught deepwater shark species: shovelnose dogfish (*Deania catcea*), Baxter's dogfish (*Etmopterus baxten*), Owston's dogfish (*Cenhoscymnus owstoni*), longnosed velvet dogfish (*Centroscymnus crepidater*), leafscale gulper shark (*Cenhophom squamosus*), and the seal shark (*Dalatias ticha*).	Complete	Blackwell & Stevenson 2003
NPB	ENV2001 -07	Reducing bycatch in scampi trawl fisheries	1. Collate and review the international literature on methods of reducing bycatch in crustacean trawl fisheries. 2. Review and analyse the data from New Zealand studies. 3. Develop recommendations on future approaches to reducing bycatch in the New Zealand scampi	Complete	Hartill et al. 2006

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			fishery, including some general thoughts on the experimental design of field trials.		
NPB	PAT2000–01	Review of rattail and skate bycatch, and analysis of rattail standardised CPUE from the Ross Sea toothfish fishery in Subarea 88.1, from 1997-1998 to 2001–02	Report on review of rattail and skate bycatch, and analysis of rattail standardised CPUE from the Ross Sea toothfish fishery in Subarea 88.1, from 1997-1998 to 2001–02.	Complete	Fenaughty et al. 2003 Marriot et al. 2003
NPB	ENV99-02	Estimation of non-target fish catch and both target and non-target fish discards in selected New Zealand fisheries	1. To estimate the quantity of non-target fish species caught in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99 using data from Scientific Observers, commercial fishing returns and from research trawl surveys. 2. To estimate the quantity of target and non-target fish species discarded in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99 using data from Scientific Observers, commercial fishing returns and from research trawl surveys. 3. To explore the effects of various factors on the total catch of non-target fish species and the discards of target and non-target fish species in the trawl fisheries for hoki and orange roughy for the fishing years 1990-91 to 1998-99. 4. To recommend appropriate levels of observer coverage for estimation of non-target fish catch and discards of target and non-target fish species in the hoki and orange roughy fisheries.	Complete	Anderson et al. 2001
NPB	ENV99-05	To identify trends in abundance of associated or dependent species from selected commercial fisheries	To estimate trends in abundance of associated and dependent species, including invertebrates, from deepwater and middle depth fisheries on the Chatham Rise.	Complete	Livingston et al. 2003
NPB	ENV98-02	Pelagic shark bycatch in the New Zealand tuna longline fishery	To determine pelagic shark bycatch in the New Zealand tuna longline fishery.	Complete	Francis et al. 2001
NPB	No project number	Fish bycatch in New Zealand tuna longline fisheries	To report on fish bycatch in New Zealand tuna longline fisheries.	Complete	Francis et al. 1999, 2000

Theme	Project code	Project title	Specific objectives	Status	Citation/s
NPB	ENV97-01	Estimation of nonfish bycatch in New Zealand fisheries	To estimate non-fish bycatch in New Zealand fisheries.	Complete	Doonan 1998; Baird 1999a; Baird et al. 1999
NPB	SCI97-01	Scampi stock assessment for 1998 and an analysis of the fish and invertebrate bycatch of scampi trawlers	1. To summarise catch, effort, observer, and research information for scampi fisheries in QMAs 1,2,3,4 (east and western portions), and 6A in 1998.	Complete	Cryer et al. 1999
BEN	BEN2017-01	Monitoring of deepwater trawl footprint	1. To help MPI groom data, develop summary statistics, for Tier 1 deepwater fisheries and the aggregate of all Tier 1 and Tier 2 deepwater fisheries, of the extent and frequency of fishing by year, by depth zone, by fishable area, and by predicted BOMEC habitat class, and to identify any trends or changes to meet management needs. 2. To update any relevant sections in the Aquatic Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.	Complete	Baird & Mules (2019) AEBR 229
BEN	SEA2016-08	Power Analysis - Benthic Fauna in Spirits Bay	Using previous survey results, conduct a power analysis to estimate the likelihood of a range of survey designs consistent with the monitoring programme from project ENV2005/23 detecting changes in key indicators of the state of the benthic communities in Spirits Bay and Tom Bowling Bay since the last survey.	Complete	
BEN	SEA2016-12	SEA2016-12 GLM Spat composition	Half funding of GLM spat composition study for 90 mile beach (aquaculture unit funding the other half).	Complete	
BEN	DAE2016-05	Monitoring the trawl footprint for deepwater fisheries	1. To estimate the trawl footprint and map the spatial and temporal distribution of trawling on or near the seabed throughout the EEZ between 1989/90 and the most recent completed fishing year. 2. To produce summary statistics, for Tier 1 deepwater fisheries and the aggregate of all Tier 1 and Tier 2 deepwater fisheries, of the extent and frequency of fishing by year, by depth zone, by fishable area, and by predicted BOMEC habitat class, and to identify any trends or changes to meet management needs. 3. To update any relevant sections in the Aquatic	Complete	Baird & Wood (2018) AEBR 193

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			Environment and Biodiversity Annual Review and Environmental and Ecosystem considerations sections of the Fisheries Assessment Plenary documents with new results from this work.		
BEN	BEN2014-01	Risk assessment for benthic habitats, biodiversity, and production	1. To review the design and implementation of management frameworks, including objectives and targets, to manage the effects of mobile bottom fishing methods on vulnerable benthic taxa and habitats. 2. To complete spatially explicit quantitative impact assessments for benthic taxa and/or habitats affected by bottom fisheries, within spatially distinct or overlapping zones within the New Zealand EEZ, consistent with available databases and the outputs of existing projects. 3. To compile and combine impact assessments from Objective 2, to inform a spatially explicit quantitative risk assessment with reference to potential management targets for benthic taxa and/or habitats (from Objective 1) combined across all bottom fisheries in the New Zealand EEZ. 4. To conduct spatially explicit Management Strategy Evaluation to simulate and evaluate the effects of alternate fisheries management scenarios on benthic taxa and/or habitats in the EEZ.	Complete	Mormede et al. (2021) AEBR 274
BEN	BEN2014-02	Monitoring recovery of benthic fauna on the Graveyard complex	1. To repeat the quantitative photographic survey of benthic invertebrate communities on the Graveyard complex. 2. To assess changes in benthic communities since the first survey in 2001.	Ongoing analysis	
BEN	BEN2014-03	Monitoring recovery of benthic fauna in Spirits Bay	1. Using previous survey results, conduct a power analysis to estimate the likelihood of a range of survey designs consistent with the monitoring programme from project ENV2005/23 detecting changes in key indicators of the state of the benthic communities in Spirits Bay and Tom Bowling Bay since the last survey. 2. To survey Spirits Bay and Tom Bowling Bay benthic invertebrate communities in accordance with an agreed design from Objective 1. 3. To assess changes in benthic communities inside and outside of the closed area since 1997.	Complete	Tuck et al. (2019) AEBR 206

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BEN	SEA2014-09	Review of New Zealand's SPRFMO VME protocol	1. To prepare a review of the scientific basis for the 'biodiversity component' of the move-on-rule thresholds comprising the current New Zealand Vulnerable Marine Ecosystem Evidence Process.	Complete	Penney 2014
BEN	BEN2012-02	Spatial overlap of mobile bottom fishing methods and coastal benthic habitats	1. To use existing information and classifications to describe the distribution of benthic habitats throughout New Zealand's coastal zone (0–200 m depth). 2. To rank the vulnerability to fishing disturbance of habitat classes from Objective 1. 3. To describe the spatial pattern of fishing using bottom trawls, Danish seine nets, and shellfish dredges and assess overlap with each of the habitat classes developed in Objective 1.	Complete	Baird et al. 2015
BEN	DEE2010-06	Design a camera / transect study	1. To design and provide indicative costs for a programme to monitor trends in deepwater benthic habitats and communities. 2. To explore the feasibility of using existing trawl and acoustic surveys to capture data relevant to monitoring trends in deepwater benthic habitats and communities.	Complete	Bowden et al. (2015) AEBR 143
BEN	DAE2010-04	Monitoring the trawl footprint for deepwater fisheries	1. To estimate the 2009/10 trawl footprint and map the spatial and temporal distribution of bottom contact trawling throughout the EEZ between 1989/90 and 2009/10. 2. To produce summary statistics, for major deepwater fisheries and the aggregate of all deepwater fisheries, of the spatial extent and frequency of fishing by year, by depth zone, by fishable area, and by habitat class, and to identify any trends or changes.	Ongoing analysis	Black et al. (2013) AEBR 110 Black & Tilney (2015) AEBR 142
BEN	Internally funded 1	SPRFMO	1. To develop detection criteria for measuring trawl impacts on vulnerable marine ecosystems in high sea fisheries of the South Pacific Ocean.	Complete	Parker and Bowden 2010
BEN	Internally funded 2	SPRFMO	1. To document protection measures implemented by New Zealand for vulnerable marine ecosystems in the South Pacific Ocean.	Complete	Penney et al. 2009
BEN	Internally funded 3	CCAMLR	1. An Impact Assessment Framework for Bottom Fishing Methods in the CCAMLR Convention Area.	Complete	Sharp et al. 2009

Theme	Project code	Project title	Specific objectives	Status	Citation/s
BEN	Internally funded 4	SPRFMO	1. To develop a bottom Fishery Impact Assessment: Bottom Fishing Activities by New Zealand Vessels Fishing in the High Seas in the SPRFMO Area during 2008 and 2009.	Complete	Ministry of Fisheries 2008
BEN	BEN2009-02	Monitoring recovery of benthic communities in Spirits Bay	1. To survey Spirits Bay and Tom Bowling Bay benthic invertebrate communities according to the monitoring programme designed in ENV2005/23. 2. To assess changes in benthic communities inside and outside the closed area since 1997.	Complete	Tuck & Hewitt (2013) AEBR 111
BEN	IFA2008-04	Guide for the rapid identification of material in the process of managing Vulnerable Marine Ecosystems	To produce a guide for the rapid identification of material in the process of managing Vulnerable Marine Ecosystems.	Complete	Tracey et al. 2008
BEN	BEN2007-01	Assessing the effects of fishing on soft sediment habitat, fauna, and processes	1. To design and test sampling and analytical strategies for broad-scale assessments of habitat and faunal spatial structure and variation across a variety of seafloor habitats. 2. To design and carry out experiments to assess the effects of bottom trawling and dredging on benthic communities and ecological processes important to the sustainability of fishing at scales of relevance to fishery managers.	Complete	Tuck et al. 2016
BEN	IFA2007-02	Development of a Draft New Zealand High-Seas Bottom Trawling Benthic Assessment Standard	1. To generate data summaries and maps of New Zealand's recent historic high-seas bottom trawling catch and effort in the proposed convention area of the South Pacific Regional Fisheries Management Organization (SPRFMO). 2. To map vulnerable marine ecosystems (VMEs) in the SPRFMO area. 3. To develop a draft standard for assessment of benthic impacts of high-seas bottom trawling on VMEs in the proposed SPRFMO convention area.	Complete	Parker (2008) FRR
BEN	BEN2006-01	Mapping the spatial and temporal extent of fishing in the EEZ	1. To update maps and develop GIS layers of fishing effort from project ENV2000/05 to show the spatial and temporal distribution of mobile bottom fishing throughout the EEZ between 1989/90 and 2004/05. 2. To produce summary statistics of major fisheries and the aggregate of all bottom impacting fisheries in terms of the extent and frequency of fishing by year, by depth zone, by fishable area, and, to the extent possible, by habitat type.	Complete	Baird et al. 2009 Baird et al. (2011) AEBR 73 Baird & Wood 2010

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			3. To identify and document any major trends or changes in fishing effort or fishing behaviour. 4. To identify, discuss the implications of, and make recommendations on data quality and other problems with current reporting systems that complicate characterisation and quantification of bottom fishing effort. 5. To integrate information on the distribution, frequency, and magnitude of fishing disturbance with habitat characteristics throughout the EEZ, using information stored in national databases, expert opinion, and the MEC.		Baird & Wood (2012) AEBR 89 Leathwick et al. 2010 Leathwick et al. (2012) AEBR 88
BEN	ENV2005 -15	Information for managing the Effects of Fishing on Physical Features of the Deep-sea Environment	1. To provide an updated database that identifies all known seamounts in the 'New Zealand region', encompassing the area from 24°00' – 57°30'S, 157°00'E – 167°00'W. The database will catalogue relevant data (e.g., physical, biological, location, fishing effort) for individual seamounts. 2. To identify indicators and measures suitable for the assessment of risk pertaining to the effects of fishing disturbance on the benthic biota of seamounts, and review suitable ecological risk assessment methods, that can be derived or utilise information contained within the seamount database.	Complete	Rowden et al. 2008 Clark et al. 2010b
BEN	ENV2005 -16	Investigate the Effects of Fishing on Physical Features of the Deep-sea Environment	1. To monitor changes in fauna and habitats over time on selected UTFs in the Chatham Rise area that have a range of fishing histories. 2. To continue development of the risk assessment model to predict the effects of fishing, and provide options for the management of UTF ecosystems.	Complete	Clark et al. (2010a) AEBR 46, 2010b, 2010c, Clark et al. (2011) AEBR 74
BEN	ENV2005 -20	Benthic invertebrate sampling and species identification in trawl fisheries	1. To produce identification guides for benthic invertebrate species encountered in the catches of commercial and research trawlers.	Complete	Tracey et al. 2007; Williams et al. 2010; Clark et al. 2009
BEN	ENV2005 -23	Monitoring recovery of the benthic community between North Cape and Cape Reinga	1. To design a monitoring programme that will provide the following quantitative estimates: i) Estimates of the nature and extent of past fishing impacts on the benthic community between North Cape and Cape Reinga; ii) Estimates of change over time in areas previously fished but subsequently closed to fishing. Estimated parameters will include indices representing biodiversity, community composition, and biogenic structure; iii) Estimates of change over time in areas	Complete	Tuck et al. (2010) AEBR 53

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			environmentally comparable to those assessed in (ii), above, but subject to ongoing fishing impacts; and iv) Estimates of change over time in areas comparable to those above, but not impacted by fishing (if any such areas can be found).		
BEN	ZBD2005 -04	Information on benthic impacts in support of the Foveaux Strait Oyster Fishery Plan	1. To assess the distribution, vulnerability to disturbance, and ecological importance of habitats in Foveaux Strait, and describe the spatial distribution of the Foveaux Strait oyster fishery relative to those habitats. 2. To assemble and collate existing information on the Foveaux Strait system between the Solander Islands and Ruapuke Island or other area to be agreed with MFish. 3. To map, using best available information, substrate type, bathymetry, wave energy, and tidal flow in this area. 4. To assess the extent to which these data can be used to define useful functional categories that might serve as habitat classes. 5. To rank the vulnerability to fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times. 6. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3, including an assessment of the relative importance of each to overall ecosystem function and productivity. 7. To describe the spatial pattern and intensity of dredge fishing for Foveaux Strait oysters over the past 10 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3. 8. To carry out a qualitative video survey of benthic habitats in Foveaux Strait, both within the established commercial oyster fishery area and areas outside the fishery area but within OYU 5.	Complete	Michael et al. 2006
BEN	ZBD2005 -15	Information on benthic impacts in support of the Coromandel Scallops Fishery Plan	1. To assemble and collate existing information on the coromandel Scallop Fishery between cape Rodney and Town Point or other, wider area to be agreed with Mfish. 2. To map, using best available information, substrate type, bathymetry, wave energy, and tidal flow in this area. 3. To assess the extent to which data can be used to define useful functional categories that might serves as habitat classes. 4. To rank the vulnerability of fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times.	Complete	Tuck et al. 2006a, 2006b

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			5. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3, including an assessment of the relative importance of each to overall ecosystem function and productivity. 6. To describe the spatial pattern and intensity of dredge and trawl fishing within the Coromandel scallop fishery over the past 15 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3.		
BEN	ZBD2005 -16	Information on benthic impacts in support of the Southern Blue Whiting Fishery Plan	1. To assemble and collate existing information on the Southern Blue Whiting fishery in SBW6A, SBW6B, SBW6I, and SBW6R or other wider area to be agreed with MFish 2. To map, using best available information, substratum type, bathymetry, wave energy, tides, and ocean currents in these areas 3. To assess the extent to which these data can be used to define useful functional categories that might serve as habitat categories. 4. To rank the vulnerability to fishing disturbance of habitat classes developed in Objective 3 using approximate regeneration times. 5. To describe the functional role and ecosystem services provided by each habitat class developed in Objective 3, including an assessment of the relative importance of each to overall ecosystem function and productivity. 6. To describe the spatial pattern and intensity of trawl fishing within the Southern Blue Whiting fishery over the past 10 fishing years and relate this to natural disturbance regimes and habitat classes developed in Objective 3.	Complete	Cole et al. 2007
BEN	ENV2003 -03	Determining the spatial extent, nature and effect of mobile bottom fishing methods	1. To determine the spatial extent, nature and time between disturbances of mobile bottom fishing methods in the Chatham Rise trawl fisheries.	Complete	Baird et al. 2006
BEN	ENV2002 -04	Benthic invertebrate sampling and specific identification in trawl fisheries	1. To quantify and map the benthic invertebrate species incidental catch in commercial and research trawling throughout the New Zealand EEZ.	Complete	Tracey et al. 2005
BEN	ENV2001 -09	The effects of mobile bottom fishing gear on bentho-pelagic coupling	To describe any effects of fishing that might modify bentho-pelagic coupling (a complex, interlinked suite of processes transferring energy, oxygen, carbon, and nutrients between pelagic and benthic systems), to consider the scale of such possible	Complete	Cryer et al. 2004

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			effects, and to put the summary in a New Zealand context.		
BEN	ENV2001-15	The effects of bottom impacting trawling on seamounts	1. To design a programme in New Zealand waters previously trawled and now closed to trawling to monitor the rate of regeneration of benthic communities on seamounts.	Complete	Clark & O'Driscoll 2003 Clark & Rowden 2009
BEN	OYS2001-01	Foveaux Strait oyster stock assessment	1. To carry out a survey and determine the distribution and absolute abundance of pre-recruit and recruited oysters in both non-commercial and commercial areas of Foveaux Strait. The target coefficient of variation (c.v.) of the estimate of absolute recruited abundance is 20%. 2. To estimate the sustainable yield for the areas of the commercial oyster fishery in Foveaux Strait for the year 2002 oyster season. 3. To identify and count benthic macro-biota collected during the dredge survey.	Complete	Rowden et al. 2007
BEN	ENV2000-05	Spatial extent, nature and impact of mobile bottom fishing methods in the New Zealand EEZ	1. To determine the spatial extent, nature and impact of mobile bottom fishing methods within the New Zealand EEZ.	Complete	Cryer and Hartill 2002 Baird et al. 2002
BEN	ENV2000-06	Review of technologies and practices to reduce bottom trawl bycatch and seafloor disturbance in New Zealand	To review technologies and practices to reduce bottom trawl bycatch and seafloor disturbance in New Zealand.	Complete	Booth et al. 2002 Beentjes & Baird 2004
BEN	ENV98-05	The effects of fishing on the benthic community structure between North Cape and Cape Reinga	1. To determine the effects of fishing on the benthic community structure between North Cape and Cape Reinga.	Complete	Cryer et al. 2000
ECO	ANT2017-03	Antarctic Research - Ross Sea region MPA	1. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee to advise the Commission on: (i) the degree to which the specific objectives of the MPA are being achieved. 2. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee to advise the Commission on the degree to which the MPA objectives are still relevant in different areas of the MPA. 3. To provide advice and scientific knowledge to MPI that would allow the CCAMLR Scientific Committee	Approved	

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			to advise the Commission on what management actions may be required to improve the achievement of the objectives for this MPA.		
ECO	MDC2015-01	MDC Benthic coring	To support benthic coring in the Marlborough Sounds by the Marlborough District Council (MDC). The results of this will provide historical information to support environmental restoration and reporting goals. It will also support potential restoration using empty mussel shell disposal.	Complete	
ECO	ENV2014-09	Spatial decision support tools for multi-use and cumulative effects	To provide a customised GIS decision support tool to help assess the cumulative effects of fishing.	Ongoing	
ECO	SEA2013-01	Provision of identification guides (sea pens and black corals)	To produce identification guides for sea pens and black corals electronically as AEBR (including MPI review).	Complete	Tracey et al. (2014) FRR 2014/6 Williams et al. (2014) AEBR 132 Opresko et al. (2014) AEBR 131 Clark et al. (2015) AEBR 149
ECO	ENV2012-01	A literature review of nitrogen levels and adverse ecological effects in embayments in temperate regions.	1. To complete a literature review of nitrogen levels and adverse ecological impacts from temperate embayments in order to assist aquaculture consenting authorities in determining at what concentration of nitrogen adverse effects may be expected.	Complete	Hartstein & Oldman (2015) AEBR 159
ECO	SEA2012-17	NPOA Sharks extension work	NPOA Sharks extension work.	Complete	Clarke et al. (2013) FAR 2013/65
ECO	ZBD2012-02	Tier 1 statistic: Ocean	1. To identify candidate oceanographic variables for potential development as part of the proposed Tier 1 Statistic, Atmospheric and Ocean Climate Change.	Complete	Pinkerton et al. (2015a) AEBR 151
ECO	DAE2010-01	Taxonomic identification of benthic specimens	1. To identify benthic invertebrates in samples taken during research trawls and by Observers on fishing vessels. 2. To update relevant databases recording the catch	Complete	Mills et al. (2013) FRR 2013/17

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			of invertebrates in research trawls and commercial fishing.		Tracey et al. (2014) FRR 2014/17
ECO	DEE2010-04	Development of a methodology for Environmental Risk Assessments for deepwater fisheries	1. To review approaches to Ecological Risk Assessments (ERA) and methods available for deepwater fisheries both QMS and non-QMS. 2. To develop and recommend a generic, cost effective, method for ERA in deepwater fisheries by using or modifying methods identified in Objective 1.	Complete	Clark et al. submitted; Mormede & Dunn (2013) AEBR 106
ECO	DEE2010-05	Development of a suite of environmental indicators for deepwater fisheries	1. To review the literature and hold a workshop to recommend a suite of ecosystem and environmental indicators that will contribute to assessing the performance of deepwater fisheries within an environmental context. 2. To examine available data and design a data collection programme to enable future calculation of the indicators identified in Specific Objective 1.	Complete	Tuck et al. (2014) AEBR 127
ECO	ENV2010-03	Habitats of particular significance for inshore finfish fisheries management	1. To review the literature to determine the most important juvenile or reproductive (spawning, pupping or egg-laying) areas for inshore finfish target species. 2. To use a gap analysis to prioritise areas for future research concerning the important juvenile or reproductive (spawning, pupping or egg-laying) areas for target inshore finfish fisheries.	Complete	Morrison et al. (2014b) AEBR 125
ECO	ENV2010-05A&B and SEA2010-15	Habitats of particular significance for fisheries management: shark nursery areas	1. Identify, from the literature, important nursery grounds for rig in estuaries around mainland New Zealand. 2. Design and carry out a survey of selected estuaries and harbours around New Zealand to quantify the relative importance of nursery ground areas. 3. Identify threats to these nursery ground areas and recommend mitigation measures.	Complete	Francis et al. (2012) AEBR 95 Jones et al. (2015) AEBR 150
ECO	ZBD2010-42	Development of a National Marine Environment Monitoring Programme	1. To design a Marine Evnvironment Monitoring Programme (MEMP) to track the physical, chemical and biological changes taking place across New Zealand's marine environment over the long term. 2. To prepare an online inventory (metadatabase) of repeated (time series) biological and abiotic marine observations/datasets in New Zealand. 3. To review, evaluate fitness for purpose, and identify gaps in the utility and interoperability of	Complete	Hewitt et al. (2014) AEBR 141

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			these datasets for inclusion in MEMP from both science and policy perspectives. 4. To design a MEMP that includes relevant existing data collection and proposed new time series.		
ECO	ENV2009-04	Trends in relative mesopelagic biomass using time series of acoustic backscatter data from trawl surveys	1. To evaluate relative changes in abundance of mesopelagic fish and other biological components from acoustic records collected during Chatham Rise and Sub-Antarctic trawl surveys. 2. To explore links between trends in mesopelagic biomass and climate variables and variations, and condition indices of commercial species in the Chatham Rise and Sub-Antarctic areas.	Complete	O'Driscoll et al. (2011) AEBR 76
ECO	ENV2009-07	Habitats of particular significance for fisheries management: Kaipara harbour	1. Collate and review information on the role and spatial distribution of habitats in the Kaipara Harbour that support fisheries production. 2. Assess historical, current, and potential anthropogenic threats to these habitats that could affect fisheries values, including fishing and land-based threats. 3. Design and implement cost-effective habitat mapping and monitoring surveys of habitats of particular significance for fisheries management in the Kaipara Harbour.	Complete	Morrison et al. (2014d) AEBR 129
ECO	GMU2009-01	Spatial Mixing of GMU1 using Otolith Microchemistry	1. To determine the level of spatial mixing and connectivity of grey mullet (*Mugil cephalus*) populations using otolith microchemistry. 2. To collect and analyse the chemical composition of grey mullet otoliths. 3. To analyse the otoliths collected under Objective 1 to determine if the samples can be spatially separated.	Complete	Morrison et al. (2016) FAR 2016/15
ECO	IPA2009-11	Trophic studies publication of review	1. To publish the comprehensive review of New Zealand-wide trophic studies completed in 2000 that was prepared by NIWA.	Complete	Stevens et al. (2011) AEBR 85
ECO	FLA2009-01	Assess the feasibility of using juvenile netting surveys to predict adult yellow-belly & sand flounder	1. Assess the feasibility of using juvenile netting surveys to predict adult yellow-belly and sand founder abundance in the Manukau Harbour and Firth of Thames (this also examined correlations between juvenile catch and environmental factors).	Complete	McKenzie et al. (2013) FAR 2013/10

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ECO	AQE2008-02	Review of ecological effects of farming shellfish and other species	1. To collate and review information on the ecological effects of farming mussels (*Perna canaliculus*), including offshore mussel farming and spat catching, in the New Zealand marine environment. 2. To collate and review information on the ecological effects of farming oysters in the New Zealand marine environment. 3. To collate and review information on the ecological effects of farming species other than mussels (*Perna canaliculus*), oysters, and finfish, in the New Zealand marine environment.	Complete	Keeley et al. 2009
ECO	IFA2008-08	Inputs to the Ross Sea bioregionalisation	1. To produce one or more benthic invertebrate classifications of the Ross Sea region. 2. To use fishery catch data to examine spatial distributions of major demersal fish species. 3. To prepare other biological or environmental spatial data layers for use in the Ross Sea workshop.	Complete	Pinkerton et al. 2009a
ECO	TOH2007-03	Toheroa Abundance	1. To investigate variations in the abundance of toheroa. 2. To investigate sources of mortality of toheroa and factors affecting the recruitment of toheroa	Complete	Williams et al. (2013) AEBR 114
ECO	BEN2007-05	Risk assessment framework for assessing fishing & other anthropogenic effects on coastal fisheries	1. To collate existing information on the distribution, intensity, and frequency of anthropogenic disturbances in the coastal zone that could be used in a risk assessment model to estimate their likely aggregate effect on ecosystem function across habitats and over different scales of ecosystem functioning and biological organisation. 2. To develop a risk assessment framework in conjunction with a variety of stakeholders and environmental scientists.	Complete	MacDiarmid et al. (2012) AEBR 93
ECO	ENH2007-01	Stock enhancement of blackfoot paua	1. To assess the survival rate of enhanced paua from introduction into the wild through to harvest. 2. To assess the genetic diversity of hatchery spawned juvenile paua bred for enhancement purposes. 3. To assess interactions between introduced and wild paua populations and to recommend research and monitoring to quantify those impacts that are potentially adverse.	Complete	McCowan 2013

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ECO	ENV2007-04	Climate and Oceanographic Trends Relevant to New Zealand Fisheries	1. To summarise, for fisheries managers, climatic and oceanographic fluctuations and cycles that affect productivity, fish distribution and fish abundance in New Zealand.	Complete	Hurst et al. (2012) AEBR 90
ECO	ENV2007-06	Trophic Relationships of Commercial Middle Depth Species on the Chatham Rise	1. To quantify the inter-annual variability in the diets of hoki, hake and ling on the Chatham Rise 1992–2007. 2. To quantify seasonal dietary cycles for hoki, hake and ling that have been collected from the commercial fleet throughout the year.	Complete	Horn & Dunn (2010) AEBR 54
ECO	HAB2007-01	Biogenic habitats as areas of particular significance for fisheries management	1. To collate and review available information on the location, value, functioning, threats to, and past and current status of biogenic habitats that may be important for fisheries production in the New Zealand marine environment. 2. To identify information gaps, in the New Zealand context, and recommend measures to address those important to an ecosystem approach to fisheries management.	Complete	Morrison et al. (2014a) AEBR 130
ECO	IPA2007-07	Land Based Effects on Costal Fisheries	1. To review and collate scientific knowledge and research on the impacts of land-based activities on coastal fisheries and biodiversity.	Complete	Morrison et al. (2009) AEBR 37
ECO	ENV2006-04	Ecosystem indicators for New Zealand fisheries	1. To carry out a literature review of potential fish-based ecosystem indicators and identify a suite of indicators to be tested in Objective 2. 2. To test a suite of fish-based ecosystem indicators (identified by Objective 1) on existing trawl survey time series in New Zealand. The utility of these indicators for monitoring the effects of fishing in New Zealand should also be evaluated.	Complete	Tuck et al. 2009
ECO	GBD2006-01	DNA database for commercial marine fish and invertebrates	1. To collect DNA sequences for vouchered specimens of commercially important marine fishes and submit the DNA data to the international Barcode of Life Database (BOLD). 2. To collect DNA sequences for vouchered specimens of commercially important marine invertebrates and submit the DNA data to the international Barcode of Life Database (BOLD). Note: The funding was limited to $60 000 for this Objective. Therefore, MFish agreed to omit the invertebrate species (Objective 2) from this project and reduce the number of fish species sequenced from 100 to 80 (up to 5 specimens per species).	Complete	No reports specified as required output

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			During the course of the project MFish staff asked NIWA to identify smoked eel product, suspect shark fillets, and possible paua slime with DNA markers, consequently the project was modified to accommodate these requests.		
ECO	IPA2006-08	Review of the Ecological Effects of Marine Finfish aquaculture: Final Report	1. Summarise and review existing information on ecological effects of finfish farming on the marine environment in New Zealand and overseas.	Complete	Forrest et al. 2007
ECO	SAP2006-06	West Coast South Island review	1. To publish a review document summarising oceanic and environmental research information particularly relevant to hoki, but also other fisheries, that spawn off Westland in winter. 2. Update the draft chapters prepared in 2004 by oceanographers, modellers, and scientists towards the overall objective. 3. Incorporate a section on other west coast spawning fisheries.	Complete	Bradford-Grieve & Livingston (2011) AEBR 84
ECO	ENV2005-08	Experimental design of a programme of indicators	1. To assess the utility/feasibility of using demographic information to assess the effects of fishing on seabird populations. 2. To identify population indicators and to provide sampling protocols and experimental design for selected high to medium priority seabird populations. 3. To recommend experimental protocols for sampling of selected seabird populations in New Zealand influenced by fisheries mortality, employing robust design methodology and including recommendations for inclusions of data into Ministry of Fisheries databases.	Complete	MacKenzie & Fletcher (2010) FRR
ECO	IPA2005-02 and MOF2003-03A	A guide to common offshore crabs in New Zealand Waters	1. Develop a guide to common offshore crabs in New Zealand waters	Complete	Naylor et al. 2005
ECO	SAM2005-02	Effects of climate on commercial fish abundance	To examine the possible effects of climate on fishery yields and abundance indices for commercial fisheries around New Zealand.	Complete	Dunn et al. 2009a
ECO	HOK2004-01	Hoki Population modelling and stock assessment	To investigate the prediction of year class strength from environmental variables.	Complete	Francis et al. 2005

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ECO	AQE2003-01	Effects of aquaculture and enhancement stock sources on wild fisheries resources and the marine environment.	1. To identify, discuss the effects and qualitatively assess the risks of aquaculture and enhancement stocks improved by hatchery technology on New Zealand's wild fisheries resources and the marine environment. 2. To identify, discuss the effects and qualitatively assess the risks associated with the translocation of aquaculture and enhancement stocks on New Zealand's wild fisheries resources and the marine environment. 3. To make recommendations on priority issues, risks, or research to be undertaken, as a result of information discussed and evaluated in Objectives 1–2.	Complete	Speed 2005
ECO	EEL2003-01	Non-fishing mortality of freshwater eels	1. To undertake a feasibility study on establishing an estimate of the mortality of eels caused by hydroelectric turbines and other point sources of mortality caused by human activity.	Complete	Beentjes et al. 2005
ECO	MOF2003-01	The implications of marine reserves for fisheries resources and management in the New Zealand context	Investigations of the implications of marine resources for fisheries resources and management in the New Zealand context.	Complete	Speed et al. 2006
ECO	ENV2002–03	Beach cast seaweed review	1. To collate existing information on the role of beach-cast seaweed in coastal ecosystems to assess the nature and extent of the impacts that the removal of beach cast seaweed may have on the marine environment. 2. On the basis of the review in Specific Objective 1 above, to identify key research gaps related to any marine environment impacts that the removal of beach cast seaweed may have.	Complete	Zemke-White et al. (2005) FAR2005/44
ECO	ENV2002-07	Energetics and trophic relationships of important fish and invertebrate species	1. To quantify food webs supporting important fish and invertebrate species.	Complete	Livingston 2004
ECO	CRA2000–01	Rock lobster stock assessment	Objective 11: To conduct a desktop study to identifi and explore data needs associated with managing the effects of rock lobster fishing on the environment.	Complete	Breen 2005
ECO	ENV2000-04	Identification of areas of habitat of particular significance for fisheries	1. To review literature and existing data for all significant fish species, including all QMS species, encountered from the 200-1500 m contour within	Complete	O'Driscoll et al. 2003

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		management within the New Zealand EEZ	the New Zealand EEZ to: a) determine areas of important juvenile fish habitat; b) determine areas of importance to spawning fish populations; and c) determine areas of importance for shark populations for pupping or egg laying. 2. To review literature and existing data for all significant pelagic fish species (excluding highly migratory species) encountered within the New Zealand EEZ to: a) determine areas of important juvenile fish habitat; b) determine areas of importance to spawning fish populations; and c) determine areas of importance for shark populations for pupping or egg laying 3. To review literature and existing data for all significant marine invertebrate species encountered within the New Zealand EEZ to: a) determine areas of important juvenile habitat; and b) determine areas of importance to spawning populations.		
ECO	MOF2000-02A	Future research requirements for the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery.	To recommend future research requirements for the Ross Sea Antarctic toothfish (*Dissostichus mawsoni*) fishery.	Complete	Hanchet 2000
ECO	ENV99-03	Identification of areas of habitat of particular significance for fisheries management within the NZ EEZ.	1. To determine areas of habitat of importance to fisheries management within the New Zealand EEZ for selected fish species in selected areas.	Complete	Hurst et al. 2000
ECO	ENV99-04	A framework for evaluating spatial closures as a fisheries management tool	To design a framework for evaluating spatial closures as a fisheries management tool.	Complete	Bentley et al. 2004
ECO	No project number	The fishery for freshwater eels (*Anguilla* spp.) in New Zealand	To review the fishery for freshwater eels (*Anguilla* spp.) in New Zealand.	Complete	Jellyman 1994
ZBD	ZBD2017-02	Linking primary and secondary production in the sea	1. Investigate the role that the p-ratio and the z-ratio play in modifying the relationship between	Withdrawn	

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			primary and secondary productivity at fishery-relevant scales in New Zealand 2. Improve and refine the methodology for the projection of climate change scenarios on primary and secondary productivity.		
ZBD	ZBD2017-04	Implications of ocean acidification on the capacity of carbonates in sediments to buffer eutrophication effects	1. Determine how the carbonate content of coastal benthic sediments affects sediment biogeochemistry and processes such as nitrogen recycling and removal. 2. Improve understanding of what the loss of carbonate materials from sediments in an acidified world (through dissolution) will mean for critical marine ecosystem functions.	Withdrawn	
ZBD	ZBD2016-07	Multiple stressors on coastal ecosystems – in situ	To assess the effects of global warming and ocean acidification on coastal productivity processes in New Zealand	Complete	Law et al. (2021) AEBR 277
ZBD	ZBD2016-11	Quantifying benthic biodiversity	1. Collect quantitative data about seabed habitats and fauna by undertaking a survey of unsampled areas on Chatham Rise. 2. Process and compile seabed habitat and fauna data from the survey and merge these with comparable data from previous quantitative surveys on Chatham Rise. 3. Use merged data to assess the utility of existing community and species distribution models for Chatham Rise. 4. Use merged data to build new community and species distribution models for Chatham Rise.	Completed	Bowden et al. (2019a) AEBR 221 Bowden et al. (2019b) AEBR 235
ZBD	DAE2015-05	Taxonomic ID of benthic samples	To identify benthic invertebrates in samples taken during research trawls and by observers on fishing vessels.	Complete	Tracey & Mills (2016) FRR
ZBD	ZBD2014-01	Live corals: Age and growth study of deepsea coral in aquaria.	Ocean acidification and temperature manipulation are now underway to look at the physiological responses (e.g., growth) of deepsea corals to future predicted environmental conditions.	Complete	Tracey et al. 2016

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ZBD	ZBD2014-03	Sublethal effects of environment change on fish populations	Co-funded by MBIE, this project explores the effects of ocean acidification on the behaviour of young snapper with a view to scaling up these effects to model long-term effects on the snapper population.	Complete	Parsons et al. (2021) AEBR 275
ZBD	ZBD2014-04	Isoscapes for Trophic and Animal Studies	1. To generate a new tool for fisheries management and conservation. Specifically, to produce a validated, modelled, south-west Pacific and Southern Ocean carbon and nitrogen isotopic map, referred to as an 'isoscape', which will improve our understanding of trophic interactions and its relationship with marine animals.	Complete	Graham & Bury (2019) AEBR 218
ZBD	ZBD2014-05	Modelling the effects of ocean acidification.	1. Determine how much the aragonite and calcite saturation horizons (ASH and CSH) have changed over the industrial era for the southwest Pacific, including New Zealand's EEZ.	Complete	Mikaloff-Fletcher et al. (2017) AEBR 187
ZBD	ZBD2014-06	Macroalgae mapping and potential as national scale indicators	Many countries use seaweeds to monitor the state of the marine environment, however this approach has not been explored in New Zealand. In this project, seaweeds will be selected according to their mapped distribution and availability, and assessed for their indicator potential.	Complete	D'Archino et al. (2019) AEBR 207
ZBD	ZBD2014-07	Southern coralline algae shellfish habitat	Coralline algae are a structurally important component of coastal habitats and play an important role in ecosystem processes. They produce chemicals which promote the settlement of the larvae of certain herbivorous invertebrates, particularly paua. Coralline algae appear to enhance larval metamorphosis and the survival of larvae through the critical settlement period. The first objective is to document critical baseline information on the diversity of coralline algae in southern New Zealand using morphological and molecular identification.	Complete	Nelson et al. (2019) AEBR 232
ZBD	ZBD2014-09	Climate change risks and opportunities	1. To prepare a technical report that explains the most up to date issues and hypotheses with regard to observed and predicted changes to the physical, chemical and biological properties of New Zealand coastal and offshore waters. 2. To prepare a synthesis of the Technical Report that will be informative and provide guidance on	Complete	Cummings et al. (2021) AEBR 261

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			what can be done, for stakeholders, policymakers and resource managers.		
ZBD	ZBD2014 -10	BPA Biodiversity	1. To update the inventory of benthic samples and biodiversity data available within BPA and Seamount Closure areas. 2. To process and identify undescribed samples and material in selected BPAs and for selected taxonomic groups. 3. To identify gaps in sample coverage, evaluate priority areas and design a sampling programme to collect appropriate data. 4. To undertake an objective spatial management planning exercise to assess the effectiveness of the current BPAs to protect biodiversity.	Complete	Clark et al. (2019) AEBR 227
ZBD	ZBD2013 -02	VME Genetic Connectivity	This project addresses the critical lack of data concerning deep sea genetic connectivity of VME indicator taxa, and will clarify the spatial relationships and distribution of biodiversity of several protected invertebrate VME species within New Zealand's EEZ and beyond.	Complete	Holland et al. (2020) AEBR 245 Rowden et al. 2015
ZBD	ZBD2013 -03	Continuous Plankton Recorder - Phase 2	The overall objective of the CPR programme is to map changes in the quantitative distribution of epipelagic plankton, including phytoplankton, zooplankton and euphausiid (krill) life stages, in New Zealand's EEZ and transit to the Ross Sea, Antarctica. To enable trend analysis, the Contractor will continue the annual time series for a further 5-year period (years 6–10).	Complete	Robinson et al. (2021) AEBR 257
ZBD	ZBD2013 -08	NZ-Ross sea connectivity Humpback whales	1. To determine the migration path and Antarctic feeding grounds for New Zealand humpback whales.	Complete	Riekkola et al. 2018, 2019
ZBD	ZBD2013 -06	Shell generation and maintenance of aquaculture species	Shells of individuals of NZ paua, flat oysters and cockles will undergo detailed analysis to determine how the decreased pH/increased temperature modified their shell (i) thickness, (ii) mineralogy and (iii) construction.	Complete	Cummings et al. 2013

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ZBD	ZBD2013-07	Interactive keys for easy identification keys of amphipods	Generate interactive identification keys for marine Amphipoda families Synopiidae and Epimeriidae for easy and free use online.	Complete	
ZBD	ZBD2012-01	Development of a Tier 1 National Reporting Statistic for New Zealand's Marine Biodiversity	To perform a preliminary investigation of the utility and feasibility of developing the variables published by Costello et al. (2010) as a Tier 1 statistic.	Complete	Lundquist et al. (2015) AEBR 147
ZBD	ZBD2012-03	Chatham Rise Benthos - Ocean Survey	1. In relation to the Fishing Intensity Effects Survey, determine whether there are quantifiable effects of variations in seabed trawling intensity on benthic communities. 2. In relation to the Crest Survey, conduct seabed mapping and photographic surveys in previously un-sampled areas on the central crest of the Chatham Rise.	Complete	Bowden & Leduc (2017) AEBR 183 Pinkerton et al. 2016
ZBD	SRP2011-02	IDG 2009-01 MPI fish ID field guide	1. IDG 2009-01 field guide.	Complete	McMillan 2011a, 2011b, 2011c
ZBD	ZBD2011-01	Evaluation of ecotrophic and environmental factors affecting the distribution and abundance of highly migratory species in NZ waters	Evaluation of ecotrophic and environmental factors affecting the distribution and abundance of highly migratory species in New Zealand waters.	Complete	Horn et al. (2013) AEBR 116 McGregor & Horn (2015) AEBR 146
ZBD	ZBD2010-39	Improved benthic invertebrate species identification in trawl fisheries	1. To revise and update the document 'A guide to common deepsea invertebrates in New Zealand waters (second edition)' to allow a third edition of this guide to be printed.	Complete	Tracey et al. 2011a
ZBD	ZBD2010-40	Predictive modelling of the distribution of vulnerable marine ecosystems in the South Pacific Ocean region.	1. To develop and test spatial habitat modelling approaches for predicting distribution patterns of vulnerable marine ecosystems in the convention Area of the South Pacific Regional Fisheries Management Organisation with agreed international partners. 2. To collate datasets and evaluate modelling approaches which are likely to be useful to predict the distribution of vulnerable marine ecosystmes in the South Pacific Ocean region.	Complete	Rowden et al. (2013b) AEBR 120

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ZBD	ZBD2010-41	Ocean acidification in fisheries habitat	1. To assess the risks of ocean acidification to deep-sea corals and deepwater fishery habitat. 2. To determine the carbonate mineralogy of selected deep-sea corals found in the New Zealand region. 3. To assess the distribution of deep-sea coral species in the New Zealand region relative to improved knowledge of current and predicted aragonite and calcite saturation horizons, assessment of potential locations vulnerable to deep water upwelling. 4. Through a literature search and analysis, determine the most appropriate tools to age and measure the effects of ocean acidification on deep sea habitat-forming corals, and recommend the best approach for future assessments of the direct effects.	Complete	Tracey et al. (2013) AEBR 117
ZBD	ZBD2009-25	Predicting impacts of increasing rates of disturbance on functional diversity in marine benthic ecosystems	1. Further develop the landscape ecological model of disturbance/recovery dynamics in marine benthic communities, incorporating habitat connectivity, based on existing model by Lundquist, Thrush, and Hewitt. 2. Predict impacts of increasing rates of disturbance on rare species abundance, functional diversity, relative importance of biogenic habitat structure, and ecosystem productivity. 3. Use literature and expert knowledge to quantify rare species abundance, biomass, functional diversity, habitat structure, and productivity of various successional community types in the model. 4. Field test predictions of the model in appropriate marine benthic communities where historical rates of disturbance are known, and benthic communities have been sampled.	Complete	Lundquist et al. 2010, (2013) AEBR 118
ZBD	IPA2009-14	Bryozoan identificaiton guides	1. For each of ~50 species of common bryozoans, provide photos and text to allow for identification. Provide information on distribution and habitat (as far as is known) and further references for each species and on bryozoans as a whole. 2. Submit these data for publication in the Ministry of Fisheries series New Zealand Aquatic Environment and Biodiversity Research.	Complete	Smith & Gordon (2011) AEBR 75
ZBD	ZBD2009-03	To evaluate the vulnerability of New Zealand rhodolith species to environmental stressors and to characterise diversity of rhodolith beds.	1. To characterise the distribution and physical characteristics of two New Zealand rhodolith beds and characterise the associated biodiversity. 2. To measure the growth rates and evaluate the	Complete	Nelson et al. 2012

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			vulnerability of New Zealand species of rhodoliths to environmental stressors.		
ZBD	ZBD2009–10	Multi-species analysis of coastal marine connectivity	1. Determine overall patterns of regional connectivity in a broad range of NZ coastal marine organisms to define the geographic units of genetic diversity for protection and the dispersal processes that maintain this diversity. 2. Review previous studies of marine connectivity and population genetics in NZ coastal organisms to determine the preliminary range of patterns observed and the principal gaps (taxonomic geographic and ecological) in our understanding. 3. In a range of invertebrate and vertebrate marine organisms determine geographic patterns of genetic variation using standardised sampling and molecular techniques. 4. Analyse data across past and present studies to reveal both common and unique patterns of connectivity around the NZ coastline and the locations of common barriers to dispersal.	Complete	Gardner et al. (2010) AEBR 58 Hannan et al. (2016) AEBR 172
ZBD	ZBD2009-13	Ocean acidification impact on key NZ molluscs	1. Controlled laboratory experiments will be used to determine the effect of pCO2 levels that are predicted to occur in NZ waters over the next few decades on appropriate life history stages of at least two key NZ mollusc species. A number of response variables will be assessed. 2. Implications of these responses to the local and broader ecosystems will be assessed.	Complete	Cummings 2011; Cummings et al. 2011, 2013
ZBD	ZBD2008-01	Biogenic large–habitat–former hotspots in the near-shore coastal zone (50–250 m); quantifying their location, identity, function, threats and protection	1. To collect and integrate existing knowledge on biogenic habitat-formers in the <5–150 m depth zone of New Zealand's continental shelf, from sources including structured fisher interviews, primary and grey literature, and other sources as available. 2. Using the findings of Objective 1, design and deploy a series of sampling voyages to selected locations, to map and characterise locations of significant biogenic structure (either still existing, or historical), and collect relevant biological samples (both through visual census, and physical collection). 3. Process and analyse the samples collected in Objective 2, to provide a hierarchical, quantitative description of the biogenic habitats and associated species encountered. 4. Using the findings from Objective 1–3, assess the present status, likely extent, ecological role, and threats to, biogenic habitat formers in the <5–150	Complete	Jones et al. (2016) AEBR 174 Jones et al. (2018) AEBR 202

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			m depth zone. This should include a spatial modelling and risk assessment framework. Integrate (as appropriate) with other information sources and/or approaches that may exist by the year 2010/11.		
ZBD	ZBD2008-05	Macroalgal diversity associated with soft sediment habitats	1. Conduct a targeted collection programme across diverse soft sediment environments to develop a permanent reference collection of representative macroalgae. 2. Examine algal distribution in soft sediment habitats in relation to selected environmental variables. 3. Prepare an annotated checklist of macroalgae found in soft sediment environments in the New Zealand region.	Complete	Neill et al. (2012) AEBR 87
ZBD	ZBD2008-07	Carbonate sediments: the positive and negative effects of land-coast interactions on functional diversity	1. To quantify shifts in community structure and functional diversity in mollusc dominated habitats along gradients associated with an estuary-coast interface in two locations. 2. To characterise the influence of estuary-derived food sources across these gradients for key species. 3. To measure changes in growth of key species in relation to changes in food supply and land-derived sediment impacts. 4. To quantify carbon and nitrogen uptake and tissue turnover rates of key species in laboratory experiments.	Complete	Thrush et al. 2011 Savage et al. 2012
ZBD	ZBD2008-11	Predicting changes in plankton biodiversity and productivity of the EEZ in response to climate change-induced ocean acidification	1. To document the spatial and inter-annual variability of coccolithophore abundance and biomass- and assess in terms of the phytoplankton abundance- biomass and community composition in sub-tropical and sub-Antarctic water. 2. To document the seasonal and inter-annual variability of foraminifera and pteropod abundance and biomass at fixed locations in sub-tropical and sub-Antarctic water by analysis of sediment trap material from time-series data collection. 3. To document the spatial and seasonal distribution of the key coccolithophore species, *Emiliana huxleyi*, using both archived and ongoing ingestion of satellite images of Ocean Colour, and ground-truth the reflectance. 4. To determine the sensitivity and response of *E. huxleyi* and other EEZ coccolithophores to pH under a range of realistic atmospheric CO2 concentrations in perturbation experiments, using monocultures and mixed populations from in situ sampling. 5. To document the spatial variability of diazotrophs (nitrogen-fixing organisms) and associated nitrogen	Complete	Law et al. (2012) FRR Boyd & Law 2011

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			fixation rate, and assess in terms of phytoplankton abundance, biomass and community composition in sub-tropical waters north of the STF. 7. To determine the sensitivity of, and response of *Trichodesmium* spp. and other diazotrophs to pH under a range of realistic atmospheric CO2 concentrations in perturbation experiments using monocultures.		
ZBD	ZBD2008-14	What and where should we monitor to detect long-term marine biodiversity and environmental changes, remote sensing, biota, context, inshore offshore workshop	1. Identify the key questions to be addressed by long-term monitoring of marine biodiversity and environment. 2. Identify appropriate monitoring indices, how they should be spatially distributed and their sampling frequency. 3. Identify relevant existing monitoring programmes across the range of New Zealand agencies and science providers and identify gaps. 4. Provide those agencies setting environmental goals/ standards or research needs (MoRST, FRST, MFish, DOC, MfE, Commissioner for the Environment) with a thorough situational analysis, including a list of priority monitoring projects/plans.	Complete	Livingston 2009
ZBD	ZBD2008-15	Continuous plankton recorder project: implementation and identification	1. To set up a time series of annual CPR data collection by deployment from a toothfish vessel on the annual summer transit between New Zealand and the Ross Sea. 2. To identify phytoplankton and zooplankton according to strict observation protocols determined by the SAHFOS[1] CPR Survey and SO-CPR[2]. 3. To enter species data, frequency and location along the transect into a spreadsheet that will allow spatial mapping of the plankton density and distribution. 4. To analyse the full dataset after 5 years of data collection to: (a) determine trends in the dataset and (b) compare results with Australian datasets available through SO-CPR. 5. To evaluate the continuation of the programme.	Complete	Robinson et al. (2014) AEBR 128
ZBD	ZBD2008-20	Ross sea benthic ecosystem function: predicting consequences of shifts in food supply	1. To increase understanding of Ross Sea coastal benthic ecosystem function. 2. Conduct in situ investigations into responses to and utilisation of primary food sources by key species, at two contrasting coastal Ross Sea locations.	Complete	Cummings & Lohrer 2011 Cummings et al. 2011 Lohrer et al. 2013

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ZBD	ZBD2008-22	Acidification and ecosystem impacts in NZ and southern ocean waters (data collected during IPY).	1. To assess the response of cocolithophorids, and their replacement by non-calcifying organisms during incubation under a range of dissolved CO2 concentrations. 2. To describe and characterise changes in abundance and biodiversity of microbial components of the samples incubated at sea under a range of dissolved CO2 concentrations. 3. To predict the likely impacts of higher acidity on foodwebs and on carbon fixation under scenarios to be encountered in the Southern Ocean under forecasted trends associated with climate change.	Complete	Maas et al. 2010b
ZBD	ZBD2008-23	Macroalgae diversty and benthic community structure at the Balleny Islands	1. To describe and characterise macroalgae diversity from the Balleny Islands and the Western Ross Sea. 2. To describe and quantify benthic community structure from one location at the Balleny Islands 3. To complete anatomical and morphological investigations & molecular sequencing required for the identification of macroalgae samples from the Balleny Islands & western Ross Sea coastline to describe & characterise macroalgae diversity in Balleny Islands. 4. To process and analyse samples collected at the Balleny Islands, to analyse them using ICECUBE methodology and compare results with those from other ICECUBE sampling locations along the Ross Sea coastline	Complete	Nelson et al. (2010) AEBR 55
ZBD	ZBD2008-27	Scoping investigation into New Zealand abyss and trench biodiversity	1. Review what is already known of abyssal, canyon and trench faunas in NZ. 2. Review what is already known of abyssal, canyon and trench faunas around the world. 3. Prioritise science questions and locations for exploration. 4. Assess NZ capacity to sample at the required depths; identify sampling equipment needs. 5. Design a suitable vessel-based sampling programme.	Complete	Lörz et al. (2012b) AEBR 92
ZBD	ZBD2008-50	OS2020 Chatham Rise Biodiversity Hotspots	1. To improve understanding of the effects of trawl fishing in New Zealand on the biodiversity of seamounts, knolls and hills. 2. To describe differences in benthic biodiversity between northwestern and eastern regions of the Chatham Rise. 3. To continue the time series of observations in the NW Chatham Rise to demonstrate recovery in terms of biodiversity. 4. To extend the observations on fished-unfished contrasts and recovery of fauna on protected	Complete	Clark et al. 2009

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			seamounts to an oceanographically distinct location.		
ZBD	IPY2007-01	International polar year census of antarctic marine life post-voyage analysis: Ross Sea - Southern Ocean Biodiversity	1. To measure seabed depth and rugosity using the multibeam system to identify topographic features such as bottom type, iceberg scouring, seamounts etc and to determine areas for targeted benthic faunal sampling. 2. To continue the analysis of opportunistic seabird and marine mammal distribution observations from this and previous BioRoss voyages and published records, and in relation to environmental variables. 3. To identify and determine near-surface spatial distribution, diversity and abundance of phytoplankton, and zooplankton, based on Continuous Plankton Recorder samples collected during transit to and from the Ross Sea. 4. To collect & analyse data collected both underway, & at stations for salinity, temperature nutrient and chlorophyll-a data, spot optical measurements with the SeaWiFS. 5. To identify and determine the spatial distribution, abundance (biomass), diversity, and size structure of epipelagic, mesopelagic (and possibly bathypelagic) species using acoustics and net sampling. 6. To identify and measure diversity, distribution & densities of mesozooplankton, macrozooplankton & meroplankton (as collected by all plankton sampling methods except transit CPR samples). 7. To determine diversity, distribution & densities of viral, bacterial, phytoplankton & microzooplankton species in the water column. 8. To determine the spatial distribution, abundance (biomass), diversity, and size structure of shelf and slope demersal fish species and associated invertebrate species using a demersal survey. 9. To determine the diversity, abundance/density, spatial distribution, and physical habitat associations of benthic assemblages across a body size spectrum from megafauna to bacteria, for shelf, slope, seamounts, and abyssal sites in Ross Sea. 10. To describe trophic/ecosystem relationships in the Ross Sea ecosystem (pelagic and benthic, fish and invertebrates). 11. Assess molecular taxonomy and population genetics of selected Antarctic fauna and flora to estimate evolutionary divergence within and among ocean basins in circumpolar species. Provide DNA barcoding.	Complete	Allcock et al. 2009, 2010, submitted; Alvaro et al. 2011; Baird & Mormede 2014; Bowden et al. 2011a, in prep; Clark et al. 2010a; Dettai et al. 2011; Eakin et al. 2009; Eléaume et al. 2011, in prep; Ghiglione et al. 2012; Gordon 2000; Grotti et al. 2008; Hanchet et al. 2008a, 2008b, 2008c, 2008d, 2013 ; Hanchet 2009, 2010; Heimeier et al. 2010; Hemery et al. in prep; Koubbi et al. 2011; Leduc et al. 2012a, 2012b, 2012c, 2013, 2014; Linse et al. 2007; Lörz 2009, 2010a, 2010b; Lörz & Coleman 2009; Lörz et al. 2007, 2009, 2012a, 2012b, 2012c, in prep; Maas et al. 2010a; McMillan et al. 2012; Mitchell 2008; Nielsen et al. 2009; Norkko et al. 2005; O'Driscoll 2009; O'Driscoll et al. 2009, 2010, 2012; O'Loughlin et al. 2011; Pakhomov et al. 2011; Pinkerton

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					et al. 2007a, 2009b, 2010, 2013; Schiaparelli et al. 2006, 2008, 2010; Smith et al. 2011a, 2011b; Stein 2012; Strugnell et al. 2012
ZBD	IPY2007-02	International polar year census of antarctic marine life post-voyage analysis: Ross Sea - Southern Ocean Biodiversity	1. To measure and describe key elements of species distribution, abundance (density or biomass) & biodiversity for the Ross Sea and Southern Ocean for main habitats and key functional ecosystem roles, for major groups, viruses, bacteria, and archaea. 2. To report on the diversity of Antarctic Cephalopoda (Octopus and Squid), including a complete inventory of taxa, & reports on ontogenetic & sexual variation in species, their systematics, diversity, distribution, life histories, & trophic importance. 3. To Beak/Biomass Regression Equations. 4. Life cycle determination.	Complete	Garcia 2010
ZBD	ZBD2007-01	Chatham-Challenger Oceans 20/20 Post-Voyage	1. To quantify in an ecological manner, the biological composition and function of the seabed at varying scales of resolution, on the Chatham Rise and Challenger Plateau. 2. To elucidate the relative importance of environmental drivers, including fishing, in determining seabed community composition and structure. 3. To determine if remote-sensed data (e.g., acoustic) and environmentally derived classification schemes (e.g., marine environmental classification system) can be utilised to predict bottom community composition, function, and diversity. 4. To count, measure, and identify to species level (where possible, otherwise to genus) all macro invertebrates (> 2 mm) and fish collected during Oceans 20/20 voyages. 5. To count, measure and identify to species level (where possible, otherwise to genus or family) all meiofauna (> 2 mm) from multicore samples collected during the Oceans 20/20 voyages. 6. To count, measure and identify to species level (where possible, otherwise to genus or family) all fauna collected by hyper-benthic sled during the Oceans 20/20 voyages. 7. To count, measure, and identify to species level all macrofauna observed on DTIS images collected during the Oceans 20/20 voyages. The number of biogenic features (burrows/mounds) and habitat	Complete	Bowden (2011) AEBR 65 Bowden et al. (2011b) AEBR 77 Bowden et al. (2014) AEBR 126 Bowden & Hewitt 2012 Coleman & Lörz 2010 Compton et al. 2012 Floerl et al. (2012) AEBR 97 Hewitt et al. (2011a) AEBR 81 Hewitt et al. (2011b) AEBR 83 Lörz 2011a, 2011b

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			(spatial) complexity should also be estimated. 8. To count, measure, and identify to species level (where possible, otherwise to genus or family) all macrofauna observed on DTIS video footage collected during the Oceans 20/20 voyages. 9. To calculate and compare the performance of a suite of diversity measures (species and taxonomic-based) at varying levels of resolution. 10. To estimate particle size composition and organic content of sediment samples. Sediment samples should be aggregated over the top 5 cm of sediment. 11. To measure the bacterial biomass (top 2 cm) of the sediment and in the sediment surface water samples, collected during the Oceans 20/20 voyages. 12. To elucidate the relationships, patterns and contrasts in species composition, assemblages, habitats, biodiversity and biomass (abundance) both within and between stations, strata and areas. 13. To define habitats (biotic) encountered during the survey and assess their relative sensitivity to modification by physical disturbance, their recoverability, and their importance to ecosystem function/production. 14. To quantify the productivity, energy flow (trophic networks) and the energetic coupling (bentho pelagic or otherwise) of the area surveyed areas at various levels of resolution. 15. To assess the extent to which patterns of species distributions and communities can be predicted using environmental data (including fishing) collected during the Ocean 20/20 voyages or held in other databases. 16. To provide an interactive, high resolution mapping facility for displaying & plotting all data collected & derived indices. Includes environmental data, the abundance of species, indices of biomass or diversity, and statistically derived groupings. 17. To assess the extent to which acoustic, environmental or other remote-sensed data can provide cost-effective reliable means of assessing biodiversity at the scale of the Oceans 20/20 surveys. 18. To assess the extent to which the 2005 MEC and subsequent variants can provide cost-effective reliable means of assessing biodiversity at the scale of the Oceans 20/20 surveys. 19. Collating all information and analysis from all objectives, devise a series of statistically supported recommendations for surveying marine biodiversity in the future. Including – but may not be limited to – statistical analyses and modelling.		Nodder et al. (2011) AEBR 70

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ZBD	ZBD2006-02	Ongoing NABIS development	As part of NABIS, users will be able to identify spatial information relating to the annual distribution (average distribution over the period of a year) of particular species within the waters around New Zealand and in the terrestrial environment (including offshore islands) of New Zealand. Users will also be able to interrogate metadata and attribute data related to the information layers presented. Users will employ NABIS to identify where a particular species is found, to identify what species are found within an area of interest, and be able to compare the spatial distribution of a particular species with other information layers. 2. Some species may have notable changes in their spatial distribution throughout a year. For such species, users of NABIS will be able to view spatial information relating to the seasonal distribution of particular species within the waters around New Zealand and in the terrestrial environment (including offshore islands) of New Zealand. Users will also be able to interrogate metadata and attribute data related to the information layers presented. For species with a seasonal component to their biological distribution, users will employ NABIS to identify where a particular species is found within the waters around New Zealand and in the terrestrial environment (including off shore islands) of New Zealand at a particular time of the year, to identify what species are found within an area of interest at a particular time of year, or be able to compare the distribution of a particular species at a particular time of year, with other information layers. 3. To provide analysis of the data used in determining the hotspot distribution.	Complete	Anderson 2007
ZBD	ZBD2006-03	Antarctic coastal marine systems	1. Quantify patterns in benthic community structure and function at two coastal Ross Sea locations (Terra Nova Bay and Cape Evans). 2. Quantify benthic community structure and function at selected locations in Terra Nova Bay and Cape Evans.	Complete	Cummings et al. 2003, 2006b, 2008 Thrush & Cummings 2011 Thrush et al. 2010
ZBD	ZBD2006-04	Chatham/Challenger oceans 20/20	1. To collect seabed fauna, sediment samples and photographic images along transects in the Chatham Rise and the Challenger Plateau, as determined by the sampling protocol described in the Voyage Programmes for Voyages 2 and 3 of the project. Multibeam data should be collected	Complete	Nodder 2008 Nodder et al. 2011

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			opportunistically as time allows. 2. To describe the distribution of broad macro epifauna groups (ID level to be determined at sea during Surveys 2 & 3), their relative abundance, the substrate and habitat types, including representative photographic images of each sea-bed habitat and associated fauna along transects in the survey areas. 3. To provide a description of the observed evidence of fishing along transects. 4. To provide indicative measures of alpha biodiversity (richness, number of taxonomic groups) at appropriate scales within and between transects, and between the Chatham Rise and the Challenger Plateau. 5. To determine broad scale variability in sea-bed habitats and associated biodiversity within and between MEC classes at 20 class levels. 6. To process and archive biological samples and data into databases and collections for future analysis in meeting the Overall Objectives above.		
ZBD	ZBD2005-01	Balleny Islands Ecology Research, Tiama Voyage (2006)	1. To characterise shallow benthic communities across a range of habitat settings around the Balleny Islands, utilising a range of data collection methodologies (including SCUBA-based rock-wall suspension feeder photo quadrats, SCUBA-based linear video transects, and drop camera photography), and to analyse community patterns with reference to possible physical/oceanographic, biological, and/or biogeographic influences on community structure. 2. To characterise aspects of the marine food web of the Balleny Islands area, using stable isotope analysis of specimens from important functional groups, and to make inferences about factors affecting ecosystem-scale trophodynamics in the Balleny Islands area and potential implications for the function of the wider ecosystem. 3. To characterise the spatial and temporal distributions of higher-level consumer species (birds, seals and whales) and of dominant pelagic prey (i.e., krill swarms) by opportunistically recording all at-sea sightings, and by systematic observation of landbased top predators (birds and seals) while sailing along the coast of the islands. 4. To collect and photograph and/or retain fish specimens from shallow benthic environments using a range of fishing methods, including food-baited fish traps, lightbaited fish traps, rotenone sampling, and/or baited lines. 5. To continuously collect bathymetric data and water-column acoustic data (i.e., mesopelagic acoustic marks) throughout the voyage, using an	Terminated	Smith 2006

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			acoustic sounder. 6. To opportunistically collect a variety of data/materials during shore-based landings, including wherever possible: i) breast feathers from living penguins; ii) tissue samples/feathers/bones from dead seals/penguins/other sea birds; iii) seal scats; iv) visual estimates of adult and juvenile penguin numbers; v) visual assessments of penguin colony status; vi) photographs of penguin colonies; vii) sediment excavations of occupied and abandoned colonies. (Where appropriate these data will contribute to Objective 2).		
ZBD	ZBD2005-02	Marine Environment Classification Project	1. Co-fund the Marine Environment Classification Project (being done by NIWA) with the Department of Conservation.	Complete	Snelder et al. 2005, 2006 Leathwick et al. 2006a, 2006b, 2006c
ZBD	ZBD2005-03	Tangaroa Ross Sea voyage	1. To test the feasibility of obtaining estimates of demersal fish relative abundance using cameras with and without flood lights in areas of high importance for the Ross Sea toothfish fishery (principally 800–1200 m). 2. To utilise deepwater camera transects, supported by other direct sampling methods, to characterise the relative abundance, distribution, and diversity of demersal fish species (assuming Objective 1 yields satisfactory results) and of benthic macro-invertebrates, and to examine relationships between demersal fishes and benthic habitats/communities. Camera transects will be deployed opportunistically, with focus on the following high-priority areas (in order of high to low priority) wherever possible: i) Areas of the continental shelf break at depths of high importance for the toothfish fishery (principally 800-1200 m but also 600-800m & 1200-1500 m if time permits), ii) Shallow (50-200 m) water in the immediate vicinity of the Balleny Islands; iii) Deeper water in the vicinity of the Balleny Islands; iv) seamounts around and between Scott Island and the Balleny Islands; and v) at other locations (<600 m) as opportunity arises (e.g., around Scott Island, western Ross Sea, south-eastern Ross Sea). 3. To collect specimens/tissues of selected benthic and pelagic organisms with priority in the vicinity of the Balleny Islands (and to the east/southeast, for pelagic specimens especially Antarctic krill species) and deliver specimens to other projects for stable	Complete	MacDiarmid & Stewart (2015) AEBR 153 Mitchell & MacDiarmid 2006

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			isotope analysis in order to contribute to understanding of trophic relationships. 4. To acquire a continuous acoustic survey of the water column, opportunistically undertake species verification of acoustic marks, integrate the acoustic marks and produce a GIS map of verified and unverified distributions of functionally important mesopelagic species (e.g., krill, Antarctic silverfish). 5. To undertake routine identification and abundance estimates of marine mammal and seabird species and deliver raw and GIS summarised data to other related projects in order to generate spatially and temporally explicit population biomass and foraging distribution estimates for top air-breathing predators in the Ross Sea. 6. To undertake automated water sampling in order to monitor the identities and spatial and temporal distributions of plankton in the Ross Sea region and to allow ground-truthing of data collection from satellites (e.g., surface seawater temperature, and chlorophyll-a concentration).		
ZBD	ZBD2005-05	Long-term effects of climate variation and human impacts on the structure and functioning of New Zealand shelf ecosystems	1. To estimate changes in marine productivity via fluctuations in ocean climate and terrestrial nutrient input over the last 1000 years. 2. To assess and collate existing archaeological, historical and contemporary data (including catch records and stock assessments) on relevant components of the marine ecosystem to provide a detailed description of change in the shelf marine ecosystem in two areas of contrasting human occupation over last 1000 years. 3. To collect additional oral histories from Māori and non-Māori fishers and shellfish gathers regarding the distribution, sizes and relative abundance (compared to present availability) of key fish and invertebrate stocks in both regions during the first half of the 20th century before the start of widespread modern industrial fishing. 4. To build mass-balance ecosystem models (e.g., Ecopath) of the coastal and shelf ecosystem in each area for five critical time periods: now, 60 years BP (before modern industrial fishing), 250 years BP (before European whaling and sealing), 600 y BP (early Māori phase) and 1000 years BP (before human settlement). 5. To use qualitative modelling techniques to determine the critical interactions amongst species and other ecosystem components in order to identify those that should be a priority for future research.	Complete	Carroll et al. 2015 Lalas et al. 2014 Lalas & MacDiarmid 2014 Lorrey et al. 2013 MacDiarmid et al. (2016a) AEBR 171 MacDiarmid et al. (2016b) AEBR 170 MacDiarmid et al. (2018) AEBR 194 Maxwell & MacDiarmid (2016) AEBR 173 McKenzie & MacDiarmid submitted Neil et al. 2012

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					Parsons et al. 2011 Paul (2012) AEBR 94 Paul (2014) AEBR 124 Pinkerton et al. (2015b) AEBR 160 Smith (2011) AEBR 82
ZBD	ZBD2005-09	Rocky reef ecosystems - how do they function? Integrating the roles of primary and secondary production, biodiversity and connectivity across coastal habitats	1. To develop a qualitative numerical model of how New Zealand's rocky reef systems are functionally structured. 2. To quantify the effects of human predation, and environmental degradation across reef gradients – top-down, or bottom-up functioning? 3. To advance our understanding of how subtidal reef systems are fuelled through primary and secondary production (from a range of sources), the role that biodiversity plays, and how this varies across different reef settings. 4. To quantify how subtidal reef systems are linked with other habitats and ecosystems at broader spatial scales, including the connectivity of MPAs with other habitats and areas.	Complete	Beaumont et al. 2011
ZBD	ZBD2004-01	Baseline information on the diversity and function of marine ecosystems	1. To quantify, and compare, the macro-invertebrate assemblage composition of a number of seamounts at the southernmost end of the Kermadec volcanic arc. 2. To compare the macro-invertebrate diversity of the southernmost end of the Kermadec volcanic arc with that of seamounts already sampled and reported on.	Complete	Rowden & Clark (2010) AEBR 62 Smith et al. 2008
ZBD	ZBD2004-02	Ecosystem-scale trophic relationships: diet composition and guild structure of middle-depth fish on the chatham rise	1. To quantitatively characterise the diets of abundant middle-depth fish species on the Chatham Rise, by analysis of fish stomach contents collected from the January 2005, January 2006 and January 2007 Chatham Rise middle-depths trawl surveys. 2. To quantitatively characterise Chatham Rise fish diets throughout the year, for a period of 24 months, by analysis of fish stomach contents collected opportunistically aboard industry vessels.	Complete	Connell et al. 2010 Dunn 2009 Dunn et al. 2009b, 2010a, 2010b, 2010c

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			3. To describe and examine patterns of diet variation within each fish species as a function of spatial, temporal, and environmental variables, and of fish size. 4. To define and characterise trophic guilds for abundant fish species on the Chatham Rise, using multivariate analysis of fish diet data, and to analyse the nature and relative strength of potential trophic interactions between guilds. 5. To create and populate a diets database to store all of the dietary information collected under Objectives 1 and 2, and for use in subsequent dietary studies.		Forman & Dunn 2010 Horn et al. 2010 Stevens & Dunn 2010
ZBD	ZBD2004 –05	Assessment and definition of the biodiversity of coralline algae of northern New Zealand	1. To assess and define the biodiversity of coralline algae in northern New Zealand. 2. To develop rapid identification tools for coralline algae using molecular sequencing data. 3. To contribute representative material to the national Coralline Algal Collections. 4. To produce ID guides to common coralline algae of northern New Zealand.	Complete	Farr et al. 2009
ZBD	ZBD2004 -08	Sea-grass meadows as biodiversity and connectivity hotspots	1. Quantify the biodiversity values and functioning of New Zealand sea-grass assemblages. 2. Complete national bio-geographic assessment of sea-grass associated biodiversity. 3. Quantify sea-grass connectivity with surrounding marine landscapes through nursery functions and detritus export. 4. Quantify sea-grass replication connectivity mechanisms. 5. Develop a risk assessment and appraisal model for sea-grass systems.	Complete	Morrison et al. (2014c) AEBR 137
ZBD	ZBD2004 -10	Development of bioindicators in coastal ecosystems	1. Investigate linkages between land use patterns in catchments and nitrogen loading to recipient estuaries and coastal ecosystems. 2. Characterise isotopic signatures of selected bioindicator organisms in relation to different terrestrial nutrient loads. 3. Validate the use of bioindicators using controlled laboratory and field experiments.	Complete	Savage 2009
ZBD	ZBD2004 -19	Ecological function and critical trophic linkages in New Zealand soft-sediment habitats	1. Define the interactive effects of two functionally important benthic species in maintaining critical trophic linkages in soft-sediment systems from a series of integrated field experiments. 2. Quantify effects of heart urchins (*Echinocardium australe*) on sediment properties, benthic primary	Complete	Lohrer et al. 2010

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			production, and macrofaunal diversity through manipulative field experiments in Mahurangi Harbour. 3. Test for interactions between pinnid bivalves (*Atrina zelandica*) and heart urchins (*Echinocardium australe*) in field experiments, and measure their respective and combined contributions to sediment properties, benthic primary production, and macrofauna. 4. Determine the dependence of results from objectives 1 and 2 (functional contributions of *Echinocardium* and *Atrina*) in an environmental context by conducting experiments along an estuarine-coastal gradient.		
ZBD	ZBD2003 -02	Biodiversity of Coastal Benthic Communities of the North Western Ross Sea.	1. Quantify patterns in biodiversity and community structure in the coastal Ross Sea region. 2. Quantify biodiversity in benthic communities at selected locations in the Ross sea north of Terra Nova Bay. 3. Describe ecosystem function at selected locations in the Ross Sea north of Terra Nova Bay.	Complete	Cummings et al. 2003, 2006a, 2010 De Domenico et al. 2006 Guidetti et al. 2006 Norkko et al. 2004
ZBD	ZBD2003 -03	Biodiversity of deepwater invertebrates and fish communities of the north western Ross Sea	1. To describe, and quantify the diversity of, the benthic macroinvertebrates and fish assemblages of the Balleny Islands and adjacent seamounts, and to determine the importance of certain environmental variables influencing assemblage composition.	Complete	Rowden et al. (2012a) AEBR 101 Rowden et al. (2013a) AEBR 115 Mitchell & Clark 2004
ZBD	ZBD2003 –04	Fiordland Biodiversity Research Cruise	1. How can ecotone boundaries be defined? 2. If you have an ecotone boundary defining the edge of a commercial exclusion zone, how wide is the transition zone across the boundary? 3. If you have an area delineated as a marine protected area or a commercial exclusion zone, does it adequately represent the different habitats or biodiversity of the whole region?	Complete	Wing 2005
ZBD	ZBD2003 -09	Macquarie Ridge Complex Research Review	To review and summarise both biological and physical research carried out on or around the	Complete	Grayling 2004

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			section of the Macquarie Ridge Complex that lies between New Zealand and Macquarie Island.		
ZBD	ZBD2002-01	Ecology of Coastal Benthic Communities in Antarctica	To research the ecology of coastal benthic Communities in Antarctica.	Complete	Cummings et al. 2003 Schwarz et al. 2003, 2005 Sharp et al. 2010 Sutherland 2008 Thrush et al. 2006 Thrush & Cummings 2011
ZBD	ZBD2002-02	Whose larvae is that? Molecular identification of planktonic larvae of the Ross Sea.	1. To use molecular sequencing tools in the taxonomic identification of cryptic/invasive marine species. 2. To provide a molecular description and characterisation of gobies that are introduced (*Arenigobius bifrenatus* and *Acentrogobius pflaumii*) cryptogenic (*Parioglossus marginalis*) or native (e.g., *Favonigobius lentiginosus* and *F. expuisitus*). 3. To describe the molecular diversity of the above species throughout their native and introduced distributions, and characterise a range of the greatest potential invasive gobioid and blennioid species from the Australasian region. 4. To develop molecular criteria to rapidly identify invasive or cryptogenic gobioid and blennioid fish.	Complete	Sewell 2005, 2006 Sewell et al. 2006
ZBD	ZBD2002-06A	Impacts of terrestrial run-off on the biodiversity of rocky reefs	1. Conduct field and laboratory experiments to determine relationships between sediment loading, epifaunal assemblages, and mortality of filter feeding invertebrates. 2. Conduct field and laboratory experiments to identify the influence of sediment on early life stages of key grazers. 3. Determine photosynthetic characteristics and survival of large brown seaweeds and understorey algal species in relation to a sediment gradient.	Complete	Schwarz et al. 2006
ZBD	ZBD2002-12	Molecular identification of cryptogenic/invasive marine species – gobies.	1. To use molecular sequencing tools in the taxonomic identification of cryptic/invasive marine species 2. To provide a molecular description and characterisation of gobies that are introduced	Complete	Lavery et al. 2006

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			(*Arenigobius bifrenatus* and *Acentrogobius pflaumii*) cryptogenic (*Parioglossus marginalis*) or native (e.g., *Favonigobius lentiginosus* and *F. expuisitus*). 3. To describe the molecular diversity of the above species throughout their native and introduced distributions- and characterise a range of the greatest potential invasive gobioid and blennioid species from the Australasian region. 4. To develop molecular criteria to rapidly identify invasive or cryptogenic gobioid and blennioid fish.		
ZBD	ZBD2002 -16	Joint New Zealand and Australian Norfolk Ridge	1. To describe the marine biodiversity of the Norfolk Ridge and Lord Howe Rise seamount communities. 2. To survey, sample and document the marine biodiversity and environmental data from seamounts on the Norfolk Ridge and Lord Howe Rise to a depth of at least 1000 m depth. 3. To preserve samples of fishes and invertebrates and hold these in accessible curated museum collections to support biosystematic research projects. 4. To provide specimens to support projects which research the identity, diversity, relationships, distributions, and assess uniqueness and conservation value of the marine life. 5. To correlate observed distribution patterns, especially areas of high diversity and areas of endemism, with measured biological and physical parameters.	Complete	Clark & Roberts 2008
ZBD	ZBD2002 -18	Quantitative survey of the intertidal benthos of Farewell Spit Golden Bay	1. To undertake a baseline survey of intertidal macrobenthic organisms at Farewell Spit Nature Reserve and adjacent flats. 2. To undertake an initial field survey of *Zostera* distribution at Farewell Spit Nature Reserve and adjacent intertidal flats. 3. To undertake a preliminary survey of sediment characteristics of the intertidal flats at Farewell Spit Nature Reserve and adjacent flats.	Complete	Battley et al. 2005
ZBD	ZBD2001 –02	Documentation of New Zealand Seaweed	1. To publish a regional algal flora of Fiordland based on voucher herbarium specimens. 2. To assemble a database of references and to review the current state of knowledge about New Zealand macroalgae.	Complete	Nelson et al. 2002
ZBD	ZBD2001 -03	Ecology and biodiversity of coastal benthic communities in Antarctica.	1. To develop sampling protocols for estimating the relative abundance of algae and benthic invertebrates. 2. To quantify patterns in biodiversity and benthic community structure at two locations in McMurdo	Complete	Norkko et al. 2002

Theme	Project code	Project title	Specific objectives	Status	Citation/s
			Sound. 3. To analyse Ross Island Sea-Level data.		
ZBD	ZBD2001-04	'Deep Sea New Zealand'	To help publish the book 'Deep Sea New Zealand'.	Complete	Batson 2003
ZBD	ZBD2001-05	Crustose coralline algae of New Zealand	1. To assess the biodiversity of crustose coralline algae in NZ using modern taxonomic methods and molecular sequence tools. 2. To establish the NZ National Coralline Algal Collection. 3. To produce identification guides to NZ species.	Complete	Harvey et al. 2005 Farr et al. 2009 Broom et al. 2008
ZBD	ZBD2001-06	Biodiversity of New Zealand's soft-sediment communities	1. To review the current knowledge of the biodiversity of macroinvertebrates and macrophytes living in and on soft-sediment substrates in New Zealand's harbours, estuaries, beaches and to 1000 m water depth. 2. To review existing published and unpublished sources of information on soft-sediment marine assemblages around New Zealand. 3. Using the results of Objective 1, identify gaps in the knowledge, hotspots of biodiversity, areas of particular vulnerability, and make recommendations on areas or assemblages that could be the subject of directed research in future years.	Complete	Rowden et al. 2012b
ZBD	ZBD2001-10	Additional Research on Biodiversity of Seamounts	1. To determine the macro-invertebrate assemblage composition on Cavalii seamount, and adjacent seamount W1, by photographic transects and epibenthic sled sampling. 2. To determine the distribution of macro-invertebrate assemblages on the seamounts. 3. To compare the macro-invertebrate species diversity of neighbouring seamounts. 4. To evaluate and collect samples from suitable macro-invertebrate species for genetic analysis. 5. To map bathymetry and habitat characteristics of the seamounts. 6. To compare macro-invertebrate assemblage composition of the seamounts with nearby hard bottom low relief (under 100 m) on the slope, if suitable areas can be located.	Complete	Rowden et. al 2004
ZBD	MOF2000–01	Bryozoan thickets off Otago Peninsula	To research the bryozoan thickets off the Otago Penninsula.	Complete	Batson & Probert 2000

Theme	Project code	Project title	Specific objectives	Status	Citation/s
ZBD	ZBD2000 –01	A review of current knowledge describing the biodiversity of the Ross Sea region	1. To review and document existing published and unpublished information describing the biodiversity of the Ross Sea region. 2. To identify and document Ross Sea region marine communities that are under high pressure or likely to come under high pressure from human activities in the near future.	Complete	Bradford-Grieve & Fenwick 2001a, 2001b, 2002 Fenwick & Bradford-Grieve 2002a, 2002b Varian 2005
ZBD	ZBD2000 -02	Exploration and description of the biodiversity, in particular the benthic macrofauna, of the western Ross Sea	1. To utilise sampling opportunities provided by the presence of RV *Tangaroa* in the western Ross Sea in February/March 2001 to make collections of (primarily) benthic organisms as a contribution to the understanding of biodiversity in the region. 2. To identify and document the organisms collected and provide for their proper storage in national collections. 3. To describe the logistic constraints of working in the Ross Sea region, and make recommendations for future research to improve understanding of biodiversity in the Ross Sea.	Complete	Page et al. 2001
ZBD	ZBD2000 -03	The spatial extent and nature of the bryozoan communities at Separation Point, Tasman Bay	1. To assess the present state and extent of bryozoan communities around Separation Point. 2. To characterise the bryozoan communities around Separation Point.	Complete	Grange et al. 2003
ZBD	ZBD2000 -04	Supplementary Research on Biodiversity of Seamounts	1. To determine the biodiversity of seamounts of the southern Kermadec volcanic arc (Rumble V, Rumble 111, Brothers). 2. To describe the distribution of fauna, with an emphasis on mapping the nature and extent, of biodiversity associated with hydrothermal vents. 3. To compare the biodiversity of the three seamounts, and adjacent slope. 4. To collect samples from near the vent sources (if possible, as these are thought to be very localised) to measure chemical and thermal aspects of the environment.	Complete	Rowden et al. 2002, 2003 Clark & O'Driscoll 2003
ZBD	ZBD2000 -06	'The Living Reef: The Ecology of New Zealand's Rocky Reefs'	1. Funding to support the publication of this book.	Complete	Andrew & Francis 2003
ZBD	ZBD2000 -08	A review of current knowledge describing New	1. To review and document existing published and unpublished reports and data describing New Zealand's deepwater benthic biodiversity.	Complete	Key 2002

Theme	Project code	Project title	Specific objectives	Status	Citation/s
		Zealand's Deepwater Benthic Biodiversity	2. To make recommendations on representative communities and potentially impacted communities that could be the subject of directed research.		
ZBD	ZBD2000-09	Antarctic fish taxonomy	1. Ross Sea fishes processing and identification.	Complete	Roberts & Stewart 2001

19.7 APPENDICES FROM CHAPTER 9 FISH AND INVERTEBRATE BYCATCH

Appendix 19.7.1: Bycatch trends for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2013–14). Regression slopes for each species/species group and fishery. Slopes indicating a decline in bycatch over time are highlighted in red, and slopes indicating an increase in bycatch over time are highlighted in green. Species/species groups are ordered alphabetically; blank cells = not estimated; LLL = ling longline fishery; HHL = hoki/hake/ling fishery. NB: These linear regression slopes should be considered only a simple indicator of general changes as relationships may be non-linear; some trends may be strongly influenced by changes in observer recording of species over time. The main purpose of the highlighted cells is to draw attention to species for which closer examination of trends may be warranted. [Continued on next pages]

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
ACS		0.01	0.22			0.06		0.19	Actinostolidae
ADT			0.02						*Aphrodita* spp.
AER			0.02						*Aeneator recens*
AFO			0.04						*Aristaeomorpha foliacea*
AGR								-0.18	*Agrostichthys parkeri*
AIR			-0.01						*Argyripnus iridescens*
ALB		0.01			0.45			0.00	*Thunnus alalunga*
ALL			0.07						*Alcithoe larochei*
ANC					🟩				*Engraulis australis*
ANT		0.00	-0.09	-0.02		0.03	-0.01	0.11	Anthozoa
ANZ		0.02							*Ecionemia novaezelandiae*
API		-0.03	0.04					0.00	*Alertichthys blacki*
APR		0.01	0.06	-0.02		0.02	0.02	0.09	*Apristurus* spp.
ARE			0.02						*Apatopygus recens*
ASR	0.01	0.11	0.08	-0.04		0.01	-0.02	0.16	Asteroid
AST			-0.02	-0.02					Astronesthinae
ATT					0.49				*Arripis trutta*
AWI			0.05						*Alcithoe wilsonae*
BAC						-0.03			*Bathygadus cottoides*
BAM			0.04						*Bathyplotes* spp.
BAR	0.00	-0.01	-0.01		-0.04			-0.11	*Thyrsites atun*
BAS		-0.01	-0.20	-0.10				0.06	*Polyprion americanus*
BAT		-0.01				0.00	-0.01		*Rouleina* spp.
BBE	-0.02	0.03	-0.03			-0.04	0.03	0.03	*Centriscops humerosus*
BCA		0.00						-0.09	*Magnisudis prionosa*
BCD		0.19	-0.01	-0.12				-0.01	*Paranotothenia magellanica*
BCO	-0.03	0.11	-0.01	-0.04	🟥			0.00	*Parapercis colias*
BCR			-0.01					-0.03	*Brotulotaenia crassa*
BDA								-0.01	*Sphyraena novaehollandiae*
BEE			0.00			-0.09	0.11	0.04	*Diastobranchus capensis*
BEL		0.06	-0.01			-0.01		0.13	*Centriscops* spp.

Species	Fishery								Scientific name
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	
BEN					🟩			0.20	*Benthodesmus* spp.
BER			-0.06					0.00	*Typhlonarke* spp.
BES			0.02					0.03	*Benthopecten* spp.
BFE						0.00			*Bathysaurus ferox*
BFI		🟩				0.00		0.00	*Bathophilus filifer*
BFL		0.01							*Rhombosolea retiaria*
BGZ		0.11							*Kathetostoma binigrasella*
BIG		0.01				🟥	🟩	-0.02	*Thunnus obesus*
BJA					🟩	-0.02	0.03	🟥	*Mesobius antipodum*
BKM					🟩			-0.04	*Makaira indica*
BNE		🟥	🟥	🟥	🟥	🟥	🟩	-0.01	*Benthodesmus elongatus*
BNS		-0.07	-0.28	-0.31		-0.19	0.01	-0.10	*Hyperoglyphe antarctica*
BNT	🟥		🟩					-0.01	*Benthodesmus tenuis*
BOA	-0.03	🟩	0.02					-0.01	*Paristiopterus labiosus*
BOC		0.01	0.11			🟥		🟩	*Bolocera* spp.
BOE						-0.20		0.05	*Allocyttus niger*
BOO						0.01			*Keratoisis* spp.
BOT		0.00			🟥			-0.01	Bothidae
BPE			-0.02					-0.01	*Caesioperca lepidoptera*
BPI		🟥			🟥			0.02	*Benthopecten pikei*
BRA		-0.01	🟥	🟩		🟥		0.01	*Dasyatis brevicaudata*
BRC			-0.08	0.01		-0.02		0.01	*Pseudophycis breviuscula*
BRG	🟥					0.08			Brisingida
BRS	-0.01		🟩					-0.01	*Echinorhinus brucus*
BRZ	🟥	🟥	0.04	🟩		🟥	🟩		*Xenocephalus armatus*
BSH	-0.01	-0.06	-0.14	-0.11		-0.11	-0.03	-0.01	*Dalatias licha*
BSK		0.16				-0.02	🟩	-0.16	*Cetorhinus maximus*
BSL						-0.12	0.03	0.11	*Xenodermichthys* spp.
BSP	🟥			-0.02		🟥		0.02	*Taractichthys longipinnis*
BSQ	-0.02		🟩			-0.03		-0.08	*Sepioteuthis australis*
BTA	🟥	🟩	0.09	🟩		🟩	🟩	0.07	*Brochiraja asperula*
BTH	-0.03	0.01	0.05	0.01		0.04	0.01	0.04	*Notoraja* spp.
BTS			-0.02		🟩			0.10	*Brochiraja spinifera*
BWH		🟩	0.10	🟥	🟩	0.00		🟥	*Carcharhinus brachyurus*
BWS		0.03		-0.08				-0.06	*Prionace glauca*
BYD			🟩	🟩		🟩		0.13	*Beryx decadactylus*
BYS		0.00	0.06	0.01		0.08		0.19	*Beryx splendens*
BYX		0.01	-0.22	-0.04		-0.25		-0.10	*Beryx splendens* & *B. decadactylus*
CAL			0.07						*Caenopedina porphyrogigas*
CAM		🟩	0.10	🟩	🟥	🟥		🟩	*Camplyonotus rathbunae*
CAR		0.24	0.12	0.02		-0.02		0.14	*Cephaloscyllium isabellum*
CAS		0.06	0.05				🟩	-0.04	*Coelorinchus aspercephalus*
CAY		🟩	🟩			🟩	0.01		*Caryophyllia* spp.
CBB		0.02	0.02			0.08	🟩		Coral rubble
CBD		0.08			🟩	0.01	0.02	🟩	Coral rubble - dead
CBE		0.04	-0.03					0.03	*Notopogon lilliei*
CBI	🟥	🟩					0.00	-0.02	*Coelorinchus biclinozonalis*
CBO	-0.04	0.00	-0.02			0.00		-0.02	*Coelorinchus bollonsi*
CBX								-0.01	*Cubiceps baxteri*
CCA			🟩					0.00	*Cubiceps caeruleus*
CCO			0.01					0.02	*Coelorinchus cookianus*
CCR		0.00						🟩	*Cetonurus crassiceps*
CCX						🟥		0.07	*Coelorinchus parvifasciatus*
CDL		🟩	🟩		🟩	-0.20	-0.01	0.02	Epigonidae
CDO		0.05	0.05		-0.41			0.18	*Capromimus abbreviatus*
CDX			0.13					-0.01	*Coelorinchus maurofasciatus*
CDY			0.01			🟥			*Cosmasterias dyscrita*

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
CEN			🟩	-0.01		-0.04			Squalidae
CFA			0.03					🟩 0.02	*Coelorinchus fasciatus*
CHA		🟩						0.01	*Chauliodus sloani*
CHC		0.02		🟩			🟩	🟩	*Chaceon bicolor*
CHG				🟩 0.01		🟩 0.02	🟥 0.09	🟥 0.05	*Chimaera lignaria*
CHI			-0.03	-0.05		0.04	0.00	-0.06	*Chimaera* spp.
CHM						🟩	🟩	-0.01	Chiasmodontidae
CHP		🟩		-0.04		0.01	0.04	🟩 -0.01	*Chimaera* sp.
CHQ		0.06					🟩	0.02	Cranchiidae
CHR							0.03	🟩	*Chrysogorgia* spp.
CHX			🟩 -0.04			0.00		🟩 0.01	*Chaunax pictus*
CJA		🟩	0.09					0.12	*Crossaster multispinus*
CMA							🟩	0.02	*Coelorinchus matamua*
CMT		0.02				🟩			Comatulida
CMU						🟩	🟥 0.01	-0.02	*Coryphaenoides murrayi*
COB		🟩				0.01		🟩	Antipatharia
COD			🟩			🟥 0.01	-0.01	🟩 -0.02	Cod
COF	🟥	🟩 0.01	🟩	🟥	🟥	🟥		🟩 0.01	*Flabellum* spp.
COL			0.02			🟩 -0.01		0.13	*Coelorinchus oliverianus*
CON	-0.02	🟥 0.07	0.00	-0.31		🟥 -0.04	🟥 0.00	🟩 0.11	*Conger* spp.
COR			🟩 -0.01	🟩		0.00	0.00	🟩	Stylasteridae
COU		-0.01	-0.01			-0.04	-0.05	0.01	Alcyonacea, Scleractinia, Antipatharia, Stylasteridae
CPA		🟥	0.10					0.06	*Ceramaster patagonicus*
CPD		🟥	🟥			🟥		🟩 -0.03	Centrolophidae
CRA		🟩 -0.02						-0.02	*Jasus edwardsii*
CRB		🟩 -0.16	-0.12	-0.01		-0.03	0.00	0.02	Crab
CRM		0.08				🟥		0.02	*Callyspongia* cf *ramosa*
CRN		🟥 0.02	🟥						Sea lily, stalked crinoid
CRS		🟩	🟩	🟩	🟥	🟥 -0.01	🟥	🟩	*Callyspongia ramosa*
CRU		🟥 -0.04	-0.07					-0.01	Crustacea
CSH	🟥	🟥 0.04	🟩 0.00	🟩 0.01		🟩 -0.04	🟩 -0.01	🟩 0.13	Catshark
CSP		-0.01							*Coelorinchus spathulatus*
CSQ		-0.01	0.02	0.08		🟩 0.11	0.04	0.10	*Centrophorus squamosus*
CST								-0.01	*Caristius* sp.
CSU						0.02			*Coryphaenoides subserrulatus*
CTU		🟥	🟥 -0.01					-0.01	*Cookia sulcata*
CUB			🟩				-0.01	-0.01	*Cubiceps* spp.
CUC		-0.02	-0.07			🟩		🟩 0.00	*Paraulopus nigripinnis*
CVI			0.02			🟩		🟩	*Pycnoplax victoriensis*
CYL			🟩	🟩		🟩 0.14	🟩	🟩 0.15	*Centroscymnus coelolepis*
CYO			🟩	-0.03		0.14		0.11	*Centroscymnus owstoni*
CYP			🟩 0.01	-0.02		0.16	0.13	0.13	*Centroscymnus crepidater*
DAP			🟩 0.15						*Dagnaudus petterdi*
DAS			🟥 0.01	🟩		🟥			*Pteroplatytrygon violacea*
DCO			🟩 0.02			🟩	🟩		*Notophycis marginata*
DCS			-0.03	0.00	🟩	-0.02		🟥 -0.04	*Bythaelurus dawsoni*
DDI			0.06			🟥 0.02	0.01		*Desmophyllum dianthus*
DEA	0.00		🟩					🟩 -0.12	*Trachipterus trachypterus*
DEQ			🟩			-0.02		-0.02	*Deania quadrispinosum*
DHO			🟩 0.01	🟩		0.02		🟩 0.01	*Dermechinus horridus*
DIR			0.07						*Diacanthurus rubricatus*
DIS		🟩	🟥			🟩 0.00	🟥	🟩	*Diretmus argenteus*
DMG	🟥	🟩	0.11					0.09	*Dipsacaster magnificus*
DPO								-0.02	*Desmodema polystictum*
DSK		0.01	-0.13	0.00		🟥 0.01	🟥 -0.04	🟩 0.10	*Amblyraja hyperborea*
DSP	-0.02	0.03						🟩	*Congiopodus coriaceus*
DSS						🟥 0.00		-0.01	*Bathylagus* spp.

Species	Fishery								Scientific name
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	
DWE			-0.04	-0.04		-0.04	-0.01	0.14	Whelks
DWO			🟥			🟩		0.18	*Graneledone spp.*
ECH		🟩	-0.05	-0.01		-0.01	0.00	-0.04	Echinodermata
ECN			0.01	-0.01		-0.01	-0.02	0.01	Echinoid
EEL			-0.16	-0.01	🟩	0.00	-0.01	-0.09	Eels
EEX		0.03			🟩				*Enypniastes eximia*
EGA			0.02	🟩					*Euciroa galatheae*
EGR					0.20			🟥	*Myliobatis tenuicaudatus*
ELE								🟩	*Callorhinchus milii*
ELT								🟩	*Electrona spp.*
EMA	0.00	0.03	🟩		-0.04	🟥	🟥	-0.20	*Scomber australasicus*
EMO			-0.02			-0.01		0.01	*Etmopterus molleri*
EPD			🟩			🟩		0.02	*Epigonus denticulatus*
EPL		0.03	0.01			-0.11	-0.03	0.21	*Epigonus lenimen*
EPO			-0.02			🟩			*Melanostigma gelatinosum*
EPR	🟥	🟩	0.02	🟩		0.06	🟩	0.13	*Epigonus robustus*
ERA		0.02	-0.01	🟩	0.24	🟥	🟩	0.04	*Torpedo fairchildi*
ERO		🟥		🟥		0.04	🟩	🟥	*Enallopsammia rostrata*
ETB	-0.02	0.05	0.06	0.03		0.08	0.26	0.24	*Etmopterus baxteri*
ETL		0.03	0.08	-0.18		-0.14	0.04	0.06	*Etmopterus lucifer*
ETM		-0.03	-0.04	-0.09		-0.11	0.01	-0.24	*Etmopterus* sp.
ETP						-0.04	-0.01	-0.01	*Etmopterus pusillus*
EUC		🟩	0.05			-0.02		0.13	*Euclichthys polynemus*
EZE		0.07	0.07						*Enteroctopus zealandicus*
FAN		🟩						-0.01	*Pterycombus petersii*
FHD		0.03	0.03					0.08	*Hoplichthys haswelli*
FLA		0.13	-0.03					-0.03	Flatfish
FLO		0.01			🟩	🟥		-0.02	Flounder
FMA		0.01	0.18			🟥		0.19	*Fusitriton magellanicus*
FOR								-0.02	*Forsterygion* spp.
FRO	0.00	0.08	-0.05		-0.01	-0.03		-0.10	*Lepidopus caudatus*
FRS						-0.05		-0.02	*Chlamydoselachus anguineus*
FRX			🟩	🟩				-0.01	Trichiuridae
FTU		0.01	🟩						*Auxis thazard*
GAO			🟩			0.00	🟩		*Gadomus aoteanus*
GAS		🟩	0.19					0.05	Gastropoda
GAT		🟩	0.03		🟥	🟥			*Gastroptychus* spp.
GDU	🟩		0.02			0.16	0.09	🟩	*Goniocorella dumosa*
GFL		0.14	🟩						*Rhombosolea tapiri*
GIZ	0.00	0.07	-0.08	-0.01	0.16	-0.03		0.00	*Kathetostoma giganteum*
GLS	0.01					0.03	🟥	0.12	Hexactinellida
GMC		0.04	0.23					0.02	*Leptomithrax garricki*
GMU		-0.01						🟩	*Mugil cephalus*
GOB			🟩			🟩	-0.01		*Mitsukurina owstoni*
GON		0.25	🟩					0.08	*Gonorynchus forsteri* & *G. greyi*
GOR						🟩	🟩	0.05	*Gorgonocephalus* spp.
GOU			0.02			0.03	🟩	🟩	*Goniocidaris umbraculum*
GPA			0.07						*Goniocidaris parasol*
GRC		🟩	🟩	🟩		0.01	0.04	-0.01	*Tripterophycis gilchristi*
GRM	🟥	🟩	🟩	🟥	🟥	🟥	0.02	0.04	*Gracilechinus multidentatus*
GSA	🟩	🟩	🟩	🟩		🟩	🟩	-0.01	*Hoplostethus gigas*
GSC		0.38	0.13	-0.03		0.00		0.09	*Jacquinotia edwardsii*
GSH	-0.09	0.10	0.03	-0.28	0.25	-0.18	-0.18	-0.09	*Hydrolagus novaezealandiae*
GSP	0.13	0.14	0.14	0.07		0.09	0.17	0.16	*Hydrolagus bemisi*
GSQ	0.00	0.00	🟥	🟩		0.01		0.02	*Architeuthis* spp.
GUR	🟥	-0.01	0.00	🟩	-0.07	🟥	🟥	0.03	*Chelidonichthys kumu*
GVO			0.09					0.01	*Provocator mirabilis*

Species	Fishery								
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
HAG		🟩	-0.08	-0.03	🟩			0.18	*Eptatretus cirrhatus*
HAK	-0.05	0.06	-0.06	-0.06		-0.03	-0.02		*Merluccius australis*
HAL	🟥	🟩	🟩	🟩		🟥		0.01	*Halosauropsis macrochir*
HAP		0.04	-0.05	-0.18				-0.03	*Polyprion oxygeneios*
HAT				🟩				🟩	Sternoptychidae
HCO		0.02	0.01	0.09		-0.01		-0.01	*Bassanago hirsutus*
HEC			0.02			🟩			*Henricia compacta*
HEP			-0.04					0.06	*Heptranchias perlo*
HEX		0.05	-0.06	0.01		🟩	🟩	0.15	*Hexanchus griseus*
HGB			🟩	🟩		0.01		0.00	*Hydrolagus* sp. D
HIS	🟥	🟩	0.02	🟩	🟥	🟥	🟩		*Histocidaris* spp.
HJO			0.00			0.05	0.12	0.01	*Halargyreus johnsonii*
HMT			0.21					0.07	Hormathiidae
HOK	-0.12	0.04	-0.06	0.01	-0.19	-0.09	0.13	🟥	*Macruronus novaezelandiae*
HOL			🟩					0.01	*Holtbyrnia* sp.
HOR		🟥	🟩			🟩	🟩	-0.01	*Atrina zelandica*
HPB		-0.08	-0.20	-0.18				-0.16	*Polyprion oxygeneios & P americanus*
HSI	🟩	🟩	0.13			🟩		🟩	*Haliporoides sibogae*
HTH		-0.02	0.04	🟩		0.07	0.02	0.05	Holothurian unidentified
HTR			0.07				🟩	0.08	*Hippasteria phrygiana*
HYA	0.03	0.03	0.07			0.02		0.31	*Hyalascus* sp.
HYB				0.00					*Hydrolagus homonycteris*
HYD						0.00	0.01	-0.01	*Hydrolagus* sp.
HYM			0.07				🟩		*Hymenocephalus* spp.
HYP	🟩	🟩		🟩	🟩	0.00		🟩	*Hydrolagus trolli*
IBR						0.05	0.02	0.00	*Isistius brasiliensis*
ISI		🟥	🟥		🟩	🟩	0.01		Isididae
JAV	0.06	0.20	-0.01	-0.03	0.18	0.00	0.08	0.04	*Lepidorhynchus denticulatus*
JDO		🟥	🟥		0.01	🟥		-0.02	*Zeus faber*
JFI		0.00	-0.06		0.04	0.01	0.01	0.05	Jellyfish
JGU		-0.01	-0.03		🟩			0.00	*Pterygotrigla picta*
JMA	0.00	-0.16	-0.14			-0.03		-0.25	*Trachurus declivis, T. murphyi, T. novaezelandiae*
KIC			-0.02			0.04	-0.02	0.04	*Lithodes murrayi, Neolithodes brodiei*
KIN			-0.02		0.12			0.01	*Seriola lalandi*
KWH		🟩	0.01		🟩		🟩	0.01	*Austrofucus glans*
LAE			0.00			-0.03	-0.01		*Laemonema* spp.
LAG	🟥	🟩	0.09		🟩	🟥			*Laetmogone* spp.
LAN		0.14	0.00		🟥	0.00	0.01	0.07	Myctophidae
LCH	0.02		0.01			-0.02	0.03	0.03	*Harriotta raleighana*
LDO	-0.01	0.06	-0.04			-0.05	0.00	0.00	*Cyttus traversi*
LEA		-0.01	🟩		-0.19			🟩	*Meuschenia scaber*
LEG	🟥	🟩	🟥		🟥	-0.06	0.04	0.00	*Lepidion schmidti & Lepidion inosimae*
LHE		🟩	🟩					-0.02	*Lampanyctodes hectoris*
LHO			0.08					0.02	*Lipkius holthuisi*
LIN	-0.04	0.05	-0.12			-0.08	-0.06	🟩	*Genypterus blacodes*
LLC		0.07	0.02					0.02	*Leptomithrax longipes*
LMI			0.03						*Leptomithrax* spp.
LMU						0.01	🟥	0.03	*Lithodes murrayi*
LNV		🟩	🟩			🟥		0.05	*Lithosoma novaezelandiae*
LPI		🟥	🟩				0.02	🟩	*Lepidion inosimae*
LPS						0.02	-0.01		*Lepidion schmidti*
LSK	🟩	0.01	0.08	🟩	🟩	-0.02		0.10	*Arhynchobatis asperrimus*
LSO	🟥	-0.02	0.04			🟩		0.00	*Pelotretis flavilatus*
LUC			-0.04			-0.02	🟩	-0.02	*Luciosudus* sp.
MAK	0.03	0.03	-0.01	0.00	0.36	-0.05		-0.06	*Isurus oxyrinchus*
MAN	-0.04	-0.02				0.02		-0.07	*Neoachiropsetta milfordi*
MCA	🟥		🟥			0.11	0.26	0.00	*Macrourus carinatus*

Species	Fishery								Scientific name
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	
MDO		0.02	0.02			-0.03		0.03	*Zenopsis nebulosa*
MIC	-0.02					🟩	🟩		*Microstoma microstoma*
MIQ	-0.05		-0.07	🟩		-0.09	-0.01	0.07	*Onykia ingens*
MNI		🟥	0.13			🟥		🟥	*Munida* spp.
MOC						0.06	0.02		*Madrepora oculata*
MOD	🟥	🟩	0.00		🟥	0.15	0.16	0.16	Moridae
MOK		-0.02	-0.01			-0.02		-0.09	*Latridopsis ciliaris*
MOL			-0.02					-0.02	Molluscs
MOO	-0.14	0.01	🟩					-0.18	*Lampris guttatus*
MOR						0.00	0.00	-0.01	Muraenidae
MRL						🟥		0.00	Muraenolepididae
MRQ								0.04	*Onykia robsoni*
MSL		🟩	0.05						*Mediaster sladeni*
MST		🟩				0.03		0.02	*Melanostomiidae*
MUR						-0.02		🟩	*Muraenolepis marmoratus*
MUU		0.00							Mullet
NCA		0.03	🟥	🟩	🟥			🟥	*Nectocarcinus antarcticus*
NCB		0.47						0.01	*Nectocarcinus bennetti*
NEB						0.07	🟩	0.01	*Neolithodes brodiei*
NEX		🟥		🟥				0.00	Nemichthyidae
NMP		0.12	-0.06	🟩	0.20			-0.08	*Nemadactylus macropterus*
NOC		🟩						0.01	*Notacanthus chemnitzi*
NOR							0.01		*Normichthys yahganorum*
NOT		-0.06	-0.03	-0.26		0.00			Nototheniidae
NSD		0.01	0.00	-0.02		0.00		0.21	*Squalus griffini*
NTO		0.01	🟩						*Notomithrax* spp.
NTU		🟩				🟥	🟥	-0.02	*Thunnus thynnus*
NUD			0.04	🟩					Nudibranchia
OAR						🟥		-0.08	*Regalecus glesne*
OCO			0.02	🟩		🟩			*Octopus* spp.
OCP			0.01					-0.02	Octopod
OCT	0.00	0.05	-0.04			0.02	-0.01	-0.05	*Pinnoctopus cordiformis*
ODO	🟥	🟩	0.01					-0.01	*Odontaspis ferox*
OEO		🟩	🟥			-0.13		-0.09	*P. maculatus, A. niger, & N. rhomboidalis*
OFH			-0.05			0.01		0.00	*Ruvettus pretiosus*
OLY			0.02					🟩	*Ophiomusium lymani*
ONG	-0.03	0.15	0.10	0.00		0.07	-0.01	0.06	Porifera
OPA	-0.02	0.15	0.06			🟥	🟩	0.02	*Hemerocoetes* spp.
OPE		0.17	-0.02		0.20	-0.01		-0.04	*Lepidoperca aurantia*
OPH			🟩			-0.02		🟩	Ophiuroid
OPI			0.11				🟩	0.26	*Opisthoteuthis* spp.
OPL		0.01				🟩	🟩		Opheliidae
ORH		🟥	-0.02				-0.01	-0.08	*Hoplostethus atlanticus*
OSE			🟩					0.00	*Ophisurus serpens*
OSK	🟩		0.20			0.04		0.18	Rajidae
OSP							0.01	0.00	*Crassostrea gigas*
PAB			🟩			0.03	0.07		*Paragorgia arborea*
PAD		-0.30	🟩					🟩	*Ovalipes catharus*
PAG			0.04						Paguroidea
PAH	0.23		🟥					0.00	*Lampris immaculatus*
PAL		🟩				🟥		-0.01	*Paralepididae*
PAM			0.05					🟩	*Pannychia moseleyi*
PAO			0.02					0.01	*Pillsburiaster aoteanus*
PCH			0.05						*Penion chathamensis*
PCO		🟩	-0.04			🟥		🟩	*Auchenoceros punctatus*
PDG		0.06	0.00			-0.06		0.05	*Oxynotus bruniensis*
PDO	🟥	🟩	0.00					🟩	*Paphies donacina*

Species	Fishery								
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
PDS					🟩			0.02	*Paradiplospinus gracilis*
PED			-0.03					🟩	*Aristaeopsis edwardsiana*
PFL			0.02	🟩				🟩	*Pseudechinus flemingi*
PHO		0.03		🟩		-0.01	🟩	0.02	*Phosichthys argenteus*
PHW		0.02	🟩					🟩	*Psammocinia cf hawere*
PIG	-0.08	0.20	0.03					0.05	*Congiopodus leucopaecilus*
PIL			🟥		0.00				*Sardinops sagax*
PIN			🟩			-0.01		0.01	*Idiolophorhynchus andriashevi*
PKN			🟩					0.08	*Plutonaster knoxi*
PLS		0.02	0.00	0.05		0.04	0.01	0.04	*Proscymnodon plunketi*
PLT			0.01					0.02	*Plutonaster spp.*
PLY			0.02						*Polycheles* spp.
PLZ	🟩		-0.05						*Pleuroscopus pseudodorsalis*
PMO			0.03		🟩			0.02	*Pseudostichopus mollis*
PMU	🟥	🟥	0.08					🟥	*Paramaretia peloria*
PNE	🟩	🟩	0.09		🟥			🟥	*Proserpister neozelanicus*
PNN			0.02						*Pennatula spp.*
PNO			0.03						*Pteropeltarion novaezelandiae*
POM	0.01								Bramidae
POP			🟩		0.02			🟩	*Allomycterus jaculiferus*
POR	-0.02	-0.04					🟥	-0.23	*Nemadactylus douglasii*
POS	0.02	0.01	0.02	-0.11				-0.08	*Lamna nasus*
PRA		🟩	0.08	🟩		🟩		0.00	Prawn
PRK			0.19			🟥	🟩		*Ibacus alticrenatus*
PRU			0.04					0.01	*Pseudechinaster rubens*
PSE			0.00				-0.01	🟩	*Pseudechinus* spp.
PSI			0.22			🟩		0.14	*Psilaster acuminatus*
PSK		0.01	0.06	0.00		0.07	-0.01	0.15	*Bathyraja shuntovi*
PSL				🟩		-0.01	0.01		*Paralomis dosleini*
PSO								-0.02	*Psolus* spp.
PSP		🟩						0.01	*Psenes pellucidus*
PSQ						0.02	🟩	0.09	*Pholidoteuthis massyae*
PSY	🟥	🟩	-0.04	🟩	🟥	0.02	0.00	-0.02	*Psychrolutes microporos*
PTO			🟥	-0.02	🟥		0.00	🟩	*Dissostichus eleginoides*
PZE	🟩	🟥		🟥	🟩	🟥		0.01	*Paralomis zealandica*
QSC		0.15	🟩		🟩			🟩	*Psychrochlamys delicatula subantactica*
RAG		🟩	🟥			0.03	0.01	-0.09	*Pseudoicichthys australis*
RAT	-0.07	0.07	-0.02	-0.12	0.32	-0.03	0.08	0.02	Macrouridae
RAY			-0.06				-0.02	0.02	Torpedinidae, Dasyatidae, Myliobatidae, Mobulidae
RBM	0.08	-0.11	🟥	-0.24	0.09	-0.01		-0.05	*Brama brama*
RBT	0.00	0.01	0.01		-0.05			0.05	*Emmelichthys nitidus*
RBY		0.01	-0.08			🟩		-0.18	*Plagiogeneion rubiginosum*
RCH		🟩	🟥			0.05	🟥	0.05	*Rhinochimaera pacifica*
RCK			0.00					🟩	Acanthoclinidae
RCO	0.04	0.05	-0.07	-0.24	0.38			-0.07	*Pseudophycis bachus*
RDO		0.11	-0.01	🟩	0.07	🟩	🟩	0.06	*Cyttopsis roseus*
RHY			0.15		0.02	0.06		0.18	*Paratrachichthys trailli*
RIB		0.04	-0.20	-0.27		-0.06	-0.03	0.00	*Mora moro*
RIS						🟩		0.05	*Bathyraja richardsoni*
RMU	🟩	🟩	🟩	🟩	🟥	🟩		-0.02	*Upeneichthys lineatus*
ROC		0.01	-0.02		🟥	0.02	0.02	0.00	*Lotella rhacinus*
RPE		🟥	-0.03	🟩	🟥				Red perch
RPI		🟩				🟥			*Bodianus vulpinus*
RSC			🟥			0.00	🟥		*Scorpaena papillosa*
RSK	0.02	0.23	0.14	0.11		0.01		0.11	*Zearaja nasuta*
RSN			-0.01					-0.02	*Centroberyx affinis*
RSO		-0.11	-0.09	-0.01	0.11			-0.01	*Rexea solandri*

Species								Fishery	
	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
RSQ		0.02				-0.07		0.00	*Ommastrephes bartrami*
RUD	🟩		-0.07			-0.06	-0.03	-0.02	*Centrolophus niger*
SNA		-0.05	-0.02		0.07	-0.04		-0.09	*Pagrus auratus*
SAF	🟥		🟩			0.01		🟩	*Synaphobranchus affinis*
SAI			🟥	🟩		🟥	🟩	0.01	*Istiophorus platypterus*
SAR		🟩	🟩			0.00	🟩	🟩	*Squilla armata*
SAW		🟩						-0.02	*Serrivomer* spp.
SBI	0.02	🟩				-0.11	-0.02	-0.03	*Alepocephalus australis*
SBK			-0.04		🟩	-0.03		0.05	*Notacanthus sexspinis*
SBO	-0.04	-0.01	0.03	🟩	🟥	0.00		0.06	*Pseudopentaceros richardsoni*
SBR		0.02	-0.06			-0.03	0.01	0.08	*Pseudophycis barbata*
SBW		0.13	-0.01			🟩	0.02	0.20	*Micromesistius australis*
SCA		0.02		🟩		🟩		🟩	*Pecten novaezelandiae*
SCD		0.11						0.01	*Notothenia microlepidota*
SCG			-0.06		0.20			0.01	*Lepidotrigla brachyoptera*
SCH		0.11	-0.05	-0.23	0.07	0.03		0.04	*Galeorhinus galeus*
SCI			🟩					0.10	*Metanephrops challengeri*
SCM		0.00	0.00	-0.04	🟩	0.03	0.01	0.08	*Centroscymnus macracanthus*
SCO			0.02	-0.02		0.01		0.13	*Bassanago bulbiceps*
SDE				🟩			0.02	-0.02	*Cryptopsaras couesii*
SDF			0.04					0.02	*Azygopus pinnifasciatus*
SDL		🟩	🟩			0.01		🟩	*Scorpaena cardilis*
SDM		🟩	0.14					0.01	*Sympagurus dimorphus*
SDO		0.38	-0.02	🟩	-0.18	🟥	🟥	0.12	*Cyttus novaezealandiae*
SDR						-0.01		0.01	*Solegnathus spinosissimus*
SEE			-0.03	0.04				0.07	*Gnathophis habetus*
SER			0.02						*Sergestes* spp.
SEV		0.04	0.03	-0.02		🟩	🟩	0.11	*Notorynchus cepedianus*
SFL	🟥	0.05	🟥	🟥	🟥	🟥	🟥	🟥	*Rhombosolea plebeia*
SHA	0.00	0.06	-0.12	-0.11		-0.15	-0.12	-0.04	Shark
SHE						-0.01		-0.04	*Scymnodalatias sherwoodi*
SHL			-0.06			🟥			*Scyllarus* sp.
SHR			0.00						*Aplysiomorpha*
SIA						0.12	0.06	🟩	Scleractinia
SKA	-0.05	-0.08	-0.38	-0.37	0.00	-0.08	-0.04	-0.33	Rajidae & Arhynchobatidae
SKJ		0.01			0.12				*Katsuwonus pelamis*
SLB			🟩			🟥	🟩	0.03	*Scymnodalatias albicauda*
SLC		🟩				-0.02		🟩	*Slosarczykovia circumantarctica*
SLG		🟩	-0.04			0.00			*Scutus breviculus*
SLK		🟩				0.01	0.15	0.14	Alepocephalidae
SLR			-0.04			0.00			*Optivus elongatus*
SLS		0.00			🟩	🟥		🟥	*Peltorhamphus tenuis*
SMA		0.01	🟥	🟩		🟩	🟩		*Stigmatophora macropterygia*
SMC			0.02			-0.04	0.06	-0.04	*Lepidion microcephalus*
SMI		0.02				0.00		0.07	*Somniosus microcephalus*
SMK		0.02	0.23			🟩		🟩	*Teratomaia richardsoni*
SMO		0.05		🟩		🟥	🟩	🟩	*Sclerasterias mollis*
SMT			0.04						*Spatangus mathesoni*
SND		0.03	-0.11	-0.01		0.03	0.06	-0.01	*Deania calcea*
SNE	🟩					🟩		0.02	*Simenchelys parasitica*
SNI		-0.03	-0.01			🟥		0.01	*Macroramphosus scolopax*
SNO			🟩	🟩		0.01		0.02	*Sio nordenskjoldii*
SNR	🟩	🟩	🟩	0.01	🟥	-0.03	0.03	0.00	*Deania histricosa*
SOL		🟩	0.02		🟥	🟥		🟩	Sole
SOM						0.02			*Somniosus rostratus*
SOP	0.02	🟥	🟥			-0.01	🟥	-0.03	*Somniosus pacificus*
SOR						-0.09		-0.01	*Neocyttus rhomboidalis*

Species	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
SOT			0.02					0.03	*Solaster torulatus*
SPD	0.02	🟩 0.03	🟥 0.09	-0.18	🟥 0.02	-0.18	-0.03	🟥 -0.01	*Squalus acanthias*
SPE		0.06	-0.01	-0.25	🟩 0.20	-0.09		0.00	*Helicolenus* spp.
SPF								-0.01	*Pseudolabrus miles*
SPI	0.01	-0.06	🟩 -0.13	0.00		-0.03	-0.01	🟩 -0.08	Spider crab
SPK			-0.01					🟥 -0.01	*Macrorhamphosodes uradoi*
SPL				🟩		🟩	🟩	🟩 0.00	*Scopelosaurus* sp.
SPO		🟥 0.03	-0.09	-0.04				-0.08	*Mustelus lenticulatus*
SPP	🟥		🟩		🟥	🟥	🟥	0.00	*Callanthias* spp.
SPR	🟩		🟥		🟩	🟩	🟥	🟩 0.00	*Sprattus antipodum, S. muelleri*
SPT	🟩		0.19					🟩 0.01	*Spatangus multispinus*
SPZ			🟩 -0.01			🟥		🟩 -0.05	*Genyagnus monopterygius*
SQA				0.01		🟩 0.02	0.03	🟩 0.03	*Squalus spp.*
SQI		🟥 -0.03	🟥					🟩	*Pristilepis oligolepis*
SQU	-0.01	🟩	0.03	🟩	-0.08	-0.11	-0.03	🟩 0.01	*Nototodarus sloanii & N. gouldi*
SQX	🟩 0.02	🟩	🟩 -0.06			🟩 0.00	🟥 -0.03	🟩 0.11	Squid
SRB	0.02		🟩	🟩	🟥	🟥	🟩	🟩 0.02	*Brama australis*
SRH			0.02			🟥 -0.01	🟩	0.13	*Hoplostethus mediterraneus*
SRI						🟥 0.01		0.03	*Scymnodon ringens*
SSC		-0.13	-0.04					0.01	*Leptomithrax australis*
SSH		🟥 0.03	🟩	-0.03				🟥 0.17	*Gollum attenuatus*
SSI	🟩 0.00	🟩 0.19	0.04		0.24	0.01	-0.04	🟩 0.04	*Argentina elongata*
SSK	0.00	-0.01	0.03	-0.15		-0.01	0.06	🟩 0.04	*Dipturus innominatus*
SSM		🟩			🟩	0.00	0.06	-0.01	*Alepocephalus antipodianus*
SSO	🟥	🟥			🟩	-0.19		🟥 0.00	*Pseudocyttus maculatus*
SSP								-0.02	*Pecten novaezelandiae*
STG		🟥 -0.01	🟩 0.03		🟩	🟩		-0.11	Stargazer
STM			🟥					0.02	*Tetrapturus audax*
STN		0.03	-0.02			🟩	🟩	0.06	*Thunnus maccoyii*
STO	🟩	🟩	🟥			🟥		0.01	*Stomias* spp.
STR		0.01	-0.05		🟩	🟥		-0.01	Stingray
STU	-0.02	-0.06			0.52	🟥		-0.10	*Allothunnus fallai*
SUH								🟩 -0.01	*Schedophilus huttoni*
SUN		-0.01	0.01		0.20	🟥 0.01		🟩 0.01	*Mola mola*
SUR			🟩 -0.06			🟩 0.00	🟩	🟩 -0.03	*Evechinus chloroticus*
SVA			🟩			0.05	0.08	🟩	*Solenosmilia variabilis*
SWA	0.06	0.05	-0.18		0.07	-0.03		🟩 -0.05	*Seriolella punctata*
SWO			🟩			-0.04		0.00	*Xiphias gladius*
SWR		🟥		🟩		-0.02		🟥 0.00	*Coris sandageri*
SYD								🟩 0.01	*Systellaspis debilis*
SYN			🟩 0.00			-0.03		0.01	Synaphobranchidae
TAM		🟩	0.06	🟩		🟩 0.04	0.08	🟩 0.23	*Echinothuriidae & Phormosomatidae*
TAY		🟩	🟩 0.10					🟩 0.04	*Typhlorke aysoni*
TDQ	🟥			🟩		🟩		🟩 0.04	*Taningia danae*
TFA		🟩	0.19					🟩	*Trichopeltarion fantasticum*
THR		-0.09			-0.02			-0.12	*Alopias vulpinus*
TLD					🟩			0.03	*Tetilla leptoderma*
TLO			0.01			🟥			*Telesto* spp.
TOA		0.10	-0.03	-0.03		0.05	0.00	0.08	*Neophrynichthys* sp.
TOD		0.05	0.03			🟩		🟩 0.06	*Neophrynichthys latus*
TOP	-0.02		🟩 0.03			0.00		0.13	*Ambophthalmos angustus*
TOR		🟩 0.06						0.14	*Thunnus orientalis*
TRA								🟩 -0.01	Trachichthyidae
TRE		🟩	🟩		0.06				*Pseudocaranx georgianus*
TRS						🟩 -0.02	🟩		*Trachyscorpia eschmeyeri*
TRU		0.00		0.00			🟥	-0.02	*Latris lineata*
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name

	Fishery								
Species	SBW	SQU	SCI	LLL	JMA	ORH	OEO	HHL	Scientific name
TSQ						0.03		0.13	*Todarodes filippovae*
TTA			0.03			🟩		🟩	*Typhlonarke tarakea*
TUR		0.02			🟥			🟥	*Colistium nudipinnis*
TVI								0.02	*Trachonurus villosus*
UFISH	-0.07	-0.22	-0.40	0.02		-0.31	-0.08	-0.37	Unidentified fish
URP		0.02	0.02			🟩		🟩	*Uroptychus* spp.
VCO	🟥	🟩	🟩			0.04	0.09	0.00	*Antimora rostrata*
VIT						🟥	-0.01		*Vitjazmaia latidactyla*
VNI		🟩			🟩	🟩		0.02	*Lucigadus nigromaculatus*
VOL			0.00					0.01	Volutidae
VSQ						0.03		0.19	*Histioteuthis* spp.
WAR		-0.01	🟩		-0.13	0.00	🟩	-0.18	*Seriolella brama*
WHE	🟥	🟩	0.01				🟩	0.02	Witch
WHR						-0.05		-0.05	*Trachyrincus longirostris*
WHX			0.01			0.07		0.18	*Trachyrincus aphyodes*
WIT	-0.01	0.11	0.10			0.04		0.12	*Arnoglossus scapha*
WOE		🟥				-0.05	-0.04		*Allocyttus verrucosus*
WPS		0.05				0.01		0.01	*Carcharodon carcharias*
WRA			🟩					0.03	*Dasyatis thetidis*
WSE			🟩						Labridae
WSQ	-0.02	0.09	0.02			0.03	0.15	-0.03	*Onykia* spp.
WWA	-0.04	0.05	-0.05	-0.04		0.00	0.03	0.06	*Seriolella caerulea*
YBF			🟩	🟩				0.02	*Rhombosolea leporina*
YBO			0.14	🟩				0.12	*Pentaceros decacanthus*
YCO		0.08	🟩	🟩				🟩	*Parapercis gilliesi*
YEM		-0.02	🟩	🟩				🟩	*Aldrichetta forsteri*
YFN		0.00	🟩	🟩				0.00	*Thunnus albacares*
YSG			0.01	🟩				🟩	*Pterygotrigla pauli*
YSP			0.02	🟩				🟩	*Yaldwynopsis spinima*
ZAS			🟩	🟩		0.03		🟩	*Zameus squamulosus*
ZOR			0.14	🟩				0.09	*Zoroaster* spp.

Appendix 19.7.2: BYCATCH: Total annual bycatch by fishery area for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2016–17). Where data have not yet been updated for recent years for a fishery, figures from the last available year have been assumed, in order for annual totals to be calculated. LLL = ling longline fishery; HHL = hoki/hake/ling fishery. [Continued on next pages]

									AUCKLAND ISLANDS
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	12	642	24	66	7	0	0	0	750
1992	1 318	300	24	60	0	0	0	0	1 701
1993	1 009	422	24	38	0	24	48	0	1 564
1994	882	497	0	158	0	9	585	0	2 132
1995	866	189	21	127	0	64	2 079	0	3 346
1996	853	5 445	36	166	57	17	996	0	7 570
1997	1 954	1 641	51	198	17	11	503	0	4 374
1998	893	209	15	550	0	18	554	0	2 240
1999	1 040	276	21	255	0	33	1 042	0	2 667
2000	1 084	416	65	130	0	61	290	0	2 046
2001	751	353	64	304	0	12	277	0	1 762
2002	1 191	362	2	546	0	15	62	0	2 177
2003	1 225	1 326	0	501	0	23	20	0	3 095
2004	1 063	2 877	27	262	0	10	47	0	4 286
2005	1 035	2 010	1	187	0	0	16	0	3 249
2006	576	3 283	0	36	0	3	0	0	3 898

AUCKLAND ISLANDS									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2007	745	1 105	0	39	0	3	0	0	1 892
2008	597	1 349	0	130	0	13	20	0	2 109
2009	876	1 914	0	109	0	18	20	0	2 937
2010	631	1 790	24	29	0	11	53	0	2 538
2011	705	2 574	111	147	0	40	17	0	3 594
2012	604	1 675	10	43	0	7	3	0	2 343
2013	586	1 688	10	56	0	0	0	0	2 340
2014	609	751	10	485	0	54	0	0	1 909
2015	329	713	10	514	0	13	50	0	1 629
2016	519	989	10	207	0	13	50	0	1 788
2017	519	989	10	342	0	13	50	0	1 923

CHATHAM RISE									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	344	1 919	1 831	8 349	1 111	1 430	15 909	0	30 893
1992	4 343	4 000	1 831	8 993	72	356	15 677	0	35 272
1993	5 603	1 781	1 831	6 467	549	592	6 877	0	23 700
1994	2 644	5 273	1 795	3 974	1 019	499	15 043	0	30 247
1995	4 422	2 954	1 870	7 551	1 855	145	12 057	0	30 854
1996	3 865	5 328	1 970	11 340	272	781	8 664	0	32 221
1997	1 191	4 350	1 881	15 042	1 472	444	7 808	0	32 188
1998	585	3 537	2 392	16 757	1 515	1 137	5 685	0	31 609
1999	777	4 819	1 020	17 707	1 944	1 744	2 913	0	30 925
2000	1 548	3 798	1 277	19 485	2 603	414	2 763	0	31 887
2001	1 987	10 277	1 240	16 745	106	236	6 016	0	36 606
2002	1 798	8 014	2 781	16 872	698	153	2 917	0	33 232
2003	1 581	6 876	1 430	20 002	1 549	111	3 265	0	34 814
2004	1 006	2 096	1 239	19 822	168	348	3 873	0	28 552
2005	2 658	4 642	1 518	13 797	73	441	3 134	0	26 263
2006	1 111	6 886	1 576	12 470	5 771	203	2 870	0	30 887
2007	2 458	4 863	849	9 970	1 476	515	2 944	0	23 075
2008	1 635	2 241	1 364	11 551	1 859	199	2 908	0	21 757
2009	1 209	511	1 366	8 766	685	312	3 204	0	16 053
2010	1 572	984	1 587	14 552	677	560	2 854	0	22 786
2011	1 150	2 282	1 751	8 130	528	134	458	0	14 433
2012	1 538	979	1 041	8 772	1 139	316	615	0	14 400
2013	1 953	221	1 041	9 879	1 885	260	314	0	15 553
2014	2 737	72	1 041	15 554	2 378	141	604	0	22 528
2015	2 261	778	1 041	24 659	2 378	340	553	0	32 010
2016	1 300	2 133	1 041	17 227	2 378	340	553	0	24 972
2017	1 300	2 133	1 041	18 530	2 378	340	553	0	26 275

COOK STRAIT									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	0	9	1 501	4	9	1 059	0	2 582
1992	0	0	9	1 068	9	7	1 089	0	2 182
1993	0	58	9	1 090	0	2	469	0	1 628
1994	0	0	5	2 621	0	12	878	0	3 517
1995	33	1	9	2 400	0	0	1 060	0	3 504
1996	0	19	8	3 274	0	3	373	0	3 677

									COOK STRAIT
									Fishery
FY	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1997	0	0	3	4 525	0	8	396	0	4 933
1998	0	1	20	2 249	113	9	221	0	2 612
1999	2	90	56	2 035	423	3	181	0	2 791
2000	2	2	67	3 244	155	12	132	0	3 614
2001	0	47	221	1 561	312	12	72	0	2 225
2002	53	0	40	882	26	10	0	0	1 012
2003	7	2	22	3 197	0	6	0	0	3 234
2004	0	0	54	1 396	0	1	0	0	1 452
2005	36	0	103	1 825	0	2	0	0	1 965
2006	0	0	128	507	0	4	0	0	639
2007	1	2	126	1 070	0	1	0	0	1 200
2008	4	0	201	790	0	0	30	0	1 025
2009	0	0	29	660	0	1	0	0	690
2010	0	0	21	700	0	0	0	0	721
2011	0	0	28	325	0	0	20	0	373
2012	0	0	11	532	0	0	0	0	543
2013	0	0	11	645	0	0	0	0	656
2014	0	0	11	706	0	0	0	0	717
2015	0	0	11	650	0	0	23	0	684
2016	0	2	11	239	0	0	23	0	275
2017	0	2	11	231	0	0	23	0	267

									EAST COAST NORTH ISLAND
									Fishery
Fyr	**SCI**	**SQU**	**LLL**	**HHL**	**JMA**	**OEO**	**ORH**	**SBW**	**ALL**
1991	2 390	0	69	725	12	22	1 438	0	4 655
1992	916	0	69	540	9	22	992	0	2 548
1993	1 466	0	69	561	16	15	509	0	2 636
1994	1 233	0	196	1 563	10	12	732	0	3 747
1995	1 255	0	96	1 137	0	1	1 619	0	4 109
1996	1 508	0	111	2 192	0	34	446	0	4 292
1997	720	0	108	3 402	4	73	1 092	0	5 400
1998	883	0	160	2 960	0	105	3 514	0	7 623
1999	1 043	0	95	1 387	0	31	804	0	3 359
2000	956	2	161	1 146	0	73	1 560	0	3 898
2001	1 365	0	398	1 313	45	45	535	0	3 700
2002	3 733	0	204	736	53	26	65	0	4 816
2003	1 282	10	83	1 035	0	50	182	0	2 642
2004	795	0	247	1 031	0	11	0	0	2 084
2005	845	0	252	680	0	10	168	0	1 956
2006	336	0	399	386	0	8	0	0	1 129
2007	591	0	159	543	0	2	0	0	1 294
2008	391	0	84	401	0	1	142	0	1 019
2009	236	0	283	442	0	1	142	0	1 105
2010	619	0	258	421	0	1	285	0	1 584
2011	441	0	269	730	0	1	169	0	1 610
2012	251	0	229	458	0	0	93	0	1 031
2013	235	0	229	507	0	0	82	0	1 052
2014	321	0	229	998	0	0	0	0	1 548
2015	332	0	229	758	0	0	54	0	1 373
2016	234	0	229	570	0	0	54	0	1 087
2017	234	0	229	697	0	0	54	0	1 214

NORTHLAND									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1 422	0	75	24	0	0	70	0	1 591
1992	919	36	75	120	4	0	12	0	1 166
1993	667	13	75	104	22	0	19	0	901
1994	456	0	152	83	6	0	50	0	747
1995	1 208	0	81	303	0	0	42	0	1 635
1996	803	0	58	393	0	0	450	0	1 705
1997	584	8	67	1 205	0	0	3 240	0	5 104
1998	392	0	109	959	0	0	569	0	2 030
1999	230	0	29	359	8	0	93	0	718
2000	482	0	24	318	0	0	132	0	956
2001	603	6	70	200	179	0	30	0	1 087
2002	1 085	0	14	270	0	0	14	0	1 383
2003	811	12	29	189	0	0	41	0	1 082
2004	746	8	18	243	0	0	15	0	1 030
2005	948	1	64	155	0	0	23	0	1 191
2006	347	11	37	275	0	0	19	0	689
2007	580	11	36	232	0	0	13	0	872
2008	445	2	46	216	0	0	15	0	724
2009	255	5	38	183	0	0	26	0	507
2010	576	0	38	396	0	0	5	0	1 015
2011	390	0	63	277	0	0	40	0	770
2012	394	0	44	244	0	0	36	0	718
2013	560	9	44	383	0	0	16	0	1 012
2014	566	0	44	216	0	0	26	0	852
2015	479	0	44	302	0	0	16	0	841
2016	318	8	44	404	0	0	16	0	790
2017	318	8	44	395	0	0	16	0	781

PUYSEGUR									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	6	30	318	0	82	429	0	865
1992	4	118	30	418	143	53	1 689	0	2 456
1993	35	145	30	144	10	78	909	0	1 352
1994	1	12	60	246	14	59	2 343	0	2 736
1995	8	29	39	414	5	26	444	0	964
1996	0	32	59	153	0	107	921	0	1 272
1997	28	125	103	515	4	35	493	0	1 302
1998	37	38	219	535	0	417	333	0	1 580
1999	212	58	123	658	8	27	53	0	1 138
2000	40	74	93	1 050	0	200	27	0	1 484
2001	148	543	217	2 887	0	134	503	0	4 432
2002	0	887	99	660	0	124	193	0	1 964
2003	165	2 708	81	637	0	252	27	0	3 870
2004	21	487	62	328	0	45	0	0	943
2005	8	497	77	368	0	39	112	0	1 101
2006	0	538	87	971	0	26	131	0	1 753
2007	0	33	71	236	0	7	0	0	347
2008	0	32	190	206	0	64	0	0	492
2009	0	13	25	160	0	5	10	0	214
2010	0	63	5	120	0	1	14	0	204
2011	0	152	95	275	0	7	0	0	529
2012	0	47	21	302	0	3	15	0	388

PUYSEGUR									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2013	0	66	21	513	0	11	0	0	611
2014	0	127	21	1 609	0	75	42	0	1 874
2015	0	90	21	1 170	0	3	68	0	1 352
2016	0	107	21	1 702	0	3	68	0	1 901
2017	0	107	21	1 269	0	3	68	0	1 468

STEWART-SNARES SHELF									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	14 215	28	2 108	65	55	68	0	16 539
1992	1	11 222	28	2 592	536	40	122	0	14 542
1993	11	14 924	28	1 707	114	14	102	0	16 901
1994	0	4 669	85	859	159	16	126	0	5 914
1995	5	1 309	205	1 250	298	8	10	0	3 085
1996	2	5 029	37	1 752	204	49	231	0	7 304
1997	10	7 993	143	1 195	257	124	880	0	10 600
1998	19	5 719	219	3 464	506	114	692	0	10 733
1999	69	14 713	508	2 499	1 559	288	154	0	19 790
2000	8	7 685	198	4 860	4 621	184	90	0	17 646
2001	3	8 536	165	6 069	877	66	165	0	15 881
2002	0	15 158	164	7 148	1 567	248	2	0	24 288
2003	156	15 326	75	2 336	2 063	158	0	0	20 115
2004	40	19 606	166	2 833	723	34	0	0	23 401
2005	6	20 599	112	1 876	527	48	0	0	23 168
2006	0	28 634	48	2 067	1 369	62	0	0	32 180
2007	0	10 859	343	3 883	360	8	0	0	15 454
2008	0	10 356	179	1 866	183	9	5	0	12 598
2009	0	9 670	91	1 673	996	15	0	0	12 445
2010	0	9 100	54	2 211	1 144	27	0	0	12 536
2011	0	13 547	60	2 092	1 081	23	0	0	16 803
2012	0	10 295	514	1 495	1 622	42	0	0	13 968
2013	0	9 000	514	2 269	935	64	0	0	12 782
2014	0	8 482	514	9030	1 135	31	0	0	19 192
2015	0	7 126	514	9369	1 135	1	0	0	18 145
2016	0	5 625	514	4196	1 135	1	0	0	11 471
2017	0	5 625	514	7029	1 135	1	0	0	14 304

SUBANTARCTIC									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1	38	1 078	433	0	3	33	533	2 120
1992	14	0	1 078	226	0	2	40	1 479	2 839
1993	95	5	1 078	98	0	4	35	206	1 521
1994	85	6	705	139	0	12	31	382	1 360
1995	3	0	549	95	0	26	76	178	928
1996	8	20	641	106	0	49	538	63	1 426
1997	154	0	637	72	0	44	2 986	203	4 096
1998	66	0	132	306	0	152	1 762	296	2 714
1999	27	0	759	166	0	124	231	283	1 590
2000	9	15	905	442	0	312	147	283	2 113
2001	4	21	912	415	0	59	104	223	1 739
2002	53	16	537	1 223	0	63	0	364	2 256

SUBANTARCTIC									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2003	41	486	609	2 003	0	596	133	230	4 098
2004	58	548	882	1 901	0	69	74	390	3 922
2005	0	103	212	449	0	101	75	250	1 190
2006	0	110	139	86	0	301	172	190	997
2007	0	194	109	197	0	57	66	40	663
2008	0	13	575	820	0	85	49	40	1 582
2009	0	2	474	514	0	126	52	40	1 208
2010	0	9	274	83	0	145	61	40	612
2011	0	18	149	335	0	54	32	40	628
2012	0	5	139	95	0	37	5	40	321
2013	0	24	139	133	0	0	0	40	336
2014	18	4	139	1 214	0	0	0	40	1 415
2015	35	7	139	723	0	0	0	40	944
2016	5	4	139	238	0	0	0	40	426
2017	5	4	139	730	0	0	0	40	918

WEST COAST NORTH ISLAND									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	2	2	27	1 839	0	15	0	1 885
1992	4	4	2	28	6 978	0	160	0	7 175
1993	64	9	2	15	8 761	0	474	0	9 324
1994	0	23	2	14	4 924	2	2 397	0	7 362
1995	3	2	0	21	1 791	1	116	0	1 933
1996	0	0	18	100	3 621	0	33	0	3 771
1997	1	2	79	33	1 814	3	200	0	2 132
1998	13	0	1	45	5 784	0	268	0	6 111
1999	11	168	36	24	2 010	0	210	0	2 460
2000	5	553	4	61	360	0	158	0	1 140
2001	0	1 155	16	89	1 826	2	489	0	3 577
2002	10	836	5	48	5 693	0	40	0	6 632
2003	0	89	0	110	7 366	0	72	0	7 637
2004	0	6	1	43	13 310	0	61	0	13 421
2005	8	0	1	20	7 292	0	43	0	7 364
2006	0	0	5	61	10 312	0	51	0	10 429
2007	0	1	36	33	11 015	0	71	0	11 156
2008	0	0	29	94	8 975	0	41	0	9 139
2009	0	0	36	77	6 891	0	39	0	7 043
2010	0	0	25	24	10 760	0	29	0	10 838
2011	0	0	56	26	4 484	0	23	0	4 589
2012	0	0	14	39	5 674	0	60	0	5 788
2013	4	2	14	51	5 696	0	49	0	5 816
2014	0	0	14	80	5 281	0	64	0	5 440
2015	0	0	14	70	5 281	0	30	0	5 396
2016	0	0	14	42	5 281	0	30	0	5 368
2017	0	0	14	99	5 281	0	30	0	5 425

WEST COAST SOUTH ISLAND									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1	36	164	8 589	2 206	0	516	0	11 513
1992	0	23	164	6 335	2 864	0	1 082	0	10 468

WEST COAST SOUTH ISLAND									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1993	95	2	164	3 431	800	0	147	0	4 640
1994	2	5	320	9 426	2 129	9	213	0	12 103
1995	68	1	223	13 187	3 253	0	509	0	17 240
1996	12	0	401	12 211	3 050	2	370	0	16 045
1997	7	2	480	11 341	3 535	5	419	0	15 789
1998	26	2	406	10 964	4 376	24	276	0	16 074
1999	38	5	292	7 028	9 536	8	426	0	17 333
2000	15	0	281	7 168	3 906	14	654	0	12 037
2001	0	38	463	8 222	9 191	2	161	0	18 076
2002	15	5	221	7 245	3 903	0	0	0	11 389
2003	8	1 590	282	6 077	1 902	0	0	0	9 859
2004	0	369	158	6 830	706	0	0	0	8 063
2005	8	764	310	4 554	452	0	7	0	6 095
2006	0	185	255	4 051	1 271	0	0	0	5 762
2007	0	418	264	2 557	3 304	0	0	0	6 543
2008	0	0	364	3 083	1 540	0	0	0	4 987
2009	0	37	370	2 750	1 287	0	18	0	4 462
2010	0	41	386	1 696	506	0	4	0	2 633
2011	0	0	332	3 439	524	0	14	0	4 308
2012	14	0	428	2 755	588	0	23	0	3 808
2013	21	0	428	3 827	889	0	41	0	5 206
2014	31	0	428	5 864	1 003	0	11	0	7 337
2015	34	450	428	7 567	1 003	0	148	0	9 630
2016	39	0	428	6 303	1 003	0	148	0	7 921
2017	39	0	428	5 410	1 003	0	148	0	7 028

Appendix 19.7.3: DISCARDS: Total annual discards by fishery area for seven deepwater trawl fisheries and one longline fishery (1990–91 to 2015–16). Where data have not yet been updated for recent years for a fishery figures from the last available year have been assumed, in order for annual totals to be calculated. LLL = ling longline fishery; HHL = hoki/hake/ling fishery. [Continued on next pages]

AUCKLAND ISLANDS									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	4	74	17	5	0	0	0	0	100
1992	90	18	17	5	0	0	0	0	130
1993	278	68	17	9	0	9	2	0	383
1994	539	37	0	12	0	1	12	0	601
1995	707	32	12	10	0	47	20	0	829
1996	716	55	18	13	6	5	17	0	830
1997	1 383	183	34	15	2	6	10	0	1 634
1998	517	332	8	42	0	4	14	0	917
1999	645	101	13	19	0	27	15	0	820
2000	790	55	52	47	0	18	8	0	970
2001	421	115	58	69	0	5	6	0	674
2002	610	191	1	41	0	4	14	0	861
2003	832	1 162	0	205	0	7	7	0	2 046
2004	270	1 741	12	71	0	3	5	0	2 052
2005	508	1 234	1	57	0	0	1	0	1 758
2006	414	2 574	0	29	0	0	0	0	2 991
2007	448	812	0	30	0	0	0	0	1 263
2008	316	538	0	3	0	0	1	0	858
2009	412	1 331	0	3	0	1	0	0	1 744
2010	520	1 079	12	3	0	0	0	0	1 614

AUCKLAND ISLANDS									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2011	545	1 386	66	3	0	0	1	0	1 999
2012	453	662	6	3	0	0	0	0	1 124
2013	439	804	6	4	0	0	0	0	1 253
2014	464	259	6	9	0	1	2	0	740
2015	250	389	6	20	0	0	2	0	667
2016	421	1 130	6	6	0	0	2	0	1 565
2017	421	1 130	6	22	0	0	2	0	1 581

CHATHAM RISE									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	154	241	605	3 239	861	454	2 419	0	7 973
1992	1 364	303	605	6 387	100	201	1 053	0	10 015
1993	2 314	610	605	2 028	360	240	1 186	0	7 344
1994	1 402	1 718	862	1 650	48	302	937	0	6 919
1995	809	453	1 019	2 817	278	184	946	0	6 505
1996	4 126	110	1 107	3 737	24	423	671	0	10 199
1997	783	889	765	3 066	68	372	720	0	6 662
1998	657	1 436	1661	8 244	1 845	364	840	0	15 046
1999	361	990	587	5 314	110	272	1 020	0	8 654
2000	271	825	690	9 829	125	297	787	0	12 825
2001	1 366	2 721	874	6 981	18	151	1 012	0	13 123
2002	1 303	1 698	1 709	9 545	77	48	471	0	14 850
2003	1 056	7 324	1 027	14 762	80	68	478	0	18 598
2004	218	902	806	9 658	5	99	538	0	5 750
2005	2 117	2 156	1 080	3 348	5	95	520	0	6 843
2006	898	2 681	955	4 042	271	93	1 399	0	7 260
2007	2 249	2 392	546	3 089	52	46	491	0	7 590
2008	1 417	864	1 016	7 160	47	28	129	0	5 064
2009	1 041	327	875	12 681	11	26	195	0	5 085
2010	1 418	419	950	3 848	3	27	76	0	9 442
2011	1 002	1 083	879	5 611	3	25	29	0	5 706
2012	1 138	395	517	3 039	4	61	115	0	3 522
2013	1 743	105	517	1 718	32	9	71	0	5 105
2014	2 245	29	517	1 801	3	30	18	0	4 643
2015	1 718	478	517	2 776	3	7	32	0	5 532
2016	1 262	1 859	517	984	3	7	32	0	4 665
2017	1 262	1 859	517	924	3	7	32	0	4 605

COOK STRAIT									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	0	3	1 081	0	3	76	0	1 164
1992	0	0	3	769	0	1	84	0	857
1993	0	4	3	786	0	2	56	0	852
1994	0	0	2	917	0	1	71	0	991
1995	9	0	3	1 729	0	2	84	0	1 826
1996	0	0	2	3 896	0	1	24	0	3 923
1997	0	0	1	3 259	0	6	32	0	3 299
1998	0	0	6	2 033	0	3	15	0	2 057
1999	3	9	16	1 871	31	4	35	0	1 968
2000	1	0	21	2 724	16	3	16	0	2 780

									COOK STRAIT
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2001	0	7	65	1 312	18	6	6	0	1 414
2002	36	0	13	718	0	5	5	0	778
2003	3	2	4	1 982	0	3	3	0	2 585
2004	0	0	15	1 330	0	0	2	0	994
2005	28	0	31	651	0	0	3	0	656
2006	1	0	36	659	0	2	7	0	253
2007	1	0	48	652	0	0	3	0	903
2008	2	0	35	838	0	0	3	0	617
2009	0	0	8	943	0	0	4	0	522
2010	0	0	7	333	0	0	2	0	376
2011	0	0	8	690	0	0	3	0	758
2012	0	0	5	643	0	0	2	0	347
2013	0	0	5	346	0	0	1	0	184
2014	0	0	5	550	0	0	2	0	557
2015	0	0	5	213	0	0	4	0	222
2016	0	2	5	366	0	0	4	0	377
2017	0	2	5	303	0	0	4	0	314

									EAST COAST NORTH ISLAND
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1 535	0	24	178	0	9	664	0	2 411
1992	696	0	24	131	0	4	692	0	1 548
1993	579	0	24	130	0	7	555	0	1 295
1994	540	0	41	351	3	1	689	0	1 625
1995	418	0	27	316	0	13	893	0	1 666
1996	1 083	0	30	582	0	13	969	0	2 677
1997	405	0	31	726	0	57	873	0	2 092
1998	624	0	42	3	0	32	669	0	1 369
1999	1 240	0	24	218	0	36	627	0	2 145
2000	348	0	43	181	0	20	275	0	866
2001	702	0	114	228	3	23	1 093	0	2 163
2002	2 284	0	59	174	6	13	31	0	2 567
2003	699	11	23	71	0	23	0	0	979
2004	161	0	58	57	0	4	5	0	344
2005	585	0	64	26	0	4	7	0	700
2006	247	0	96	24	0	3	56	0	453
2007	470	0	66	32	0	1	59	0	655
2008	289	0	84	23	0	0	59	0	470
2009	186	0	78	40	0	0	83	0	396
2010	601	0	81	33	0	0	100	0	870
2011	423	0	73	55	0	0	22	0	665
2012	209	0	79	39	0	0	32	0	376
2013	238	0	79	37	0	0	57	0	478
2014	301	0	79	71	0	0	37	0	488
2015	285	0	79	11	0	0	13	0	388
2016	255	0	79	70	0	0	13	0	417
2017	255	0	79	104	0	0	13	0	451

									NORTHLAND
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	615	0	23	13	0	0	5	0	657

									NORTHLAND
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1992	405	0	23	91	0	0	1	0	520
1993	303	1	23	127	0	0	2	0	457
1994	231	0	44	53	0	0	4	0	332
1995	386	0	23	191	0	0	3	0	604
1996	732	0	18	215	0	0	8	0	974
1997	431	1	22	520	0	0	40	0	1 015
1998	214	0	33	2	0	0	37	0	286
1999	137	0	7	144	0	1	118	0	406
2000	262	0	8	117	0	0	16	0	404
2001	383	1	17	70	9	0	2	0	482
2002	832	0	4	124	0	0	0	0	961
2003	367	35	9	70	0	0	4	0	496
2004	235	2	5	72	0	0	1	0	292
2005	746	0	12	32	0	0	1	0	774
2006	281	4	13	86	0	0	0	0	358
2007	507	5	12	63	0	0	1	0	568
2008	355	1	24	55	0	0	1	0	412
2009	231	2	11	67	0	0	4	0	282
2010	505	0	13	71	0	0	0	0	659
2011	335	0	19	93	0	0	1	0	448
2012	268	0	23	98	0	0	6	0	359
2013	505	3	23	116	0	0	9	0	674
2014	454	0	23	104	0	0	7	0	588
2015	342	0	23	75	0	0	3	0	444
2016	321	6	23	138	0	0	3	0	492
2017	321	6	23	158	0	0	3	0	512

									PUYSEGUR
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	1	21	225	0	26	34	0	306
1992	1	17	21	512	0	34	139	0	724
1993	13	31	21	208	0	115	118	0	505
1994	0	5	27	248	3	32	268	0	582
1995	2	5	22	227	0	63	38	0	358
1996	0	1	34	567	0	75	117	0	794
1997	21	25	60	429	0	100	34	0	669
1998	27	24	122	293	0	90	8	0	564
1999	196	16	66	270	0	130	5	0	683
2000	15	12	42	307	0	109	3	0	488
2001	84	162	106	253	0	167	76	0	848
2002	0	282	60	297	0	65	32	0	735
2003	32	3 204	45	784	0	172	18	0	3 667
2004	4	122	34	276	0	15	9	0	222
2005	5	196	39	681	0	15	4	0	554
2006	0	147	54	2 074	0	11	1	0	401
2007	0	12	45	484	0	3	0	0	189
2008	0	10	116	622	0	11	1	0	252
2009	0	4	14	152	0	2	2	0	110
2010	0	19	4	267	0	1	2	0	92
2011	0	49	57	199	0	3	3	0	263
2012	0	12	16	417	0	0	2	0	163
2013	0	8	16	378	0	0	0	0	241

PUYSEGUR									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2014	0	37	16	249	0	2	5	0	309
2015	0	34	16	275	0	0	0	0	326
2016	0	71	16	515	0	0	0	0	603
2017	0	71	16	460	0	0	0	0	548

STEWART-SNARES SHELF									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	144	15	1 471	0	14	1	0	1 645
1992	0	83	15	1 531	1 405	21	2	0	3 056
1993	3	1 344	15	1 942	105	11	2	0	3 423
1994	0	99	39	585	13	7	3	0	745
1995	1	79	171	211	64	16	0	0	542
1996	3	4	15	459	502	28	4	0	1 014
1997	6	1 659	55	1 017	51	91	57	0	2 936
1998	14	1 826	96	1 167	0	50	10	0	3 163
1999	64	1 335	132	1 430	40	174	6	0	3 180
2000	2	162	55	1 114	612	140	2	0	2 088
2001	2	928	58	1 852	39	69	3	0	2 950
2002	0	3 145	46	2 221	277	146	1	0	5 836
2003	44	3 383	34	2 853	25	141	0	0	4 661
2004	9	2 858	58	2 528	4	24	0	0	3 452
2005	3	3 277	70	2 179	7	23	0	0	3 768
2006	0	4 713	18	2 008	10	37	1	0	6 077
2007	0	1 892	218	4 774	3	5	0	0	2 553
2008	0	1 096	74	594	1	5	0	0	1 258
2009	0	1 816	34	1 384	7	2	0	0	1 995
2010	0	1 656	23	512	2	4	0	0	1 962
2011	0	2 690	23	724	2	4	0	0	3 526
2012	0	1 513	252	1 019	3	4	0	0	2 362
2013	0	1 544	252	1 288	32	6	0	0	2 449
2014	0	1 152	252	1 525	17	6	0	0	2 952
2015	0	961	252	1 987	17	0	0	0	3 218
2016	0	1 284	252	876	17	0	0	0	2 430
2017	0	1 284	252	2 006	17	0	0	0	3 560

SUBANTARCTIC									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1	0	317	364	0	1	3	746	1 431
1992	2	0	317	114	0	1	3	1 218	1 656
1993	47	0	317	40	0	2	5	537	948
1994	34	1	235	33	0	4	3	483	793
1995	1	0	318	22	0	38	7	303	689
1996	16	0	346	26	0	20	40	406	854
1997	125	0	427	17	0	22	97	270	958
1998	38	0	10	73	0	55	34	392	602
1999	19	0	471	54	0	46	25	471	1 086
2000	2	0	537	278	0	155	10	471	1 454
2001	2	2	557	147	0	230	10	137	1 085
2002	37	3	296	655	0	38	7	159	1 195
2003	19	497	267	1 002	0	91	14	250	1 661
2004	13	129	329	773	0	58	19	90	847

									SUBANTARCTIC
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
2005	0	43	136	765	0	78	5	160	530
2006	0	42	66	221	0	121	27	240	518
2007	0	96	53	949	0	51	4	120	371
2008	0	2	62	115	0	32	4	120	343
2009	0	1	151	222	0	33	6	120	322
2010	0	5	115	47	0	36	3	120	299
2011	0	6	20	23	0	12	2	120	175
2012	0	2	86	90	0	9	0	120	218
2013	0	6	86	67	0	0	0	120	224
2014	14	1	86	121	0	0	0	120	342
2015	25	0	86	102	0	0	0	120	333
2016	4	1	86	9	0	0	0	120	220
2017	4	1	86	223	0	0	0	120	434

									WEST COAST NORTH ISLAND
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	0	0	0	14	0	0	5	0	19
1992	1	0	0	22	100	0	27	0	150
1993	24	1	0	18	915	0	333	0	1 291
1994	0	4	1	9	1 285	0	141	0	1 439
1995	1	0	0	14	317	4	161	0	497
1996	0	0	10	55	109	0	45	0	220
1997	0	1	32	14	0	2	106	0	155
1998	8	0	1	0	0	0	43	0	53
1999	10	18	17	9	837	0	62	0	953
2000	1	26	2	22	16	0	129	0	196
2001	0	148	6	31	51	1	246	0	483
2002	7	181	3	22	230	0	7	0	450
2003	0	140	0	13	183	0	6	0	376
2004	0	2	0	10	133	0	7	0	151
2005	5	0	1	3	98	0	15	0	121
2006	0	0	2	30	77	0	11	0	103
2007	0	1	19	19	165	0	2	0	193
2008	0	0	15	32	166	0	13	0	207
2009	0	0	17	32	97	0	3	0	131
2010	0	0	11	4	264	0	6	0	290
2011	0	0	23	8	76	0	4	0	112
2012	0	0	9	10	140	0	26	0	184
2013	3	1	9	18	113	0	21	0	164
2014	0	0	9	14	165	0	60	0	247
2015	0	0	9	8	165	0	10	0	192
2016	0	0	9	12	165	0	10	0	196
2017	0	0	9	26	165	0	10	0	210

									WEST COAST SOUTH ISLAND
									Fishery
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1991	1	0	115	5 904	76	0	195	0	6 291
1992	0	0	115	3 382	201	0	223	0	3 921
1993	36	0	115	4 889	495	0	192	0	5 727
1994	1	1	139	6 671	118	1	422	0	7 353

WEST COAST SOUTH ISLAND									
	Fishery								
FY	SCI	SQU	LLL	HHL	JMA	OEO	ORH	SBW	ALL
1995	18	0	127	8 327	461	4	147	0	9 084
1996	12	0	220	7 314	363	1	102	0	8 013
1997	5	1	270	7 520	964	3	181	0	8 946
1998	19	1	219	3 587	0	7	145	0	3 979
1999	35	1	223	3 177	520	8	290	0	4 254
2000	6	0	155	2 035	16	4	301	0	2 517
2001	0	6	231	2 421	482	1	71	0	3 211
2002	9	1	128	2 926	100	0	2	0	3 166
2003	3	212	172	3 250	7	0	1	0	3 158
2004	0	9	107	2 882	1	0	1	0	1 773
2005	6	29	165	1 509	3	0	0	0	668
2006	0	5	153	2 676	29	0	7	0	1 608
2007	0	14	157	3 740	84	0	1	0	780
2008	0	0	236	2 167	7	0	0	0	869
2009	0	1	204	2 402	5	0	0	0	689
2010	0	1	234	659	4	0	0	0	719
2011	0	0	183	2 302	1	0	2	0	1 231
2012	9	0	209	3 606	5	0	3	0	1 256
2013	14	0	209	2 727	7	0	6	0	2 319
2014	19	0	209	2 491	9	0	10	0	2 739
2015	21	20	209	2 681	9	0	48	0	2 989
2016	28	0	209	2 106	9	0	48	0	2 401
2017	28	0	209	2 802	9	0	48	0	3 097

19.8 APPENDICES FROM CHAPTER 10 CHONDRICHTHYANS

Appendix 19.8.1: List of New Zealand chondrichthyans, with details of their fisheries management classification, IUCN and Department of Conservation threat classes, and their distribution area. IUCN threat classes: EN, Endangered; VU, Vulnerable; NT, Near Threatened; LC, Least Concern; DD, Data Deficient. DOC threat classes: DD, Data Deficient; NE, Nationally Endangered; NV, Threatened – Nationally Vulnerable; ARNU, At Risk – Naturally Uncommon; MI, Migrant; VA, Vagrant; NOT, Not Threatened. DOC qualifiers: CD, Conservation Dependent; DP, Data Poor; Inc, Increasing; SO, Secure Overseas; S?O, Uncertain Whether Secure Overseas; TO, Threatened Overseas; T?O, Uncertain Whether Threatened Overseas. Sources: IUCN Redlist classes as at July 2013 (L. Harrison, Shark Specialist Group IUCN, pers. comm.); DOC threat classes 2018 (Duffy et al. 2018). [Continued on next page]. NB: IUCN Redlist classes for New Zealand chondrichthyans were last reviewed in 2018.

Group	Family	Species	Authority	Common name	Code	Management class	IUCN Red List class	DoC Threat class	DoC qualifer	Location	Notes
Chimaera	Callorhinchidae	*Callorhinchus milii*	Bory de Saint-Vincent, 1823	Elephantfish	ELE	QMS	LC	NOT	CD, Inc	NZ EEZ	
Chimaera	Rhinochimaeridae	*Harriotta haeckeli*	Karrer, 1972	Smallspine spookfish	HHA	Non-QMS	LC	NOT		NZ EEZ	
Chimaera	Rhinochimaeridae	*Harriotta raleighana*	Goode & Bean, 1895	Longnose spookfish	LCH	Non-QMS	LC	NOT		NZ EEZ	
Chimaera	Rhinochimaeridae	*Rhinochimaera pacifica*	(Mitsukuri, 1895)	Pacific spookfish	RCH	Non-QMS	LC	NOT	DP	NZ EEZ	
Chimaera	Chimaeridae	*Chimaera carophila*	Kemper, Ebert, Naylor & Didier, 2014	Brown chimaera, longspine chimaera	CHP	Non-QMS	LC	NOT		NZ EEZ	
Chimaera	Chimaeridae	*Chimaera lignaria*	Didier, 2002	Purple chimaera, giant chimaera	CHG	Non-QMS	LC	NOT	SO	NZ EEZ	
Chimaera	Chimaeridae	*Chimaera panthera*	Didier, 1998	Leopard chimaera	CPN	Non-QMS	DD	NOT	DP	NZ EEZ	
Chimaera	Chimaeridae	*Hydrolagus bemisi*	Didier, 2002	Pale ghost shark	GSP	QMS	LC	NOT	CD	NZ EEZ	
Chimaera	Chimaeridae	*Hydrolagus homonycteris*	Didier, 2008	Black ghost shark	HYB	Non-QMS	LC	NOT	SO	NZ EEZ	
Chimaera	Chimaeridae	*Hydrolagus novaezealandiae*	(Fowler, 1911)	Dark ghost shark	GSH	QMS	LC	NOT		NZ EEZ	
Chimaera	Chimaeridae	*Hydrolagus trolli*	Didier and Seret, 2002	Pointynose blue ghost shark	HYP	Non-QMS	LC	NOT	SO	NZ EEZ	
Chimaera	Chimaeridae	*Hydrolagus* cf. *affinis*		Giant black ghost shark	HGB	Non-QMS		DD	CD	NZ EEZ	
Shark	Chlamydoselachidae	*Chlamydoselachus anguineus*	Garman, 1884	Frilled shark	FRS	Non-QMS	LC	ARNU	DP,SO	NZ EEZ	

Group	Family	Species	Authority	Common name	Code	Management class	IUCN Red List class	DoC Threat class	DoC qualifer	Location	Notes
Shark	Hexanchiidae	*Heptranchias perlo*	(Bonnaterre, 1788)	Sharpnose sevengill shark	HEP	Non-target	NT	ARNU	DP,SO	NZ EEZ	
Shark	Hexanchiidae	*Hexanchus griseus*	(Bonnaterre, 1788)	Sixgill shark	HEX	Non-QMS	NT	NOT	DP,SO	NZ EEZ	
Shark	Hexanchiidae	*Notorynchus cepedianus*	(Peron, 1807)	Broadnose sevengill shark	SEV	Non-QMS	DD	NOT	DP,SO	NZ EEZ	
Shark	Echinorhinidae	*Echinorhinus brucus*	(Bonnaterre, 1788)	Bramble shark	BRS	Non-QMS	DD	ARNU	DP,SO	NZ EEZ	
Shark	Echinorhinidae	*Echinorhinus cookei*	Pietschmann, 1928	Prickly shark	ECO	Non-QMS	DD	ARNU	DP,SO	NZ EEZ	
Shark	Squalidae	*Cirrhigaleus australis*	White, Last & Stevens, 2007	Southern mandarin dogfish	MSH	Non-QMS	DD	ARNU	DP,TO	NZ EEZ	
Shark	Squalidae	*Squalus acanthias*	Linnaeus, 1758	Spiny dogfish	SPD	QMS	VU	NOT	SO	NZ EEZ	
Shark	Squalidae	*Squalus griffini*	Phillipps, 1931	Northern spiny dogfish	NSD	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Squalidae	*Squalus raoulensis*	Duffy & Last, 2007	Kermadec spiny dogfish		Non-QMS	LC	DD		NZ EEZ	
Shark	Squalidae	*Squalus* sp.		Shortspine dogfish		Non-QMS		DD		NZ EEZ	
Shark	Centrophoridae	*Centrophorus harrissoni*	McCulloch, 1915	Harrisson's dogfish		Non-QMS	EN	DD	TO	NZ EEZ	
Shark	Centrophoridae	*Centrophorus squamosus*	(Bonnaterre, 1788)	Leafscale gulper shark	CSQ	Non-QMS	VU	NOT	SO	NZ EEZ	
Shark	Centrophoridae	*Deania calcea*	(Lowe, 1839)	Shovelnose dogfish	SND	Non-QMS	LC	NOT		NZ EEZ	
Shark	Centrophoridae	*Deania hystricosa*	(Garman, 1906)	Rough longnose dogfish	SNR	Non-QMS	DD	DD		NZ EEZ	
Shark	Centrophoridae	*Deania quadrispinosa*	(McCulloch, 1915)	Longsnout dogfish	DEQ	Non-QMS	NT	DD	SO	NZ EEZ	
Shark	Etmopteridae	*Centroscyllium kamoharai*	Abe, 1966	Fragile dogfish		Non-QMS	LC	DD		NZ EEZ	
Shark	Etmopteridae	*Etmopterus granulosus*	(Günther, 1880)	Baxter's dogfish	ETB	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Etmopteridae	*Etmopterus lucifer*	Jordan & Snyder, 1902	Lucifer dogfish	ETL	Non-QMS	LC	NOT	DP,SO	NZ EEZ	
Shark	Etmopteridae	*Etmopterus molleri*	(Whitley, 1939)	Moller's lantern shark	EMO	Non-QMS	DD	DD	S?O	NZ EEZ	
Shark	Etmopteridae	*Etmopterus pusillus*	(Lowe, 1839)	Smooth lantern shark	ETP	Non-QMS	LC	ARNU	DP,SO	NZ EEZ	

Group	Family	Species	Authority	Common name	Code	Management class	IUCN Red List class	DoC Threat class	DoC qualifer	Location	Notes
Shark	Etmopteridae	*Etmopterus unicolor*	(Engelhardt, 1912)	Bristiled lantern shark	ETU	Non-QMS	DD	NOT	SO	NZ EEZ	
Shark	Etmopteridae	*Etmopterus viator*	Straube, 2011	Blue-eye lantern shark	EVI	Non-QMS	LC	DD		NZ EEZ	
Shark	Somniosidae	*Centroscymnus coelolepis*	Barbosa du Bocage & de Brito Capello, 1864	Portuguese dogfish	CYL	Non-QMS	NT	NOT	DP	NZ EEZ	
Shark	Somniosidae	*Centroscymnus owstonii*	Garman, 1906	Owston's dogfish	CYO	Non-QMS	VU	NOT		NZ EEZ	
Shark	Somniosidae	*Centroselachus crepidater*	(Barbosa du Bocage & de Brito Capello, 1864)	Longnose velvet dogfish	CYP	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Somniosidae	*Scymnodalatias albicauda*	Taniuchi & Garrick, 1986	Whitetail dogfish	SLB	Non-QMS	DD	DD	S?O	NZ EEZ	
Shark	Somniosidae	*Scymnodalatias sherwoodi*	(Archey, 1921)	Sherwood's dogfish	SHE	Non-QMS	DD	DD	S?O	NZ EEZ	
Shark	Somniosidae	*Centroscymnus macracanthus*	Regan, 1906	Plunket's dogfish	PLS	Non-QMS	DD	NOT	T?O	NZ EEZ	
Shark	Somniosidae	*Scymnodon plunketi*	(Waite, 1910)	Knifetooth dogfish	SRI	Non-QMS	VU	DD	S?O	NZ EEZ	
Shark	Somniosidae	*Somniosus antarcticus*	Whitley, 1939	Southern sleeper shark	SSS	Non-QMS	LC	NOT	DP,S?O	NZ EEZ	
Shark	Somniosidae	*Somniosus longus*	(Tanaka, 1912)	Little sleeper shark	SOM	Non-QMS	DD	DD	S?O	NZ EEZ	
Shark	Somniosidae	*Zameus squamulosus*	(Günther, 1877)	Velvet dogfish	ZAS	Non-QMS	DD	DD	S?O	NZ EEZ	
Shark	Oxynotidae	*Oxynotus bruniensis*	(Ogilby, 1893)	Prickly dogfish	PDG	Non-QMS	NT	NOT	DP,SO	NZ EEZ	
Shark	Dalatiidae	*Dalatias licha*	(Bonnaterre, 1788)	Seal shark	BSH	Non-QMS	VU	NOT	SO	NZ EEZ	
Shark	Dalatiidae	*Euprotomicrus bispinatus*	(Quoy & Gaimard, 1824)	Pygmy shark	EBI	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Dalatiidae	*Isistius brasiliensis*	(Quoy & Gaimard, 1824)	Cookie cutter shark	IBR	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Heterodontidae	*Heterodontus portusjacksoni*	(Meyer, 1793)	Port Jackson shark	PJS	Non-QMS	LC	VA	SO	NZ EEZ	
Shark	Rhincodontidae	*Rhincodon typus*	Smith, 1828	Whale shark	WSH	Protected	EN	MI	SO	NZ EEZ	
Shark	Odontaspididae	*Odontaspis ferox*	(Risso, 1810)	Deepwater (smalltooth) sand tiger shark	ODO	Protected	VU	ARNU	TO	NZ EEZ	

Group	Family	Species	Authority	Common name	Code	Management class	IUCN Red List class	DoC Threat class	DoC qualifer	Location	Notes
Shark	Pseudocarchariidae	*Pseudocarcharias kamoharai*	(Matsubara, 1936)	Crocodile shark	CRC	Non-QMS	LC	DD	SO	NZ EEZ	
Shark	Mitsukurinidae	*Mitsukurina owstoni*	Jordan, 1898	Goblin shark	GOB	Non-QMS	LC	ARNU	DP,SO	NZ EEZ	
Shark	Alopiidae	*Alopias superciliosus*	Lowe, 1841	Bigeye thresher	BET	Non-QMS	VU	NOT	TO	NZ EEZ	
Shark	Alopiidae	*Alopias vulpinus*	(Bonnaterre, 1788)	Thresher shark	THR	Non-QMS	VU	NOT	DP,TO	NZ EEZ	
Shark	Cetorhinidae	*Cetorhinus maximus*	(Gunnerus, 1765)	Basking shark	BSK	Protected	EN	NV		NZ EEZ	
Shark	Laminidae	*Carcharodon carcharias*	(Linnaeus, 1758)	White shark, white pointer	WPS	Protected	VU	NE	DP,TO	NZ EEZ	
Shark	Laminidae	*Isurus oxyrinchus*	Rafinesque, 1810	Mako shark, shortfin mako	MAK	QMS	EN	NOT	S?O	NZ EEZ	
Shark	Laminidae	*Lamna nasus*	(Bonnaterre, 1788)	Porbeagle shark	POS	QMS	VU	NOT	TO	NZ EEZ	
Shark	Pentanchidae	*Apristurus albisoma*	Nakaya & Seret, 1999	Grey roundfin catshark		Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus ampliceps*	Sasahara, Sato & Nakaya, 2008	Roughskin catshark	AAM	Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus exsanguis*	Sato, Nakaya & Stewart, 1999	Pale catshark	AEX	Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus garricki*	Sato, Stewart & Nakaya, 2013	Garrick's catshark	AGK	Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus melanoasper*	Iglésias, Nakaya & Stehmann, 2004	Flesynose catshark	AML	Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus pinguis*	Deng, Xiong & Zhan, 1983	Bulldog catshark	APN	Non-QMS	LC	DD		NZ EEZ	
Shark	Pentanchidae	*Apristurus* cf *sinensis*	Chu & Hu, 1981	Freckled catshark	ASI	Non-QMS		DD		NZ EEZ	
Shark	Scyliorhinidae	*Bythaelurus dawsoni*	(Springer, 1971)	Dawson's catshark	DCS	Non-QMS	LC	NOT	DP	NZ EEZ	
Shark	Scyliorhinidae	*Cephaloscyllium isabellum*	(Bonnaterre, 1788)	Carpet shark	CAR	Non-QMS	LC	NOT		NZ EEZ	
Shark	Scyliorhinidae	*Cephaloscyllium* cf *variegatum*	Last & White, 2008	Swellshark		Non-QMS		DD		NZ EEZ	
Shark	Pentanchidae	*Parmaturus macmillani*	Hardy, 1985	McMillan's catshark	PCS	Non-QMS	VU	DD	S?O	NZ EEZ	

Group	Family	Species	Authority	Common name	Code	Management class	IUCN Red List class	DoC Threat class	DoC qualifer	Location	Notes
Shark	Pentanchidae	*Parmaturus* sp.		Rough-backed catshark		Non-QMS		DD		NZ EEZ	
Shark	Pseudotriakidae	*Gollum attenuatus*	(Garrick, 1954)	Slender smooth hound	SSH	Non-QMS	LC	NOT	SO	NZ EEZ	
Shark	Pseudotriakidae	*Pseudotriakis microdon*	de Brito Capello, 1868	False catshark	PMI	Non-QMS	LC	DD	SO	NZ EEZ	
Shark	Triakidae	*Galeorhinus galeus*	(Linnaeus, 1758)	School shark	SCH	QMS	VU	NOT	CD,TO	NZ EEZ	
Shark	Triakidae	*Mustelus lenticulatus*	Phillipps, 1932	Rig	SPO	QMS	LC	NOT	CD	NZ EEZ	
Shark	Triakidae	*Mustulus* sp.		Kermadec rig		Non-QMS		NOT		NZ EEZ	
Shark	Carcharhinidae	*Carcharhinus brachyurus*	(Günther, 1870)	Bronze whaler	BWH	Non-QMS	NT	NOT	CD,DP,SO	NZ EEZ	
Shark	Carcharhinidae	*Carcharhinus galapagensis*	(Snodgrass & Heller, 1905)	Galapagos shark	CGA	Non-QMS	LC	NOT	CD,SO	NZ EEZ	
Shark	Carcharhinidae	*Carcharhinus longimanus*	(Poey, 1861)	Oceanic whtietip shark	OWS	Non-QMS	CR	MI	SO	NZ EEZ	
Shark	Carcharhinidae	*Carcharhinus obscurus*	(Le Sueur, 1818)	Dusky shark	DSH	Non-QMS	EN	MI	SO	NZ EEZ	
Shark	Carcharhinidae	*Carcharhinus plumbeus*	(Nardo, 1827)	Sandbar shark		Non-QMS	VU	DD		NZ EEZ	
Shark	Carcharhinidae	*Galeocerdo cuvier*	(Peron & Lesueur, 1822)	Tiger shark	TIS	Non-QMS	NT	MI	SO	NZ EEZ	
Shark	Carcharhinidae	*Prionace glauca*	(Linnaeus, 1758)	Blue shark	BWS	QMS	NT	NOT	SO	NZ EEZ	
Shark	Carcharhinidae	*Triaenodon obesus*	(Rüppell, 1837)	Whitetip reef shark	TRB	Non-QMS	NT	VA		NZ EEZ	
Shark	Sphyrnidae	*Sphyrna zygaena*	(Linnaeus, 1758)	Hammerhead shark, smooth hammerhead	HHS	Non-target	VU	NOT	SO	NZ EEZ	
Batoid	Narkidae	*Typhlonarke aysoni*	(Hamilton, 1902)	Blind electric ray	TAY	Non-QMS	LC	NOT	DP	NZ EEZ	
Batoid	Torpedinidae	*Tetronarce nobiliana*	(Bonaparte, 1835)	Electric ray	ERA	Non-QMS	DD	DD		NZ EEZ	
Batoid	Torpedinidae	*Tetronarce* cf. *tokionis*	(Tanaka, 1908)	Slender electric ray		Non-QMS		DD		NZ EEZ	
Batoid	Arhynchobatidae	*Arhynchobatis asperrimus*	Waite, 1909	Longtail skate	LSK	Non-QMS	DD	DD		NZ EEZ	

Group	**Family**	**Species**	**Authority**	**Common name**	**Code**	**Management class**	**IUCN Red List class**	**DoC Threat class**	**DoC qualifer**	**Location**	**Notes**
Batoid	Arhynchobatidae	*Bathyraja* cf. *eatonii*		Antarctic allometric skate	BEA	Non-QMS				Southern Ocean	
Batoid	Arhynchobatidae	*Bathyraja maccaini*	Springer, 1971	MacCain's skate	MCS	Non-QMS	NT			Southern Ocean	
Batoid	Arhynchobatidae	*Bathyraja pacifica*	Last, Stewart & Seret, 2016	Pacific blond skate		Non-QMS	LC	NOT	DP	NZ EEZ	
Batoid	Arhynchobatidae	*Bathyraja richardsoni*	(Garrick, 1961)	Richardon's skate	RIS	Non-QMS	LC	NOT	DP	NZ EEZ	
Batoid	Arhynchobatidae	*Bathyraja shuntovi*	Dolganov, 1985	Longnose deepsea skate	PSK	Non-QMS	DD	NOT		NZ EEZ	
Batoid	Arhynchobatidae	*Bathyraja* sp.		Antarctic dwarf skate	BHY	Non-QMS				NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja albilabiata*	Last & McEachran, 2006	Whitemouth skate		Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja asperula*	(Garrick & Paul, 1974)	Smooth deepsea skate	BTA	Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja heuresa*	Last & Seret, 2012	Eureka skate		Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja leviveneta*	Last & McEachran, 2006	Blue skate	BRL	Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja microspinifera*	Last & McEachran, 2006	Dwarf skate	BMI	Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja spinifera*	(Garrick & Paul, 1974)	Prickly deepsea skate	BTS	Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Brochiraja vittacauda*	Last & Seret, 2012	Ribbontail skate		Non-QMS	DD	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Notoraja alisae*	Seret & Last, 2012	Velcro skate	NAL	Non-QMS	LC	DD		NZ EEZ	
Batoid	Arhynchobatidae	*Notoraja sapphira*	Seret & Last, 2009	Sapphire skate		Non-QMS	DD	DD		NZ EEZ	
Batoid	Rajidae	*Amblyraja georgiana*	(Norman, 1938)	Antarctic starry skate	SRR	Non-QMS	DD			Southern Ocean	
Batoid	Rajidae	*Amblyraja hyperborea*	(Collett, 1879)	Deepwater spiny skate	DSK	Non-QMS	LC	NOT		NZ EEZ	
Batoid	Rajidae	*Dipturus innominatus*	(Garrick & Paul, 1974)	Smooth skate	SSK	QMS	LC	NOT	CD	NZ EEZ	
Batoid	Rajidae	*Zearaja nasutus*	(Müller & Henle, 1841)	Rough skate	RSK	QMS	LC	NOT	CD	NZ EEZ	Now *Dipturus nasutus*

Group	**Family**	**Species**	**Authority**	**Common name**	**Code**	**Management class**	**IUCN Red List class**	**DoC Threat class**	**DoC qualifer**	**Location**	**Notes**
Batoid	Dasyatidae	*Bathytoshia brevicaudata*	(Hutton, 1875)	Shorttail stingray	BRA	Non-QMS	LC	NOT	SO	NZ EEZ	
Batoid	Dasyatidae	*Bathytoshia lata*	(Garman, 1880)	Longtail stingray	WRA	Non-QMS	LC	NOT	SO	NZ EEZ	
Batoid	Dasyatidae	*Pteroplatytrygon violacea*	(Bonaparte, 1832)	Pelagic stingray	DAS	Non-QMS	LC	NOT	SO	NZ EEZ	
Batoid	Myliobatidae	*Myliobatis tenuicaudatus*	Hector, 1877	Eagle ray	EGR	Non-QMS	LC	NOT	DP,SO	NZ EEZ	
Batoid	Mobulidae	*Mobula birostris*	(Walbaum, 1792)	Manta ray	RMB	Protected	VU	DD	TO	NZ EEZ	
Batoid	Mobulidae	*Mobula mobular*	(Bonnaterre, 1788)	Spinetail devil ray	MJA	Protected	EN	DD	SO	NZ EEZ	

Appendix 19.8.2: Indicative information on status of stocks for the eleven shark species subject to the QMS.

Species name	Plenary stock	Last assessment date	At or above target levels?	Below the soft limit?	Below the hard limit?	Overfishing?	Corrective management action
Blue shark*	BWS1	2014					-
Elephant fish	ELE2	-					-
Elephant fish	ELE3	2016	●	●●	●●●	■	May be considered for review in 2020
Elephant fish	ELE5	2017	●	●●	●●●	■	-
Elephant fish	ELE7	2019	●	●●	●●●	■	TAC set in 2019
Ghost shark - dark	GSH1, GSH2, GSH7, GSH8	-					-
Ghost shark - dark	GSH3	-					-
Ghost shark - dark	GSH4, GSH5, GSH6	-					-
Ghost shark - pale	GSP1, GSP5	2011		●●	●●●		-
Ghost shark - pale	GSP7	-					-
Hammerhead sharks (smooth)	HHS1	-					-
Mako shark*	MAK1	2014					TAC reduced in 2012
Porbeagle shark*	POS1	2017				●●●●	TAC reduced in 2012
Rig	SPO1, SPO8	2019					-
Rig	SPO2	2019	●●	●●	●●●	●●	-
Rig	SPO3	2019	●	●●●	●●●	■	-
Rig	SPO7	2019	●●	●●●	●●●	●●	-
School shark	SCHN/1E, SCH7, SCH8, SCH1W	2018	●	●●	●●●	■	-
School shark	SCH2/3N, SCH4	2018					-
School shark	SCH3S/5	2018	■■	●●	●●●	■■	-
Skate - rough	RSK1, RSK3, RSK7, RSK8	2007					-
Skate - smooth	SSK1, SSK3, SSK7, SSK8	2007					-
Spiny dogfish	SPD1, SPD3, SPD7, SPD8	-					-
Spiny dogfish	SPD4, SPD5	-					-

***** denotes Highly Migratory Species, for which stock status cannot be determined for the portion of the stock found within New Zealand waters.

NOTES

At or above target levels? The ‘at or above target levels’ indicator describes the present status of the stock relative to its target (usually B_{MSY}, the average biomass associated with a maximum sustainable yield (MSY) strategy, or F_{MSY}, the associated fishing mortality, or appropriate surrogates or proxies for these metrics, or alternative reference points that will result in higher average biomass – see Maximum Sustainable Yield Harvest Strategies for definitions and explanations of these terms).

Below the soft limit? Below the hard limit? Overfishing? In April 2009, the Ministry’s Stock Assessment Methods Working Group adopted a probabilistic scale for categorising the ‘at or above target levels’, ‘below the soft limit’, ‘below the hard limit’ and ‘overfishing’ indicators (based on the scale developed by the Intergovernmental Panel on Climate Change (IPCC) in 2007). While these probability categories are best applied in situations where models give appropriate quantitative outputs, they can also be used subjectively, based on expert opinion, when such model outputs are not available, or are highly uncertain.

The stock status table uses the IPCC criteria, coded according to the following key:

At or above target levels?	Probability	Description	Below the soft limit? Below the hard limit? Overfishing?
●●●●	> 99 %	Virtually Certain	■■■■
●●●	> 90 %	Very Likely	■■■
●●	> 60 %	Likely	■■
●	40 - 60 %	About as Likely as Not	■
■■	< 40 %	Unlikely	●●
■■■	< 10 %	Very Unlikely	●●●
■■■■	< 1 %	Exceptionally Unlikely	●●●●

Note that green circles indicate a favourable status, while orange squares indicate an unfavourable status, with the number of circles or squares indicating the degree to which the status is favourable or unfavourable.

Whether or not a stock is likely to be at or above the target level, or to be below the soft or hard limits, or subject to overfishing, is based on the most recent stock assessment summarised in the Ministry’s Fishery Assessment Plenary Reports. The current (2018) stock status may be better or worse than that indicated by the most recent stock assessment. Where several alternative assessment runs are reported (as is frequently the case), or if the assessment results are contentions, the result reported represents the best judgement on the part of the Chair of the appropriate Fisheries Assessment Working Group, and the Ministry’s Principal Advisor Fisheries Science.

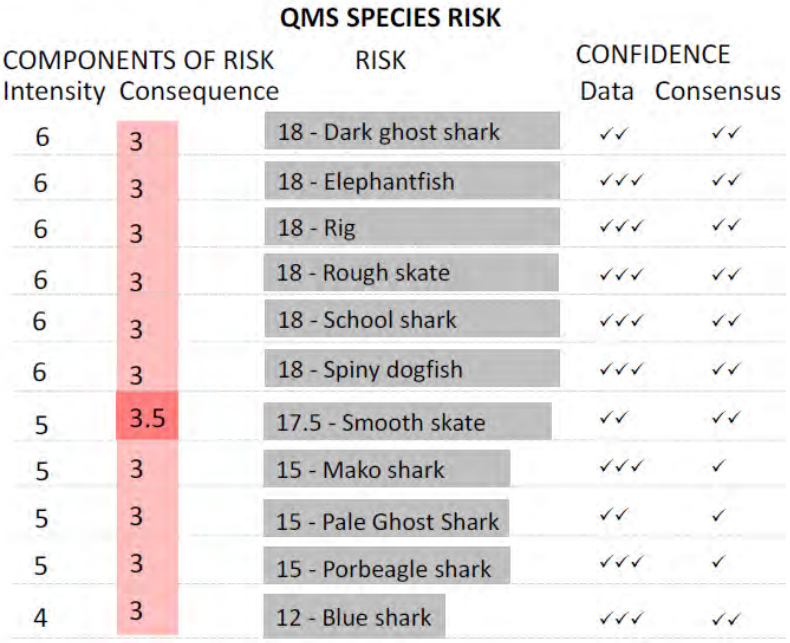
Corrective management action: This column describes corrective management action underway for those stocks believed to be below the target level, or the soft or hard limits, or subject to overfishing.

Grey shading indicates that stock status is unknown, because an appropriate quantitative analysis to ascertain stock status relative to a target or limit has not been undertaken, or because such an analysis was not definitive, generally because of insufficient or inadequate data.

Source: based on the Status of the Stocks 2017 data published by the Ministry for Primary Industries on its website (https://www.mpi.govt.nz/growing-and-harvesting/fisheries/fisheries-management/fish-stock-status/).

Appendix 19.8.3: QMS chondrichthyan risk assessment results from Ford et al. (2018). [Continued on next pages]

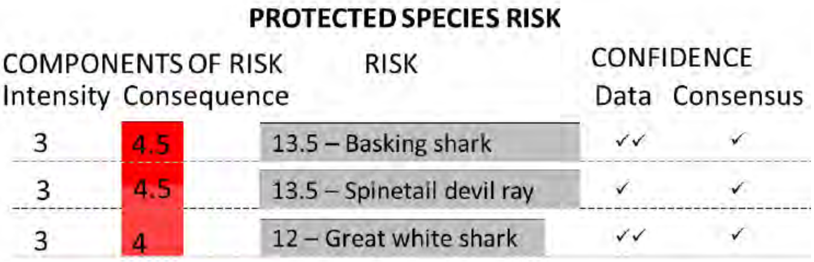
QMS species risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus (Two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented in descending order of consequences and then alphabetically



QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
6	3	18 - Dark ghost shark	✓✓	✓✓
6	3	18 - Elephantfish	✓✓✓	✓✓
6	3	18 - Rig	✓✓✓	✓✓
6	3	18 - Rough skate	✓✓✓	✓✓
6	3	18 - School shark	✓✓✓	✓✓
6	3	18 - Spiny dogfish	✓✓✓	✓✓
5	3.5	17.5 - Smooth skate	✓✓	✓✓
5	3	15 - Mako shark	✓✓✓	✓
5	3	15 - Pale Ghost Shark	✓✓	✓
5	3	15 - Porbeagle shark	✓✓✓	✓
4	3	12 - Blue shark	✓✓✓	✓✓

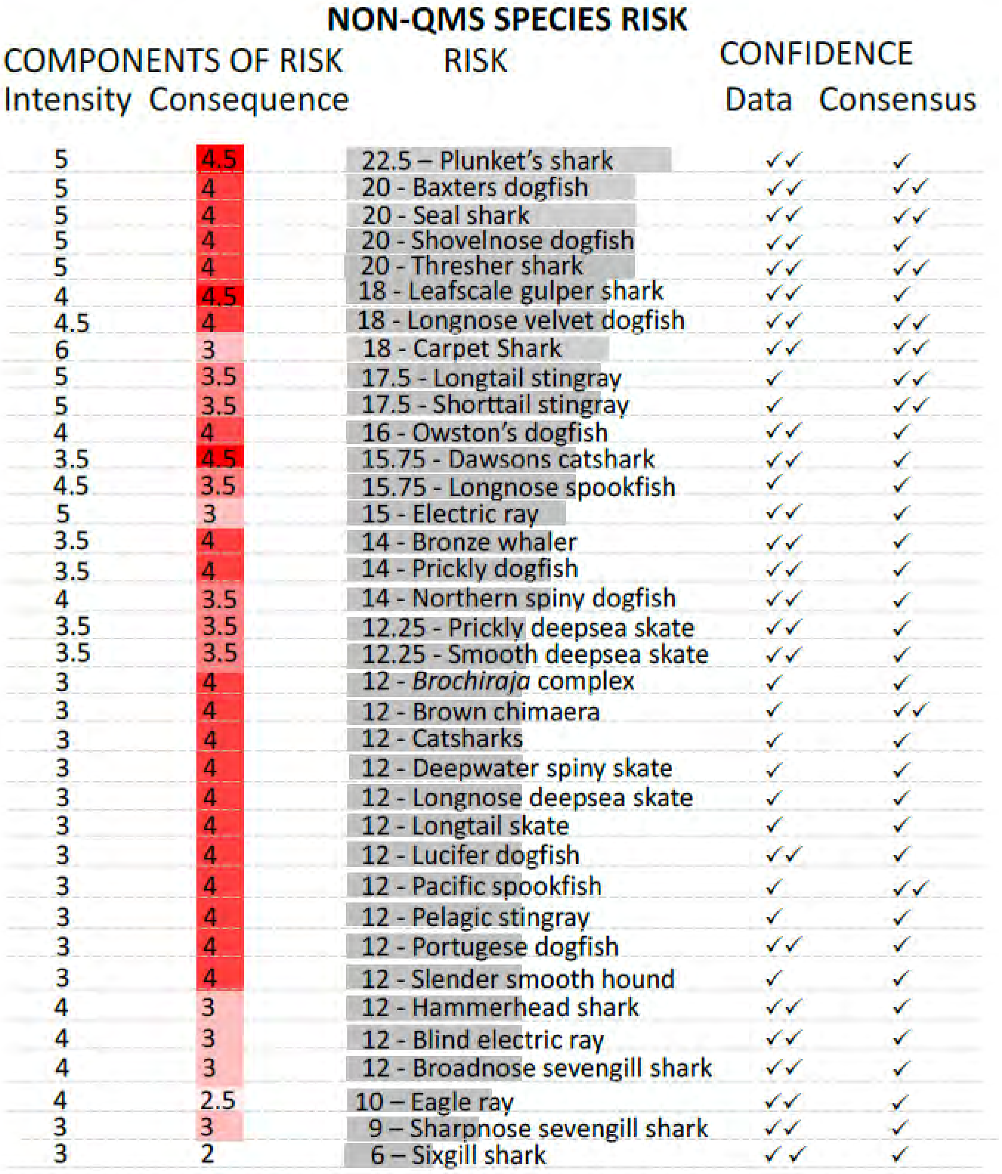
Protected Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus and a cross indicates a lack of consensus (Two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented so that higher consequences are reported first and then taxa are in alphabetical order. Taxa that scored less than three for consequence were not scored further, see Section 2.3 for more details. See Ford et al. (2015) for available data on shark species not listed in the table above.



PROTECTED SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
3	4.5	13.5 – Basking shark	✓✓	✓
3	4.5	13.5 – Spinetail devil ray	✓	✓
3	4	12 – Great white shark	✓✓	✓

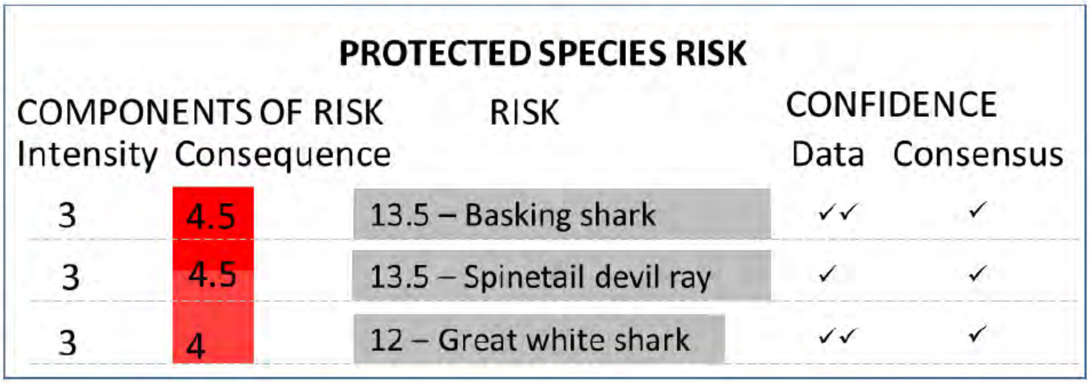
Non-QMS Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus (Two ticks in the consensus column indicate full consensus). Where taxa risk scores were identical they are presented so that higher consequences are reported first and then in alphabetical order. Taxa that scored less than three for consequence were not scored further, see Section 2.3 for more details. See Ford et al. (2015) for available data on shark species not listed in the table above.



NON-QMS SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
5	4.5	22.5 – Plunket's shark	✓✓	✓
5	4	20 - Baxters dogfish	✓✓	✓✓
5	4	20 - Seal shark	✓✓	✓✓
5	4	20 - Shovelnose dogfish	✓✓	✓
5	4	20 - Thresher shark	✓✓	✓✓
4	4.5	18 - Leafscale gulper shark	✓✓	✓
4.5	4	18 - Longnose velvet dogfish	✓✓	✓✓
6	3	18 - Carpet Shark	✓✓	✓✓
5	3.5	17.5 - Longtail stingray	✓	✓✓
5	3.5	17.5 - Shorttail stingray	✓	✓✓
4	4	16 - Owston's dogfish	✓✓	✓
3.5	4.5	15.75 - Dawsons catshark	✓✓	✓
4.5	3.5	15.75 - Longnose spookfish	✓	✓
5	3	15 - Electric ray	✓✓	✓
3.5	4	14 - Bronze whaler	✓✓	✓
3.5	4	14 - Prickly dogfish	✓✓	✓
4	3.5	14 - Northern spiny dogfish	✓✓	✓
3.5	3.5	12.25 - Prickly deepsea skate	✓✓	✓
3.5	3.5	12.25 - Smooth deepsea skate	✓✓	✓
3	4	12 - *Brochiraja* complex	✓	✓
3	4	12 - Brown chimaera	✓	✓✓
3	4	12 - Catsharks	✓	✓
3	4	12 - Deepwater spiny skate	✓	✓
3	4	12 - Longnose deepsea skate	✓	✓
3	4	12 - Longtail skate	✓	✓
3	4	12 - Lucifer dogfish	✓✓	✓
3	4	12 - Pacific spookfish	✓	✓✓
3	4	12 - Pelagic stingray	✓	✓
3	4	12 - Portugese dogfish	✓✓	✓
3	4	12 - Slender smooth hound	✓	✓
4	3	12 - Hammerhead shark	✓✓	✓
4	3	12 - Blind electric ray	✓✓	✓
4	3	12 - Broadnose sevengill shark	✓✓	✓
4	2.5	10 – Eagle ray	✓✓	✓
3	3	9 – Sharpnose sevengill shark	✓✓	✓
3	2	6 – Sixgill shark	✓✓	✓

Appendix 19.8.3 [Continued]:



PROTECTED SPECIES RISK

COMPONENTS OF RISK		RISK	CONFIDENCE	
Intensity	Consequence		Data	Consensus
3	4.5	13.5 – Basking shark	✓✓	✓
3	4.5	13.5 – Spinetail devil ray	✓	✓
3	4	12 – Great white shark	✓✓	✓

Figure 7: Protected Species Risk scores. For the COMPONENTS OF RISK higher numbers indicate greater intensity or consequence of impact (for more details see Table 3 and Table 4). For RISK longer bars and larger numbers indicate higher risk, and for CONFIDENCE more ticks indicate higher confidence in the data, or greater consensus and a cross indicates a lack of consensus (Two ticks in the consensus column indicate full consensus). Where species scored identical risk scores they are presented so that higher consequences are reported first and then taxa are in alphabetical order. Taxa that scored less than three for consequence were not scored further, see Section 2.3 for more details. See Ford et al. (2015) for available data on shark species not listed in the table above.